The Effects of Clearcut Logging on the Stream Biology of the North Fork of Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA

-- 1986 to 1994 --

Final Report

by

Richard L. Bottorff and Allen W. Knight
University of California, Davis

Prepared for
California Department of Forestry and Fire Protection
Contract No. 8CA63802

May, 1996

TABLE OF CONTENTS

Page

I. INTRODUCTION

II. NORTH CASPAR CREEK BASIN

III. METHODS

IV. RESULTS

A. Macroinvertebrates (MI)

C. Algae

I. INTRODUCTION

The dense coniferous forests of the North Coast Range of California have been harvested for valuable redwood (*Sequoia sempervirens*), Douglas fir (*Pseudotsuga menziesii*), and other tree species for more than 100 years. Initially, the primary focus of logging activities was to efficiently fall the trees and transport them to the mill site without much concern for the sustained productivity of the renewable forest resource and other environmental components within the drainage basin. However, for some time now, it has been recognized that logging activites can have significant impacts, both short- and long-term, within the drainage basin (Salo & Cundy 1987, Meehan 1991, Naiman 1992). These impacts are not localized just to the specific areas of tree cutting, but extend downstream into the network of streams draining the logged sites. Understanding how to minimize these downstream impacts is especially vital because many North Coast streams serve as habitat for valuable salmonid fishes. There is also interest in whether cumulative effects of logging are impacting stream biological communities.

Presently, there exists a good understanding of many of the most important, potential, logging impacts on drainage basins and the value to be gained from management practices that avoid or minimize adverse impacts. However, improvements in forestry management practices and new insights into logging impacts are an ongoing evolving process, with the goal of maintaining the productivity of forested lands and the high quality of streams draining the basin. Management practices have changed from a primary focus on timber yield and harvest techniques to further considerations of slope stability, sediment production, water quality, and protection of valuable fisheries and biodiversity. Recent evidence of this change can be seen in the dramatically different forestry practices used in the N. Caspar Creek basin versus that in the S. Caspar Creek basin (Burns 1972, Krammes & Burns 1973, Tilley & Rice 1977, Rice et al. 1979, Pearce 1987).

The studies of logging impacts and management practices in the Caspar Creek basin should not be viewed in isolation, for there exists intense interest by many forest managers and researchers throughout North America to improve their knowledge of forested ecosystems (Salo & Cundy 1987, Meehan 1991, Naiman 1992). The large number of watershed projects in western North America demonstrates this interest (Callaham 1990, Meyer et al. 1993). Long-term watershed studies within the Pacific Northwest ecoregion, in addition to Caspar Creek, include the Alsea Study and H.J. Andrews Experimental Watershed, Oregon (Hall et al. 1987); Clearwater River, Washington (Cederholm & Reid 1987); and Carnation Creek, British Columbia (Chamberlin 1988, Hartman & Scrivener 1990). Similar long-term studies are being conducted in the South Fork Salmon River, Idaho (Platts et al. 1989), and in the eastern United States at the Hubbard Brook Ecosystem Study, New Hampshire (Likens et al. 1977) and Coweeta Hydrological Laboratory, North Carolina (Swank & Crossley 1988).

A. REVIEW OF POTENTIAL LOGGING EFFECTS ON THE STREAM BIOTA

Many physical, hydrological, geological, chemical, and biological factors within a drainage basin determine the nature and health of a stream, and logging activities have the potential to significantly change at least some of these (Campbell & Doeg 1989, Naiman 1992, Waters 1995). Because of the number of factors involved, the subject of logging effects on the stream biota is complex and may involve both positive and negative impacts. Regional differences in these logging effects often exist. Most stream biological studies have focused on logging effects on fishes, with fewer studies on aquatic macroinvertebrates, algae, or basic measures of stream structure and function.

One of the most important insights gained during the past 20 years of research on stream biology has been the realization that the structure and function of stream ecosystems are closely linked with their drainage basins and the terrestrial vegetation, especially plants in the riparian zone (Cummins 1974, 1992; Hynes 1975; Vannote et al. 1980, Haefner & Wallace 1981, Knight & Bottorff 1984, Gregory et al. 1991, Naiman et al. 1992). Thus, logging activities which significantly change these linkages will alter the stream communities present. Seven potential impacts of logging on streams are discussed below: (1) changes in the stream substrate composition from introduced fine inorganic sediments, (2) changes in the amount of solar radiation reaching benthic algal communities, (3) changes in the water temperature, (4) changes in the amount and type of particulate organic matter (detritus) entering the stream, (5) changes in the nutrient flows into streams, (6) changes in water discharge, and (7) changes in riparian habitat for aquatic insect adults.

1. Changes in Stream Substrate from Introduced Fine Sediments

The effects of introduced, fine, inorganic sediment (silt and sand; 0.004-2mm) on the stream biota is a complex subject which has received increased attention and study in the past 30 years. Waters (1995) provides an excellent review of the biological impacts of fine sediments in streams.

Increases in soil erosion, slope instability, and fine sediment transport may cause the most serious logging impacts on streams, with the potential for long-term damage to forest productivity, downstream water quality, and biotic communities. Basin disturbances during tree cutting, road building, tractor skidding, and cable yarding expose the soil to erosion and may trigger mass soil movements, resulting in higher levels of fine sediments entering streams (Brown & Krygier 1971, Beschta 1978, Webster et al. 1983), including those in the Caspar Creek basin (Burns 1972, Krammes & Burns 1973, Rice et al. 1979, Lisle 1989). Road construction can be an especially large source of these fine sediments (Rice & Lewis 1991, Franklin 1992, Rice 1992, Waters 1995). The increased susceptibility for landslides from clearcuts and logging roads lasts many years (Franklin 1992). The potential for adverse sediment impacts increase greatly on steep hillslopes and erodible soils, such as existed in the N. Caspar Creek basin. The effects of increased fine sediments on the stream biota can be divided into two components: suspended sediments and deposited sediments.

Suspended fine sediment has several biotic effects in small streams, though these may be transitory, during and shortly after storm events. Suspended sediment reduces light to the benthic alage, decreasing primary production, algal diversity, and the food supply for grazing invertebrates (Ryan 1991). Stream invertebrate numbers and biomass are decreased by suspended sediment (Wagener & LaPerriere 1985, Culp et al. 1986, Ryan 1991). Those forms which strain food particles from the flowing water are harmed when their filtering apparatus is overwhelmed by inorganic particles (Gammon 1970, Lemly 1982, Aldridge et al. 1987, Campbell & Doeg 1989). Suspended sediment can damage invertebrates by abrasion, and by covering gills and diffusion surfaces (Lemly 1982). Invertebrate drifting behavior is increased by suspended sediments (Rosenberg & Wiens 1975, 1978, Doeg & Milledge 1991, Ryan 1991, Waters 1995). Saltation of particles along the substrate apparently scours the invertebrate fauna (Culp et al. 1986).

Deposited, fine, inorganic sediment from logging activities may cause the most significant long-term impacts on stream invertebrates because these animals are closely linked with the channel substrate. Most stream invertebrates inhabit the bottom substrate, and the composition and character of the community is largely governed by the sizes and types of substrate particles present (Cummins et al. 1966, Cummins & Lauff 1969, Erman & Erman 1984). In typical, undisturbed, small streams, a coarse substrate (gravel, pebble, cobble; 2-256 mm) forms a three-dimensional matrix of habitats for invertebrates, especially for the abundant stream insect orders Ephemeroptera, Plecoptera, and Trichoptera (EPT). This matrix includes numerous surfaces, void spaces, cracks, crevices, narrow channels, and small openings in the upper substrate layers which invertebrates occupy according to specific preferences and adaptations. If void spaces exist deep below the substrate surface (the hyporheic), invertebrates will also inhabit this space (Williams 1984). Three-dimensional, complex, coarse substrates allow diverse invertebrate communities to exist (Bottorff 1990). Coarse substrates also collect organic detritus which is an important food source for many invertebrates (Rabeni & Minshall 1977).

In contrast, the entry of fine sediments into streams fills the network of small pore spaces occupied by invertebrates (especially by the EPT) and converts the substrate into a uniform, two-dimensional, unstable habitat which few invertebrates species can utilize (Chutter 1969). The impact of introduced fine sediment is especially great in low-gradient, low-flow streams with limited ability to transport sediments (Murphy & Hall 1981). The coarser particle sizes prefered by invertebrates become embedded in the fine sediment, reducing invertebrate densities. Invertebrate diversity is greatly reduced by fine sediment, particularily for the EPT (Chutter 1969, Lemly 1982, Murphy et al. 1986, Allen 1995, Waters 1995). Fine sediments infiltrate into the hyporheic habitat, reducing or sealing off its use by invertebrates in logged streams (Trayler 1996). Invertebrate drift behaviors are altered and upstream larval movements are more difficult on fine sediments (Leudtke & Brusven 1976). Invertebrate eggs and pupae which are fixed to coarse substrate particles can be smothered by sediment (Rutherford & Mackay 1986), as can organic detrital food sources (Webster & Waide 1982) and the benthic algae. These detrimental fine sediment impacts also effect stream amphibians (Corn & Bury 1989).

Nevertheless, there are a few invertebrate groups (e.g. Tubificidae and Chironomidae) adapted to burrowing in fine substrates and they can become very abundant if organic matter is also present. Thus, the literature on fine sediment impacts in streams occasionally report increased invertebrate densities (Lemly 1982, Ryan 1991), but usually report decreased densities (Tebo 1955, Cordone & Kelly 1961, Chutter 1969, Bjornn 1974, Campbell & Doeg 1989), particularily for the EPT. Deposited fine sediment causes a dramatic shift in the taxonomic composition of the invertebrate community, from the larger-sized EPT which are accessible to fish as food, to smaller-sized burrowing forms which are largely inaccessible to fish (Grubaugh & Wallace 1995, Waters 1995).

Tebo (1955) found that the invertebrate fauna in a North Carolina logged stream recovered after a flood removed the fine sediments. Waters (1995) and Ryan (1991) also report that if the entry of fine sediment is stopped, flood events will flush the fine deposits from high gradient streams and the invertebrate populations (EPT) will recover. Therefore, stream gradient partly determines if the sediment impacts are short- or long-term. Aquatic insect habitats in heavily sedimented streams can also be improved by various in-stream alterations which increase sediment transport Luedtke et al. 1976).

2. Changes in Solar Radiation: Effects on Algae

Riparian vegetation often determines the amount of solar radiation reaching the benthic algal community in small headwater streams, and if nutrients are not limiting, the level of primary production possible (Minshall 1978, Gregory et al. 1987, 1991). This is especially true of the light-limited, small streams flowing through the dense evergreen forests of the North Coast of California (Triska et al. 1983, Hill & Knight 1988). These streams often receive less than 5% of total solar radiation (Burns 1972, Gregory et al. 1991). Logging activity adjacent to these streams can reduce or eliminate the overlying canopy of vegetation, immediately increasing the solar radiation reaching the stream and its substrate. This effect is most pronounced when clearcut logging removes all trees from the riparian buffer zone.

Normally, the algal community of light-limited, nutrient poor, streams is dominated by microscopic diatoms (Hansmann & Phinney 1973, Hill & Knight 1988, Naiman et al. 1992). Opening the forest canopy over small streams has been shown in many field and laboratory studies to increase the primary production and standing crop of benthic algae (Lyford & Gregory 1975, Murphy & Hall 1981, Murphy et al. 1981, Hawkins et al. 1982, Webster et al. 1983, Bilby & Bisson 1992, Ulrich et al. 1993), and often leads to changes in species composition, most visibly seen as large growths of green filamentous algae (McIntire & Phinney 1965, Brocksen et al. 1968, Brown & Krygier 1970, Likens et al. 1970, Hansmann & Phinney 1973, Graynoth 1979, Shortreed & Stockner 1983, Lowe et al. 1986, Noel et al. 1986, Campbell & Doeg 1989, Holopainen et al. 1991, Lamberti et al. 1991). Filamentous algal mats have been observed in N. Caspar Creek after logging (Cafferata 1990a). The diatom community also responds to light changes by shifting its species composition (Hansmann & Phinney 1973). In some cases, riparian vegetation removal can completely change stream metabolism from heterotrophic to autotrophic (Gelroth & Marzolf 1978, Holopainen et al. 1991). There is also some evidence that increased light causes the number of algal species to increase (Holopainen et al. 1991).

Increased algal production in open streams has often been found to translate directly into higher trophic levels, including both invertebrates and fish. When streams with open or closed canopies are compared, invertebrate densities or biomass are higher in open canopy streams (Erman et al. 1977, Newbold et al. 1980, Murphy et al. 1981, 1986, Murphy & Hall 1981, Hawkins et al. 1982, 1983, Silsbee & Larson 1983, Gurtz & Wallace 1984, Duncan & Brusven 1985, Murphy et al. 1986, Noel et al. 1986, Carlson et al. 1990, Burton & Ulrich 1994), especially for macroinvertebrates which feed on algae (Cushing et al. 1983, Webster et al. 1983, Wallace & Gurtz 1986, Bilby & Bisson 1992). However, invertebrate diversities in open canopy streams may be lower (Graynoth 1979, Newbold et al. 1980, Burton & Ulrich 1994), higher (Murphy & Hall 1981, Silsbee & Larson 1983), or unchanged (Noel et al. 1986, Carlson et al. 1990, Growns & Davis 1991). Lower biodiversity may result when a stream is simplified, and the increased light and nutrients stimulates increased production by a few taxa (Bisson et al. 1992). Winterbourne (1986) found few differences in macroinvertebrates between streams with and without riparian buffer zones; however, macroinvertebrates in open streams made greater use of autochthonous production.

3. Changes in Solar Radiation: Effects on Water Temperatures

Shade from riparian vegetation moderates stream temperatures, often preventing excessive summer temperatures that may be lethal to aquatic invertebrates and fish. Opening the forest canopy by logging near small streams has been found to significantly increase summer water temperatures (Levno & Rothacher 1967, Brown & Krygier 1970, Likens et al. 1970, Swift & Messer 1971, Burns 1972, Krammes & Burns 1973, Patten 1973, Gelroth & Marzolf 1978, Graynoth 1979, Rishel et al. 1982, Webster & Waide 1982, Webster et al. 1983, Barton et al. 1985, Noel et al. 1986, Beschta et al. 1987, Holtby 1988a, b, Holopainen et al. 1991), including those in N. Caspar Creek (Cafferata 1990a, b). The effect on winter water temperatures is usually less pronounced, with some studies reporting slight increases (Likens et al. 1970, Holtby 1988b) or decreases (Aubertin & Patric 1974, Lee & Samuel, Graynoth 1979, Naiman et al. 1992) from normal values. In clearcut areas where small streams are protected with riparian buffer zones, the water temperatures remain essentially the same as in undisturbed basins, or change only slightly (Brown & Krygier 1970, Swift & Messer 1971, Graynoth 1979, Rishel et al. 1982).

Water temperature affects many important stream functions, such as processing rates of organic matter, chemical reactions, metabolic rates of macroinvertebrates, and cues for life cycle events. Because of these complex interactions it is difficult to assess the ultimate effects of shade removal and water temperature changes on stream animals. Stream invertebrates have different tolerances for water temperature variations, but most species in headwater streams are narrowly adapted for cool temperatures (Hynes 1970), and increased summer temperatures may stress or eliminate some taxa.

4. Changes in Particulate Organic Matter

The entry of particulate organic matter (detritus) into streams is important to the aquatic biota for two entirely different reasons, (a) as food for many invertebrates and (b) as physical structure for the stream. Understanding the importance of particulate organic matter to streams has been a major result of research in the past two decades (Cummins 1974, Harmon et al. 1986, Bisson et al. 1987, Cummins et al. 1989, Merritt & Cummins 1996).

Riparian vegetation often supplies large amounts of dead organic matter (leaves, needles, branches, logs, etc.) to streams, which is utilized as a food source by many stream invertebrates (Anderson et al. 1978, Anderson & Sedell 1979, Cummins & Klug 1979, Naiman & Sedell 1980, Cummins et al. 1989, Wotton 1994). Both woody and non-woody detritus is consumed, with different invertebrates adapted to each. The detrital food source often accounts for most of the energy flow in stream ecosystems, especially in small headwater streams of forested basins. The organic detritus is commonly divided by size into two categories, coarse particulate organic matter (CPOM, > 1mm) and fine particulate organic matter (FPOM, < 1mm).

Many of the stream invertebrates possess complex anatomical structures, behaviors, and life cycle events which are adapted spatially and temporally for using different kinds and sizes of organic detritus as food. Invertebrates known as Shredders consume CPOM, while Collectors consume FPOM (Cummins 1973, Cummins and Klug 1979, Anderson & Sedell 1979, Cummins et al. 1989). Collectors are further divided into Collector-gatherers which use deposited FPOM, while Collector-filterers strain suspended FPOM from the water. Logging practices which alter the normal flow of organic detritus into streams will shift the trophic structure and relative abundances of invertebrate feeding guilds (Haefner & Wallace 1981, Webster et al. 1983). Logging can decrease the quantities of CPOM input and change the kind of inputs to earlier successional stages, causing changes in the Shredders and Collectors (Wallace et al. 1970, Wallace & Merritt 1980, Haefner & Wallace 1981, Webster & Waide 1982, Culp 1988, Culp et al. 1983, Culp & Davies 1983, Webster et al. 1983, Gurtz & Wallace 1984).

Attitudes about the value of large woody debris in streams have changed considerably in recent years (Bisson et al. 1992). Under past forestry practices, woody debris was routinely removed from streams to aid fish movements and to improve water quality; however, these actions simplified the channels, altered the flow patterns, decreased the ability of streams to retain nutrients, detrital food, and sediments, and reduced macroinvertebrate and fish densities (Bilby 1984, Elliot 1986, Harmon et al. 1986, Bisson et al. 1992). The input of large woody debris into streams, especially into smaller streams, adds complexity and physical structure which can divert or slow the current, create pools and riffles, and retain debris and sediment in small debris dams (Bryant 1983, Harmon et al. 1986, Andrus et al. 1988). In small headwater streams which lack large cobbles and boulders, large woody debris is very important in determining the character of the streamflow and the habitats available to invertebrates (Maser & Sedell 1994). Some of these stream macroinvertebrates are adapted to consuming wood debris (Anderson et al. 1978).

Small streams in coniferous basins of the Pacific Northwest, including N. Caspar Creek, are especially influenced by large woody debris (Murphy & Hall 1981, Bryant 1983, Bilby 1984, Andrus et al. 1988, O'Connor & Ziemer 1989). At N. Caspar Creek, coarse woody debris originates from streambank erosion and windthrow (O'Connor & Ziemer 1989), predominantly of Douglas fir (*P. menziesii*) and grand fir (*Abies grandis*). This debris is important in forming pools in N. Caspar Creek.

5. Changes in Nutrient Flow

Forest cutting changes the biogeochemistry of a drainage basin, often resulting in higher concentrations of various ions and nutrients in the streams draining the cut areas (Likens et al. 1970, 1977; Brown et al. 1973, Vitousek & Melillo 1979, Webster & Waide 1982, Swank 1988, Hartman & Scrivener 1990, Holopainen et al. 1991). Some of these return to pre-cutting levels within a few years (Bormann et al. 1974, Likens et al. 1978); however, other increases are delayed months or years from the disturbance and remain elevated for many years (Swank 1988).

Increased levels of nutrients such as nitrates and phosphates can stimulate algal growths, especially in logged areas with increased light levels and water temperatures (Triska et al. 1983, Hill & Knight 1988). Nitrates and phosphates are often the limiting nutrients in the coastal ecoregion (Stockner & Shortreed 1976, 1978, Shortreed & Stockner 1983, Hem 1985). Streams along the north coast of California often experience large increases in nitrate concentrations during fall storm runoffs (Triska et al. 1983). The combination of increased nutrients, water temperature, and light probably cause the mats of green filamentous algae that commonly are observed in many streams after forest cutting (Brown & Krygier 1970, Likens et al. 1970, Graynoth 1979).

Increases in nutrients and benthic algae usually translates into higher overall invertebrate densities, not only in the Scraper feeding group, but also in the Collectors and Predators (see references in Section 3 above). Increased nutrients also tend to increase leaf decay rates by stimulating instream bacterial and fungal communities which colonize organic matter (Kaushik & Hynes 1971, Webster & Benfield 1986), though small increases in nitrate may not always increase rates (Triska & Sedell 1976). Dissolved organic matter originating from the forested basin also enters streams in surface or groundwater and is a valuable food source for microbes and some stream invertebrates (Wotton 1994). Dissolved organic carbon in streams draining clearcut basins can be higher (Holopainen et al. 1991) or lower (Meyer & Tate 1983, Meyer et al. 1988) than that in undisturbed basins.

6. Changes in Water Discharge

The hydrologic characteristics of streams are governed mainly by the climate of a region, modified by topography, geology, vegetation, and other factors. The hydrologic regime of a stream typically follows a somewhat repeatable annual cycle, with periods of high and variable discharge and periods of low and constant discharge. Logging activites are known to alter at least some aspects of the hydrologic regime of receiving streams, often increasing streamflow (Harr et al. 1975, Webster et al. 1983), but the effects vary by region and sometimes the changes are minimal or conflicting (Ziemer 1981, Golladay et al. 1987, Swank et al. 1988, Campbell & Doeg 1989). As partly demonstrated in the Caspar Creek basin, road building and logging can increase flow volume and peak discharges, and decrease lag times for peak flows (Ziemer 1981, Wright et al 1990, Keppeler & Ziemer 1990). Some of these effects may be of short-term duration.

Invertebrate populations in streams reflect the hydrologic conditions. It is well known that stream invertebrates vary widely in their preferred current velocities, riffle or pool habitats, and abilities to withstand or recover from floodflows (Hynes 1970, Merritt & Cummins 1996). During periods of very high discharges, invertebrates have been observed to move into slower currents near stream banks or to penetrate deeper into the hyporheic zone. Some species appear to time their life cycle so they are in dormant stages during stressful periods, while others have the reproductive ability to rapidly recover from flood washouts (Lamberti et al. 1991).

Although logging is known to alter the hydrologic regime, the overall effect on the invertebrate population is unclear, partly because the population has the abililty under natural conditions to withstand or recover from very large fluctuations in flow. However, changes in flow do change velocity patterns, channel dimensions, frequency of pools and riffles, and substrate composition, all of which are important components of invertebrate habitat. Since each invertebrate species has a preferred habitat, some species may benefit while others are harmed by flow changes. Alterations in flow are closely linked with the impacts of fine sediment transport discussed above.

7. Changes in Riparian Habitat: Effects on Aquatic Insect Adults

Most aquatic insects, including those in N. Caspar Creek, utilize terrestrial vegetation during part of their life cycle, most commonly as adults. Normally, the vegetation immediately adjacent to the stream is most heavily used, but winged adults can disperse considerable distances away from the stream. Riparian vegetation is heavily used by some aquatic insects for emergence and pupation, and by adults for feeding, resting, hiding from predators, mating, and egg laying (Erman 1984). The greatest use of streamside vegetation occurs from spring to autumn, but some use continues through winter. Stonefly adults use riparian vegetation as a drumming platform to locate mates, the vibrational signals being transmitted along the plant. Some stonefly adults must feed on the thin layer of lichens, fungi, and mosses encrusting tree bark before they are able to produce viable eggs. Some insect adults lay eggs on plants overhanging the stream so that hatching larvae will fall back into the stream. This method of egg laying is used by the alderfly (*Sialis*) in Caspar Creek. It is difficult to predict what effect logging might have on these myriad and complex life history events; however, it should at least be known that the riparian vegetation is heavily used by aquatic insects, and it may be that some of these insect-plant interactions are highly evolved and determine the success of particular species.

B. OBJECTIVES OF THE N. CASPAR CREEK BIOLOGICAL STUDY

The primary objective of the N. Caspar Creek biological study was to determine if logging treatments (1989-1991) within the drainage basin caused changes in three components of stream structure and function, (1) the benthic macroinvertebrate community, (2) leaf litter processing rates, and (3) the benthic algal community. This report describes the results of eight years of study (1987-1994) on the stream biology of N. Caspar Creek, including three pre-treatment years and five post-treatment years. Additional results were obtained during a preliminary study year in 1986.

The numerous interacting factors govering the character of a stream made it impossible to predict *a priori* the actual biological changes that would occur in N. Caspar Creek from specific logging treatments. Because of the steep hillslopes and erodible soils in the N. Caspar Creek basin, we expected the main logging impacts to be caused by increased quantities of fine inorganic sediments entering the stream. Other possible logging impacts were expected from light, water temperature and nutrient increases, and in changes to allochthonous inputs of detritus; however, the presence of riparian buffer zones was expected to moderate all of these changes. Additionally, the expected changes due to fine sediment may be partly or totally nullified by opposing changes from opening the stream to increased solar radiation.

Based upon our review of the literature on potential logging effects and our judgement of which logging impacts were most likely to affect the stream biota of N. Caspar Creek, we made several predictions of how the stream biota would respond. These predictions formed the specific objectives of this report because data bearing upon each prediction was examined and statistically tested.

A. Increased Fine Inorganic Sediments from Logging
    1. Macroinvertebrate density will decrease.
    2. Macroinvertebrate number of taxa will decrease.
    3. EPT density will decrease.
    4. EPT number of taxa will decrease.
    5. Chironomidae density will decrease.

B. Increased Light, Water Temperature, & Nutrients from Logging
    6. Macroinvertebrate density will increase.
    7. Chironomidae density will increase.
    8. *Baetis* density will increase.
    9. Scraper relative abundance will increase.
    10. Leaf decay rates (k) will increase.
    11. Algal chlorophyll-a will increase.
    12. Algal biomass will increase.
    13. Algal species composition will change.

C. Decreased Allochthonous Detritus (CPOM) from Logging
    14. Shredder relative abundance will decrease.

II. NORTH CASPAR CREEK BASIN

A. BASIN CHARACTERISTICS

The Caspar Creek drainage basin is located about 10 km south-southeast of Fort Bragg, California, within the Jackson Demonstration State Forest managed by the California Department of Forestry and Fire Protection. Caspar Creek flows about 12 km from east to west, discharging into the Pacific Ocean near the town of Caspar. The drainage basin is forested with dense stands of second growth redwood, Douglas-fir, hemlock, grand fir, and several hardwoods. Clearcut logging and burning first occurred at Caspar Creek in the late 1800's, giving the present forest an age of 80-100 years. The Caspar Creek basin lies within the Pacific Northwest coastal ecoregion (Naiman et al. 1992) and has forests, climates, and topography somewhat similar to coastal Oregon and Washington.

Research has been conducted at Caspar Creek since 1962 mainly by the California Department of Forestry and Fire Protection, Fort Bragg, and the US Forest Service, Redwood Sciences Laboratory, Arcata, but also by others from the California Department of Fish and Game and Humboldt State University. Two smaller watersheds within the Caspar Creek drainage, North Fork Caspar Creek (473.3 ha) and South Fork Caspar Creek (423.7 ha), have been studied to determine the effects of different logging practices on streamflow characteristics, sediment transport, slope stability, and several environmental factors, especially anadromous runs of steelhead and salmon. Both research watersheds have been fitted with numerous instruments to collect basic data on hydrology, climate, and sediment transport (Krammes & Burns 1973, Tilley & Rice 1977, Rice et al. 1979, Ziemer 1981, Pearce 1987, Lisle 1989, 1995, Cafferata 1990a, Keppeler & Ziemer 1990, Thomas 1990, Wright et al. 1990, Keppeler et al. 1994).

The North and South Caspar Creek drainage basins were originally selected for study because of their closeness and similarity, which allowed the untreated watershed to serve as a control for the treated watershed. The South Caspar Creek watershed received logging treatments (60% of timber volume removed) from 1967-1973, while the North Caspar Creek watershed served as a control.

For the second phase of research from 1985-1994, ten smaller units (about 50% of the total area) within the North Caspar Creek watershed were clearcut logged and treated with additional forestry practices. In addition to the continuing studies of logging treatment impacts on streamflow characteristics and sediment transport, this phase of the project also examined the treatment effects on the stream biology of N. Caspar Creek.

Dr. Allen W. Knight, Department of Land, Air, and Water Resources, University of California, Davis, was initially contacted by the California Department of Forestry and Fire Protection in 1985 about doing stream biological research at N. Caspar Creek. A visit to the drainage basin was made in April, 1985, and a research proposal prepared. Preliminary stream biology work started in 1986 in N. Caspar Creek; however, drought conditions caused several study tributaries to cease flowing, requiring changes in the research design. By 1987 a methodology for sampling three different components (aquatic macroinvertebrates, leaf decay rates, and algae) of the stream was developed and the sampling was continued until 1994.

The portion of the N. Caspar Creek drainage basin studied in this report ranged in elevation from about 82m to 305m. Five biological sampling stations (B, C, G, H, and I) were located along 1450 m of N. Caspar Creek, varying in elevation from 96 m to 123 m (Fig. 1). Within the study reach, N. Caspar Creek had a gradient of about 1.9%. During the low flow period of summer-autumn, typical stream velocities were 0.2-0.4 m/sec, stream widths were 0.5-2 m, and stream depths were 5-30 cm. Soils in the basin originated from Cretaceous sedimentary rocks (Tilley & Rice 1977) and were erodible.

The hydrologic regime of N. Caspar Creek was determined by the regional climate which alternates between cool, wet winters and mild, dry summers. The mean annual precipitation was about 1100 mm, almost all of which fell between October and April (Fig. 3). Essentially all of the precipitation was from rainfall and very little was from snowfall. The streamflows of N. Caspar Creek reflected the wet-dry pattern of precipitation. Normally, flows were low (< 0.03 cfs) and stable between May and October; however, between October and April flows increased and become more variable (Fig. 4). Low flows often extended into November, and sometimes into December. During the 1991 drought water year, low flows extended though many of the winter months (December-February).

High flows transporting sediment occurred during and after heavy rainstorms in winter and early spring . An unusual, aseasonal, high runoff event occurred in late-May, 1990. About 12 inches of rainfall on 21-27 May 1990 resulted in a peak discharge of 150 cfs on 27 May. Normal discharges in May were < 1 cfs. The 9-year mean annual runoff (1986-1994) at Station NFC of N. Caspar Creek was 1593 acre-feet (Fig. 5). Water temperatures varied seasonally from a summer maximum of about 14°C (monthly mean) to a winter minimum of 6-8°C (Fig. 6). Water temperatures for 1992-1993 appeared to be slightly higher than the previous three years.

Water chemistry was not measured as part of this biological study, except once in the mainstem and side tributary of N. Caspar Creek at Station H (nirate-nitrogen < 0.1 ppm and phoshorus < 0.15 ppm at both locations). Dr. Randy Dalhard, University of California, Davis, is conducting more detailed studies of water chemistry.

N. Caspar Creek within the study area is normally considered to be a second order stream (based on US Geological Survey 7 1/2' topographic maps), though the reach downstream of tributary D-E might be considered third order. North Caspar Creek flowed through a dense coniferous forest and received little direct solar radiation (< 5%). Organic detritus, both woody and non-woody, was abundant in and adjacent to the stream channel. In the study area, the riparian zone contained few alder trees, though these were more common further downstream and along S. Caspar Creek.

B. WATERSHED TREATMENTS

Prior to the logging treatments starting in 1989, the N. Caspar Creek watershed had not been disturbed for 80-100 years since being logged in the late 1800's. Although 80-100 years would appear to be sufficient time for the watershed to recover to natural conditions, it was not certain if this watershed should be considered pristine. Past logging practices at N. Caspar Creek included flushing logs downstream by rapidly releasing high discharges from a splash dam located upstream of Station I. This practice must have caused considerable stream bank erosion and high sediment transport. Mass soil slides were also evident in the N. Caspar Creek watershed. Rice et al. (1979) reported a mass soil movement of 4234 cubic yards in March, 1974, originating downstream of the old splash dam. Thus, it was possible that the logging of 80-100 years ago may still be causing sediment impacts on the stream. Data such as these, and other pre-logging studies (Krammes & Burns 1973) suggest that the resident aquatic fauna of N. Caspar Creek was long ago impacted by high sediment loads and these conditions may have continued for many years. Because of these past influences, new treatments which increase sediment loads may not be detectable in the aquatic fauna. This uncertainty emphasized the value of undisturbed control watersheds for interpreting results.

In 1985 and 1986, Units Y and Z in the lower part of the N. Caspar Creek watershed were clearcut logged (Fig. 1); however, runoff from these two areas entered N. Caspar Creek downstream from all five stations of our study. Because logging in Y-Z had no impact on our biological stations, these logging treatments were not part of this study. Therefore, excluding Y-Z, the total watershed area of N. Caspar Creek was 384.3 ha at the Arfstein stream gauging station, rather than the 473.3 ha at the NFC gauging station.

No logging treatments occurred in the N. Caspar Creek watershed from 1986 to 1989, and it was this short period when pre-treatment biological sampling occurred. We believe that logging was not originally proposed by the Department of Forestry for the upper basin, and, thus, Stations H and I were initially selected to serve as controls for the study. Nevertheless, these plans were changed, and road construction and clearcut logging commenced in 1989-1990 on Units J, K, and L (Table 1, Fig. 1). This headwater logging made it much more difficult to interpret the stream biological results because all five stations, including the two proposed control stations, were located downstream from logging. All stations could have been impacted by sediment or other factors (water temperature, nutrients, solar radiation, etc.) for the remainder of the study (1989-1994).

A total of eight treatments were applied to the N. Caspar Creek watershed during 1989-1991, each treatment consisting of relatively similar methods of road construction and clearcut logging. Starting in the upper basin in 1989 and proceeding downstream until 1991, these logging treatments were applied to the following units: B, C, E, G, J, K, L, and V (Table 1, Fig. 1). The time when potential impacts of these treatments might first be observed at the 5 stations varied with the impact and location (Table 2-3). The overall treatment design was planned to determine if cumulative watershed effects were occurring in the N. Caspar Creek basin (Table 4).

Access roads to the logging unit were first constructed, followed by felling the trees. Most logging access roads were constructed near watershed boundaries and far from N. Caspar Creek, except for a few roads which penetrated the basin along ridges. Access to the research stations along N. Caspar Creek was by foot trail, thus, there were no roads near N. Caspar Creek and no road crossings. In addition, no tractor skidding occurred in or near the creek. These features of the treatment design minimizing mechanical disturbance near the creek were opposite to that used in the S. Caspar Creek basin (Burns 1972, Krammes & Burns 1973).

All treated units were clearcut, except for buffer zones along Class 1 or 2 streams, where selective cutting removed up to about 50% of the trees (Fig. 1). The widths of buffer zones varied from 50 feet to 200 feet depending on the hillslope and stream size. Class 1 streams (fish bearing) had buffer zones of either 200 feet (>70% slope), 150 feet (50-70%), 100 feet (30-50%), or 50 feet (<30%), while Class 2 streams (aquatic insect bearing) had buffer zones of either 150 feet (>70%), 100 feet (50-70%), 50 feet (30-50%), or 50 feet (<30%). N. Caspar Creek is a Class 1 stream from its lower reaches upstream to near the old splash dam above station I.

After felling, the trees were removed from the steeper slopes by cable yarding or from gentler slopes by tractor skidding. The vast majority of trees in all N. Caspar Creek units were removed by cable yarding, which causes less disruption of the forest floor than does tractor logging. Additional disturbances to the units were caused by tractor skidtrails and landing areas. Some of the units (G, E, J, and L) were burned after logging, followed by herbicide applications in 1993 or 1994 (Table 1). All of these watershed treatments were recorded by personnel of the California Department of Forestry and Fire Protection or US Forest Service, Redwood Sciences Laboratory. The condition of treated and untreated watershed units of N. Caspar Creek have been continuously monitored during this study, especially for sediment transport and slope stability during and after major rainstorms. All soil slides of more than 10 cubic yards in the basin were recorded. In conclusion, many efforts have been made in the N. Caspar Creek treatment design to minimize logging impacts on basin streams.

In the past few years since logging was completed in the N. Caspar Creek watershed, numerous trees in the buffer zones of Units B, V, and L have been blown down, many falling across N. Caspar Creek. These three buffer zones on N - NW facing slopes with exposed reaches to the south were especially vulnerable to windthrown trees. The problem of blown down trees in buffer zones has been reported by others (Steinblums 1984, Murphy et al. 1986).

C. SAMPLING SITES

During most seasons, N. Caspar Creek had a relatively small channel width of 0.5-2 m and a depth of 5-30 cm. Relatively constant, low, clear, discharges flowed in the channel from late spring until late autumn or early winter (Fig. 4). Higher and more variable discharges occurred during and shortly after rainstorm events in winter and early spring. It was during the higher storm flows when the streamflow was turbid from increased sediment transport (Rice et al. 1979). Lower winter flows had a slightly "milky" appearence. All sampling in this study occurred during the stable, low flow period of spring-fall.

The creekbed substrate was usually composed of pebbles, gravel, and smaller particles, with a few scattered cobbles. Prior to logging in N. Caspar Creek, 70-80% of streambed material was < 2.76 cm (pebble) and considerable sediment was being transported (Krammes & Burns 1973, Lisle 1989, 1995). Pools had finer substrates than riffles, and coarser substrates occurred in reaches with exposed bedrock. Many of the pebbles and cobbles were embedded in finer sediments (Fig. 2).

This study was conducted at 5 main stations located along a 1450m reach of N. Caspar Creek, each station located at the confluence of a side tributary (Figs. 1, 7-11). The five tributaries were labelled B, C, G, H, and I from downstream to upstream, respectively. North Caspar Creek drops 27m in elevation in the 1450m distance between station I and B. Samples were collected in this study immediately below and above each of the five tributary confluences with N. Caspar Creek. Starting from the lowest site and proceeding upstream, the sampling sites are described as they existed after all watershed treatments had been completed.

In this report, the term "Station" is only used to designate one of the five main collect points (B, C, G, H, I), while the term "Site" is used to designate one of the ten specific collection locations (B below, B above, C below, C above, G below, G above, H below, H above, I below, and I above).

B below (96m elevation) -- This site had about 50% pools and 50% riffles. The substrate was composed primarily of pebbles, gravel, and finer particles, with some scattered cobbles and boulders. Clearcut Unit B to the south of B was easily visible through the buffer zone, which was thinner than normal because trees have been blown down by the wind. Many of these trees have fell across the channel of N. Caspar Creek. Some of these wind felled trees have curved bases, showing the effects of past soil creep on the steep hillslopes. The forest canopy directly overhead the creek was fairly open. The clearcut area and wind toppled trees have opened up this site to more sunlight than before the logging, and direct sunlight reached the creek during parts of the day. To the north of this site, the forest was dense and dark.

B above (97m elevation) -- This site had about 70% pools and 30% riffles. The substrate was composed primarily of pebbles, gravel, and finer particles, with a few scattered cobbles. There were also several logs across the creek which provides some structure. To the south of this site, the sky was partially open from Clearcut Unit B and the thinned buffer strip. Direct sunlight reached the lower end of this site during parts of the day. On the north side, the forest was dense and dark.

C below (98m elevation) -- This site had about 60% pools and 40% riffles. The substrate was composed primarily of pebbles, gravel, and finer particles, and appeared eroded. Both the south and north sides of the creek were partly open from Clearcut Units B and C, though less so than at station B. Directly overhead, the forest canopy was very open. The south buffer zone at this site had many trees blown down by the wind, some lying across the creek and creating physical structure in the channel.

C above (99m elevation) -- This site had about 90% pools and 10% riffles. The substrate was composed primarily of pebbles, gravel, and finer particles, with a few scattered cobbles. To the south of this site, the sky was very open from Clearcut Unit B and the thinned buffer strip. Likewise, the forest canopy was very open directly overhead. Direct sunlight reached the creek during parts of the day. However, north of this site, the forest was closed and dark. Only a few windthrown trees crossed the channel. The direct sunlight often stimulated visible growths of green filamentous algae on the creek substrate.

G below (116m elevation) -- This site had about 40% pools and 60% riffles. Because of bedrock outcrops, this site had relatively coarse substrate composed of pebbles and cobbles, and these were less embedded. The creekbed was incised 2-3 m into the valley with steep channel banks. To the south of this site, the sky was fairly open from Clearcut Unit V, but little or no direct sunlight reached the creekbed because of the buffer zone, steep hillslope, and incised channel. The buffer zone had lost many trees to high winds and some of these crossed the channel. To the north, the sky was somewhat open from Clearcut Unit G, but less than from the south. Directly overhead the forest canopy was about 50% open.

G above (117m elevation) -- This site had about 50% pools and 50% riffles. It had relatively coarse substrate composed of pebbles and cobbles, and these were less embedded. The creekbed was incised 2-3 m into the valley with steep channel banks. To the south, this site was more open than at G below. The sky was fairly open from Clearcut Unit V and the thinned buffer zone. To the north, the sky was somewhat open from Clearcut Unit G, but much less than from the south. Directly overhead the forest canopy was about 40% open. The buffer zone had lost many trees to high winds and many of these crossed and shaded the channel. Like G below, this site also received little direct sunlight because it was incised into the valley and shaded by downed trees.

H below (120m elevation) -- This site had about 75% pools and 25% riffles. The substrate was relatively fine being composed of pebbles and smaller particles, and these were embedded. No cobbles were present. This site was generally shaded to the south, but Clearcut Unit V to the southwest allowed some direct sunlight to reach the creekbed late in the day. To the north, the sky was dark and closed off by the dense forest. Directly overhead the forest canopy was 20% open. Only a few small trees have been blown down by the wind.

H above (121m elevation) -- This site had about 30% pools and 70% riffles. The substrate was coarser at this site than at H below. It was composed of pebbles or slightly larger particles, some of these being embedded. This site was generally shaded and closed to the south, but Clearcut Unit V to the southwest allowed some direct sunlight to reach the creekbed late in the day. To the north, the sky was dark and closed off by a dense forest. Directly overhead the forest canopy was 15% open. No trees have been blown down at this site.

I below (122m elevation) -- This site had about 50% pools and 50% riffles. The substrate was composed of pebbles or slightly larger particles, plus a few cobbles, some of these embedded. This site was fairly open to the south from Clearcut Unit L. Directly overhead the forest canopy was about 40% open. Although the streambed was incised about 2m deep into the valley on the south side, direct sunlight reached the streambed during part of the day. To the north, the sky was closed off and dark from the dense forest. Only a few trees have been blown down across the creek.

I above (123m elevation) -- This site had about 50% pools and 50% riffles. Because of exposed bedrock, this site had a coarser substrate than at I below. It was composed of pebbles, gravel, and cobbles, some of these being embedded. This site was somewhat open to the south from Clearcut Unit L, but less than at I below. Directly overhead the forest canopy was about 40% open. Although the streambed was incised about 1-3m into the valley, direct sunlight reached the streambed during part of the day. To the north, the sky was closed off and dark from the dense forest. Only a few trees have been blown down across the creek.

D. OBSERVED STREAM DISTURBANCES

Many disturbances to N. Caspar Creek were directly observed during this study, both natural and logging-related. Some of the most obvious disturbances are summarized here.

North Caspar Creek completely dried at Station I in the fall, 1987, destroying all rock pack and clay tile samples. Thereafter, N. Caspar Creek had perennial flow at all stations, although it was very low in late-summer or autumn in some years.

In summer, 1989, during the clearcut logging in Unit L (Caspar East '89), terrestrial detritus from the cable yarding operation accumulated in the main creek channel at Station I. Some trees were felled or dragged across the channel.

Heavy rains on 22-23 October 1989 increased stream discharge at all stations on 23-24 Oct., just prior to collecting the rock pack, leaf pack, and clay tile samples. Thus, some samples were lost and the 1989 leaf decay field studies were halted after this higher runoff. During the storm, fine sediment accumulated at I above and I below.

An unusual series of late rainstorms of 12 inches in 19-30 May 1990 caused flood discharges (peak flow = 150cfs, 27 May). Normally, May discharges were < 1 cfs. This aseasonal runoff destroyed all rock pack and clay tile samples, plus the solar radiation equipment of Harrison Phipps. The channel characteristics at several sites were altered by the flood. Site I below was dramatically changed because a small debris dam downstream of the bridge was washed out. The pool downstream was broadened and filled with fine sediments. Overall, the channel became straighter, wider, and the depth decreased from sediment deposition.

In summer-fall, 1990, during the clearcut logging in Unit V (Rice '90), terrestrial detritus from the cable yarding operation accumulated in the main creek channel at Station G and just downstream of Station H. The detritus mainly was branches, limbs, bark, and needles broken from nearby trees by the cable as the logs were transported to the landing sites. This terrestrial detritus was present in the stream channel when the fall, 1990, samples were collected. Some of the rock pack and clay tiles were buried under this debris at Station G.

A heavy rainstorm on 19-20 January 1993 caused the highest peak discharge (242 cfs, 20 Jan) during this study. We did not directly observe this event and samples were not in the stream at the time; however, considerable sediment must have been transported.

Side Tributary YZ draining two areas clearcut prior to this study contributed sediment to the lower section of N. Caspar Creek.

Table 1. Timing and area of watershed treatments in N. Caspar Creek basin, Jackson Demonstration State Forest, Fort Bragg, CA.

	Road Construction[1]			Clearcutting			Other Treatments	
Unit	Start	Finish	Area (ha)	Start	Finish	Area (ha)	Burning	Herbicides
B	9/91	10/91	0.84	9/91	11/91	25.90		
C	9/91	10/91	1.06	9/91	1/92	27.11		
E	4/90	6/90	1.32	9/90	10/91	27.92	11/91	3/93 3/94
G	3/90	9/90	2.36	5/91	9/91	36.42	11/91	3/94
H		NONE			NONE		NONE	
I		NONE			NONE		NONE	
J			0.66	8/89	4/90	25.98	11/90	2-3/93
K	5/89	6/89	1.06	5/89	11/89	17.24		
L	5/89	6/89	0.24	5/89	9/89	12.46	11/90	3/93
V	4/90	6/90	0.54	6/90	10/90	14.97		
Total Area (ha)			8.08			188.00		
(%)[2]			2.1			48.9		

---

[1] Road construction = Roads, Skid Trails and Landings

[2] % of basin upstream of Station A (area = 384.3 ha)

Table 2. First possible sampling dates when impacts from treatments within a watershed unit might be expected to be detected at the five main stations of N. Caspar Creek, Jackson Demonstration Forest, Fort Bragg, CA. These are treatments that might be expected to cause differences between Above and Below sites at a station.

Station	Sediment Impacts		Water Temperature Impacts
	Road Construction	Clearcutting	
B	Spring 92	Spring 92	Spring 92
C	Spring 92	Spring 92	Spring 92
G	Spring 90	Spring 92	Fall 1991
H	-- No Treatments in Unit --		
I	-- No Treatments in Unit --		

Table 3. First possible sampling dates when solar radiation impacts from watershed treatments might be expected to be detected at the five main stations of N. Caspar Creek, Jackson Demonstration Forest, Fort Bragg, CA.

Station	Logging Unit Causing Impact	First Sampling Date to Expect Solar Radiation Impacts
B	B	Spring 92
C	B, C	Spring 92
G	G, V	Fall 90
H	V	Fall 90
I	L	Fall 89

Table 4. Possible cumulative impacts from roadbuilding and clearcut logging treatments at the five main stations of N. Caspar Creek, Jackson Demonstration Forest, Fort Bragg, CA. The earliest possible sampling date when impacts might be detected was Fall, 1989.

Station	Drainage Basin Area (ha)	Upstream Logging Units Causing Impact	Cumulative Upstream Roadbuilding (ha)	Cumulative Upstream Clearcutting (ha)
I	120	J, K, L	1.96	55.68
H	157	J, K, L	1.96	55.68
G	205	G, V, J, K, L	4.86	107.07
C	348	B, C, E, G, V, J, K, L	8.08	188.00
B	361	B, C, E, G, V, J, K, L	8.08	188.00

III. METHODS

This report describes the results of nine years of study (1986-1994) on the stream biology of N. Caspar Creek, including one preliminary year (1986), three pre-treatment years (Spring 1987 - Spring 1989), and five treatment or post-treatment years (Fall 1989 - Spring 1994). These studies were conducted under the guidance of Dr. Allen W. Knight and several of his graduate students, Michael S. Parker (1986-1990), Dean F. Messer (1991-1992), and Richard L. Bottorff (1993-1994). Catherine L. (Arvey) Donaldson worked on all aspects of the project for many years, both in field collections and laboratory processing of data. Other graduate students involved with the N. Caspar Creek biological studies included Edward J. Connor (original proposal and preliminary sampling), Walter R. Hill (identification of algae), Angela Kost (identification of algae), Jessica Lacey (identification of algae), Harrison Phipps (light studies), John Martin (drift studies), and William J. Trush (original proposal). Many undergraduate students (100±) also helped with field collections and laboratory processing of data.

Although this biological study of N. Caspar Creek changed somewhat from that originally proposed (see Summary of Early Biological Studies), by 1987 a methodology for sampling three different components of the stream was developed and the sampling was continued until 1994. This investigation of N. Caspar Creek (1987-1994) focused on three major aspects of stream structure and function, (1) the benthic macroinvertebrate community, (2) leaf litter processing rates, and (3) the benthic algal community.

All samples were collected from five main stations (B, C, G, H, and I) along N Caspar Creek (Fig. 1), each consisting of 25-50 m of stream length. These five stations were located where smaller side tributaries entered N. Caspar Creek, each side tributary draining a watershed with different logging or forestry treatments. All samples in this study were taken directly from N Caspar Creek, both above and below the confluence point, but not from the side tributary. In this report, the term "Station" refers to the five main collecting locations in N. Caspar Creek, while the term "Site" refers to the upstream (above) or downstream (below) locations at each station. For example, "Ba" and "Bb" designate the above and below site locations at Station B, respectively.

A. <u>MACROINVERTEBRATES (MI)</u> -- Rock Packs

The abundance, diversity, and trophic composition of stream MI's was determined with the use of artificial substrate stream samplers. Artificial substrates have been advocated for sampling benthic MI's in a wide variety of aquatic habitats and have several important advantages over other methods (Rosenberg and Resh 1982). These advantages include (1) allowing samples to be collected from habitats where standard sampling methods aren't feasible, (2) standardizing sampling protocols to reduce variability caused by different people collecting the benthic samples (especially important in this long term study incorporating many people), (3) providing a simple, low cost sampling device, and (4) decreasing disturbance of the stream environment when compared with standard sampling devices.

In this study, the artificial substrates consisted of rock packs made of plastic mesh bags (15 cm x 15 cm; mesh size = 1.9 cm), each bag containing 20-25 pebbles (mean diameter = 4.6 ± 0.4 cm) gathered from the substrate of N. Caspar Creek. Twenty rock packs were placed in the stream at each of the five main study stations along N. Caspar Creek (B, C, G, H, I), ten of these being placed upstream of the side tributary confluence and ten being placed downstream. Thus, for each sampling period, a total of 100 rock packs were placed in N. Caspar Creek. Two sample periods were included in each year, one in the spring, another in the autumn. During both of these sampling periods, streamflow in N Caspar Creek was relatively low and stable, although spring flows tended to be decreasing somewhat, while fall flows were increasing. No sampling was done during the higher variable discharges of winter months because the rock packs would have been eroded and washed downstream, or buried under transported sediment. Water temperatures were increasing during the spring samples and were decreasing during the fall samples.

To minimize variability in physical conditions between sample stations, an effort was made to place rock packs in similar velocities and depths. Because stream levels fluctuated somewhat, rock packs were placed at locations where they would remain submerged during the colonization period. Each rock pack was placed in a shallow depression prepared in the streambed, and similar-sized pebbles were placed around the rock pack until its surface was level with the surrounding substrate. Rock packs were spaced at least 0.5m apart to minimize disturbance to adjacent rock packs during collection. The rock packs remained in the stream for about 60 days (Table 10), the colonization time for stream benthic MI's to reach equilibrium with that of the surrounding streambed.

After the colonization period, each rock pack was lifted into the opening of a D-frame dip net (0.6 mm mesh) placed directly downstream. After the rock pack was inside the net, the substrate within the depression was stirred to a depth of 2-3 cm, suspending the detritus and organisms present and collecting these also in the net. The rock pack and net contents were placed into a bucket containing 5-6 liters of stream water. Each pebble was taken from the rock pack and washed in the bucket to remove all MI's. Also, the plastic mesh was washed in the bucket. The accumulated detritus and MI's in the bucket were then poured through a sieve (0.33mm mesh) and preserved in 80% ethyl alcohol. This completed the field collection.

In the laboratory, all MI's in each preserved rock pack sample were separated from the inorganic and organic debris under 10X magnification, counted, and identified to the lowest possible taxonomic level. The main identification manual for the aquatic insects was Merritt and Cummins (1984, 1996) and for the non-insects was Pennak (1989). These two references were also used to place each MI taxon into a Functional Feeding Group (FFG).

Most statistical analyses of the data were done by Hong Zhou, Statistician, Statistical Laboratory, University of California, Davis, using the SAS computer package. Statistical analyses were done for MI densities and number of taxa, and on subgroupings such as (1) EPT density and number of taxa, (2) chironomid density, (3) *Baetis* density, and (4) FFG densities and relative abundance. Analysis of variance (ANOVA) was performed on these variables (see Appendix for data sets) to determine the effects of logging treatments, site position, station, season, and year. Tukey's Studentized Range (HSD) Test was used to compare pre- and post-treatment means, while the Least Squares Means Test was used to compare the means for site position, station, season, and year.

All density values were first transformed by $\log_e$ (x + 1), but none of the taxa number data were transformed. The FFG relative abundances were first transformed by arcsine (square root of FFG relative abundance (as a fraction)). All collection dates from 1987 to 1994 were used in the statistical analyses of density, taxa, and relative abundance analyses, eventhough spring, 1992, and spring, 1994, appeared to have abnormally high densities. A particularily energetic person collected the spring, 1994, samples and stirred deep into the substrate under the rock pack. This increased sample numbers, especially of chironomids at Stations B and C. The reasons for high densities in spring, 1992, are unknown, but again, chironomid densities were especially large at Stations B and C. We considered excluding these two dates from the analyses, but did not believe that sufficient reasons existed. Because season was a highly significant factor in most data sets, exclusion of two spring samples would have produced a fall bias in the results.

All ANOVA's were general linear models, with the following sources of variation:

Source of Variation
- Model
    - Pre/Post Logging (pre-treatment vs. post-treatment)
    - Site Positions (Above site vs. Below site)
    - Station (B, C, G, H, and I)
    - Season (spring vs. fall)
    - Year (1987, 1988, 1989, 1990, 1991, 1992, 1993, 1994)
- Error

For all ANOVA's, the pre-treatment period included five sample collections from spring, 1987, to spring, 1989, while the post-treatment period included ten collections from fall, 1989, to spring, 1994. Although the treatment period extended for about three years after logging first started in 1989 (Table 1), all five stations were located downstream of the 1989 logging and potentially were impacted after that date. Further division of the study into pre-treatment, during treatment, and post-treatment may yield additional insights; however, this was not done in this report.

In addition to the ANOVA's, additional statistical analyses were made (linear regressions and t-tests) and are designated in the text.

B. LEAF DECAY RATES -- Leaf Packs

The rate at which leaf litter decomposes in N. Caspar Creek was determined by monitoring the weight loss of premeasured leaf packs for 6-10 weeks in the autumn. Red alder leaves (*Alnus rubra*) were collected during the first week of September directly from trees in the riparian zone of N. Caspar Creek (along Forest Road 0620). Leaf packs were constructed by sewing 15-20 leaves together with monofilament line, dried to constant weight at room temperature, and given a numbered identification tag. The initial dry weight of leaf packs were 5-10 g.

Leaf packs were placed on the stream bottom at 3 sampling stations (B, G, and I) during the third week of September. To keep the leaf packs in place, they were attached to 2m ropes with monofilament line and the rope was anchored to the bottom. A total of either 4 or 5 leaf packs were attached to each rope and 6 leaf pack rope sets were placed at each station, three above the tributary confluence and three below. Thus, a total of 18 leaf pack sets and either 72 or 90 leaf packs were placed in the creek.

Approximately every two weeks from late-September through November (Table 18), one leaf pack was removed from each of the 18 leaf pack sets, stored in individual zip-loc bags, placed on ice, and transported to the laboratory where all of the associated macroinvertebrates were immediately separated from the remaining leaf pack. The remaining leaf pack tissue was dried at room temperature and its final weight determined.

Decomposition rates (k) of leaf packs were determined from the decay equation $W_t = W_0 e^{-kt}$ by regressing percent of initial leaf mass remaining against the time the leaf pack was in the creek. In this equation, W_t is the weight remaining, W_0 is the initial weight, t is the time (days), and k is the decay coefficient (days^{-1}) (Petersen & Cummins 1974).

Preserved macroinvertebrates were processed as described for the rock packs, counted, identified, and placed in FFG's. FFG densities were expressed as the number of individuals per gram of leaf mass.

Leaf decay rates were analyzed using a general linear model ANOVA, with the following sources of variation:

Source of Variation
- Model
  - Year (1988, 1989, 1990, 1991, 1992, 1993)
  - Station (B, G, and I)
  - Site Positions (Above site vs. Below site)
- Error

Tukey's Studentized Range (HSD) Test was used to compare year means, while the Least Squares Means Test was used to compare the means for station, and site position. Season was not a factor in the analyses because rates were only determined in the fall.

C. ALGAE -- Clay Tiles

The benthic algal community was sampled by placing unglazed clay quarry tiles (15cm x 15cm) on the stream bottom at each of the 5 main sampling stations and allowing algae to colonize for 60 days. Placement and removal of clay tiles during this study were usually made on the same dates as for the MI rock packs (Table 21). At each of the 5 main stations along N. Caspar Creek, 5 tiles were placed immediately upstream of the tributary confluence, while 5 tiles were placed immediately downstream. Thus, 10 clay tiles were present at each of the 5 main stations, giving a total of 50 algal samples on each sampling date. Tiles were dispersed to experience the full range of light levels at each station.

After the colonization period, each tile was removed from the creek and three algal samples were taken from each tile in the field, one each for determination of algal biomass, chlorophyll-a, and taxonomic composition. Algal samples were collected using a circular (diameter about 32mm), hard rubber template pressed firmly onto the tile surface. The rubber template size varied somewhat during the 1987-1994 study (Area = 7.07 or 7.79 or 9.62 cm^2). Algae within the template were loosened using a fine bristle brush and squirt bottle, then sucked into a pipet.

The pipet contents were processed differently in the field depending on whether the sample was to be used to determine algal biomass, chlorophyll-a, or taxonomic composition. For the algal biomass and chlorophyll-a samples, the pipet contents were filtered through a pre-ashed filter (Gelman A/E glass fiber, pore size = 0.45 µm), then the filter was placed in a pre-labelled clean vial for further processing in the laboratory. The chlorophyll-a sample was frozen immediately after collection in the field by placing it in a chest of dry ice and held in darkness until analyzed in the laboratory. Taxonomic composition samples were placed directly into a clean vial with an algal preservative (6 parts water, 3 parts ethyl alcohol, and 1 part formalin). This completed the field collection of algae.

In the laboratory, algal biomass values were determined by drying the biomass sample filters to a constant weight at 60°C, weighing them on an analytical balance (to nearest 0.01mg), ashing them at 500°C in a muffle furnace (4 hours), and reweighing them. Biomass values were reported as mg ash-free dry mass (AFDM) per cm^2 of tile surface sampled.

Chlorophyll-a concentrations were determined in the laboratory by placing each chlorophyll-a filter in a separate vial with 90% reagent-grade acetone (24 hours at 4°C in dark). The extracted chlorophyll-a was measured with a Turner (Model 111) fluorometer using the procedures of Strickland and Parsons (1968). Chlorophyll-a concentrations were reported as µg chl-a per cm^2 of tile surface sampled.

Chlorophyll-a and algal biomass were analyzed using a general linear model ANOVA, with the same sources of variations as described for the MI analyses. All data (1987-1994) were used without transformation. Tukey's Studentized Range (HSD) Test was used to compare pre- and post-treatment means, while the Least Squares Means Test was used to compare the means for site position, station, season, and year.

The taxonomic composition of benthic algae (diatoms) was determined in the laboratory by transferring the preserved samples onto permanent microscope slides. Each sample was shaken to disperse clumped algal cells, then allowed to settle for one hour. One drop of algal sample was mixed with one drop of clean water on a slide coverslip, dried overnight at room temperature, then heated 3 hours on a hot plate. One drop of Hyrax® mounting medium was placed on a preheated microscope slide and allowed to boil for one minute. Following this, the cover slip was inverted and placed onto the slide, tapping gently to remove air bubbles. After the slide cooled, it was viewed at 1000X magnification to identify and count the algal cells along horizontal transects. A total of 200 algal cells were identified and counted from each sample. Thus, for each of the 5 main sampling stations, a total of 2000 algal cells were identified and counted, 1000 above the tributary confluence and 1000 below the confluence. It should be noted that because the algal samples were heated to remove all organic matter, this destroyed all algal cells except the silica frustules of diatoms. Therefore, the algal identifications only included one main group of algae, the diatoms. Other groups such as the green algae would not be reported eventhough they may have been present in N. Caspar Creek. Also, the sampling procedure emphasized the presence of adnate diatoms, which occur adjacent to the tile substrate. In contrast, filamentous diatoms were not fixed to the substrate and some may have washed off the tile as it was lifted from the stream.

All diatom identifications early in this study (1987-1988) were done by Walter R. Hill and Jessica Lacey, while all later identifications (1989-1994) during the quantitative part of the algal identifications were done by Angela Kost. Angela prepared a photographic collection of all the diatoms she identified from N. Caspar Creek, and all of her identifications have been verified by Dr. J. Patrick Kociolek, Diatomist, California Academy of Sciences, Golden Gate Park, San Francisco, CA. She intends to donate microscope slides of N. Caspar Creek diatoms to the California Academy of Sciences. Walter Hill prepared an early list of diatoms identified from N. Caspar Creek, including some species not on Angel'a list; however, we have not included these species in this report because they were not verified.

D. SUMMARY OF EARLY BIOLOGICAL STUDIES

The stream biological research program at N. Caspar Creek changed somewhat during the study period (1986-1994), both in the topics of study and the methodology. Modifications were made, especially during the early years, to (1) adapt field methods to actual conditions in the N. Caspar Creek basin and changes in the proposed logging plans, (2) add topics and methods that appeared to enhance understanding of potential logging impacts, and (3) abandon topics and methods which preliminary work showed to be impracticable or poor measures of logging impacts.

Dr. Allen W. Knight and several graduate students first visited the N. Caspar Creek drainage basin in April, 1985, after which, a research proposal was prepared. This proposal focused on the biological conditions in the small tributary streams of N. Caspar Creek, as well as the main channel. The selected side tributaries drained basins with different proposed logging treatments, including some control basins without logging. Greater impacts were expected to occur in the side tributaries than in the main channel because they were closer to the actual logging. From downstream to upstream, it was proposed to study the lower 100m of eight side tributaries (B, C, D, G, H, I, K, and M). The original proposal included studies of MI density and diversity, FFG's, drift rates, stonefly diversity, algal diversity, and chlorophyll-a concentrations. Macroinvertebrates and algae were to be collected from replicates of standard-sized rocks placed at study sites. For the main channel, the proposal included studies of the physical structure and hydrology of N. Caspar Creek, macroinvertebrates, salamanders, and juvenile salmonid fishes.

Preliminary stream biological research started in the N. Caspar Creek basin in 1986-1987, and included the following:

(1) 5 June 1986 -- collection of MI's from individually placed rocks in side tributaries YZ, A, I, J, and L.

(2) 24-28 June 1986 -- collection of 24 hour drift samples (50cm x 50cm net) for five days from mainstem Sites A below and A above, and from side tributaries YZ, D, I, L, and K.

(3) alder leaf decay studies from 16 July to 4 October 1986 in mainstem Station A and side tributaries YZ and D.

(4) 21-25 May 1987 -- collection of 3-4 hour drift samples (8cm x 20cm net) for five days upstream and downstream of Stations A, B, C, G, H, and I. Collection of MI rock pack samples upstream and downstream of Station A, in addition to the normal stations at B, C, G, H, & I.

(5) autumn, 1987 -- tanoak leaf decay studies at Stations B, G, and I.

(6) 7-8 November 1987 -- collection of 3 hour drift samples at Stations B and C.

Based on this preliminary work, it quickly became obvious that changes were needed in the original study proposal. First of all, it was found that many side tributaries were not perennial, but ceased flowing in summer, even during years of normal precipitation. Thus, the focus of the biological studies was shifted to the main channel of N. Caspar Creek, with sampling sites upstream and downstream of seven tributary confluences (YZ, B, C, D, G, H, and I). Shortly thereafter, the number of main sampling stations was reduced to five (B, C, G, H, and I), and these stations were used for the remainer of the study (1987-1994). N. Caspar Creek was generally perennial, but during fall, 1987, the stream dried up at Station I.

The preliminary studies also found that using introduced individual rocks to sample the macroinvertebrate populations was impractical because the number on each rock was very low. Therefore, multi-rock substrate baskets were used instead for the remainder of the study.

Individual rocks were also impractical for collecting algal samples because surface irregularities prevented the rubber template from effectively sealing off the sampled area. Clay tiles were substituted for the individual rocks and were used for the remainder of the study (1987-1994) to gather samples for species identification, algal chlorophyll-a, and a new addition to the study, algal biomass. For the first few years (1987-1988), algal species identifications were only qualitatively, to see which species were present; however, starting in spring, 1989, and continuing to spring, 1994, quantitative measurements of algae were made.

Macroinvertebrate drift rate studies were eventually abandoned after preliminary collections in 1986 and 1987. In 1986, 24-hour drift samples were collected from seven locations (5 replicates collected over 5 days) using a 50cm x 50cm net. Because of the enormous amount of work and logistics involved in obtaining and processing these samples, the drift studies were reduced in 1987 to include 3-4 hour collections (1 hour before sunset until 2 hours after sunset) upstream and downstream of six stations on N. Caspar Creek (A, B, C, G, H, & I) using an 8cm x 20cm net. Thereafter, all drift rate studies were stopped because (1) the work required enormous amounts of time, (2) it was difficult to collect drift samples in the low current velocities and shallow water of N. Caspar Creek, (3) the results from the drift studies were of questionable value because the low currents allowed drifting organisms to crawl out of the net and benthic organisms to crawl into the net, (4) it was difficult to replicate samples, and (5) collecting night samples was dangerous.

Leaf decay rate studies were added to the proposed biological work in N. Caspar Creek because this appeared to be a potentially useful measure of overall stream functioning which may be altered by logging treatments. Preliminary leaf decay studies were done in the summer-autumn of 1986 using alder leaves placed in mainstem Station A and in side tributaries YZ and D. This was followed in the autumn of 1987 by tanoak leaf decay studies at Stations B, G, and I. In contrast to the alder leaves used in 1986, tanoak leaves were very resistant to decay, and the 1987 study was stopped when no measureable decay occurred after 8 weeks in the stream. Tanoak leaves were initially chosen for study because they naturally occurred along N. Caspar Creek and were probably used by stream macroinvertebrates. For the remaining leaf decay studies (1988-1993), only alder leaves were used.

During 1989-1991, Harrison Phipps studied light conditions along N. Caspar Creek, especially at Station G. His main interest was to determine how logging altered the amount of light reaching the stream and what effects this had on benthic algae. To aid these studies, he plane-table mapped Stations B, C, G, H, and I during the summer, 1989. Continuous measurements of light conditions at the creek were made with a recording pyranometer ($\mu E/m^2/sec$). To document changes in the forest canopy at Station G before and after logging in Unit V, Harrison photographed the forest canopy using a fisheye lens (180° exposure). Permanent locations allowed him to replace his camera exactly at the same spot at different times before and after logging from 1989-1991. Reverse images of the photographs were scanned into a computer and converted to pixels. This image was integrated to obtain a direct measure of the light area penetrating the forest canopy. Harrison is presently working on his light study results for publication.

IV. RESULTS

A. MACROINVERTEBRATES (MI)

During the entire 8 years of this biological study of N. Caspar Creek (Spring 1987 - Spring 1994), MI’s were sampled twice a year (spring and fall), each time using 100 rock packs (Table 10). This resulted in 15 separate sampling occasions and a potential maximum of 1500 samples for the entire study. However, some rock packs were lost. All 100 rock packs were destroyed in spring, 1990, because of very large, aseasonal discharges (150 cfs, 27 May 1990), and 20 rock packs were lost in fall, 1987, because portions of the stream completely dried at Station I. Additionally, smaller numbers of rock packs could not be found throughout the study because they were concealed by accumulated detritus and substrate. Thus, a total of 188 rock packs were lost or destroyed in the field during the study, giving a grand total of 1312 MI samples of the 1500 possible.

Because research at Caspar Creek has extended over many years and will likely continue into the future, collected data sets are especially valuable resources for measuring long-term changes as the treated basins recover from logging. Therefore, we have included our biological data sets as an Appendix to this report.

For each macroinvertebrate variable measured in this study (MI density, MI taxa, EPT density, EPT taxa, Chironomidae density, Baetis density, FFG density and relative abundance), the analysis of variance of the overall data (all stations, all dates) and each station data (all dates) always found significant differences ($P < 0.01$) for years.

1. MI Taxonomic Composition

During the eight years of study of the MI's in N. Caspar Creek, a total of 95 different taxa were indentified, 86 of these being aquatic insects within eight different insect orders (Table 5-6). The remaining 9 taxa were non-insects from different animal Phyla or Classes. Most of the aquatic insects (73 taxa) in N. Caspar Creek were from four insect orders, Ephemeroptera (14 taxa), Plecoptera (12 taxa), Trichoptera (23 taxa), and Diptera (24 taxa). Fewer insect taxa occurred in the Odonata, Megaloptera, and Coleoptera. The taxonomic dominance of the benthos by aquatic insects, and the importance of the four abundant orders (EPT and Diptera), is a typical feature of most small and medium streams worldwide.

One overall measure of the impact of logging on N. Caspar Creek was the total taxonomic composition of MI's in N. Caspar Creek (combining all five stations) before and after logging. Although the total number of MI taxa appeared to increase from 46 to 75 taxa between 1987 and 1994 (Fig. 12, Table 6), the slope of the regression line was not significantly different from zero and there was not a significant difference ($P > 0.05$) between pre- and post-treatment taxa numbers (t-test). Part of the observed increase may be artificial, caused by (1) different people doing the identifications, (2) identification ability increasing with experience, and/or (3) increased sample sizes at the end of the study. Nevertheless, the MI fauna at the start and end of the study contained the same major groups and usually the same genera. There were no dramatic losses in total number of taxa or in large parts of the fauna as might be expected if fine sediment deposition was severe. Therefore, comparison of faunal lists from before and after logging did not demonstrate an adverse impact on N. Caspar Creek.

It should be emphasized that the 95 taxa identified in this study does not represent the actual number of MI species present in N. Caspar Creek because most identifications could only be taken to the generic level. Species level identification requires collection of the terrestrial adult stage, rather than only the aquatic larval stage, which was the subject of this study. In fact, the true number of MI species present in N. Caspar Creek would be far higher that 95, possibly 3-5 times higher. An example of the different results which can be obtained by using different levels of taxonomic resolution can be seen with our Plecoptera (stonefly) data for N. Caspar Creek. Using aquatic larvae only for identification, 12 taxa were distinguished; however, using larvae and very limited collections of terrestrial adults, 23 taxa were distinguished (Table 7). It is likely that additional adult collections would reveal at least another 10 or 15 species. Even though only four adult collections were made, a new species of *Sweltsa* stonefly was discovered and is presently being described. Thus, this example using stoneflies demonstrated the high insect diversity contained within small streams such as N. Caspar Creek. Similar results could be demonstrated within most other insect groups.

An additional reason for expecting many more than 95 species in N. Caspar Creek was that in this study the ubiquitous and abundant dipteran family Chironomidae were not identified to genus because this was difficult and very time consuming. The chironomids alone could easily have more than 50-100 species inhabiting N. Caspar Creek.

Of the 272,409 MI specimens collected from the rock packs, 38% were Diptera, 31% were Ephemeroptera, 16% were Trichoptera, 9% were Plecoptera, 3% were Coleoptera, and 3% were non-insects (Fig. 13). The other insect orders of Collembola, Odonata, and Megaloptera made up a small percentage. The six main groups fluctuated somewhat in relative abundance throughout the study (Fig. 14), but none of the groups had regression line slopes significantly different from zero. However, the negative regression line slope for mayflies and increasing slope for dipterans may indicate some adverse logging impacts.

Two taxa dominated the macroinvertebrate numbers in N. Caspar Creek, Chironomidae (33.9%) and *Baetis* (18.1%), and this dominance was persistent on all sampling dates (Table 8). Chironomidae were the most abundant taxon (22 - 42%) on 13 of the 15 sampling dates. Both groups rapidly resond to environmental changes because they can produce more than one generation per year and can rapidly recolonize disturbed areas by their good dispersal abilities.

Seventeen additional taxa composed a smaller part (1-5%) of the total fauna (Table 9). Nine of these were relatively abundant on different samping dates throughout the study, including *Glossosoma, Paraleptophlebia,* *Hydropsyche, Malenka, Cinygmula, Simulium, Optioservus, Ironodes,* and *Lepidostoma* (Table 8-9). The 19 most common taxa made up 91.6% of the total numbers collected. Of the total 95 MI taxa, 76 were either uncommon or rare, each making up less than 1% of the total macroinvertebrates collected (Table 6).

Taxonomic Notes -- Because taxonomic identifications may have introduced variability into this study, we add here some discussion of the taxa and problems with their identification. Ideally when studying a stream community, species-level identifications are desired. This is seldom possible in North America because of the incomplete state of our taxonomic knowledge. Species descriptions usually only exist for the terrestrial adult life stage, but the aquatic immature life stages (nymphs, larvae, pupae), which were the main focus of this study, have not been associated with the adults and cannot be identified to species. Most stream MI communities, including N. Caspar Creek, are composed of immature aquatic insects. We identified the MI's in this study as far as possible, normally to the generic level, but for some of the more difficult taxa only to family or higher levels.

Collembola (springtails) -- Springtails live on the surface water film or along the stream banks and were not part of the underwater benthic community which were sampled with the rock packs. The small numbers reported were accidentally collected during rock pack retrieval or came from partially submerged rock packs. Collembola were never abundant in the samples and those few found were excluded from the 1992-1994 data.

Ephemeroptera (mayflies) -- Several genera of mayflies have a very similar appearence and unless examined carefully may be grouped in one taxon. This was especially true if the nymphs were small, early instars. It was difficult in this study to always distinguish between the following genera: (1) *Baetis - Centroptilum,* (2) *Ephemerella - Serratella,* and (3) *Cinygma - Cinygmula - Nixe*. The identifications of these may have shifted with time and identifier. During 1992-1994, whenever there was a question of identity, we arbitrarily placed the nymphs in *Baetis* and *Serratella,* and equally divided them between *Cinygmula - Nixe*. Mature nymphs were much easier to distinguish.

Plecoptera (stoneflies) -- The stonefly species of N. Caspar Creek were better known because some adult speciemens were collected in this study (Table 7). However, problems existed in identifying small stonefly nymphs. Again, several genera of stoneflies have similar appearences, including (1) *Capnia - Mesocapnia,* (2) *Paraleuctra - Moselia,* (3) *Sweltsa - Suwallia - Alloperla,* and (4) *Malenka - Soyedina - Zapada*. During 1992-1994, the 1st and 2nd generic pairs were grouped as the families Capniidae and Leuctridae, respectively. We believed that the vast majority of the third group were *Sweltsa*, but arbitrarily placed all chloroperlids, except *Kathroperla,* as "other Chloroperlidae". It is likely that more than one species of *Sweltsa* were present. The fourth group, the nemourid stoneflies, were very common in this study and were separated into three genera in 1992-1994. However, earlier in the study *Malenka* and *Zapada* were grouped, while *Soyedina* were called *Nemoura*. If further analysis of the stoneflies were desired, the data for all three genera should be grouped as the family Nemouridae.

Trichoptera (caddisflies) -- Most caddisfly larvae could be identified to genus; however, more difficulty occurred when early instars were present. These small larvae were placed arbitrarily in the more common genera of a family. If there was a difficulty in distinguishing between small Lepidostomatidae and Limnephilidae, these were arbitrarily called *Lepidostoma*.

Chironomidae (midges) -- Although the midges were the single most common MI in this study, we did not identify them beyond family level because generic determinations would have substantially increased processing time. Generic identifications would have given better resolution of FFG's and other aspects of this study.

Non-insects -- Non-insects were seldom identified to generic level; however, this had little effect on this study because non-insects were a relatively minor component of the MI fauna.

2. MI Density

From the overall total 1312 rock packs collected, 272,409 macroinvertebrate specimens representing 95 different taxa were collected and indentified from N. Caspar Creek (Table 6, 11-12). This resulted in an overall mean density of 207.6 MI per rock pack for the entire study. Densities ranged from a minimum of 77.4 (Fall 1987) to a maximum of 516.9 (Spring 1994), with most sample period means being 140-200 per rock pack (Fig. 16). The ANOVA R^2 values ranged from low to medium (Table 13), reflecting the variability of MI densities.

Pre-treatment vs. Post-treatment -- Post-treatment MI densities were significantly higher ($P < 0.01$) than pre-treatment densities when the overall data (all stations, all dates) were tested. Post-treatment MI densities were also significantly higher ($P < 0.05$) at Stations B, C, and I, but were non-significant at Stations G and H (Table 13, Fig. 17). These results were especially influenced by the large MI densities of spring, 1992 and 1994. Without these two collections, no significant differences existed between pre- and post-treatment densities in the overall or station data.

The most important conclusion of the treatment analyses was that no evidence could be found that fine sediment caused significant decreases in MI densities. In contrast, increased MI densities indicated that the infuences of light, water temperature, and nutrients on stream algae may have translated into higher MI densities.

Station -- Macroinvertebrate densities for the entire 1987-1994 study varied by sampling station and site (Table 12, Fig. 15, 17). Mean densities ranged from a low of 118.8 at Site Ga to a high of 359.6 at Site Bb, with most site densities being 130-250 per rock pack. The lowest densities occurred at Station G in the middle reaches of the N. Caspar Creek study area, and these densities were significantly different from all other stations, except Station H (Table 14). Station B and C, the furthest downstream, had significantly higher densities than the three upstream stations. All pairwise comparisons were significantly different ($P < 0.05$), except for G-H (Table 14). Higher MI densities at downstream locations, probably from increased algae, may indicate a cumulative logging effect. This result also indicated little or no adverse sediment impact.

Site Position -- Of the total 272,409 MI collected in this study, 140,645 came from Above sites (661 rock packs, mean density = 212.8), while 131,764 came from Below sites (651 rock packs, mean abundance = 202.4). These differences in means appeared to be small; however, Above site densities were significantly higher ($P < 0.01$) than Below site densities when the overall data (all stations, all dates) were tested. When each station was analyzed separately, only Stations C and I had significantly higher densities at Above than Below sites, while Stations B and H showed no differences (Table 13). Because no logging occurred in sub-basins H and I, Above-Below differences were not expected there, but this was only true at Station H.

When Above-Below site densities were compared (t-test) for each sampling date separately, only 15 of the 70 comparisons (9 higher at Above, 6 higher at Below) were significantly different (Table 15). No obvious clustering of the 15 significant differences, which might indicate logging impacts, could be detected, except for Fall 1990. Significant differences did not appear to be clustered in either pre-treatment or post-treatment periods. This suggests that the significant differences found probably do not have biological significance and have little relation to logging impacts.

Season -- Of the total number of macroinvertebrates collected, 180,508 were from 7 spring sampling dates (673 rock packs, mean density = 268.2), while 91,901 were from 7 fall sampling dates (639 rock packs, mean density = 143.8). Macroinvertebrate density was significantly higher ($P < 0.01$) in spring than in fall for the overall data. The higher spring MI density was greatly increased by the spring, 1992 and 1994 collections (Fig 16). If these two dates were excluded, the remaining 5 spring periods (480 rock packs, mean density =177.8) were still significantly higher ($P < 0.01$) than the fall densities. Analyzing the MI densities from each station separately, significantly higher ($P < 0.01$) spring densities were found at all locations, except at Station G (Table 13). Higher spring than fall MI densities are typical features of stream MI communities (Hynes 1970) primarily because of the timing of life cycle events, with high growth occurring in spring and summer, followed in the fall by many species not being collected because they are in the egg stage or young larval stage.

3. MI Number of Taxa

For the entire study, 95 different MI taxa were collected and identified from the rock pack samples (Table 5-6). The overall mean number of MI taxa varied from a minimum of 11.5 (Spring 1987) to a maximum of 26.8 (Spring 1994), with most sample dates having means of 15-20 per rock pack (Table 15). The number of MI taxa proved to be much less variable than were MI densities. Values of R^2 for the ANOVA's were generally higher than for MI density (Table 13).

Pre-treatment vs. Post-treatment -- The post-treatment number of MI taxa were significantly higher ($P < 0.01$) than pre-treatment values for the overall data (all stations, all dates) and for each of the five stations (Table 13, Fig. 18). This again suggests little or no adverse logging impacts from fine sediments, which normally decreases the number of MI taxa. Higher numbers of MI taxa may be caused by increased quantities of stream algae from the increased light levels, water temperatures, and/or nutrients in N. Caspar Creek.

Station -- The number of MI taxa for the entire 1987-1994 study varied somewhat by sampling station and site (Fig. 15, 18). Variation by site closely paralleled that of MI density (Fig. 15). MI mean taxa ranged from a low of 17.1 at Site Ga to a high of 20.4 at Site Ca, with most sites being 17-18 per rock pack. Stations G and H had low taxa numbers, and no significant differences existed between Stations G, H, and I (Table 14). Also, no significant differences existed between Stations B and C, but both stations had significantly higher MI taxa than the three upstream stations. Therefore, the MI taxa analyses divided the five stations into two classes, lower numbers upstream and higher numbers downstream. This result indicates that sediment did not adversely and cumulatively impact stations downstream. Higher MI taxa at downstream locations, probably from increased algae, may indicate a cumulative logging effect.

Site Position -- MI taxa number were significantly higher ($P < 0.01$) at Above than Below site when the overall data (all stations, all dates) were tested. When each station was analyzed separately, Stations C and I had significantly higher ($P < 0.01$) MI taxa at Above than Below sites, while the remaining three stations showed no differences (Table 13). This result parallels that found for MI density. Again, the non-significance of Above and Below sites at Station H was expected because no logging occurred in the sub-basin, but the significant difference at Station I was unexpected.

Season -- MI taxa number showed mixed seasonal results. No significant differences were found for the overall data (all stations, all dates). When each station was analyzed separately, Stations B and I had significantly higher taxa in spring, Station G in fall, and the remaining two stations showed no differences (Table 13).

4. EPT Density

From the entire 1312 rock packs collected in this study, 151,759 EPT (Ephemeroptera, Plecoptera, and Trichoptera) specimens were collected and indentified (Table 6). This resulted in an overall mean density of 115.7 EPT per rock pack. EPT mean density ranged from a minimum of 51.1 (Fall 1987) to a maximum of 247.9 (Spring 1994), with most sample dates being 60-160 per rock pack. The values of R^2 for the ANOVA's were low to medium (Table 13), reflecting the variability of EPT densities.

Pre-treatment vs. Post-treatment -- Post-treatment EPT densities were significantly higher than pre-treatment densities when the overall data (all stations, all dates) were tested (Table 13). When each station was analyzed separately, post-treatment densities were significantly higher at Stations B and I, but not for Stations C, G, and H.

Similar to the treatment results for MI density, no evidence could be found that fine sediment caused significant decreases in EPT densities. This finding was important because EPT are reported to be more sensitive to adverse sediment impacts than is MI density (Waters 1995). Again, there was evidence that EPT densities were responding to increases in stream algae.

Station -- EPT densities for the entire 1987-1994 study varied by sampling station and site (Fig. 19-20). EPT mean densities ranged from a low of 71.3 at Site Ga to a high of 170.1 at Site Bb, with most site densities being 75-140 per rock pack. The lowest EPT densities occurred at Station G in the middle reaches of the N. Caspar Creek study area, and these densities were significantly different from all other stations, except Station H (Table 14). Station B and C, the furthest downstream, had significantly higher ($P < 0.01$) EPT densities than the three upstream stations, but they were not significantly different from each other (Table 14). Higher EPT densities at downstream locations, probably from increased algae, may indicate a cumulative logging effect.

Site Position -- Of the total 151,759 EPT collected in this study, 77,856 came from Above sites (661 rock packs, mean density = 117.8), while 73,903 came from Below sites (651 rock packs, mean abundance = 113.5), a non-significant difference. When each station was analyzed separately, Stations C and I had significantly higher EPT densities at Above than Below sites, while Stations B, G, and H showed no differences (Table 13). Again, the non-significance of Above and Below sites at Station H was expected because no logging occurred in the sub-basin, but the significant difference at Station I was unexpected.

Season -- Of the total 151,759 EPT collected in this study, 99,325 were from 7 spring sampling dates (673 rock packs, mean density = 147.6), while 52,434 were from 7 fall sampling dates (639 rock packs, mean density = 82.1). Spring EPT densities were significantly higher ($P < 0.01$) than fall densities for the overall data (all stations, all dates) and for each of the 5 stations, except Station G (Table 13). Again, higher spring densities reflected the typical life cycles of most EPT.

5. EPT Number of Taxa

Of the total 95 macroinvertebrate taxa identified in this study, there were 49 EPT taxa (52%). The overall mean number of EPT taxa varied from a minimum of 8.6 (Spring 1987) to a maximum of 18.9 (Spring 1994), with most sample means being 10-15. Again, the number of EPT taxa proved to be much less variable than EPT densities, and the ANOVA R^2 values were higher than for EPT density (Table 13).

Pre-treatment vs. Post-treatment -- The post-treatment number of EPT taxa were significantly higher ($P < 0.01$) than pre-treatment values for the overall data (all stations, all dates) and for each of the five stations tested separately (Table 13). This again strongly indicated that no adverse logging impacts from fine sediments occurred to the highly sensitive EPT taxa number. In contrast, the EPT taxa results showed significant increases as would occur from increased quantities of stream algae.

Station -- The number of EPT taxa for the entire 1987-1994 study varied somewhat by sampling station and site (Fig. 19, 21). Mean values ranged slightly from a low of 12.4 at Site Ga, to a high of 14.8 at Site Ca, with most sites being 12.5-13.5 per rock pack. Both Stations G and H had lower numbers of EPT taxa. No significant differences existed in EPT taxa number between Stations G, H, and I (Table 14), but Stations B and C were significantly higher ($P < 0.05$). Once again, a distinct division was apparent between downstream (B and C) and upstream (G, H, and I) stations. This result again indicates that fine sediment did not adversely and cumulatively impact downstream stations. Higher EPT taxa at downstream locations, probably from increased algae, may indicate a positive cumulative logging effect.

Site Position -- Above site numbers of EPT taxa were significantly higher ($P < 0.01$) than Below site taxa for the overall data (all stations, all dates). When each station was analyzed separately, Stations C and I had significantly higher taxa number at Above than Below sites, while the remaining three stations had no differences (Table 13). This result parallels that found for EPT density and MI taxa. Again, the non-significance of Above and Below sites at Station H was expected because no logging occurred in the sub-basin, but the significant difference at Station I was unexpected.

Season -- The spring number of EPT taxa were significantly higher ($P < 0.01$) than fall for the overall data (all stations, all dates). When each station was analyzed separately, Stations B, C, and I had significantly higher taxa during spring than fall, while the remaining two stations showed no differences (Table 13). Again, higher spring densities reflect the typical life cycles of most EPT.

6. Chironomidae Density

Of the 272,409 macroinvertebrate specimens collected in this study, Chironomidae accounted for 92,444 (34%), by far the most abundant taxon collected in N. Caspar Creek. In the 1986-1987 preliminary drift studies, Chironomidae were also one of the commonest MI's. The overall mean density of chironomids was 70.5 for the entire study. Density ranged from a minimum of 11.6 (Spring 1987) to a maximum of 214.6 (Spring 1994), with most sample periods being 30-80 per rock pack. The values of R^2 for the ANOVA's ranged from low to medium (Table 13).

Pre-treatment vs. Post-treatment -- Post-treatment chironomid densities were significantly higher ($P < 0.01$) than pre-treatment densities when the overall data (all stations, all dates) were tested (Table 13). When each station was analyzed separately, only Stations B and C had significantly higher ($P < 0.01$) chironomid densities, while the other three stations were non-significant. Once again, a distinct division was apparent between downstream (B and C) and upstream (G, H, and I) stations. Chironomidae densities were not reduced by fine sediments from logging impacts; however, they apparently were significantly increased at the downstream stations from increased quantities of stream algae.

Station -- Chironomid densities for the entire 1987-1994 study varied by sampling station and site (Fig. 22, 24). Mean densities ranged from a low of 34.6 at Site Ga to a high of 150.8 at Site Bb, with most site chironomid densities being 40-80 per rock pack. The lowest chironomid densities occurred at Station G in the middle reaches of the N. Caspar Creek study area, and these densities were significantly different from Stations B and I, but not from Stations C and H (Table 14). Stations B and C, the furthest downstream, had the highest chironomid densities, but only Station B had significantly higher densities than all other stations. This result, with significantly higher chironomid densities downstream, again indicates that fine sediment did not adversely and cumulatively impact stations downstream. Higher chironomid densities at downstream locations, probably from increased algae, may indicate a positive cumulative logging effect.

Site Position -- Of the total 92,444 chironomids collected in this study, 48,385 came from Above sites (661 rock packs, mean density = 73.2), while 44,059 came from Below sites (651 rock packs, mean abundance = 67.7). Above site densities were significantly ($P < 0.01$) higher than Below site densities when the overall data (all stations, all dates) were tested. When each station was analyzed separately, Stations C, H, and I had significantly higher densities at Above than Below sites, while Stations B had no difference (Table 13).

Season -- Of the 92,444 chironomids collected, 63,178 were from 7 spring sampling dates (673 rock packs, mean density = 93.9), while 29,266 were from 7 fall sampling dates (639 rock packs, mean density = 45.8). Chironomid densities were significantly higher ($P < 0.01$) in the spring than in the fall for the overall data and for each station, except Station G (Table 13). The higher spring densities were greatly increased by two sample periods, spring, 1992, and spring, 1994, especially at Station B and C (Fig. 24). If these two dates were excluded from the analysis, the remaining 5 spring periods (480 rock packs, mean density = 57.5) were still significantly higher ($P < 0.01$) than in the fall. Again, higher spring densities reflect the typical life cycles of most MI.

7. *Baetis* Density

Of the 272,409 macroinvertebrate specimens collected in this study, *Baetis* accounted for 49,173 (18%). In the 1986-1987 preliminary drift studies, *Baetis* was also one of the commonest MI's. The overall mean density of *Baetis* was 37.5 per rock pack for the entire study. Density ranged from a minimum of 5.4 (Fall 1987) to a maximum of 145.0 (Spring 1992), with most sample periods being 10-70 per rock pack. The ANOVA R^2 values were generally higher than for many other variables tested (Table 13).

Pre-treatment vs. Post-treatment -- The treatment results for *Baetis* densities were mixed. Pre-treatment *Baetis* densities were significantly higher ($P < 0.01$) than post-treatment densities when the overall data (all stations, all dates) were tested (Table 13). When each station was analyzed separately, Station G and H also had significantly higher ($P < 0.05$) pre-treatment densities. In contrast, Station B had significantly higher ($P < 0.05$) post-treatment densities, while Stations C and I were non-significant.

Higher post-treatment *Baetis* densities at Station B, the furthest downstream, indicated that fine sediment was not adversely and cumulatively impacting this location, but rather that increased quantities of stream algae were causing increased densities. The change in significance from pre-treatment to none to post-treatment in a downstream direction may represent a longitudinal gradient of opposing sediment and algae effects.

Station -- *Baetis* densities for the entire 1987-1994 study varied by sampling station and site (Fig. 23, 25). Mean densities ranged from a low of 13.2 at Site Ha to a high of 72.8 at Site Bb, with most site densities being 15-60 per rock pack. Similar low densities occurred at Stations G and H; however, all station comparisons except B and C were significantly different (Table 14). Station B and C, the furthest downstream, had significantly higher *Baetis* densities than the three upstream stations (Fig. 25), again separating the study reach into two distinct regions. This result indicated that fine sediment did not adversely and cumulatively impact downstream stations. However, this result may demonstrate a positive cumulative logging effect from increased algae downstream.

Site Position -- Of the total 49,173 *Baetis* collected in this study, 23,255 came from Above sites (661 rock packs, mean density = 35.2), while 25,918 came from Below sites (651 rock packs, mean abundance = 39.8). There were no significant differences in *Baetis* densities between Above and Below sites for the overall data (all stations, all dates) or for any of the five stations. The lack of site position differences may be due to the high numbers of *Baetis* which readily and persistently disperse downstream in the stream drift.

Season -- Of the 49,173 *Baetis* collected, 40,691 were from 7 Spring sampling dates (673 rock packs, mean density = 60.5), while only 8,482 were from 7 Fall sampling dates (639 rock packs, mean density = 13.3). *Baetis* densities were significantly higher ($P < 0.01$) in spring than in fall for the overall data and for each station (Table 13). The higher spring densities were greatly increased by two sample periods, spring, 1992 and 1994, especially at Station B and C. If these two dates were excluded, the remaining 5 spring periods (480 rock packs, mean density = 44.6) were still significantly higher ($P < 0.01$) than the fall densities. The cycling of *Baetis* densities between high spring values and very low fall values was a distinct feature at all stations (Fig 25). Again, this seasonality reflected the typical life cycles of most MI, but was especially pronounced in *Baetis.*

8. MI Functional Feeding Groups

All six Functional Feeding Groups (FFG) as defined by Merritt and Cummins (1984, 1996) occurred in N. Caspar Creek during this 1987-1994 study -- Scrapers, Shredders, Collectors, Predators, Macrophyte Piercers, and Parasites. Also, both subdivisions of Collector, Filterers and Gatherers, were common during the study. Only Macrophyte Piercers and Parasites were uncommon or rare.

Of the 272,409 total MI collected in the rock packs during the entire study, 179,413 (65.9 %) were Collectors, this being subdivided into 161,025 (59.1 %)

Collector - gatherers and 18,388 (6.8 %) Collector - filterers. In addition, 44,923 (16.5 %) Scrapers, 28,708 (10.5 %) Shredders, 18,968 (7.0 %) Predators, 396 (0.1 %) Macrophyte Piercers, and 1 Parasite were collected (Fig. 26). The relative abundances of FFG's in N. Caspar Creek were typical of many North American streams and very similar to that proposed in the River Continuum Concept for headwater or medium sized streams (Vannote et al. 1980, Cummins et al. 1989).

The overall relative abundances of FFG's fluctuated somewhat between sampling dates (combining the data from all 5 stations); however, Collector-gatherers always dominated the composition (Fig. 27). Collectors and Scrapers exhibited a distinct seasonal pattern between spring and fall sampling dates, giving their graphs a "sawtooth pattern", while Shredders and Predators remained fairly stable. The overall 7.0 % abundance of Predators in N. Caspar Creek was similar to the 10 % predicted from theory and found in many stream studies (Vannote et al. 1980).

In this study of N. Caspar Creek, FFG categories were assigned to each identified taxon primarily using Merritt and Cummins (1984, 1996) and Pennak (1989). These references provided the best current information on FFG. However, it should be cautioned that FFG information is incomplete or contradictory for some MI’s, and the FFG references sometimes placed a taxon into more than one category. Thus, our assignment of taxa to FFG's (Table 5) should be considered only a first approximation, and judgement has been used in assigning these categories. This was especially true for the Chironomidae, which we have listed as Collector - gatherers. This large family contains many genera and species which undoubtably should be listed in other FFG's; however, without extremely time-consuming taxonomic work to separate this family into genera we believed that the Collector-gatherer FFG best describes this abundant family.

Another important taxon in N. Caspar Creek was the mayfly, *Baetis*. The FFG to which it belongs was unclear from the stream literature. As with this study, some have called it a Collector-gatherer (Hawkins et al. 1982, Gurtz & Wallace 1984, Duncan & Brusven 1985), while Merritt and Cummins (1984, 1996) list it as a Collector-gatherer (detritus, diatoms) and a Scraper, and Bilby and Bisson (1992) call it a Scraper. Others have found that *Baetis* is a fine particle feeder which seasonally injests algae (Haefner & Wallace 1981). *Baetis* has also been shown to be opportunistic, rapidly increasing when algae increase (Wallace & Gurtz 1986). Because *Baetis* was such an important organism in N. Caspar Creek and its FFG classification was unclear, we analyzed it separately, in addition to including it in the Collector-gatherer FFG. Our FFG results may be somewhat different if *Baetis* had been included purely as a Scraper.

Our interest in the FFG's of N. Caspar Creek was primarily focused upon Scraper and Shredder relative abundance because other studies have shown these two groups to be affected by logging, allowing us to make predictions about how they might change. However, we have also included in our analysis the Collector-filterers, Collector-gatherers, and Predators, although predictions about how they might be affected by logging were unclear.

We have also examined both the densities and relative abundances of FFG's. Density measurements gave the actual variation of individual groups with location and date, while relative abundance gave an idea of how a group related to the entire community. The results for these two measurements do not always match (Hawkins et al. 1982) because the relative abundance of a specific group depended both on its own density and the densities of the other groups. Relative abundances were useful for comparing the study results with stream theory (Vannote et al. 1980).

a. Scrapers

Scrapers feed by grazing on algae and fine detritus closely attached or adjacent to solid surfaces in streams. Previous studies indicated that if the forest canopy over a stream was reduced, admitting more light and stimulating greater algal production, that Scraper relative abundance should increase. These predictions proved to be true in N. Caspar Creek.

In N. Caspar Creek, Scrapers were the second most abundant FFG for the entire study (Fig. 26). A total of 44,923 Scrapers were collected (34.2 per rock pack, 16.5% of total MI fauna). Densities (mean) varied from a low of 11.2 (Spring 1987) to a high of 55.0 (Spring 1994). Scraper relative abundances (mean) varied with sampling date, ranging from a low of 8.4% (Spring 1988) to a high of 38.2% (Fall 1987), with most sample periods being 10-25% (Fig. 27). The ANOVA R^2 values were generally medium (Table 16).

Pre-treatment vs. Post-treatment -- Post-treatment Scraper densities and relative abundances were always significantly higher ($P < 0.01$) than pre-treatment values for the overall data (all stations, all dates) and for each of the five stations tested separately (Table 16, Fig. 28-29). This strongly indicated that logging had an impact on Scrapers, which likely responded positively to increased algal levels in N. Caspar Creek. It also suggested that increased fine sediments from logging were minimal, or that the negative sediment impacts were offset by the positive effects of increased algae.

Station -- Scraper densities and relative abundances for the entire 1987-1994 study varied by sampling station and site (Fig. 28-29). Stations B and C had the highest overall mean densities (41.0 - 50.1 per rock pack), while the three upstream stations had about half these densities (21.8 - 36.4). Overall mean relative abundances ranged slightly from a low of 11.8% at Site Bb to a high of 20.5% at Site Cb. Station B and C densities were significantly higher than all upstream stations, and the same was true for relative abundances, except for no difference between B and G (Table 17). Once again, higher Scraper densities and relative abundances at downstream locations may indicate a positive cumulative logging effect, probably from increased algae, but not from an adverse sediment impact.

Site Position -- Of the total 44,923 Scrapers collected, 23,982 were from Above sites (661 rock packs, mean density = 36.3), while 20,941 were from Below sites (651 rock packs, mean abundance = 32.2). Scraper densities were significantly higher ($P < 0.05$) at Above than Below sites for the overall data and for Stations B and C (Table 16). However, there were no significant differences between Above and Below site relative abundances for the overall data (all stations, all dates) or for the five stations, except Station B (Table 16). These results suggested that the logged side tributary at Station G had no local influence on Scrapers, but that there may have been an influence at Stations B and C. As expected, neither Station H or I showed any significant differences between Above and Below sites because no logging occurred in their sub-basins.

Season -- Of the 44,923 Scrapers collected, 20,347 were from 7 Spring sampling dates (673 rock packs, mean density = 30.2), while 24,576 were from 7 Fall sampling dates (639 rock packs, mean density = 38.5). Scraper relative abundances distinctly varied between spring and fall (Fig. 27). Significantly higher ($P < 0.01$) Scraper relative abundances occurred in fall than in spring, both when the overall data (all stations, all dates) were tested and when each station was analyzed separately. A similar but less conclusive result occurred with Scraper densities. Higher fall abundances distinctly contrasted with most results for other MI groups, which typically had higher levels during spring. Higher fall Scraper abundances suggested that they were most abundant when the standing crop of diatoms were highest (Fig. 48).

b. Shredders

Shredders feed on large pieces of organic detritus (CPOM, > 1mm), such as leaves, branches, and other plant parts, which enter the stream from the riparian zone. Logging can reduce the inputs of allochthonous detritus to the stream or alter its composition, impacting the Shredders.

The results for Shredders in N. Caspar Creek were mixed, depending upon whether densities or relative abundances were analyzed. Shredders were the third most abundant FFG for the entire study (Fig. 26). A total of 28,708 Shredders were collected (21.9 per rock pack, 10.5% of MI fauna). Densities (mean) varied from a low of 4.7 (Spring 1987) to a high of 59.4 (Spring 1994), while relative abundances ranged from 5.0% (Spring 1991) to 16.7% (Fall 1988). The ANOVA R^2 values were generally low to medium (Table 16).

Pre-treatment vs. Post-treatment -- Post-treatment Shredder densities were significantly higher for the overall data (all stations, all dates) and at Stations B and H (Table 16). However, there were no significant differences between pre- and post-treatment relative abundances for the overall data. When each station was analyzed separately, Stations G and I had no significant differences, but pre-treatment relative abundances were significantly higher ($P < 0.01$) at Stations B and C. This relative abundance result suggested that Shredders at Station B and C may have been reduced, possibly by changes in allochthonous detritus. Because this adverse impact only occurred at the downstream stations, it might be interpreted as a cumulative effect; however, this conclusion remains questionable because the relative abundance results were not substantiated by the Shredder density results.

Station -- Shredder densities and relative abundances for the entire 1987-1994 study varied by sampling station and site (Fig. 30-31). Stations B and C had similar overall mean densities (18.3 - 25.9 per rock pack) with the three upstream stations (14.3 - 31.4). However, Stations B and C had significantly lower ($P < 0.01$) relative abundances (7.0 - 10.0%) than the three upstream stations (10.2 - 16.1%). There were no significant differences between B and C (Table 17). Thus, it was possible that reduced amounts of allochthonous detritus at the downstream stations cumulatively reduced the Shredders.

Site Position -- Of the total 28,708 Shredders collected, 16,008 were from Above sites (661 rock packs, mean density = 24.2), while 12,700 were from Below sites (651 rock packs, mean abundance = 19.5). Shredder densities were significantly higher ($P < 0.01$) at Above than Below sites for the overall data (all stations, all dates) and for Stations C and I (Table 16). Significantly higher ($P < 0.05$) Shredder relative abundances also occurred at Above sites for the overall data, but the results for individual stations differed from that found using the density data (Table 16).

Season -- Significantly higher ($P < 0.01$) Shredder relative abundances occurred in fall than in spring, both when the overall data (all stations, all dates) were tested and when each station was analyzed separately, except at Station I (Table 16). This seasonal effect was generally not obvious in the density data; therefore, the seasonality in the relative abundances may be partly caused by changes in other FFG's. Higher fall abundances contrasted with most results for other MI groups, which typically had higher levels during spring. The higher Shredder relative abundances may have matched the higher fall inputs of allochthonous detritus.

c. Collector-filterers

Collector-filterers feed on fine organic detrital particles (FPOM, < 1mm) suspended and carried downstream by the current. They possess numerous anatomical structures for filtering these particles from the water. Normally, they are of low to medium abundance in streams, though they can be very abundant in local stream patches and microhabitats (e.g. Simuliidae, Hydropsychidae).

Collector-filterers were the fifth most abundant FFG in N. Caspar Creek (Fig. 26). Of the total MI collected in this study, 18,388 were Collector-filterers (14.0 per rock pack, 6.8% of total MI fauna). The overall mean density varied from < 1.0 per rock pack (Fall 1987, 1988, and 1992) to over 50 (Spring 1992 and 1994), with most sample periods having densities <10 per rock pack. Collector-filterer relative abundances (mean) varied with sampling date, ranging from a minimum of 0.2% (Fall 1988) to a maximum of 11.9% (Spring 1992), with most sample periods being < 10% (Fig. 27). The ANOVA R^2 values were generally low (Table 16).

Pre-treatment vs. Post-treatment -- Post-treatment Collector-filterer densities and relative abundances were always significantly higher ($P < 0.01$) than pre-treatment values for the overall data (all stations, all dates) and for essentially all of the five stations when tested separately (Table 16). These results strongly indicate that logging had an impact on Collector-filterers, possibly by increasing the quantity or quality of fine seston carried in the current.

Station -- Collector-filterer densities and relative abundances for the entire 1987-1994 study varied by sampling station and site (Figs. 32-33). Stations B and C usually had the highest overall mean densities (11.4 - 28.0 per rock pack), while the three upstream stations normally had lower densities (2.9 - 14.7). Overall mean relative abundances ranged slightly from a low of 4.4% at Site Ba to a high of 10.3% at Site Cb. Station pairwise comparisons for densities and relative abundances were mixed (Table 16); however, Station B and C densities were usually significantly higher than the three stations upstream. The failure of relative abundance results to corroborate density results was likely due to the overall low abundance of Collector-filterers, causing relative abundance data to vary with changes in other FFG's.

<u>Site Position</u> -- Of the total 18,388 Collector-filterers collected, 8,669 were from Above sites (661 rock packs, mean density = 13.1), while 9,719 were from Below sites (651 rock packs, mean abundance = 14.9). Collector-filterer densities and relative abundances were not significant different between Above and Below sites for the overall data, but were significantly different at Stations B and H (Table 16). The higher densities and relative abundances at Site B below may have indicated increased seston quantities from the logged sub-basin (Unit B).

<u>Season</u> -- Of the 18,388 Collector-filterers collected, 15,133 were from 7 spring sampling dates (673 rock packs, mean density = 22.5), while only 3,255 were from 7 fall sampling dates (639 rock packs, mean density = 5.1). Thus, very pronounced seasonal variations occurred in the Collector-filterers (Fig. 32, 33). Significantly higher ($P < 0.01$) densities and relative abundances occurred in spring than in fall, both when the overall data (all stations, all dates) were tested and when each station was analyzed separately. We believe these seasonal differences were caused by larger quantities of suspended FPOM carried in the higher spring discharges, while the low fall discharges carried fewer suspended particles.

d. Collector-gatherers

Collector-gatherers feed on deposits of fine organic detrital particles (FPOM, < 1mm) originating from within the stream or being imported from the riparian zone. Collector-gatherers are normally the dominant FFG in streams (Vannote et al. 1980), and this was also true for the entire study in N. Caspar Creek, where they typically made up 50-60% of the MI fauna (Fig. 26). The vast majority of Collector-gatherers were Chironomidae and *Baetis,* which have already been analyzed above.

Of the total MI collected in this study, 161,025 (59.1%) were Collector-gatherers. The overall mean density was 122.7 per rock pack, ranging from 32.6 (Fall 1987) to 318.4 (Spring 1992). Collector-gatherer relative abundances (mean) varied with sampling date, ranging from a minimum of 35.8% (Fall 1989) to a maximum of 68.6% (Spring 1991), with most sample periods being 45-60% (Fig. 27). The ANOVA R^2 values were generally medium (Table 16).

<u>Pre-treatment vs. Post-treatment</u> -- Treatment effects on Collector-gatherer densities and relative abundances were mixed and appeared to be a mixture of its two most abundant groups, the Chironomidae and *Baetis* (Table 13, 16). Post-treatment densities were significantly higher ($P < 0.01$) than pre-

treatment values at Stations B and C, but not for the other stations or the overall data. In contrast, treatment effects on relative abundance were not significant at Stations B and C, but the overall data (all stations, all dates) and Stations G, H, and I were significantly higher during the pre-treatment period (Table 16). Because of the mixed results, definite conclusions about treatment effects were not possible.

Station -- Collector-gatherer densities and relative abundances for the entire 1987-1994 study varied by sampling station and site (Fig. 34-35). Stations B and C usually had the highest overall mean densities (106.2 - 239.1 per rock pack), while the three upstream stations normally had lower densities (61.5 - 138.0). Overall mean relative abundances ranged slightly from a low of 51.3% at Site Ha to a high of 68.0% at Site Bb. Station pairwise comparisons for densities and relative abundances were mixed, except that Station B usually had significantly higher densities and relative abundances than stations upstream (Table 17).

Site Position -- Of the total 161,025 Collector-gatherers collected, 81,807 were from Above sites (661 rock packs, mean density = 123.8), while 79,218 were from Below sites (651 rock packs, mean abundance = 121.7). Collector-gatherer densities and relative abundances had no significant differences between Above and Below sites for the overall data, but the results were mixed for the individual stations (Table 16).

Season -- Of the 161,025 Collector-gatherers collected, 116,305 were from 7 spring sampling dates (673 rock packs, mean density = 172.8), while only 44,720 were from 7 fall sampling dates (639 rock packs, mean density = 70.0). Thus, Collector-gatherers had pronounced seasonal variations (Fig. 27). Significantly higher ($P < 0.01$) Collector-gatherer densities and relative abundances occurred in spring than in fall, both when the overall data (all stations, all dates) were tested and when each station was analyzed separately. These seasonal changes were likely due to the typical life cycles of aquatic insects, which reach maximum abundance in spring and summer. Possibly, FPOM food source deposits may be more abundant in spring following the higher winter discharges.

e. Predators

Predators feed on other invertebrates in the stream, where they typically make up about 10% of the total MI fauna (Vannote et al. 1989). Of the total MI collected in N. Caspar Creek, Predators were the fourth most abundant FFG with 18,968 collected (14.5 per rock pack, 7.0% of MI fauna). The overall mean densities ranged from 5.6 (Fall 1987) to 37.2 (Spring 1994). Predator relative abundances (mean) varied slightly from a minimum of 3.6% (Spring 1992) to a maximum of 12.7% (Fall 1989), with most sample periods being < 10% (Fig. 27). The ANOVA R^2 values were generally low to medium (Table 16).

Pre-treatment vs. Post-treatment -- Treatment effects on Predator densities and relative abundances were mixed (Table 16). Predator densities were not significantly different between pre- and post-treatments for the overall data and for each station, except at B where post-treatment densities were significantly higher. It would appear that the higher amounts of algae at Station B translated into higher Predator densities, a result which has been reported previously (Murphy & Hall 1981, Hawkins et al. 1982, Growns & Davis 1991, Burton & Ulrich 1994). In contrast, pre-treatment Predator relative abundances were significantly higher (P < 0.01) for the overall data (all stations, all dates) and for each of the 5 stations.

Station -- Predator densities and relative abundances for the entire 1987-1994 study varied by sampling station and site (Figs. 36-37). Stations B and C usually had the highest overall mean densities (13.7 - 19.3 per rock pack), while the three upstream stations normally had lower densities (10.4 - 16.3). Overall mean relative abundances ranged slightly from a low of 5.2% at Site Bb to a high of 9.7% at Site Hb. Station B and C densities were significantly higher than at all upstream stations, but the results for relative abundances were mixed (Table 17).

Site Position -- Of the total 18,968 Predators collected, 10,041 were from Above sites (661 rock packs, mean density = 15.2), while 8,927 were from Below sites (651 rock packs, mean abundance = 13.7). Predator densities were significantly higher at Above sites for the overall data and for Stations C and I. However, Predator relative abundances were not significantly different between Above and Below sites for the overall data and for each station (Table 16).

Season -- Of the 18,968 Predators collected, 11,881 were from 7 spring sampling dates (673 rock packs, mean density = 17.7), while 7,087 were from 7 fall sampling dates (639 rock packs, mean density = 11.1). Thus, Predator densities were significantly higher in spring (P < 0.01) for the overall data and for all stations, except at G (Table 16). In contrast, Predator relative abundances showed no significant seasonal differences.

f. Minor Functional Feeding Groups

Two other Functional Feeding Groups were found in N. Caspar Creek, Macrophyte piercers and Parasites. Only 396 Macrophyte Piercers and 1 Parasite were collected in this study. The caddisfly *Hydroptila* was the only Macrophyte Piercer, while the horsehair worm was the only Parasite. Undoubtably, Macrophyte Piercers were uncommon in N. Caspar Creek because aquatic macrophytes and large quantities of filamentous algae were generally absent.

Table 5. List of 95 macroinvertebrate taxa collected (1987-1994) using rock and leaf packs from N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA

INSECTS	FUNCTIONAL FEEDING GROUP*
COLLEMBOLA (Springtails)	(Cg)
EPHEMEROPTERA (Mayflies)	
Baetidae	
Baetis	(Cg)
Centroptilum	(Cg)
Ephemerellidae	
Drunella	(Sc)
Ephemerella	(Cg)
Serratella	(Cg)
Timpanoga hecuba	(Cg)
Heptageniidae	
Cinygma	(Sc)
Cinygmula	(Sc)
Epeorus	(Cg)
Ironodes	(Sc)
Nixe	(Sc)
Rhithrogena	(Cg)
Leptophlebiidae	
Paraleptophlebia	(Cg)
Siphlonuridae	
Ameletus	(Cg)
PLECOPTERA (Stoneflies)	
Capniidae	(Sh)
Chloroperlidae	
Kathroperla	(Cg)
Other Chloroperlidae	(P)
Leuctridae	(Sh)
Nemouridae	
Malenka	(Sh)
Soyedina	(Sh)
Zapada	(Sh)
Peltoperlidae	
Soliperla quadrispinula	(Sh)
Perlidae	
Calineuria californica	(P)
Hesperoperla	(P)

Perlodidae	(P)
Pteronarcyidae	
Pteronarcys	(Sh)
ODONATA (Dragonflies)	
Cordulegastridae	
Cordulegaster dorsalis	(P)
Gomphidae	
Octogomphus specularis	(P)
TRICHOPTERA (Caddisflies)	
Apataniidae	
Apatania	(Sc)
Brachycentridae	
Micrasema	(Sh)
Calamoceratidae	
Heteroplectron californicum	(Sh)
Glossosomatidae	
Glossosoma	(Sc)
Goeridae	
Goeracea	(Sc)
Hydropsychidae	
Hydropsyche	(Cf)
Parapsyche	(Cf)
Hydroptilidae	
Hydroptila	(Mp)
Palaeagapetus	(Sh)
Lepidostomatidae	
Lepidostoma	(Sh)
Limnephilidae	
Cryptochia	(Sc)
Ecclisomyia	(Cg)
Hydatophylax	(Sh)
Psychoglypha	(Sh)
Odontoceridae	
Namamyia plutonis	(Cg)
Nerophilus californicus	(Sh)
Parthina	(Sh)
Philopotamidae	
Wormaldia	(Cf)
Phryganeidae	
Yphria californica	(P)
Polycentropodidae	
Polycentropus	(P)
Rhyacophilidae	
Rhyacophila	(P)

Uenoidae	
Farula	(Sc)
Neophylax	(Sc)
MEGALOPTERA (Dobsonflies, Alderflies)	
Corydalidae	
Dysmicohermes	(P)
Sialidae	
Sialis californica	(P)
COLEOPTERA (Beetles)	
Dytiscidae (Predaceous Diving Beetles)	
Agabus	(P)
Elmidae (Riffle Beetles)	
Narpus	(Cg)
Optioservus quadrimaculatus	(Sc)
Zaitzevia parvula	(Cg)
Hydraenidae (Minute Moss Beetles)	
Hydraena	(Sc)
Hydrophilidae (Water Scavenger Beetles)	(P)
Psephenidae (Water Penny Beetles)	
Acneus	(Sc)
Eubrianax edwardsi	(Sc)
DIPTERA (Flies)	
Ceratopogonidae (Biting Midges)	(P)
Chironomidae (Midge Flies)	(Cg)
Culicidae (Mosquitoes)	(Cf)
Dixidae (Dixa Midges)	
Dixa	(Cg)
Meringodixa chalonensis	(Cg)
Ephydridae (Shore Flies)	(Cg)
Empididae (Dance Flies)	
Chelifera	(P)
Clinocera	(P)
Hemerodromia	(P)
Pelecorhynchidae	
Glutops	(P)
Psychodidae (Moth Flies)	
Maruina	(Sc)
Pericoma	(Cg)
Ptychopteridae (Phantom Craneflies)	
Ptychoptera	(Cg)
Simuliidae (Black Flies)	
Simulium	(Cf)

Stratiomyidae (Soldier Flies)	(Cg)
Tabanidae (Horse & Deer Flies)	(P)
Tipulidae (Craneflies)	
Antocha	(Cg)
Dicranota	(P)
Hexatoma	(P)
Limnophila	(P)
Limonia	(Sh)
Pedicia	(P)
Rhabdomastix	(Sh)
Tipula	(Sh)
NON-INSECTS	
ANNELIDA	
Oligochaeta (Aquatic Earthworms)	(Cg)
COELENTERATA	
Hydridae (Hydroids)	
Hydra	(P)
CRUSTACEA	
Cladocera (Water Fleas)	(Cf)
Copepoda (Copepods)	(Cg)
Ostracoda (Seed Shrimp)	(Cg)
HYDRACARINA (Water Mites)	(P)
NEMATODA (Roundworms)	(P)
NEMATOMORPHA (Horsehair Worms)	(Par)
TURBELLARIA (Flatworms)	
Planariidae	
Dugesia	(P)

---

*FUNCTIONAL FEEDING GROUP: Cg = Collector-gatherers; Cf = Collector-filterers; Sc = Scrapers; Sh = Shredders; P =Predators; Mp = Macrophyte Piercers; Par = Parasites

Table 7. Comparison of stonefly taxa identified using two different collecting methods in Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

Using Nymphs Only (12 taxa)	Using Nymphs & Adults (23 taxa)[1]
PLECOPTERA	PLECOPTERA
Capniidae	Capniidae
	Capnia
	Mesocapnia
Chloroperlidae	Chloroperlidae
Kathroperla	*Kathroperla*
Other Chloroperlidae	*Alloperla delicata*
	Paraperla frontalis
	Sweltsa borealis
	Sweltsa new species
Leuctridae	Leuctridae
	Despax*ia augusta*
	Moselia infuscata
	Paralecutra divisa
	Paraleuctra occidentalis
	Paraleuctra vershina
	Perlomyia collaris
Nemouridae	Nemouridae
Malenka	*Malenka cornuta*
Soyedina	*Soyedina producta*
Zapada	*Zapada cinctipes*
	Zapada frigida
Peltoperlidae	Peltoperlidae
Soliperla quadrispinula	*Soliperla quadrispinula*
Perlidae	Perlidae
Calineuria californica	*Calineuria californica*
Hesperoperla	*Hesperoperla hoguei*
	Hesperoperla pacifica
Perlodidae	Perlodidae
Pteronarcyidae	Pteronarcyidae
Pteronarcys	*Pteronarcys*

---

[1] Adults collected 17-18 April 1985, 21 April 1991, 9 May 1991 and 10 January 1996.

Table 8. Abundant macroinvertebrate taxa (>5%) collected for 15 sample periods (1987-1994) from N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

Spring 1987			Fall 1987		
	Baetis	42.4 %		Chironomidae	27.1 %
	Chironomidae	14.2		*Glossosoma*	18.0
	Simulium	10.3		*Cinygmula*	11.9
	Glossosoma	5.5		*Baetis*	7.4
		72.4		*Heteroplectron*	5.6
					70.0
Spring 1988			Fall 1988		
	Chironomidae	31.8 %		Chironomidae	36.0 %
	Baetis	27.5		*Cinygmula*	8.5
	Malenka	7.0		*Baetis*	6.4
	Lepidostoma	5.8			50.9
	Paraleptophlebia	5.0			
		77.1			
Spring 1989			Fall 1989		
	Chironomidae	41.5 %		Chironomidae	22.5 %
	Baetis	17.9		*Ironodes*	11.7
	Paraleptophlebia	5.6		*Cinygmula*	7.0
		65.0			41.2
Spring 1990			Fall 1990		
				Chironomidae	34.4 %
	All rock packs destroyed by high discharge.			*Baetis*	14.4
				Ironodes	11.5
				Malenka	6.6
				Hydropsyche	5.9
					72.8
Spring 1991			Fall 1991		
	Chironomidae	36.7 %		Chironomidae	25.5 %
	Baetis	23.1		*Glossosoma*	18.6
	Paraleptophlebia	6.8		*Dixa*	7.1
	Cinygma	6.5		*Baetis*	5.2
		73.1			56.4
Spring 1992			Fall 1992		
	Chironomidae	34.3 %		Chironomidae	27.7 %
	Baetis	30.6		*Glossosoma*	17.4
	Simulium	5.9		Baetis	9.2
	Hydropsyche	5.9		*Heteroplectron*	6.2
	Glossosoma	5.5		*Optioservus*	5.1
		82.2		*Cinygmula*	5.1
					70.7
Spring 1993			Fall 1993		
	Chironomidae	28.1 %		Chironomidae	40.0
	Baetis	22.5		*Baetis*	13.5
	Optioservus	7.9		*Optioservus*	6.6
	Hydropsyche	6.0		*Ironodes*	6.2
	Paraleptophlebia	5.9			66.3
		70.4			
Spring 1994					
	Chironomidae	40.0 %			
	Baetis	11.4			
	Paraleptophlebia	6.9			
	Hydropsyche	6.4			
		64.7			

Table 9. List of common and abundant (> 1 %) macroinvertebrate taxa collected (1987-1994) from 10 sites in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

Taxon	Composition of Total Fauna (%)
Chironomidae	33.9
Baetis	18.1
Glossosoma	4.6
Paraleptophlebia	3.9
Hydropsyche	3.4
Malenka	3.2
Cinygmula	3.0
Simulium	2.9
Optioservus	2.8
Ironodes	2.6
Lepidostoma	2.5
Hesperoperla	1.7
Heteroplectron	1.7
Hydracarina	1.5
Calineuria	1.4
Cinygma	1.3
Serratella	1.1
Farula	1.0
Zapada	1.0
Total	91.6

Table 10. Macroinvertebrate rock pack sampling dates for 15 sample periods (1987-1994) from N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

Sampling Period		Placed	Removed	Days in Creek
1	SPRING 1987		24-25 May 1987	
2	FALL 1987		7-8 Nov 1987	
3	SPRING 1988		4 June 1988	
4	FALL 1988		27-28 Oct 1988	
5	SPRING 1989		6-7 Jun 1989	
6	FALL 1989		26-27 Oct 1989	
7	SPRING 1990		-- Washed Out --	
8	FALL 1990	31 Aug	20 Oct 1990	50
9	SPRING 1991	9 Apr	9 Jun 1991	61
10	FALL 1991		12 Oct 1991	
11	SPRING 1992	7 Apr	13 Jun 1992	67
12	FALL 1992		3 Oct 1992	
13	SPRING 1993		1 Jul 1993	
14	FALL 1993	11 Aug	9 Oct 1993	59
15	SPRING 1994	4 May	9 Jul 1994	63

Table 11. Macroinvertebrate (MI) abundance for 15 sampling periods between Spring 1987 and Spring 1994 in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

Sampling Period	Total No. MI Collected	Number of Rock Packs	Mean Number/ Rock Pack
Spring 1987	8,011	98	81.7
Fall 1987	5,186	67	77.4
Spring 1988	24,787	99	250.4
Fall 1988	15,356	97	158.3
Spring 1989	16,700	90	185.6
Fall 1989	13,680	98	139.6
Spring 1990	-- ALL ROCK PACKS DESTROYED BY HIGH FLOWS --		
Fall 1990	18,076	94	192.3
Spring 1991	18,209	97	187.7
Fall 1991	12,302	86	143.0
Spring 1992	43,984	94	467.9
Fall 1992	8,377	99	84.6
Spring 1993	17,643	96	183.8
Fall 1993	18,924	98	193.1
Spring 1994	51,174	99	516.9
	272,409	1312	207.6
All Spring	180,508	673	268.2
All Fall	91,901	639	143.8

Table 12. Macroinvertebrate (MI) abundance for the entire 1987-1994 study at 10 collecting sites along N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

Site	Total No. MI Collected	Number of Rock Packs	Mean Number/ Rock Pack
Bb	47,110	131	359.6
Ba	34,025	134	253.9
Cb	27,211	135	201.6
Ca	40,900	136	300.7
Gb	18,206	130	140.0
Ga	15,438	130	118.8
Hb	17,108	130	131.6
Ha	20,746	133	156.0
Ib	22,129	125	177.0
Ia	29,536	128	230.8
	272,409	1312	207.6
All Above	140,645	661	212.8
All Below	131,764	651	202.4

B. LEAF DECAY RATES

The decay rate of organic matter which enters a stream is determined by many physical, chemical, and biological factors, most of which can be altered by logging (Webster & Benfield 1986). The decay rate integrates the influence of all these factors and provides a basic measure of stream functioning. In N. Caspar Creek, we expected leaf decay rates to increase after logging because of potential increases in water temperature and nutrient concentrations.

Alder leaf decay rates were determined at Stations B, G, and I in N. Caspar Creek during six autumn periods from 1988 to 1993 (Table 18). A total of 432 leaf packs were studied during the six years (72, 36, 90, 72, 72, & 90). In addition, preliminary leaf decay studies were completed in 1986 using alder leaves at Station A and in side tributaries D and YZ. In 1987, tanoak leaf decay studies were done at Stations B, G, and I.

Macroinvertebrate shredders were observed to rapidly colonize the alder leaf packs in N. Caspar Creek. Shredders such as the large-cased caddisfly larvae *Hydatophylax* and *Heteroplectron* were observed on the leaf pack surface within 30 minutes after they were placed in the stream. We believe that the alder leaf packs were highly attractive food substrates for shredders in N. Caspar Creek. Visible decreases in the size and condition of leaf packs were observed on successive visits to N. Caspar Creek until the leaves were skelotonized and only the resistant mid-veins were present.

During the entire study period at N. Caspar Creek and at all locations, alder leaves decayed rapidly. All leaf decay rates (k) exceeded the 0.01/day "fast rate" category (Petersen & Cummins 1974), and ranged from about 0.015 to 0.09/day (Table 19, Fig. 38). These leaf decay rates were similar to previously reported rates for alder from many locations in North America and Europe (Table 20). The 0.0912/day rate for Site Ga in 1991 exceeded the highest published rate.

Alder leaf decay rates varied substantially between 1988 and 1993 in N. Caspar Creek, with much of the variation attributable to logging impacts. Rates were generally 0.02-0.03/day for the first three years of the study from 1988 to 1990. Following this period, rates approximately doubled for two years in 1991-1992, before returning to the initial rates in 1993 (Fig. 38). Alder leaf decay rates were significantly higher ($P < 0.05$) during 1991 and 1992 than the preceding and following years. No significant differences existed in rates between 1991 and 1992 , or between 1988, 1989, 1990 and 1993.

The alder leaf decay rates within each year were relatively similar, with a few exceptions in 1988 and 1989. In 1988, the decay rate at Site Ba was exceptionally high. The reason for this high value is unknown; however, it appears that something abnormal occurred during the final leaf pack collections. For all 1988 collections prior to the final date, the leaf packs at Site Ba were decaying at a rate (about 0.047/day) similar to the other sites. However, on the final date the entire leaf pack biomass had disappeared, eventhough the flow had remained low and stable.

In 1989, the decay rate at Site Ib was also much higher than the other 5 sites, possibly for two different reasons. First, logging-related changes occurred at Station I just prior to doing the 1989 leaf decay studies (Table 1, 18). We observed that cable logging in Unit L during May-September, 1989, added much debris (needles, branches, limbs, etc) to the channel just before the leaf packs were put in the stream. Normally, we expected Sites Ib and Ia to have similar leaf decay rates; however, the differences observed in 1988 may have been caused by site specific differences in the recent logging disturbances, such as debris quantities or other environmental changes. A second reason for the higher rate in 1989 may be the shortened study period. The 1989 leaf decay studies were terminated earlier than planned by high discharges on 23 October, which washed out the remaining leaf packs. Thus, the 1989 leaf pack studies were shortented from the typical 60-70 days to only 26 days. Because only two collections were available for calculating the leaf decay rates (k), rather than the normal 4-5 dates, more variability might be expected in the results.

We believe the leaf decay rates of 1988 - 1989 represented pre-treatment conditions, eventhough logging first occurred during the summer, 1989, in the upper basin. Logging impacts on the leaf decay rates would not be expected at most stations in fall, 1989, because N. Caspar Creek was still in summer-autumn low flow conditions at the time of the leaf studies and the riparian buffer zone moderated water temperature increases.

We expected the first logging impacts on decay rates to be observed in fall, 1990, following the 1989-1990 winter rainy season and the first period of higher runoff coming from the upper basin. This runoff might be expected to have higher than normal nutrient concentrations, which would have stimulated higher decay rates. In fact, the leaf decay rates in 1990 were somewhat lower than pre-treatment rates, possibly because of drought conditions. It was only in 1991 that leaf decay rates increased greatly, following additional logging in the basin and additional years of winter runoff.

We found no evidence of cumulative logging effects on the 1988-1993 alder leaf decay rates (Fig. 39). Alder leaf decay rates did not increase or decrease in a downstream direction. There were no significant differences in decay rates (1988-1993) between the three stations or between Above and Below sites. Water temperatures, an important factor in leaf decay rates, were generally similar during the leaf decay studies in N. Caspar Creek, although 1992 may have been slightly warmer than other years (Fig. 6, 40).

Preliminary alder leaf decay rates were determined in 1986 at three locations in the N. Caspar Creek basin, Station A (0.0297/day) and side tributaries D (0.0264/day) and YZ (0.0377/day). Station A was located on the mainstem of N. Caspar Creek downstream of Station B, but upstream of the confluence with side tributary YZ. The 1986 rate at Station A was the same as the pre-treatment rates found in 1988-1989, suggesting that pre-logging rates were relatively stable. The rate in side tributary D, an unlogged stream in 1986, was also identical to the pre-treatment rates. The decay rate was higher in side tributary YZ, which drained a recently logged sub-basin. Therefore, these preliminary alder leaf decay studies reinforced the 1988-1993 main results at Stations B, G, and I.

In the autumn of 1987, leaf decay studies were attempted at Stations B, G, and I using tanoak leaves. In contrast to the alder leaf studies, tanoak leaves were very resistant to decay, and the 1987 study was stopped when no measureable decay occurred after 8 weeks in the stream. Tanoak leaves were initially chosen for study because they naturally occurred along N. Caspar Creek and were probably used by stream MI's.

Table 18. Leaf pack sampling dates (and number of days in creek) for 6 autumn collections (1988-1993) at Stations B, G, & I in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

	Placed in Creek	1st Sample	2nd Sample	3rd Sample	4th Sample	5th Sample
1988	8 Sep (0)	15 Sep (7)	29 Sep (21)	13 Oct (35)	30 Oct (52)	
1989	14 Sep (0)	21 Sep (7)	9 Oct (26)	-- Washed Out --		
1990	11 Sep (0)	25 Sep (14)	9 Oct (28)	21 Oct (40)	6 Nov (56)	20 Nov (70)
1991	13 Sep (0)	28 Sep (15)	12 Oct (29)	26 Oct (43)	10 Nov (58)	
1992	10 Sep (0)	24 Sep (14)	9 Oct (29)	22 Oct (42)	7 Nov (58)	
1993	21 Sep (0)	28 Sep (7)	9 Oct (18)	27 Oct (36)	18 Nov (58)	18 Dec (88)

Table 20. Comparison of alder leaf decay rates (k) from several locations in the Holarctic region.

	Water Temp. (^{0}C)	Alder Leaf Decay Rate, k (days^{-1})	Comment	Study No.*
CALIFORNIA (*Alnus rubra*)				
N. Caspar Creek				
1988		0.0220-0.0750		1
1989	11	0.0205-0.0604		1
1990	8-13	0.0133-0.0218		1
1991	10-12	0.0453-0.0912		1
1992	10-13	0.0413-0.0636		1
1993	8-12	0.0202-0.0326		1
ALASKA (*Alnus tenuifolia*)				
Monument Creek	0	0.0513	many shredders	2
COLORADO (*Alnus tenuifolia*)				
Colorado River	warmer	0.0462	below a dam	3
Fraser River	cooler	0.0235	unregulated	3
Little Beaver Creek	0	0.0308	many shredders	4
MICHIGAN (*Alnus rugosa*)				
Tenderfoot Creek	18	0.077	summer study	5
Ford R. & Schwartz Cr.	12	0.0126-0.0171	fresh	6
Ford R. & Schwartz Cr.	9	0.0058-0.0086	senescent	6
OREGON (*Alnus rubra*)				
Watershed 10	4	0.0124		7
Mack Creek	4	0.0168		7
FRANCE (*Alnus glutinosa*)				
Garonne River	6-15	0.0065		8
GERMANY (*Alnus glutinosa*)				
Steina	5	0.0229	fresh	9

---

* [1]Present study; [2]Cowan et al. 1983; [3]Short & Ward 1980; [4]Short et al. 1980; [5]Maloney & Lamberti 1995; [6]Stout et al. 1985; [7]Sedell et al. 1975; [8]Chauvet; & [9]Gessner et al. 1991.

C. ALGAE

The algae of N. Caspar Creek were studied from spring,1987, to spring, 1994 (Table 21). This included 15 sampling dates for chlorophyll-a and algal biomass determinations, and 11 sampling dates for quantitative counts of diatoms. For the entire study, 750 samples each of chlorphyll-a and biomass were the maximum number possible, while 550 samples were the maximum possible for diatom abundance. However, some samples were lost from floods (50 samples), droughts (10 samples), laboratory errors, and unknown reasons. Some tiles were found upside down in the creek, eventhough the flow had been low and stable, suggesting that an animal had flipped the tile looking for food. Because of these losses, there were a total of 556 chlorophyll-a samples, 631 algal biomass samples, and 474 quantitative diatom samples collected for the entire study.

Algae in north coast streams of California are normally light-limited (Burns 1972, Hill & Knight 1988), and, thus, changes in the forest canopy from logging are likely to produce significant changes in the stream algae. Canopy photographs taken by Harrison Phipps in 1989-1991 at Station G in N. Caspar Creek before and after logging clearly showed that the canopy was opened and solar radiation increased (Fig. 41). Solar Pathfinder measurements made along N. Caspar Creek near Station G before and after logging (1990) in Unit V also showed a decrease in the radiation (%) blocked (Pete Cafferata, California Department of Forestry and Fire Prevention, personal communication). Therefore, we expected to observe changes in the algae of N. Caspar Creek, although the amount of impact was unknown because of the moderating influence of the riparian buffer zone. Pre-logging canopy photographs taken a year apart at Station B showed no change (Fig. 42).

1. Algal Taxonomic Composition

Diatoms were the dominant algae in N. Caspar Creek. This group normally dominates the algal flora of light-limited streams, although other groups such as green algae, blue-green cyanobacteria, red algae and others may be present in low numbers. Diatoms are especially good at using low light levels or sun flecks, while green algae typically dominate locations with higher light levels. Stream algae can respond quickly to environmental changes in light, water temperature, nutrients, MI grazing, and other factors, causing noticeable shifts in species composition.

Algal identifications in this study of N. Caspar Creek focused primarily on diatoms. A total of 46 diatom species were identified from all stations in N. Caspar Creek during 1989-1994 (Table 22-23). Genera with many species included *Achnanthes, Gomphonema, Navicula, Nitzschia,* and *Surirella*. The total number of diatom species increased significantly with collection date (regression, $P < 0.01$) from 17 species to about 30 species (Fig. 44). Whether or not the post-treatment numbers were significantly higher than pre-treatment numbers could not be determined because only qualitative identifications were made early in this study. Nevertheless, it does appear that the total number of diatom species increased during and after logging, a finding which has been reported previously from other streams draining clearcuts (Holopainen et al. 1991). One obvious decline in species number occurred in spring, 1993, and this was undoubtably caused by a peak discharge of 242 cfs on 20 January which scoured and buried benthic diatoms. Recovery from this decline was rapid (Fig. 44).

The overall mean number of diatom species present at each collecting site was normally 11-12, a relatively small, non-significant, difference (Fig. 43). However, the number of diatom species at each collecting site varied by date, with Station B and Sites Ca and Ga having significant positive regressions ($P < 0.05$) with time (Fig. 45). At many sites, increasing species number was especially evident after spring,1990, which experienced an unusual, aseasonal, flood discharge of 150 cfs on 27 May. It appeared to take a year or more for species number to recover from this flood. Therefore, at least some of the observed increases in species number during and after logging probably was a recovery response from flood events. The spring, 1993, decline in total diatom species from the 242 cfs peak discharge (Fig. 44), was also observed at most stations (Fig. 45). This flood must have scoured and buried benthic diatoms; however, the number of diatom species recovered to previous levels within a few months (Fall 1993).

During the quantitative diatom studies at N. Caspar Creek (1989-1994), a total of 94,800 diatom cells were counted. The vast majority were composed of two species, *Achnanthes lanceolata* (47%) and *Cocconeis placentula* (38.5%), with each of the remaining 44 species only composing 5% or less of the counted cells (Table 23-24). The abundance of different diatoms varied with sampling date (Table 23). The filamentous diatom *Melosira* wasn't found early in the study, but became more common later, corroborating field observations of its increased abundance.

Shifts in the algal taxonomic composition after logging were noted by direct field observations of N. Caspar Creek. Logging in Unit L during the summer-fall,1989, opened the forest canopy and greatly increased the light penetrating to the stream at Station I. Green filamentous algal mats were observed at Station I after this logging. The green algae (possibly Zygnematales) were especially common at Site Ib along the stream margins and in low flow reaches. Filamentous diatoms (possibly *Melosira*) also were seen. During the period when green algae were obvious at Station I, no other station downstream had visible growths of filamentous algae.

Logging in Unit V during summer-fall, 1990, opened the forest canopy and greatly increased the light penetrating to the stream at Station G. Again, small patches of green filamentous algae were observed at Station G. Similar observations were made at Stations B and C after Unit B was logged in the fall, 1991. Filamentous green algae continued to be observed near B and C until the end of the study in 1994, especially at Site Ca. The green filamentous algae at this site was sparse and appeared to be near its lower light limit. This green alga was thin and had a "slimy" feel. The green filamentous alga *Cladophora,* which is typically observed in clearcut streams (Erman & Ligon ca. 1985), was not observed in N. Caspar Creek during this study.

The filamentous red alga *Batrachospermum* was seen in the lower reaches of side tributary YZ.

2. Algal Chlorophyll-a

Chorophyll-a levels, which indicate the standing stock of all algae present, were determined at 10 sites in N. Caspar Creek (1987-1994). Although diatoms predominated on the clay tile samples, chlorophyll-a measurements included all algal groups present. Normally, the clay tiles were coated with a visible brownish diatom layer adhering to the tile surface. Some tiles had MI Scraper grazing scars or trails along the surface. Green filamentous algae never attached to the tiles. Occasionally, filamentous diatoms (*Melosira* ?) not attached to the tile surface were observed, though these usually washed off the tile when it was removed from the water.

Chlorophyll-a levels in N. Caspar Creek typically ranged from less than 0.5 $\mu m/cm^2$ to over 5 $\mu m/cm^2$ (Fig. 46). Post-treatment chlorophyll-a levels were significantly higher than pre-treatment levels at all collecting sites, both when the overall data (all collecting dates and all sites combined) were tested and when the data from each station were analyzed separately. This response to logging was probably caused by increased solar radiation reaching the creek, but may also have been enhanced by increased water temperatures and nutrients in N. Caspar Creek. Direct evidence of increased solar radiation was recorded in canopy photographs taken before and after logging at Station G (Fig. 42). The increased chlorophyll-a levels after logging also indicated little or no adverse impacts from fine inorganic sediments.

Chlorophyll-a levels in N. Caspar Creek varied considerably between sampling dates (Fig. 46). One source of variation was season, with fall samples usually having significantly higher values ($P < 0.01$) than spring samples (Table 13). This seasonal trend was most obvious during the pre-treatment period (Spring 1987 - Spring 1989). Higher cholorphyll-a values in fall reflected the increased diatom growths accumulated during the low stable flows of summer and fall. During spring, the algal growths had yet to increase after being scoured and diminished by the variable, high winter flows. Some of the unexplained larger variations, such as the lower values during spring, 1993, were undoubtably caused by high peak flood discharges.

No significant differences in chlorophyll-a were found between Above and Below sites when the overall data (all collecting dates and all sites combined) were tested and when the data from Stations B, C, and G were analyzed separately (Table 13).

Chlorophyll-a levels in N. Caspar Creek increased significantly in a downstream direction from Station I to B (Fig. 47). During the pre-treatment period, small differences existed in chlorophyll-a between upstream and downstream sites, but the downstream sites still had significantly higher levels (positive regression, $P < 0.05$). During the post-treatment period, chlorophyll-a levels were much higher (Fig. 47), and the significant differences between upstream and downstream sites were large (positive regression, $P < 0.001$). These upstream-downstream differences appeared to be a cumulative logging effect, with greater impacts at downstream sites than upstream sites. However, at least some of the upstream-downstream increase occurred naturally, as demonstrated by the pre-treatment results.

Chlorophyll-a levels and MI Scraper relative abundances varied with treatment and season (Fig. 48). During the pre-treatment period, spring chlorophyll-a and Scraper relative abundances were low, but both increased during fall. During the post-treatment period, both spring and fall chlorophyll-a values greatly increased, while Scraper relative abundances were greater in the fall.

3. Algal Biomass

Algal biomass values were also determined at 10 sites in N. Caspar Creek (1987-1994), and the results were similar to that found for chlorophyll-a (Fig. 49). Biomass levels typically ranged from less than 0.1 mg/cm^2 to over 1 mg/cm^2 (Fig. 49). Post-treatment biomass levels were significantly higher than pre-treatment levels both when the overall data (all collecting dates and all sites combined) were tested and when the data from each station were analyzed separately.

Algal biomass levels in N. Caspar Creek varied considerably between sampling dates (Fig. 49). Seasonal differences were not as pronounced as for chlorophyll-a, and there was no indication of higher values in fall. Spring biomass values were significantly higher than fall values for the overall data and at Station G, but the other four stations were not significantly different (Table 13). Distinct seasonal differences in biomass may have been more difficult to observe because of our sampling methodology. Our reported biomass values included both algal biomass and any fine organic detritus which might have settled onto the tiles. Usually, tiles accumulated algal biomass during low stable flow periods when detrital transport was low. However, we did observe fine detritus on some tiles, most obviously as brown accumulations on the pre-ashed filter. Some samples were undoubtably abnormally high from deposited organic sediments after higher flows. Spring biomass values might be expected to contain greater amounts of organic detritus because of the higher discharges.

In contrast to the chlorophyll-a results, the upstream-downstream biomass regression was not significant because of higher values at Station I (Fig. 50). We believe that chlorophyll-a values provided a better measure of algal standing crop than did our biomass measures because of the confounding effect of organic detritus.

4. *Achnanthes lanceolata - Cocconeis placentula*

Two diatom species dominated the algal community in N. Caspar Creek, *Achnanthes lanceolata* and *Cocconeis placentula* (Table 23-24). Distinct shifts in the relative abundances of these two species occurred during and after logging. At almost every site, *C. placentula* significantly increased in relative abundance, while *A. lanceolata* decreased (Fig. 51-55). This change in relative abundance was most obvious at eight sites (Bb, Cb, Gb, Ga, Hb, Ha, Ib, and Ia). Station H, which we judged to be the most heavily shaded location, also showed this shift in relative abundance, but it was less pronounced and *A. lanceolata* maintained generally higher abundances throughout the study. In contrast, at Site Bb the change between these two species was very pronounced, and the pattern of change was similar to the leaf decay results, with a definite period of change, followed by a period of recovery.

Table 21. Algae clay tile sampling dates for 15 sample periods (1987-1994) from N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

Sampling Period		Placed	Removed	Days in Creek
1	SPRING 1987		23-26 May 1987	
2	FALL 1987		7-8 Nov 1987	
3	SPRING 1988		4 June 1988	
4	FALL 1988		29-30 Oct 1988	
5	SPRING 1989		5, 8 Jun 1989	
6	FALL 1989		29 Oct 1989	
7	SPRING 1990		-- Washed Out --	
8	FALL 1990		20 Oct 1990	
9	SPRING 1991		9 Jun 1991	
10	FALL 1991		12 Oct 1991	
11	SPRING 1992		13 Jun 1992	
12	FALL 1992		3 Oct 1992	
13	SPRING 1993		1 Jul 1993	
14	FALL 1993	11 Aug	9 Oct 1993	59
15	SPRING 1994	4 May	9 Jul 1994	63

Table 22. List of diatoms identified* (1989-1994) from Stations B, C, G, H, & I in N. Caspar Creek, Jackson State Demonstration Forest, Fort Bragg, CA.

Achnanthes deflexa
Achnanthes exigua var. *heterovalva*
Achnanthes lanceolata var. *dubia*
Achnanthes lanceolata var. *lanceolata*
Achnanthes laterostrata
Achnanthes microcephala
Achnanthes minutissima
Achnanthes saxonica
Amphora perpusilla
Cocconeis pediculus
Cocconeis placentula var. *lineata*
Cymbella minuta
Cymbella tumida
Diploneis elliptica
Epithemia adnata
Eunotia maior
Eunotia sp. VIII-12
Gomphonema acuminatum
Gomphonema clevei
Gomphonema constrictum
Gomphonema cf. *subclavatum*
Gyrosigma sp. XII-11
Melosira sp. XIV-1
Navicula cf. *cryptocephala*
Navicula elginensis
Navicula sublitissima
Navicula sp. XIV-11
Navicula sp. XIX-33
Navicula sp. XX-1
Navicula sp. XXIII-26
Nedium sp. XVIII-1
Nitzschia frustulum
Nitzschia inconspicua
Nitzschia sigmoidea
Nitzschia sp. I-21
Nitzschia sp. VI- 4
Nitzschia sp. XII-3
Pinnularia sp. I-33
Rhoicosphenia curvata
Rhopalodia gibberula
Rhopalodia sp. XX-17
Surirella linearis
Surirella sp. XVI-33
Surirella sp. XVII-24
Surirella sp. XIX-35
Surirella sp. XXII-19
Synedra ulna
Tabellaria sp. XXV-12

---

*Diatom species designated by Roman numerals (film roll #) and Arabic numbers (negative #) refer to the photographic reference collection of Angela Kost, Davis, CA. (var. = variety; cf = looks like). Diatom identifications verified by J. Patrick Kociolek, Diatomist, California Academy of Science, Golden Gate Park, San Francisco.

V. DISCUSSION

Based upon the results of this study, we concluded that the eight logging treatments during 1989-1991 in the N. Caspar Creek drainage basin did cause changes in the three components of stream structure and function measured, (1) the benthic macroinvertebrate community, (2) leaf litter processing rates, and (3) the benthic algal community. However, the measured biological variables responsed to logging somewhat differently than we expected.

The specific objectives of this report focused upon fourteen predictions of how the stream biota would respond to several logging impacts which we initially thought had high potential to alter the stream. The fourteen predictions and the results of the study are summarized below:

	Study Result
A. Expected Increases in Fine Sediment from Logging	
1. Macroinvertebrate density will decrease.	No
2. Macroinvertebrate number of taxa will decrease.	No
3. EPT density will decrease.	No
4. EPT number of taxa will decrease.	No
5. Chironomidae density will decrease.	No
B. Expected Increases in Light, Water Temperature, & Nutrients from Logging	
6. Macroinvertebrate density will increase.	Yes
7. Chironomidae density will increase.	Yes
8. *Baetis* density will increase.	Yes ?
9. Scraper relative abundance will increase.	Yes
10. Leaf decay rates will increase.	Yes
11. Algal chlorophyll-a will increase.	Yes
12. Algal biomass will increase.	Yes
13. Algal species composition will change.	Yes
C. Expected Decreases in Allochthonous Detritus from Logging	
14. Shredder relative abundance will decrease.	Yes ?

A. SEDIMENT EFFECTS

Because of the steep hillslopes and erodible soils in the N. Caspar Creek basin, we initially expected the greatest logging impacts would come from habitat degradation by increased amounts of fine inorganic sediments. However, little or no evidence of sediment-related impacts was found in this study, and none of the five sediment-related predictions occurred.

None of the MI, EPT, or Chironomidae densities had significant decreases for the overall data or for each station following logging (Table 13). In fact, all significant differences in density found after logging were increases, strongly indicating that fine sediment had little or no impact on densities. This result was particularily significant for EPT densities, which normally are dramatically decreased as fine sediment reduces their benthic habitat (Waters 1995). Even if the two collection dates having very large densities were excluded, there was no indication of a post-logging decline.

The stream literature is nearly unanimous that introduced fine sediment will decrease the number of MI taxa or alter its taxonomic composition, especially for the EPT (Waters 1995). Despite this potential, there was little or no evidence that MI and EPT taxa number decreased or species composition changed in N. Caspar Creek during this study. In fact, the number of taxa significantly increased at all stations after logging (Table 13), indicating that fine sediments had little or no effect. Taxa present at the start of the study were also present at the end, and there were no noticeable declines or disappearance of taxa. The six major taxonomic groups maintained similar relative abundances throughout the study, none having significant increasing or decreasing regressions with time. The exact cause for the post-logging diversity increases was unknown, but may be from the increased amounts of algae. Also, a part of the increases may reflect the direct relation between taxa number and density (Fig. 15), and it may be better to normalize taxa numbers by sample size.

The absence of distinct sediment-related MI and EPT density and taxonomic changes in N. Caspar Creek raises several interesting questions. First, it may be that fine sediment impacts have yet to occur in N. Caspar Creek; that very large inputs may occur in the future from more extreme climatic events than have been experienced to date. However, although the 1987-1994 study period contained many drought years, a high peak discharge (242 cfs) occurred in January, 1993, without noticeable decreases in the MI or EPT after that event. This question can be more fully examined when actual sediment transport data for N. Caspar Creek become available.

Our results suggest the possibility that the MI fauna of N. Caspar Creek prior to this study was already adapted to large quantities of fine sediment in the channel substrate. Large inputs of fine sediment may be the natural condition of N. Caspar Creek because of its topography, climate, and soils. Previous studies in the Caspar Creek basin in the 1960-1980's demonstrated that N. Caspar Creek transported considerable quantities of sediment, experienced large landslides, and had a relatively fine channel substrate (Krammes & Burns 1973, Tilley & Rice 1977, Rice et al. 1979, Lisle 1989). There is even a remote possibility that present sediment conditions are still being influenced by the original logging of 80-100 years ago. Comparison of stream bed compositions prior to logging (Krammes & Burns 1973, Lisle 1989) and after logging may help to explain why so few sediment-related impacts were found. If substrate compositions were essentially the same during pre- and post-logging, then no differences would be expected in the benthic fauna.

Evidence that fine sediments influenced the MI fauna prior to this study was found on our initial visit to N. Caspar Creek on April, 1985. On that date the channel substrate was similar to its present composition, most commonly being pebbles and gravel, with many embedded in sand and silt. We particluarily noticed the fine substrate because it contrasted greatly with the coarse substrates of Sierra streams. On that date, preliminary sampling found 20 MI taxa at Station A just upstream of the sediment pond, while 35 taxa occurred immediately downstream of the sediment pond in a coarser substrate. This demonstrated the potential for higher MI diversity if coarser, less embedded, substrates were present in N. Caspar Creek. Therefore, whatever the reasons for the large amounts of fine sediment in N. Caspar Creek, it is likely that the MI fauna had already been impacted prior to this study and prior to the 1989-1991 logging. Similar-sized Sierra streams commonly have a mean of 20-50 MI taxa (Bottorff 1990).

Dominance of the MI fauna throughout this study by two taxa, Chironomidae and *Baetis* (together > 50%), also suggested that N. Caspar Creek was inluenced by fine sediment prior to logging. Both taxa have life cycle and behavioral features which allow them to readily recolonize disturbed streams, while more sensitive taxa lack these abilities.

If the MI fauna were impacted by fine sediments during this study, we would expect to observe differences between Above and Below sites, especially at Stations B, C, and G, located where side tributaries drained a logged sub-basin. Below sites should have received greater sediment impacts because of the localized entry of sediments from the logged sub-basin, but the Above sites wouldn't receive that localized impact. Therefore, Above sites should have higher MI and EPT densities and number of taxa than Below sites. Also, Stations H and I should have no differences between Above and Below sites because no logging occurred in their sub-basins. Of the 18 variables of MI and algae statistically tested for site differences, 60% of the Above-Below comparisons were not significant, while 31% had Above > Below and 9% had Below > Above (Table 13, 16). This result suggests a sediment-related impact. For individual stations, C had the greatest number of Above>Below comparisons with 10 of 18 (none Below>Above). All other stations, except I, had very few Above>Below comparisons and these were offset by having a few Below>Above comparisons. Therefore, the Above-Below comparisons do give some indication of a fine sediment impact in N. Caspar Creek, with the greatest impact being located at Station C. However, this conclusion is tentative because Station I, which lacked sub-basin logging, also had many significant Above>Below site comparisons (8 of 18). As sediment transport and substrate composition data becomes available, it would be interesting to reexamine Station C and others for possible correlations.

An alternative reason for the lack of sediment-related impacts in this study is that detrimental sediment impacts may have been offset by the positive effects of opening the forest canopy. Although such a result may at first appear unlikely, evidence exists in the stream literature of this counteracting effect (Murphy & Hall 1981, Murphy et al. 1981, Hawkins et al. 1983), including in streams similar to N. Caspar Creek from western Oregon and northern California. Streams exhibiting the counteracting effects of sediment and solar radiation usually lacked riparian buffer zones, causing them to be open to full sunlight. The riparian buffer zone along N. Caspar Creek probably minimized these counteracting effects, but sufficient additional sunlight (plus increased water temperatures and nutrients) must have increased stream algae, which resulted in enhanced MI populations, thus, masking any sediment impacts.

B. LIGHT, WATER TEMPERATURE, & NUTRIENT EFFECTS

Because riparian buffer zones were used in the N. Caspar Creek basin, we initially expected solar radiation to remain relatively low and to continue restricting algal growth. However, all eight predictions of the logging impacts on MI’s, leaf decay rates, and algae occurred during this study, providing strong evidence of an increase in light, water temperature, and/or nutrients. Increases in these factors led to significant increases in benthic algae, which translated into increased densities, relative abundances, and number of taxa in the MI’s. The only MI component with a questionable response was *Baetis* density, which only increased significantly at Station B, the location furthest downstream and with the highest standing crop of algae.

The exact factor or combination of factors causing the observed increases in algae are unknown. We believe that solar radiation was a primary factor, but it wasn't the only controlling influence, as was clearly demonstrated by significant increases in chlorophyll-a and biomass, and changes in the relative abundance of diatom species prior to increases in light from local logging. This suggests that water temperature and/or nutrients were also important factors.

Algal standing crop responded to logging in the N. Caspar Creek basin by increasing about three times above its original levels, a considerable response considering that riparian buffer zones were present. Although many other studies have reported increased amounts of algae following logging, most treatments were much more extreme, with the streams being opened to full sunlight and lacking buffer zones. Even after logging, N. Caspar Creek was still fairly heavily shaded and received direct sunlight in only short bursts or in small patches. The positive response of the algae to relatively small light changes may be caused by their high efficiency in utilizing available light (Naiman & Sedell 1980).

Although algae tend to respond rapidly to new environmental conditions, it was unclear if the present levels of algal standing crop in N. Caspar Creek represented a short-term or long-term effect. Nutrients probably are still variable and elevated in the creek, and the amount of light penetrating the riparian buffer is probably increasing as more trees are blown down. The toppling of riparian trees is occurring primarily on the south side of the stream, which has a greater effect on lighting conditions than trees lost to the north. These changes in the riparian zone demonstrated that logging impacts continue to occur at N. Caspar Creek and that substantial differences from our present results may yet occur. Because the light conditions continue to change, we believe that future studies at N. Caspar Creek should reexamine the status of the benthic algae, particularily because this group is an important component of the stream's energetics and determines the success of many MI's. The problem of windthrown trees in buffer zones has been reported by others (Steinblums et al. 1984, Murphy et al. 1986) and should receive more consideration at Caspar Creek.

C. ALLOCHTHONOUS DETRITUS EFFECTS

Allochthonous detritus forms the energetic foundation of most small forested streams and many MI's depend upon a regular annual transport of organic detritus from the riparian zone and from upstream reaches (Cummins 1973, 1974, Cummins & Klug 1979, Anderson & Sedell 1979, Naiman & Sedell 1980, Wallace & Merritt 1980, Cummins et al. 1989). Alterations in the amounts, kinds, and timing of allochthonous detrital inputs will effect the MI community, especially when logging occurs without a riparian buffer zone. Shredders usually decrease, either overall in all taxa or in certain species, because their CPOM food source is greatly diminished or altered (Woodall & Wallace 1972, Graynoth 1979, Haefner & Wallace 1981, Webster et al. 1983, Gurtz & Wallace 1984, Growns & Davis 1991, Burton & Ulrich 1994), but increases or no changes have also been reported (Silsbee & Larson 1983, Hawkins et al. 1992). Evidence exists that some Shredders can change to a Collector-gatherer feeding mode if their CPOM food source is decreased (Webster et al. 1983). In addition, detritus structurally modifies the stream habitat and increases the stream's retentiveness for nutrients and food particles, affecting MI groups other than Shredders.

Shredders were common in N. Caspar Creek, composing about 10% of the MI fauna, a numerical abundance which is typical of most small streams (Cummins et al. 1989). Because of the potential for clearcut logging to decrease allochthonous inputs to streams, we predicted that Shredders would decrease in N. Caspar Creek; however, we realized that the riparian buffer zone would moderate, or possibly nullify, this impact. In fact, Shredders did have lower post-treatment relative abundances at the two downstream stations (Table 16), possibly indicating a decline in allochthonous detritus. However, this result remains questionable because it wasn't substantiated by Shredder densities.

No logging impact predictions were made in this study for the other two detritus-dependent FFG's, Collector-filterers and Collector-gatherers, because it was unclear from the stream literature how these groups would respond. Reduction in the FPOM food source might be expected to translate into fewer Collectors; however, FPOM quality appears to be more important than quantity in these organisms. Studies have shown that FPOM quality is greatly increased if it has high microbial biomass or is derived from or associated with algae, which has high food value (Cummins 1973, Anderson & Sedell 1979, Cummins & Klug 1979, Naiman & Sedell 1979a, b, Wallace & Merritt 1980, Hawkins & Sedell 1981, Hawkins et al. 1982). Apparently, some Collectors selectively feed on higher quality FPOM, while detrital particles resistant to breakdown are passed quickly through the gut. Thus, Collector abundances usually are higher in streams with an open canopy and higher amounts of algae (Hawkins et al. 1982, 1983, Webster et al. 1983, Gurtz & Wallace 1984, Growns & Davis 1991), but there are some exceptions (Graynoth 1979, Burton & Ulrich 1994).

Collector-gatherers had significantly higher post-treatment densities in N. Caspar Creek at the two downstream stations with the highest algal standing crops (Table 16), suggesting that the FPOM deposits were linked with the algal production. This response was not found at the three upstream stations which had less algae.

Collector-filterers had significantly higher post-treatment densities and relative abundances at most locations in N. Caspar Creek (Table 16). This strongly suggested that the FPOM was linked with the increased algae, but it may also be that greater amounts of suspended FPOM was present after logging. Other studies have reported increased organic seston in streams following logging (Gurtz et al. 1980, Silsbee & Larson 1983, Golladay et al. 1987, Webster et al. 1988), but few have determined if this translates into more Collector-filterers. Hawkins et al. (1982) found higher abundances of Collector-filterers in streams open to sunlight and a significant positive correlation between filterers and algal biomass.

Although some of our results for the detritus-based FFG's were unclear, it should not be concluded that detritus was unimportant in N. Caspar Creek. First of all, the riparian buffer zone undoubtably continued to supply detritus and large woody debris to the stream after logging, at least moderating the losses from clearcutting. Although the riparian zone probably was the main source before logging, detritus also originated from beyond this zone, including the smaller side tributaries which drained clearcut sub-basins. Second, allochthonous detritus and dependent FFG's may not respond immediately to logging because the stream and adjacent banks contained stored quantities of detritus which may not be depleted for years. Third, the riparian buffer zone continues to be diminished from windthrown trees. Thus, the long-term effectiveness of the buffer zone remains unknown. The fallen trees have added considerable amounts of large woody debris to the channel and adjacent stream banks (O'Connor & Ziemer 1989). Fourth, because detritus originates from the entire watershed, logging impacts on allochthonous detritus tend to be long-term, with full recovery only following the regrowth of the forest (Webster et al. 1983, Wallace 1988, Webster et al. 1988, Niemi et al. 1990, Yount & Niemi 1990). Because of these complexities of allochthonous detritus at N. Caspar Creek, the results of our study provided little insight into how logging affected this basic stream component. The three FFG's which use detritus as a food source appeared to be more closely linked with the increased sunlight and amounts of stream algae. To determine these long-term effects in N. Caspar Creek, we recommend that detritus-dependent FFG's be reexamined in 5-10 years.

D. CHIRONOMIDAE

Numerous studies worldwide document the fact that the dipteran family Chironomidae (the midges) are often the most abundant MI group in streams (Armitage et al. 1995). This ubiquitous group has many different feeding modes and food sources, utilizes many stream microhabitats (bare rock surfaces, algal mats, detritus, accumulations of fine sediments, symbioses with plants and animals), and possesses rapid, multivoltine life cycles and good dispersal abilities which allow it to readily recolonize disturbed streams. Because of these adaptations, it is not surprising that chironomids are normally the most abundant MI in streams disturbed by logging (Burns et al. 1972, Newbold et al. 1980, Noel et al. 1986, Lamberti et al. 1991, Bilby & Bisson 1992).

Chironomidae were the most abundant macroinvertebrate group overall in N. Caspar Creek (33.9%), and this was a persistent feature throughout the study. They were also a dominant part of the stream drift measured in 1986-1987. It was difficult to predict how their densities would respond to fine sediment inputs to N. Caspar Creek. Although extreme amounts of fine sediment or scouring during floods can temporarily reduce densities (Lamberti et al. 1991), chironomids have the ability to recover and sometimes exceed their original abundances, especially if fine organic matter is mixed with the sediments. Because we primarily expected inputs of inorganic sediments, without organic pollution, we concluded their densities should decrease.

In contrast to the adverse effects of fine sediment, large increases in chironomids commonly occur when algal production is stimulated by increased light, water temperature, and nutrients (Newbold et al. 1980, Noel et al. 1986). Thus, actual chironomid densities in N. Caspar Creek were determined by at least two counteracting logging effects, adverse sediment impacts and positive algal impacts. Our results showed significantly higher densities occurred after logging at Stations B and C, the stations furthest downstream and the locations with the largest quantities of chlorophyll-a. At these two locations, the positive effects of increased algae were larger than the negative effects of fine sediment.

While it appeared that chironomids in N. Caspar Creek were responding to increased algae at Stations B and C, we cannot completely eliminate the possibility that fine sediment also influenced densities. Stations B, C, and G potentially were impacted by logging in their sub-basins, and if deposits of fine sediment were important in N. Caspar Creek, we should have found distinct Above-Below site differences. Above sites should have higher densities than Below sites. However, a different response occurred at each of these three stations (Table 13); no difference at B, Above>Below at C, and Below>Above at G. Thus, only at Station C was there an indication of a fine sediment impact.

E. ***BAETIS***

Many stream studies in North America have shown *Baetis* mayflies to be particularily responsive to logging disturbances (Newbold et al. 1980, Haefner & Wallace 1981, Webster et al. 1983, Gurtz & Wallace 1984, Noel et al. 1986, Wallace & Gurtz 1986). Clearcut logging which removes the forest canopy over a stream exposes it to additional solar radiation, resulting in increased algal production and higher *Baetis* densities. *Baetis* feeds on fine organic particles in streams, and its populations are especially enhanced by increased quantities of algae or algal-based detritus. Because *Baetis* has a relatively short, multivoltine life cycle with high fecundity, it has the ability to rapidly and opportunistically increase in numbers when its food base is increased. *Baetis* also has the ability to locate food-rich patches within streams because it readily disperses by drifting with the current. Because of these abilities, it is often one of the first macroinvertebrates to recolonize disturbed streams (Lamberti et al. 1991).

Baetis was the second most abundant (18.1%) macroinvertebrate in N. Caspar Creek after the Chironomidae (33.9%). It was abundant throughout the entire study, and was also a dominant part of the stream drift as shown in our preliminary studies (1986-1987). Because it was so common in N. Caspar Creek and because of past research, we fully expected *Baetis* densities to increase in N. Caspar Creek if algal standing crops increased. We expected *Baetis* density to be a sensitive indicator of logging disturbance. Nevertheless, *Baetis* density did not show distinct post-treatment increases, except at Station B, the location furthest downstream and the location with the largest quantities of chlorophyll-a and biomass. Results from the other stations were mixed, with pre- and post-treatments either being not significantly different or with higher pre-treatment densities. These results are difficult to explain, especially because both our indicators of algal standing crop, chlorophyll-a and biomass, had significantly higher post-treatment values at all stations.

A major difference between our study at N. Caspar Creek and other published studies was that the latter often lacked a riparian buffer zone, opening the stream to full sunlight. This resulted in large increases in algal growths and dramatic increases in *Baetis* densities. The buffer zone along N. Caspar Creek minimized changes in light and algae, resulting in a different *Baetis* response at different stations. Although our results were mixed, variation in *Baetis* densities with location did appear to be related with light and chlorophyll-a levels. Subjectively, Station H appeared to have the lowest light levels reaching the creek, and this location also had the lowest *Baetis* densities. Also, the three upstream stations had lower chlorophyll-a levels and *Baetis* densities, while the two downstream stations had higher chlorophyll-a levels and *Baetis* densities.

F. LEAF DECAY STUDIES

Alder trees are an important riparian species throughout western North America, and were common in the Caspar Creek basin, though less so at N. Caspar Creek. Alder leaves have special nutritional value for microbes and MI's because their nitrogen content usually exceeds 2%, far higher than for most other species (Kaushik & Hynes 1971, Hart & Howmiller 1975, Triska et al. 1975, Triska & Sedell 1976, Pozo 1993, Basaguren & Pozo 1994). Because of their importance to streams, many studies have determined their decay rates under a variety of environmental conditions (Table 20). The alder leaf decay rates obtained at N. Caspar Creek generally fall within the published range, essentially all of which were in the "fast" category (> 0.01/day). The decay rate at Site Ga in 1991 (0.0912/day) exceeded any previously published rate.

The only previous study of alder leaf decay in California didn't determine comparable rates (k); however, white alder leaves (*Alnus rhombifolia*) decayed faster than three other species and was completely decayed in only 34 days (Hart & Howmiller 1975). This study was done in southern California, in a stream warmer than N. Caspar Creek. Normally, leaf decay is directly temperature dependent (Webster & Benfield 1986); although, studies of alder leaves have also demonstrated rapid decay even at low water temperatures if Shredder populations were high (Table 20). Shredder abundance is known to affect the leaf decay rate (Benfield & Webster 1985).

One of the most interesting results of our biological studies at N. Caspar Creek was the two year doubling of alder leaf decay rates during and following logging, followed by a return to pre-treatment levels (Fig. 38). The exact logging impacts causing the increases in rates were unknown, though increases in light, water temperature, and nutrient runoff were thought to be primary factors. Other studies have demonstrated increased leaf decay rates after logging (Webster & Waide 1982, Meyer & Johnson 1983, Benfield et al. 1991). We found a short delay between logging and increased decay rates, a result that has also been reported for a stream draining a North Carolina clearcut site (Webster and Waide 1982). They found that leaf decay rates initially decreased, before increasing, because the leaf packs were buried by sediments generated from the clearcutting and because of decreased populations of the dominant Shredder. We also observed slightly lower rates in 1990 immediately after clearcutting, but do not believe this was due to sediment deposition.

One possible explanation for delayed increases in leaf decay rates was the timing of nutrient runoff coming from clearcut basins, especially nitrogen concentrations. Usually, clearcutting or disturbing a forest increases nitrate levels in streamwater and groundwater (Brown et al. 1973, Likens et al. 1977, Vitousek & Melillo 1979, Meyer & Johnson 1983). Elevated nitrate levels usually increase leaf decay rates (Kaushik & Hynes 1971, Meyer & Johnson 1983, Webster & Waide 1986); though a few studies found no effect (Triska & Sedell 1976). The mechanisms governing the timing and amount of nitrate runoff from clearcut basins is complex, causing each site to display different nitrogen characteristics (Vitousek & Melillo 1979). Swank (1988) reported that increases in nitrate were delayed 9 months after clearcutting and didn't reach peak values until the second winter. It may be possible that increased nutrient runoff from clearcutting was delayed in N. Caspar Creek in 1989-1990, especially because of drought conditions, and this affected the leaf decay rates.

Because we did not study streamwater chemistry in N. Caspar Creek, we do not know if nitrate levels changed and influenced leaf decay rates after clearcutting. Apparently, nitrate levels were low in N. Caspar Creek, based upon our one analysis of nitrate and phosphate concentrations made about 1990 in the mainstem and side tributary at Station H. Low levels of nitrate-nitrogen (<0.1 ppm) and phosphorus (< 0.15 ppm) were found in both locations. Hill and Knight (1988) also reported very low levels of nitrogen (< 14 µg/l) in two small streams on the north coast of California, north of Fort Bragg. Because N. Caspar Creek probably had low natural levels of nitrate, leaf decay rates would be especially likely to increase if nutrient levels were increased by logging. Further insights into the possiblity that nutrients stimulated decay rates in N. Caspar Creek may be gained when stream chemistry research becomes available.

In this study, the use of leaf decay rates to assess logging impacts proved to be an especially sensitive and powerful method which integrated numerous logging impacts into a single functional measure. Further, the field and laboratory methods were straightforward, not subject to experimental bias, relatively inexpensive, easily replicated, and the results were immediately available when the field work was completed. Because of these significant advantages, we highly recommend that additional leaf decay studies be performed in the Caspar Creek basin to answer some specific logging impact questions (see FUTURE STUDIES).

G. ALGAE

Diatoms are very responsive to changes in their aquatic environment, especially to changes in light, water temperature, nutrients, and velocity (Lowe 1974, Stockner & Shortreed 1976, Horner & Welch 1981, Bothwell 1988, Hill & Knight 1988, Duncan & Blinn 1989, Horner et al. 1990, Friberg & Kjeldsen 1994). Thus, the numerous effects of logging often produce dramatic shifts in species composition, chlorophyll-a and biomass (Hansmann & Phinney 1973, Lyford & Gregory 1975, Webster et al. 1983, Lowe et al. 1986, Holopainen et al. 1991, Ulrich et al. 1993). This responsiveness was also observed in the algae of N. Caspar Creek during this study.

All chlorophyll-a values measured in N. Caspar Creek were low to medium-low in comparison with numerous other studies in a wide variety of natural and laboratory streams (Stockner & Shortreed 1976, Steinman & McIntire 1990). During this study of N. Caspar Creek, chlorophyll-a had an absolute range (all dates, all sites) of 0.06 - 8.49 µg/cm^2 and a range of means of 0.11 - 5.46 µg/cm^2. Some of the lowest chlorophyll-a levels ever recorded in the stream literature (< 0.1 µg/cm^2) occurred in N. Caspar Creek prior to logging. These low values indicate just how light-limited streams such as N. Caspar Creek are and why their algal communities exhibited such distinct responses to changes in the forest canopy, even when these changes were moderated with riparian buffer zones. Post-treatment chlorophyll-a values were about three times higher than pre-treatment levels.

Algal biomass results were generally similar to those for chlorophyll-a, with additional variability introduced because fine organic detritus settled onto the clay tiles and these additions were interpreted as algal biomass. However, even with added detritus, the algal biomass values measured in N. Caspar Creek were usually low in comparison with numerous other studies from a variety of natural and laboratory streams (Steinman & McIntire 1990). During this study of N. Caspar Creek, algal biomass had an absolute range (all dates, all sites) of 0.024 - 1.990 mg/cm^2 (excluding an abnormal value, 5.737) and a range of means of 0.050 - 0.583 mg/cm^2 (excluding two abnormal values, 1.213 & 1.626). As with chlorophyll-a, some of the lowest biomass levels ever recorded in the stream literature (< 0.05 mg/cm^2) occurred in N. Caspar Creek prior to logging.

Of course, light is not the only important algal growth factor which can be altered by logging. Increases in nutrients such as nitrates and phosphates after logging can also stimulate algal growths. Although we did not study nutrient concentrations in N. Caspar Creek, we believe that increased nutrients, possibly in combination with increased water temperatures, increased algal growths at the two locations furthest downstream (B and C) prior to changes in their light conditions from local logging (Clearcut Units B and C). This response occurred not only in the chlorophyll-a and algal biomass variables (Fig. 46, 49), but also in the species composition changes between *Achnanthes lanceolata* and *Cocconeis placentula* (Fig. 51-52). This nutrient effect on algal growth and species composition prior to light changes was not observed at the upstream stations (G, H, & I). Therefore, it would appear the initial increases in algal growth at Stations B and C represented a cumulative effect of released nutrients from upstream logging in Clearcut Units E, G, J, K, L, and V.

Although algal standing crops were generally low in N. Caspar Creek because of the low light levels, it should not be concluded that algae were unimportant in stream functioning. Evidence exists in the literature that algae is very important, even in shaded west coast streams, because of its higher food quality than terrestrially-derived organic detritus (Minshall 1978, Anderson & Sedell 1979, Cummins & Klug 1979, Hawkins & Sedell 1981, Hawkins et al. 1982). Thus, the algae present in these streams can have a greater effect on the MI's than larger quantities of resistant detritus. The importance of algae to MI's reportedly translates into salmonid fishes and higher levels of the food web (Hawkins et al 1982, Bilby & Bisson 1992).

Several results in this study of N. Caspar Creek indicated that flood discharges temporarily disrupted the algal community (Fig. 44-45). The number of diatom species present was greatly decreased by the high peak flow of January, 1993, although recovery was rapid. This flood decreased chlorophyll-a levels (Fig. 46), but not biomass values (Fig. 49), which possibly remained at higher levels because of increased deposits of detritus. It also appears that the unusual flood in May, 1990, reduced the number of diatom species, but that recovery required 1-2 years, suggesting that high flows at atypical times of the year make recovery more difficult The diatoms in N. Caspar Creek may be more susceptible to flood damage than other streams because their attachment sites were primarily fine substrates which were easily moved by the currents (Steinman & McIntire 1990). In addition to movement of attachment substrates, fine sediment particles can also scour diatoms (Horner & Welch 1981, Horner et al. 1990).

We found a significant increase (2X) in the number of diatom species during and after logging in N. Caspar Creek (Fig. 44). Few other studies have compared species number before and after logging; however, Holopainen et al. (1991) also reported a doubling of species in a Finish stream.

Two prostrate species dominated the diatom community of N. Caspar Creek during this study, *Achnanthes lanceolata* and *Cocconeis placentula*, and both are commonly reported to be abundant or dominant in many other North American streams (Hansmann & Phinney 1973, Lowe 1974, Lowe etal. 1986, Hill & Knight 1988, Duncan & Blinn 1989). Both species attach close to the substrate and are often overlain by other algal species (Hill & Knight 1988). They are usually considered to be shade-tolerant species (Steinman & McIntire 1990). In N. Caspar Creek following logging, *A. lanceolata* decreased in relative abundance while *C. placentula* increased, the same result reported by Hansmann and Phinney (1973) from the Alsea basin streams (Oregon) following logging.

The exact reasons for the observed shifts in species composition is unclear from our study and from the literature because both species are sensitive to many interacting factors which determine their abundance and competitive abilites. Season is one factor, with *A. lanceolata* dominating in winter and spring, *C. placentula* dominating in summer and fall (Hansmann & Phinney 1973, Lowe 1974). Dominance by *A. lanceolata* at lower light levels agrees with our results of continuously high abundance at Station H, which had the lowest light levels in N. Caspar Creek (Fig. 54). Duncan and Scott (1989) found *C. placentula* abundance to be positively correlated with water temperature, while *A. lanceolata* was positively correlated with light energy, and nitrate was an interacting factor. Hill and Knight (1988) demonstrated significant increases in abundance for both *A. lanceolata* (17X) and *C. placentula* (2.5X) on nitrate-enriched substrates in north coastal streams. Lowe et al. (1986) also found *A. lanceolata* abundance was significantly increased on nitrogen and phosphorus enriched substrates, but in contrast to our result, this species had a higher abundance in clearcut streams. In addition to light, water temperature, and nutrients, *A. lanceolata* and *C. placentula* also are influenced by grazing pressures and compete for substrate space (Steinman & McIntire 1990).

The change in diatom species composition demonstrated by *A. lanceolata* and *C. placentula* was an obvious example of a logging impact. Other possible changes in the diatom community included an increase in *Melosira* after logging. But perhaps the largest shift in algal species composition in N. Caspar Creek occurred from increases in green filamentous algae after logging, while none were observed before logging. These conclusions were based only upon our general observations of N. Caspar Creek before and after logging, and not from quantitative measurements, although these observations have been confirmed by others (Cafferata 1990a). Increased mats of green filamentous algae appear to be one of the most commonly observed and reported changes in streams following logging or manipulations which increased light and nutrients (McIntire & Phinney 1965, Brocksen et al. 1968, Brown & Krygier 1970, Likens et al. 1970, Hansmann & Phinney 1973, Lyford & Gregory 1975, Stockner & Shortreed 1976, Graynoth 1979, Shortreed & Stockner 1983, Erman & Ligon ca. 1985, Lowe et al. 1986, Noel et al. 1986, Lamberti et al. 1991).

H. <u>SPRING - FALL VARIATION</u>

An interesting and somewhat unexpected aspect of our study of N. Caspar Creek was the distinct spring - fall change in biological variables. Most MI variables were higher in spring than in fall (Tables 13, 16). These seasonal variations can be explained by the typical annual life cycles of most aquatic insects (Hynes 1970). Eggs often hatch in late-summer or autumn, followed by growth during autumn, winter (sometimes slowed or stopped), and spring. Transformation to terrestrial adults occurs in late-spring or early-summer, and eggs are once again laid into the stream. Therefore, MI's appear to be more abundant in spring than in fall because in spring the aquatic larvae are relatively large, while in fall they are in small egg or early instar stages which are not captured by most collecting methods.

In contrast to the spring maxima of many MI's in N. Caspar Creek, Scrapers, Shredders, and chlorophyll-a were more abundant in fall (August-September) than spring (April-May). Algal standing crops apparently were greatly diminished by scour during the higher winter discharges, but began to increase as the flows stabilized in spring. The higher light levels and continued low, stable, flow conditions from spring to fall allowed benthic algae to accumulate, resulting in high standing crops in fall. Other studies have found low standing crops of stream algae in winter and spring, with increases at least through the summer months and into early-fall (Triska et al. 1983, Shortreed & Stockner 1983, Friberg & Kjeldsen 1994). Scrapers apparently have adjusted their life cycles to take advantage of maximum fall standing crops of algae. Previous studies of Scrapers report low spring abundances and higher summer abundances (Silsbee & Larson 1983, Duncan & Brusven 1985). It is also likely that Shredders have shifted their life cycle in N. Caspar Creek to be most abundant in fall when their detrital food source was most common and in a condition to support growth. Again, other studies of Shredders report higher fall abundances (Cummins et al. 1989)

During this study of N. Caspar Creek, we have been concerned with the large variability of most MI data sets, often questioning if the variability was mostly natural or caused by sampling errors. Most stream literature reveals large, inherent, MI variablity. However, the recurrent seasonal patterns we found in most N. Caspar Creek data sets suggested that the variability was natural rather than artificial. This is a significant observation because it adds confidence to these MI data sets. If most of the variability came from experimental errors, we would not expect to find distinct seasonal patterns.

I. <u>CUMULATIVE EFFECTS</u>

Although there is interest and concern over whether logging activities are causing cumulative effects on streams, few biological studies focus directly on this question. Bisson et al. (1992) have generalized from many logging studies that " Reduced stream habitat complexity has been one of the most pervasive cumulative effects of past forest practices....." However, we found no evidence of reduced complexity at N. Caspar Creek, primarily because of (1) riparian buffer zones, (2) no roads near the stream, and (3) logging methods which minimized soil disturbance.

At N. Caspar Creek, cumulative effects were directly determined by looking for different responses to logging between upstream (G, H, and I) and downstream (B and C) stations. This gradient from upstream to downstream was subjected to an increasing number of upstream logging areas (Table 4). Many cumulative effects were found, with distinct differences between the upstream (G, H, & I) and downstream stations (B & C).

The most obvious cumulative effect in N. Caspar Creek was the highly significant downstream increase in chlorophyll-a levels following logging (Fig. 47). This increase can probably be attributed to increases in light, water temperatures, and nutrients. Downstream increases in chlorophyll-a were generally corroborated by our measurements of algal biomass, except at Station

I (Fig. 50). We believe chlorophyll-a is a better indicator of algal standing crop than algal biomass because of the confounding effect of fine detrital accumulations. The distinct cumulative effect on stream algae was directly translated into the MI's and FFG's.

Essentially all measured variables in this study (Table 13, 16) showed the cumulative effects of logging. Macroinvertebrate density, MI taxa, EPT density, EPT taxa, Chironomidae density, and *Baetis* density all had significantly higher post-treatment values at Stations B and/or C, the furthest downstream, than the three upstream stations (Table 13-14). The same downstream cumulative effect was observed for all FFG densities, Scrapers, Shredders, Collector-filterers, Collector-gatherers, and Predators. However, few FFG relative abundances demonstrated cumulative logging effects. Essentially all of the cumulative effects appeared to be related to light, water temperature, and nutrients, rather than from adverse changes in fine sediment or allochthonous detritus.

Rather than look for cumulative effects in specific measured biological variables, examination of all data sets at all locations in N. Caspar Creek revealed an additional cumulative effect. Station B, the furthest downstream, and subjected to the highest cumulative logging impacts (Table 4), had the highest number of significant treatment differences (13 of the 13 measured variables). Station C, which had slightly fewer cumulative impacts, had fewer significant treatment differences (9 of the 13 measured variables). The three upstream locations, with even fewer cumulative impacts, had fewer significant treatment differences (7-8 of the 13 measured variables).

We found no evidence of a cumulative logging effect on leaf decay rates (Fig. 39). Leaf decay rates were sensitve indicators of logging and varied substantially with time, but remained remarkably stable between locations along N. Caspar Creek. This lack of distinct upstream-downstream differences in decay rates contrasted greatly with the other results of this study and raised some interesting questions. Many factors which control algal growth are also known to influence leaf decay rates (Webster & Benfield 1986). Therefore, why did leaf decay rates fail to show cumulative effects ? We believe the different results for leaf decay and algae may be caused by their different responses to solar radiation, with light-limited algae having a large response, while decay rates change little unless water temperatures are affected. In N. Caspar Creek, light levels distinctly increased during this study, but water temperature changes were controlled by the riparian buffer zone. Future studies should directly test if light influences leaf decay rates independently of water temperature. We know of no other published study which addresses this question.

Although most of our MI and algal results showed distinct upstream-downstream differences which can be interpreted as cumulative effects, we do not know which factors or combination of factors were responsible for these results. An understanding of these factors is crucial to firmly concluding that cumulative impacts occurred at N. Caspar Creek. At this point, we cannot eliminate the possibility that the upstream-downstream results were caused by random differences in logging treatments. In particular, knowing the amount of solar radiation reaching N. Caspar Creek at each station is especially important to answering this question. For example, were the increased downstream values caused by the specific details of logging in Clearcut Unit B, which unintentially allowed more light to reach the stream, causing more algae, which translated into abundant aquatic insect populations ? Did Clearcut Unit B randomally allow more light to reach the stream than did Unit V or Unit L ? An answer to these crucial questions about light can still be determined by comparative studies of the solar radiation conditions along N. Caspar Creek.

The River Continuum Concept describes the expected longitudinal changes in stream biological communities between their forested headwaters and lower reaches (Vannote et al. 1980). Because our study extended over a relatively short reach (1450 m) of N. Caspar Creek, we do not believe that the observed upstream-downstream changes in the biota were caused by the physical and chemical gradients described in the River Continuum Concept. Within our study reach, the stream increased somewhat in size, but the stream order and many other features were the same at all locations.

A word of caution should be given here concerning cumulative effects at N. Caspar Creek. Because some logging impacts on streams may be delayed for years after the actual clearcutting, we presently cannot claim to know the entire set of cumulative effects on N. Caspar Creek. Therefore, we want to emphasize the importance of follow-up biological studies at N. Caspar Creek to add new perspectives to the findings in this report.

J. REVIEW OF STUDY DESIGN AND DATA

We believe that our study design and methods proved to be adequate for clearly demonstrating many logging impacts on the stream biota of N. Caspar Creek, and that much was learned during this long-term study. Nevertheless, it is always possible with hindsight to design a better study, and avoid unexpected problems, so that treatment effects can be clearly detected. Therefore, here we review our N. Caspar Creek study, point out problem areas, and make suggestions for improvements.

1. Study Design

The ability to unequivocally detect logging impacts on the stream biota of N. Caspar Creek was greatly limited in this study because of the lack of control sites. All study sites were located downstream of logged areas and potentially could have been impacted. The importance of control sites was clearly understood in the initial research proposal which focused on comparisons between logged and unlogged side tributary basins. The greatest logging impacts were expected to occur in these small side tributary streams because they were much closer to the actual logging than the main channel of N. Caspar Creek. However, because it was found that many side tributaries ceased flowing even during normal water years, this study shifted focus from the side tributaries to the main channel of N. Caspar Creek. Presently, it is unclear to the senior author if Stations H and/or I were originally planned to be control stations, but that later alterations to the logging plans shifted timber harvests further upstream in the basin. In any event, the power of comparing treated sites with untreated controls was lost early in this study.

The lack of control sites was particularly serious in this biological study because of the inherent complexity of determining logging impacts coming from multiple treatments differing in space (8 logged sub-basins) and time (3 years), each one, or in total, potentially affecting all of our stream study variables. Adding to this complexity was the variable timing of the logging impacts after treatment; some occurring immediately, others being delayed. Further, some logging impacts act in opposite directions, such as the adverse effects of fine sediments and the positive effects of increased solar radiation. Because of these problems and the natural variability of the stream biota, it was seldom possible to ascribe year-to-year changes unequivocally to logging treatments. This was only possible where significant changes were large and persisted for several years after logging.

While pre-treatment and post-treatment conditions provided a partial control for our study, it had less power to detect logging impacts because it was not possible to confidently know if the study results came from natural year-to-year variations or from logging impacts. It was also unknown if pre-treatment conditions represented the natural undisturbed state, or if past logging from 80-100 years ago continued to influence N. Caspar Creek. Because of these problems in our study design, all results in this biological study have some ambiguity. With this study design, it is unlikely that logging impacts producing subtle changes in the stream biota could be detected with confidence, although larger changes were detected.

2. Data Collection

Long-term data sets such as exist for the Caspar Creek drainage basin are valuable assets for understanding how forestry practices impact natural resources. However, collecting consistently high-quality data is particularily important and difficult during long-term studies. Changes in field and laboratory personnel and equipment can introduce errors and bias into studies which decrease or destroy the ability to detect the main treatment effects.

We believe some portions of our biological study at N. Caspar Creek suffered from these long-term problems and added unnaturally to data variability. To minimize some of these problems, the senior author reexamined all data collected from 1986 to 1994. All identifications were critically reviewed and all calculations were checked, starting from the original data sheets and extending through data transcription onto computer files. Many corrections were made. This process only corrected data-handling errors, not the field collection or laboratory measurement errors. Original data sheets existed for all of the MI and leaf decay studies, and for most of the algal studies. Correcting the data-handling errors in the macroinvertebrate and leaf decay data sets were straightforward because only simple counts, percentages, or differences in weight were involved.

The most difficult data sets to verify in this study were for algal chlorophyll-a and biomass because some of the original data sheets were either missing or incomplete, especially from 1991-1992. Values unsupported by original data sheets were suspect. Many of the data sheets for chlorophyll-a and biomass failed to record the sample area, and for chlorophyll-a lacked basic information on the extraction volume or fluorometer door number used. Therefore, when checking these calculations, it was necessary to make several assumptions to arrive at the published results. The most common template used to collect algal samples had an area of 9.62 cm^2 and the most common extraction volumes were either 25 ml, 35 ml or 40 ml.

3. Data Analysis

As stated previously, this biological study of N. Caspar Creek was complex, not only because multiple biological variables were measured at many locations for 8 years, but also because multiple logging impacts varied spatially and temporally. To grapple with this complexity, we simplified our analysis to include (1) a single pre-treatment period before the first logging occurred in the upper basin in 1989 and (2) a single post-treatment period after the 1989 logging. The rationale for this division is that after the first logging in 1989 in the upper basin, all five sampling stations potentially could have been impacted and we had no way of knowing exactly when the impact would be observable in the stream after logging. Some logging impacts occur immediately, while others are delayed for months or years, or only become observable during times of high discharge. As logging continued further downstream in the following two years, sub-basins closer to downstream stations were impacted. While it is likely that these closer and later impacts had more influence on the downstream stations, we had no way of knowing if observed stream changes were caused by upstream logging or nearby logging. Comparison of Above and Below sites at each station only partially indicated the influence of local logging. Because of these complexities, we used the same pre- and post-treatment periods at all stations, rather than different periods at each station downstream. Using different pre- and post-treatment periods at each station would have been possible if the logging plan had proceeded from lower to upper basin.

When evaluating stream data sets for logging impacts, greater confidence existed that a true impact had been found when the data possessed distinct and sustained changes during or following logging; less confidence existed when the data had erratic spikes. Although MI's have the ability to respond rapidly to environmental changes, exceptionally large density increases, preceded and followed by lower typical densities only months apart, raised questions about the validity of some MI data.

Prior to doing statistical analyses on the MI data, we noticed that densities were very high on two collection dates, spring, 1992, and spring, 1994, and tried to determine if sampling methods were somehow different. For spring, 1994, a particularly energetic person collected the samples and stirred deeper into the substrate below each rock pack than normally. Therefore, densities were probably increased above normal levels, especially for chironomids, but only at the two downstream stations (B & C). As best we can determine for spring, 1992, field collections were made by experienced people in a manner similar to prior collections. Again, higher densities only occurred at the two downstream stations (B & C). Although the spring, 1994, collection may have contained some sampling error, sufficient reasons did not appear to exist for excluding either collection from the overall data set. Because season was a highly significant factor in most data sets, exclusion of two spring samples would have produced a fall bias in the results. Therefore, all collection dates were used in the statistical analyses of MI's (and EPT, Chironomidae, *Baetis,* and all FFG's).

In contrast to the uncertainties in the macroinvertebrate results, we had much greater confidence in the leaf decay and algal results because both data sets had distinct and sustained changes during and following logging.

There were many ways to analyze the data sets obtained in this study. We have attempted to extract as much information as possible from the results, but realize that others with different viewpoints and different abilities may be able to gain new insights into the logging impacts by using different analyses. Because of this possibility, and as an aid to potential future studies in N. Caspar Creek, we have included all data sets in the Appendix of this report. Revisiting some of these biological data sets may prove interesting in the future as other research data from the N. Caspar Creek basin are published. For example, correlation of the biological data with detailed data on sediment transport, channel substrate composition, solar radiation, and water chemistry could reveal important insights.

4. Improvements in Methods to Decrease Variability

Substantial experience was gained during this study into the value of using different stream variables to measure logging impacts. Each variable required different field and laboratory techniques, and each technique differed in its inherent variablilty, bias, and cost. In the following, we outline these techniques and suggest some improvements.

Macroinvertebrates -- The use of broad, community-wide, MI measurements to assess logging impacts provided several worthwhile results. However, MI data was inherently variable and required major investments in time and cost in the laboratory to completely extract ("pick") the MI specimens from accumulated detritus, and to identify and count each taxonomic group.

Variability was undoubtably introduced into the data sets because of (1) different thoroughness in extracting specimens from the detritus, (2) spillage or drying of some samples in the laboratory, and (3) different identification abilities. In this study, possibly as many as 100± different people were employed to pick specimens. Attempts were made to control the thoroughness of extraction by checking the picked samples for additional specimens, especially for new employees learning to recognize the MI's. No checks were made after the employee became experienced. Considerable time and expense could have been saved by extracting specimens from sub-samples rather than entire samples. The use of sub-samples would have been appropriate in this study because the samples contained numerous small organisms.

Variability in identification of specimens could have been prevented by making a reference collection early in the study. Succeeding taxonomists could have refered to this collection for questionable taxa and maintained consistency throughout the study. The reference collection would then have become a valuable resource for future studies at Caspar Creek and could have been placed in a permanent museum collection.

We believe that the use of rock packs to sample the MI's was particularily appropriate in the small, shallow dimensions of N. Caspar Creek. This technique undoubtably minimized the variability of field collections. However, the technique would be greatly improved by eliminating one of our steps. Upon collection of the rock pack, we moved it from the stream substrate into a net held downstream, then stirred 2-3 cm into the substrate. The stirring step should be eliminated because it is applied differently by each collector, some hardly stirring at all, others vigorously stirring. This stirring action introduced variable numbers of MI’s into the sample.

Leaf Decay Rates -- The use of leaf packs to assess logging impacts proved to be a valuable technique and sensitive indicator. The methodology was straightforward, less variable, and appeared to have less bias introduced from different personel. In addition, the measurements involved much less laboratory processing time and cost, and the results were quickly available once the field work was completed.

Algae -- The use of clay tiles to sample stream algae proved to be a valuable technique in N. Caspar Creek. Algae could be removed easily and quantitatively from the uniform surface. The only field improvement would be to fix the tile in place rather than just set the tile on the substrate. This would prevent the flipping of tiles by animals (racoons ?) which destroyed some samples.

The determination of algal chlorophyll-a, biomass, and species composition all contributed important results to this study. Chlorophyll-a was a particularily sensitive and accurate measure of logging impacts. The determination of chlorophyll-a was fairly straightforward, but required attention to detail in the laboratory, including recording extraction volume, sample area, and fluorometer door number on a standard lab sheet. Chlorophyll-a measurements provided fast results, were inexpensive, and were sensitive to stream changes.

Algal biomass was also a useful, sensitive measurement in this study and generally coincided with the chlorophyll-a results. However, biomass measurements were complicated by fine organic detritus which settles onto the tiles. The reported biomass was actually a combination of algae and organic detritus.

Quantitative measurements of diatom abundance also provided useful results in this study. However, the quantitative study of algal species would be improved if other algal groups, in addition to diatoms were included. Our present techniques destroyed the non-diatom groups. This would be an important change in methods because one of the expected logging impacts was a shift in species composition from diatoms to green algae. This result could not be tested using our methods.

K. BEST MANAGEMENT PRACTICES FOR MINIMUM LOGGING IMPACTS

Many published studies of how logging affects streams demonstrate dramatic changes in the aquatic biota. However, often the logging impacts were particularily severe because clearcut logging extended completely to the stream without buffer zones, roads and skidtrails were built in or close to the stream, and logging methods greatly disturbed the soil. With such drastic alterations of the drainage basin, it is not surprising that the stream biology changed.

The above disturbance scenario does not apply to the forestry practices used in the N. Caspar Creek basin. At N. Caspar Creek (1) riparian buffer zones were used along the main stream and the lower courses of larger side tributaries, (2) all road were kept far from the stream, (3) roads, skidtrails, and landing areas were minimized, (4) most logs were retrieved by cable yarding, especially on steep hillslopes, and (5) tractor skidding of logs was limited to lower slopes near ridgelines.

Because of these forest practices implemented to minimize impacts on N. Caspar Creek, we expected biological changes to be subtle or much less obvious than previous logging impact studies, and this was proven to be true. It also meant that much of the published literature, while helpful in understanding potential changes, may not be applicable to our studies at N. Caspar Creek.

Of all the forestry practices used in the N. Caspar Creek basin, riparian buffer zones were possibly the most important in protecting the stream environment. Numerous linkages exist between the stream and the riparian forest which govern ecosystem structure and function. Buffer zones help to maintain many of these linkages. The advantages of riparian buffer zones have been well described in many logging impact studies (Erman et al. 1977, Karr & Schlosser 1978, Graynoth 1979, Newbold et al. 1980, Erman & Mahoney 1983, Culp & Davies 1983, Steinblums et al. 1984, Barton et al. 1985, Murphy et al. 1986, Golladay et al. 1987, Growns & Davis 1991, Holopainen et al. 1991). It would appear that buffer zone trees along Clearcut Units B, V, and L at N. Caspar Creek are susceptible to windthrow. Additional considerations should be given to this windthrow problem when designing clearcuts and setting buffer zone widths.

Another forestry practice that we believe was very important at N. Caspar Creek was minimizing the length of logging roads and keeping them far from the stream. This one practice may be the main reason that few or no adverse fine sediment impacts were found in this study. Waters (1995) has emphasized that forest roads are often a major source of fine inorganic sediments to streams.

L. NATURAL VARIABILITY & RESILIENCE OF THE AQUATIC BIOTA

During the nine years of this study, we have observed large variations in the natural variablity of N. Caspar Creek. For many years, drought conditions existed, causing many side tributaries and portions of N. Caspar Creek to stop flowing. Drought conditions undoubtably influenced MI and algal populations, and also leaf decay rates, as well as changing the timing of logging impacts. In contrast, N. Caspar Creek also experienced large winter and spring floods during our study, which transported sediment and scoured the channel substrate, undoubtably again influencing the stream biota. Peak flows in winter often were

> 100 cfs compared with the low summer and autumn flows of < 0.3 cfs. Thus, annual peak flows were often 100-1000 times greater than low flows.

We believe that even in its natural, pristine condition, before any logging in the watershed, MI and algal communities were continually adjusting, recovering, and recolonizing sections of N. Caspar Creek after being impacted from the natural climatic and hydrologic events which characterize this region of California. The lack of major changes in MI taxonomic composition and only moderate changes in abundance between the start and end of this study despite droughts, floods, and logging, suggest that the benthic fauna has adaptations for withstanding and recovering from disturbances. The dominance of the N. Caspar Creek MI fauna by Chironomidae and *Baetis,* both adapted for dispersal, fast growth, and rapid recolonization, also indicates that regular disturbances are a feature of this stream.

Our conclusion that the stream macroinvertebrates in N. Caspar Creek were resilient to disturbances, has been reached in other studies which have shown relatively rapid recovery (a few years) from "pulse" disturbances, especially when the physical structure and complexity of the stream and its normal energy flow are maintained (Yount & Niemi 1990, Niemi et al. 1990, Wallace 1990). Stream algae also exhibit rapid recovery from disturbances because of their short generation times and good dispersal abilities (Steinman & McIntire 1990).

Recovery of stream organisms from "press" disturbances, which include clearcut logging of watersheds, usually awaits the reforestation process (decades). This long recovery process is particularly true when the press disturbance degrades or simplifies the stream habitat, such as when large quantities of fine sediment are deposited on the channel substrate, the riparian vegetation is lost, or the transport of detritus to the stream is greatly limited (Niemi et al. 1990, Gregory et al. 1991). Once a stream returns to its pre-disturbance condition, the stream biota reestablishes itself rapidly.

In this study, we found little or no evidence that the habitat of N. Caspar Creek was degraded or simplified from the basin clearcut logging, probably because the riparian buffer zones and forestry practices used minimized disturbances. If N. Caspar Creek was simplified by inputs of fine sediment, much of this occurred prior to logging, originating from the steep hillslopes and erodible soils. However, the possibility of future habitat simplification in N. Caspar Creek remains an unanswered question. Will increased quantities of fine sediment enter the creek in the next 5-10 years ? Will normal amounts and types of allochthonous detritus continue to be transported into the stream ? Will the riparian buffer zone continue to provide long-term protection to N. Caspar Creek ? Answers to these important questions and a complete understanding of the resilience of the stream biota in N. Caspar Creek will require future studies.

VI. CONCLUSIONS

GENERAL

1. Logging impacts on N. Caspar Creek were complex, with multiple potential effects varying in space and time.

2. Two major logging impacts were initially expected to be observed in the stream biota, one from fine sediment, another from solar radiation. Few definite sediment-related impacts were found, but many light-related (or water temperature and nutrient) impacts were observed.

3. Most macroinvertebrate and algal variables increased significantly after logging. Macroinvertebrates increased because of increased stream algae. Algae increased because of increased light, water temperature, and nutrients.

4. Little or no evidence was found of adverse sediment impacts on stream macroinvertebrates, leaf decay rates, or algae.

5. Stream algae and leaf decay rates proved to be very sensitive measurement variables of logging impacts, while macroinvertebrates were somewhat less sensitive variables because of their inherent variability

6. Cumulative logging impacts were observed in most macroinvertebrate and algae variables as distinct upstream-downstream differences. Station B, the furthest downstream, demonstrated significant increases in all measured variables after logging. The cumulative impacts were linked with increased light and algae. Adverse cumulative impacts due to fine sediment were not observed. Cumulative logging impacts were not observed in leaf decay rates.

7. Logging impacts on the N. Caspar Creek biota were often not dramatic because forest practices minimized the impacts. The three most important practices which ameliorated the impacts were the presence of the riparian buffer zones, the absence of roads near the stream, and the use of cable yarding which minimized soil disturbance.

8. Logging impacts continue to change the stream environment of N. Caspar Creek. Trees are being lost in the riparian buffer zone from high winds.

MACROINVERTEBRATES

9. Macroinvertebrate taxonomic composition did not change after logging.

10. Macroinvertebrate density increased significantly at three stations following logging.

11. Macroinvertebrate diversity increased significantly at all five stations after logging.

12. EPT (Ephemeroptera, Plecoptera, and Trichoptera) density increased significantly at two stations after logging.

13. EPT diversity increased significantly at all five stations after logging.

14. Chironomid density increased significantly at the two stations furthest downstream after logging.

15. *Baetis* densities increased significantly at the furthest station downstream, but decreased at two upstream stations after logging.

16. Scraper densities and relative abundances increased significantly at all stations after logging.

17. Shredder results were mixed. Densities increased significantly at two stations after logging. Relative abundances at the two stations furthest downstream were decreased significantly after logging, suggesting that allochthonous inputs may have been reduced.

18. Most Collector-filterer densities and relative abundances increased significantly after logging, possibly from increased seston quality or quantity.

19. Collector-gatherer densities increased significantly at the two stations furthest downstream after logging.

20. Predator densities increased significantly at the furthest station downstream after logging, but relative abundances were significantly lower at all stations.

21. Most macroinvertebrate variables were significantly higher in spring than fall. In contrast, Scrapers, Shredders and chlorophyll-a were significantly higher in fall. Scrapers and Shredders probably were most abundant in fall because their algal and detrital food sources were then most abundant.

22. There were few significant differences in macroinvertebrate and algal abundances between Above and Below sites within stations, indicating few sediment impacts. Only at Station C did Above-Below abundances suggest an adverse sediment impact.

LEAF DECAY

23. Alder leaf decay rates doubled for 2 years during and after logging, then returned to pre-logging levels. The increase in decay rates lagged the logging activities by one year or more. The lag may have been partially caused by drought conditions.

ALGAE

24. Before and after photographs of the forest canopy demonstrated that logging increased the amount of light reaching the stream.

25. Algal chlorophyll-a increased significantly at all stations after logging. Chlorophyll-a increased significantly from upstream to downstream locations, indicating a positive cumulative effect. Chlorophyll-a was significantly higher in the fall, probably because the stable, low flows of summer-fall allowed benthic algae to accumulate.

26. Algal biomass increased significantly at all stations after logging.

27. Algal species composition changed significantly after logging. Direct observations showed that green filamentous algae increased after logging. Diatom species composition also changed, with the relative abundance of *Achnanthes lanceolata* decreasing and *Cocconeis placentula* increasing. These algal changes were probably caused by increased light levels, but increased water temperature or nutrients may have also been important.

VII. FUTURE STUDIES

Based upon our current understanding of N. Caspar Creek, we believe that additional biological studies are warranted to address questions raised in the current study and to further understand long-term logging impacts and the recovery process. We feel this is particularily important in the N. Caspar Creek basin for at least two reasons. First, although all logging ceased in early-1992, new and unexpected impacts continue to affect the stream. The riparian buffer zone along N. Caspar Creek continues to change, with the loss of windthrown trees on the south side, opening the stream to additional solar radiation. Second, the potential for future sediment impacts on N. Caspar Creek remains high.

1. Reexamine Stream Biology in 5-10 Years -- The 1987-1994 study of N. Caspar Creek established the biological conditions before, during, and shortly after logging. Several logging impacts were discovered, with some being distinct, while others were subtle or questionable. These initial studies, plus the continued management of the Caspar Creek basin as a research watershed, present a unique research opportunity to learn about the long-term impacts of logging and the recovery process. We highly recommend that the stream biology of N. Caspar Creek be reexamined in 5-10 years. Comparisons between the two studies would provide important insights into (1) results that were subtle, questionable, or difficult to interpret in the original study, (2) the rate of recovery of different stream components and processes, and (3) additional impacts not detected in the original study which were delayed years from the actual clearcutting. Despite their value, few studies exist which compare logging impacts on streams over a long time frame.

2. Sediment / Stream Biota Studies -- As actual data on sediment transport and channel substrate composition becomes available from current and past research at N. Caspar Creek, the macroinvertebrate and algal data should be reexamined for possible correlations. An especially valuable comparison would be between pre- and post-logging substrate compositions and how these relate to the benthic faunal results. The amount and timing of sediment transport from sub-basins B, C, and G should be compared with the densities and diversities of macroinvertebrates and algae above and below each confluence. A similar reexamination of the data would be valuable when results of nutrient studies become available.

3. Solar Radiation Studies -- Many of the logging impacts at N. Caspar Creek appeared to be caused by changes in the amount of solar radiation reaching the stream. However, our understanding of these impacts were greatly limited because specific lighting conditions along the stream were unknown. Fortunately, the opportunity still exists to determine the lighting conditions. Such a post-logging study would greatly enhance understanding of our biological results, especially the observed upstream-downstream differences.

4. Leaf Decay Studies -- Leaf decay studies at N. Caspar Creek proved to be an especially sensitive and effective measurement of logging impacts. The method integrated many logging impacts into a single measure of an important stream function. Major advantages of the method included its low cost, fast results, replication, accuracy, and stability between field personnel. We believe that additional leaf decay studies should be performed in the Caspar Creek basin and would provide valuable insights on the condition of the stream. Some possible studies include the following:

(a) Determine if alder leaf decay rates have actually returned to pre-treatment levels in N. Caspar Creek. Presently, only one year (1993) suggests a recovery.

(b) Determine alder leaf decay rates along the complete longitudinal profile of the Caspar Creek basin looking for cumulative effects, starting in the headwaters of N. Caspar Creek and extending downstream into the main channel of Caspar Creek, as far downstream as Highway 1.

(c) Determine the influence of light on leaf decay rates independently of water temperature effects. Conduct a field experiment of decay rates in a fully sunlit reach, adding several different shading treatments.

(d) Compare alder leaf decay rates between logged and unlogged sub-basins in the N. Caspar Creek basin. Or design the comparisons between several logging treatments (burning, herbicides, aspect, etc.). Do herbicides affect leaf decay rates?

(e) Compare leaf decay rates between North and South Caspar Creek. This would tell if South Caspar Creek has recovered from its previous logging. It would also be an interesting comparison because S. Caspar Creek presently has much greater quantities of alder trees.

(f) Compare leaf decay rates between Caspar Creek and other drainage basins, possibly a pristine basin such as Elder Creek.

(g) Compare decay rates of other plant species than alder.

5. Algal Studies -- Algal studies proved to be a sensitive, fast, accurate, and cost effective method for measuring logging impacts. In particular, chlorophyll-a and species composition were useful. We believe that many interesting comparative studies could be designed to investigate the differences between differently treated sub-basins within both North and South Caspar Creek. Many of the comparisons described for leaf decay rate studies would also apply to algal studies.

6. Biodiversity of N. Caspar Creek -- Surprisingly, no stream in California has ever been completely inventoried to determine its resident aquatic species. Thus, we really do not know the magnitude and extent of species diversity of any stream. This information would be extremely valuable as future attention is focused more on the biodiversity of forested ecosystems. Such a study would immediately establish Caspar Creek as a reference stream to which others would be compared. The study would be valuable within Caspar Creek because comparisons of species between logged and unlogged sub-bsains could give a real measure of logging impacts. Such an inventory would require collection of adult aquatic insects on a year round basis using well established field techniques.

7. Long-Term Ecological Research -- Research has been conducted at Caspar Creek for at least 30 years, and valuable long-term data sets have been established. A special effort should be made to continue stream ecological research at Caspar Creek and to add the site to the network of Long-Term Ecological Research sites (Meyer et al. 1993). The value and rationale for these types of studies have been presented by Likens (1983). A fundamental question that needs to be answered at N. Caspar Creek is, "how resilient is the stream biota to logging disturbances"? Does recovery occur within a few years or decades?

VIII. REFERENCES

Aldridge, D.W., B.S. Payne & A.C. Miller. 1987. The effects of intermittent exposure to suspended solids and turbulence on three species of freshwater mussels. Environ. Poll. 45: 17-28.

Allan, J.D. 1995. Stream ecology: structure and function of running waters. Chapman & Hall, London. 388 pp.

Anderson, N.H., J.R. Sedell, L.M. Roberts & F.J. Triska. 1978. The role of aquatic invertebrates in processing of wood debris in coniferous forest streams. Am. Midl. Nat. 100: 64-82.

Anderson, N.H. & J.R. Sedell. 1979. Detritus processing by macroinvertebrates in stream ecosystems. Ann. Rev. Entomol. 24: 351-377.

Andrus, C.W., B.A. Long & H.A. Froelich. 1988. Woody debris and its contribution to pool formation in a coastal stream 50 years after logging. Can J. Fish. Aquat. Sci. 45: 2080-2086.

Armitage, P.D., P.S. Cranston & L.C.V. Pinder. 1995. The Chironomidae: biology and ecology of non-biting midges. Chapman & Hall, New York. 572 pp.

Aubertin, G.M. & J.H. Patric. 1974. Water quality after clearcutting a small watershed in western Virginia. J. Environ. Qual. 3: 243-249.

Barton, D.R., W.D. Taylor & R.M. Biette. 1985. Dimensions of riparian buffer strips required to maintain trout habitat in southern Ontario streams. N. Am. J. Fish. Manage. 5: 364-378.

Basaguren, A. & J. Pozo. 1994. Leaf litter processing of alder and eucalyptus in the Aguera stream system (North Spain). II. Macroinvertebrates associated. Arch. Hydrobiol. 128: 57-68.

Benfield, E.F., J.R. Webster. 1985. Shredder abundance and leaf breakdown in an Appalachian mountain stream. Freshwat. Biol. 15: 113-120.

Benfield, E.F., J.R. Webster, S.W. Golladay, G.T. Peters & B.M. Stout. 1991. Effects of forest disturbance on leaf breakdown in southern Appalachian streams. Verh. Internat. Verein. Limnol. 24: 1687-1690.

Beschta, R.L. 1978. Long-term patterns of sediment production following road construction and logging in the Oregon Coast Range. Water Resour. Res. 14: 1011-1016.

Beschta, R.L., R.E. Bilby, G.W. Brown, L.B. Holtby & T.D. Hofstra. 1987. Stream temperature and aquatic habitat: fisheries and forestry interactions (pp. 191-232). *In:* E.O. Salo & T.W. Cundy (eds.), Streamside management: forestry and fishery interactions. Univ. Washington, Institute of Forest Resources, Contribution 57.

Bilby, R.E. 1984. Removal of woody debris may affect stream channel stability. J. Forestry 82: 609-613.

Bilby, R.E. & P.A. Bisson. 1992. Allochthonous vs. autochthonous organic matter contributions to the trophic support of fish populations in clear-cut and old-growth forested streams. Can. J. Fish. Aquat. Sci. 49: 540-551.

Bisson, P.A., R.E. Bilby, M.D. Bryant, C.A. Dolloff, G.B. Grette, R.A. House, M.L. Murphy, K.V. Koski & J.R. Sedell. 1987. Large woody debris in forested streams in the Pacific Northwest: past, present, and future. *In:* E.O. Salo & T.W. Cundy (eds.), Streamside management: forestry and fishery interactions. Univ. Washington, Institute of Forest Resources, Contribution 57.

Bisson, P.A., T.P. Quinn, G.H. Reeves & S.V. Gregory. 1992. Best management practices, cumulative effects, and long-term trends in fish abundance in Pacific Northwest river systems (Chapter 7, pp. 189-232). *In:* Naiman, R.J. (ed.), Watershed management: balancing sustainability and environmental change. Springer-Verlag, New York.

Bjornn, T.C. 1974. Sediment in streams and its effects on aquatic life. Univ. Idaho, Idaho Water Resources Research Institute, Research Technical Completion Report, Project B-025-IDA, July 1972-October 1974. 47 pp.

Bormann, F.H., G.E. Likens, T.G. Siccama, R.S. Pierce & J.S. Eaton. 1974. The export of nutrients and recovery of stable conditions following deforestation at Hubbard Brook. Ecol. Monogr. 44: 255-277.

Bothwell, M.L. 1988. Growth rate response of lotic periphytic diatoms to experimental phosphorus enrichment: the influence of temperature and light. Can. J. Fish. Aquat. Sci. 45: 261-270.

Bottorff, R.L. 1990. Macroinvertebrate functional organization, diversity, and life history variation along a Sierra Nevada river continuum, California. Ph.D. Thesis, University of California, Davis. 179 pp.

Brocksen, R.W., G.E. Davis & C.E. Warren. 1968. Competition, food consumption, and production of sculpins and trout in laboratory stream communities. J. Wild. Manage. 32: 51-75.

Brown, G.W. & J.T. Krygier. 1970. Effects of clearcutting on stream temperature. Water Resour. Res. 5: 1133-1139.

Brown, G.W. & J.T. Krygier. 1971. Clear-cut logging and sediment production in the Oregon coast range. Water Resour. Res. 7: 1189-1198.

Brown, G.W., A.R. Gahler & R.B. Marston. 1973. Nutrient losses after clearcut logging and slash burning in the Oregon Coast Range. Water Resour. Res. 9: 1450-1453.

Bryant, M.D. 1983. The role and management of woody debris in west coast salmonid nursery streams. N. Am. J. Fish. Manage. 3: 322-330.

Burns, J.W. 1972. Some effects of logging and associated road construction on northern California streams. Trans. Am. Fish. Soc. 101: 1-17.

Burton, T.M. & K.E. Ulrich. 1994. The effects of whole-tree harvest on insects associated with leaf packs in small streams in New Hampshire. Verh. Internat. Verein. Limnol. 25: 1483-1491.

Cafferata, P. 1990a. Temperature regimes of small streams along the Mendocino coast. California Dept. Forestry, Jackson Demonstration State Forest, JDSF Newsletter No. 39: 1-4.

Cafferata, P.H. 1990b. Water temperature evaluation guide. California Dept. Forestry & Fire Protection.

Callaham, R.Z. (ed.) 1990. Case studies and catalog of watershed projects in western provinces and states. Univ. Calif., Berkeley, Wildland Resources Center Report 22.

Campbell, I.C. & T.J. Doeg. 1989. Impact of timber harvesting and production on streams: a review. Aust. J. Mar. Freshwat. Res. 40: 519-539.

Carlson, J.Y., C.W. Andrus & H.A. Froelich. 1990. Woody debris, channel features, and macroinvertebrates of streams with logged and undistrurbed riparian timber in northeastern Oregon, U.S.A. Can. J. Fish. Aquat. Sci. 47: 1103-1111.

Cederholm, C.J. & L.M. Reid. 1987. Impact of forest management on coho salmon (*Oncorhynchus kisutch*) populations of the Clearwater River, Washington: a project summary (pp. 373-398). *In:* E.O. Salo & T.W. Cundy (eds.), Streamside management: forestry and fishery interactions. Univ. Washington, Institute of Forest Resources, Contribution 57.

Chamberlin, T.W. (ed.). 1988. Proceedings of the workshop: applying 15 years of Carnation Creek results. Workshop held January 13-15, 1987, Nanaimo, British Columbia. Carnation Creek Steering Committee, Pacific Biological Station, Nanaimo, BC.

Chauvet, E. 1987. Changes in the chemical composition of alder, popular and willow leaves during decomposition in a river. Hydrobiol. 148: 35-44.

Chutter, F.M. 1969. The effects of silt and sand on the invertebrate fauna of streams and rivers. Hydrobiol. 34: 57-76.

Cordone, A.J. & D.W. Kelley. 1961. The influence of inorganic sediment on the aquatic life of streams. Calif. Fish & Game 47:189-228.

Corn, P.S. & R.B. Bury. 1989. Logging in western Oregon: responses of headwater habitats and stream amphibians. For. Ecol. Manage. 29: 39-57.

Cowan, C.A., M.W. Oswood, C.A. Buttimore & P.W. Flanagan. 1983. Processing and macroinvertebrate colonization of detritus in an Alaskan subarctic stream. Holarctic Ecol. 6: 340-348.

Culp, J.M. 1988. The effects of streambank clearcutting on the benthic invertebrates of Carnation Creek, British Columbia (pp. 87-92). *In:* T.W. Chamberlin, Proceedings of the workshop: applying 15 years of Carnation Creek results. Workshop held January 13-15, 1987, Nanaimo, British Columbia. Carnation Creek Steering Committee, Pacific Biological Station, Nanaimo, BC.

Culp, J.M. & R.W. Davies. 1983. An assessment of the effects of streambank clear-cutting on macroinvertebrate communities in a managed watershed. Can. Tech. Rep. Fish. Aquat. Sci. No. 1208. 96 pp.

Culp, J.M., S.J. Walde & R.W. Davies. 1983. Relative importance of substrate particle size and detritus to stream benthic macroinvertebrate microdistribution. Can. J. Fish. Aquat. Sci. 40: 1568-1574.

Culp, J.M., F.J. Wrona & R.W. Davies. 1986. Response of stream benthos and drift to fine sediment deposition versus transport. Can. J. Zool. 64: 1345-1351.

Cummins, K.W. 1973. Trophic relations of aquatic insects. Ann. Rev. Entomol. 18: 183-206.

Cummins, K.W. 1974. Structure and function of stream ecosystems. BioScience 24: 631-641.

Cummins, K.W. 1992. Catchment characteristics and river ecosystems (Chapter 8, pp. 125-135). *In:* P.J. Boon, P. Calow & G.E. Petts (eds.), River conservation and management, John Wiley & Sons.

Cummins, K.W., C.A. Tryon, Jr. & R.T. Hartman. 1966. Organism-substrate relationships in streams. Symposium held July 16-17, 1964. Univ. Pittsburgh, Pymatuning Laboratory of Ecology, Special Pulbication No. 4. 145 pp.

Cummins, K.W. & G.H. Lauff. 1969. The influence of substrate particle size on the microdistribution of stream macrobenthos. Hydrobiol. 34: 145-181.

Cummins, K.W. & M.J. Klug. 1979. Feeding ecology of stream invertebrates. Ann. Rev. Ecol. Syst. 10: 147-172.

Cummins, K.W., M.A. Wilzbach, D.M. Gates, J.B. Perry & W.B. Taliaferro. 1989. Shredders and riparian vegetation. BioScience 39: 24-30.

Cushing, C.FE., K.W. Cummins, G.W. Minshall & R.L. Vannote. 1983. Periphyton, chlorophyll a, and diatoms of the Middle Fork of the Salmon River, Idaho. Holarctic Ecology 6: 221-227.

Doeg, T.J. & G.A. Milledge. 1991. Effects of experimentally increasing concentrations of suspended sediment on macroinvertebrate drift. Aust. J. Mar. Freshwat. Res. 42: 519-526.

Duncan, S.W. & D.W. Blinn. 1989. Importance of physical variables on the seasonal dynamics of epilithic algae in a highly shaded canyon stream. J. Phycol. 25: 455-461.

Duncan, W.F.A. & M.A. Brusven. 1985. Benthic macroinvertebrates in logged and unlogged low-order southeast Alaskan streams. Freshwat. Invert. Biol. 4: 125-132.

Elliot, S.T. 1986. Reduction of Dolly Varden population and macrobenthos after removal of logging debris. Trans. Am. Fish. Soc. 115: 392-400.

Erman, D.C. & F. Ligon. ca 1985. The response of algal communities in streams of the Jackson Demonstration State Forest to timber harvest activities. California Department of Forestry, Final Report. 39 pp.

Erman, D.C. & N.A. Erman. 1984. The response of stream macroinvertebrates to substrate size and heterogeneity. Hydrobiol. 108: 75-82.

Erman, D,C., J.D. Newbold & K.B. Roby. 1977. Evaluation of streamside bufferstrips for protecting aquatic organisms. Univ. California, Davis, Water Resources Center, Contribution No. 165. 48 pp.

Erman, D.C. & D. Mahoney. 1983. Recovery after logging in streams with and without bufferstrips in northern California. Univ. California, Davis, Water Resources Center, Contribution No. 186.

Erman, N.A. 1984. The use of riparian systems by aquatic insects (pp. 177-182). *In:* R.E. Warner & K.M. Hendrix (eds.), California Riparian Systems: ecology, conservation, and productive management. Conference held September 17-19, 1981, Davis, CA. Univ. California Press, Berkeley. 1035 pp.

Franklin, J.F. 1992. Scientific basis for new perspectives in forests and streams (pp. 25-72). *In:* Naiman, R.J. (ed.), Watershed management: balancing sustainability and environmental change. Springer-Verlag, New York.

Friberg, N. & K. Kjeldsen. 1994. Development of benthic algal biomass in six Danish beech, mixed and coniferous forest streams. Verh. Internat. Verein. Limnol. 25: 1534-1538.

Gammon, J.R. 1970. The effects of inorganic sediment on stream biota. Environmental Protection Agency, Water Quality Office, Water Pollution Control Research Series 18050DWC12/70.

Gelroth, J.V. & G.R. Marzolf. 1978. Primary production and leaf-litter decomposition in natural and channelized portions of a Kansas stream. Am. Midl. Nat. 99: 238-243.

Gessner, M.O., E. Meyer & J. Schwoerbel. 1991. Rapid processing of fresh leaf litter in an upland stream. Verh. Internat. Verein. Limnol. 24: 1846-1850.

Golladay, S.W., J.R. Webster & E.F. Benfield. 1987. Changes in stream morphology and storm transport of seston following watershed disturbance. J. N. Am. Benthol. Soc. 6: 1-11.

Graynoth, E. 1979. Effects of logging on stream environments and faunas in Nelson. New Zealand J. Mar. Freshwat. Res. 13: 79-109.

Gregory, S.V., G.A. Lamberti, D.C. Erman, K.V. Koski, M.L. Murphy & J.R. Sedell. 1987. Influences of forest practices on aquatic production (pp. 233-255). *In:* E.O. Salo & T.W. Cundy (eds.), Streamside management: forestry and fishery interactions. Univ. Washington, Institute of Forest Resources, Contribution 57.

Gregory, S.V., F.J. Swanson, W.A. McKee & K.W. Cummins. 1991. An ecosystem perspective of riparian zones. BioScience 41: 540-551.

Growns, I.O. & J.A. Davis. 1991. Comparison of the macroinvertebrate communities in streams in logged and undisturbed catchments 8 years after harvesting. Aust. J. Mar. freshwat. Res. 42: 689-706.

Grubaugh, J.W. & J.B. Wallace. 1995. Functional structure and production of the benthic community in a Piedmont river: 1956-1957 and 1991-1992. Limnol. Oceanogr. 40: 490-501.

Gurtz, M.E. & J.B. Wallace. 1984. Substrate-mediated response of stream invertebrates to disturbance. Ecology 65: 1556-1569.

Gurtz, M.E., J.R. Webster & J.B. Wallace. 1980. Seston dynamics in southern Appalachian streams: effects of clear-cutting. Can. J. Fish. Aquat. Sci. 37: 624-631.

Haefner, J.D. & J.B. Wallace. 1981. Shifts in aquatic insect populations in a first-order southern Appalachian stream following a decade of old field succession. Can J. Fish. Aquat. Sci. 38: 353-359.

Hall, J.D., G.W. Brown & R.L. Lantz. 1987. The Alsea watershed study: a retrospective (pp. 399-416).). *In:* E.O. Salo & T.W. Cundy (eds.), Streamside management: forestry and fishery interactions. Univ. Washington, Institute of Forest Resources, Contribution 57.

Hansmann, E.W. & H.K. Phinney. 1973. Effects of logging on periphyton in coastal streams of Oregon. Ecology 54: 194-199.

Harmon, M.E., J.F. Franklin, F.J. Swanson, P. Sollins, S.V. Gregory, J.D. Lattin, N.H. Anderson, S.P. Cline, N.G. Aumen, J.R. Sedell, G.E. Lienkaemper, K. Cromack, Jr. & K.W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research 15: 133-302.

Harr, R.D., W.C. Harper & J.T. Krygier. 1975. Changes in storm hydrographs after road building and clear-cutting in the Oregon Coast Range. Water Resour. Res. 11: 436-444.

Hart, S.D. & R.P. Howmiller. 1975. Studies on the decomposition of allochthonous detritus in two southern California streams. Verh. Internat. Verein. Limnol. 19: 1665-1674.

Hartman, G.F. & J.C. Scrivener. 1990. Impacts of forestry practices on a coastal stream ecosystem, Carnation Creek, British Columbia. Can Bull. Fish. Aquat. Sci. 223. 148 pp..

Hawkins, C.P. & J.R. Sedell. 1981. Longitudinal and seasonal changes in functional organization of macroinvertebrate communities in four Oregon streams. Ecology 62: 387-397.

Hawkins, C.P., M.L. Murphy & N.H. Anderson. 1982. Effects of canopy, substrate composition, and gradient on the structure of macroinvertebrate communities in Cascade Range streams of Oregon. Ecology 63: 1840-1856.

Hawkins, C.P., M.L. Murphy, N.H. Anderson & M.A. Wilzbach. 1983. Density of fish and salamanders in relation to riparian canopy and physical habitat in streams of the northwestern United States. Can. J. Fish. Aquat. Sci. 40: 1173-1185.

Hem, J.D. 1985. Study and interpretation of the chemical characteristics of natural water. U.S. Geological Survey, Water-Supply Paper 2254.

Hill, W.R. & A.W. Knight. 1988. Nutrient and light limitation of algae in two northern California streams. J. Phycol. 24: 125-132.

Holopainen, A.L., P. Huttunen & M.Ahtiainen. 1991. Effects of forestry practices on water quality and primary productivity in small forest brooks. Verh. Internat. Verein. Limnol. 24: 1760-1766.

Holtby, L.B. 1988a. The effects of logging on stream temperatures at Carnation Creek (pp. 118-122). *In:* T.W. Chamberlin, Proceedings of the workshop: applying 15 years of Carnation Creek results. Workshop held January 13-15, 1987, Nanaimo, British Columbia. Carnation Creek Steering Committee, Pacific Biological Station, Nanaimo, BC.

Holtby, L.B. 1988b. Effects of logging on stream temperatures in Carnation Creek, British Columbia, and associated impacts on the coho salmon *(Oncoryhnchus kisutch).* Can J. Fish. Aquat. Sci. 45: 502-515.

Horner, R.R. & E.B. Welch. 1981. Stream periphyton development in relation to current velocity and nutrients. Can. J. Fish. Aquat. Sci. 38: 449-457.

Horner, R.R., E.B. Welch, M.R. Seeley & J.M. Jacoby. 1990. Responses of periphyton to changes in current velocity, suspended sediment and phosporus concentration. Freshwat. Biol. 24: 215-232.

Hynes, H.B.N. 1970. The ecology of running waters. Univ. Toronto Press, Toronto, Canada. 555pp.

Hynes, H.B.N. 1975. The stream and its valley. Verh. Internat. Verein. Limnol. 19: 1-15.

Karr, J.R. & I.J. Schlosser. 1978. Water resources and the land-water interface. Science 201: 229-234.

Kaushik, N.K. & H.B.N. Hynes. 1971. The fate of the dead leaves that fall into streams. Arch. Hydrobiol. 68: 465-515.

Keppler, E.T. & R.R. Ziemer. 1990. Logging effects on streamflow: water yield and summer low flows at Caspar Creek in northwestern California. Water Resour. Res. 26: 1669-1679.

Keppler, E.T., R.R. Ziemer & P.H. Cafferata. 1994. Changes in soil moisture and pore pressure after harvesting a forested hillslope in northern California (pp. 205-214). *In:* Am. Water Resour. Assoc., Effects of human-induced changes on hydrologic systems.

Knight, A.W. & R.L. Bottorff. 1984. The importance of riparian vegetation to stream ecosystems (pp. 160-167). *In:* R.E. Warner & K.M. Hendrix (eds.), California Riparian Systems: ecology, conservation, and productive management. Conference held September 17-19, 1981, Davis, CA. Univ. California Press, Berkeley. 1035 pp.

Krammes, J.S. & D.M. Burns. 1973. Road construction on Caspar Creek watersheds ..... 10-year report on impact. US Forest Service Research Paper PSW-93.

Lamberti, G.A., S.V. Gregory, L.R. Ashkenas, R.C. Wildman & K.M.S. Moore. 1991. Stream ecosystem recovery following a catastrophioc debris flow. 48: 196-208.

Lee, R.& D.E. Samuel. 1976. Some thermal and biological effects of forest cutting in West Virginia. J. Environ. Qual. 5: 362-366.

Lemly, A.D. 1982. Modification of benthic insect communities in polluted streams: combined effects of sedimentation and nutrient enrichment. Hydrobiol. 87: 229-245.

Levno, A. & J. Rothacher. 1967. Increases in maximum stream temperatures after logging in old-growth Douglas fir watersheds. US Forest Service, Pacific Northwest Forest and Range Experiment Station, Research Note PNW-110. 7 pp.

Likens, G.E. 1983. A priority for ecological research. Bull. Ecol. Soc. Am. 64: 234-243.

Likens, G.E., F.H. Bormann, N.M. Johnson, D.W. Fisher & R.S. Pierce. 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed-ecosystem. Ecol. Monogr. 40: 23-47.

Likens, G.E., F.H. Bormann, R.S. Pierce, J.S. Eaton & N.M. Johnson. 1977. Biogeochemistry of a forested ecosystem. Springer-Verlag, New York.146 pp.

Likens, G.E., F.H. Bormann, R.S. Pierce & W.A. Reiners. 1978. Recovery of a deforested ecosystem. Science 199: 492-496.

Lisle, T.E. 1989. Sediment transport and resulting deposition in spawning gravels, north coastal California. Water Resour. Res. 25: 1303-1319.

Lisle, T.E. 1995. Particle size variations between bed load and bed material in natural gravel bed channels. Water Resour. Res. 31: 1107-1118.

Lowe, R.L. 1974. Environmental requirements and pollution tolerance of freshwater diatoms. Environmental Protection Agency, Environmental Monitoring Series, EPA-670/4-74-005.

Lowe, R.L., S.W. Golladay & J.R. Webster. 1986. Periphyton response to nutrient manipulation in streams draining clearcut and forested watersheds. J. N. Am. Benthol. Soc. 5: 221-229.

Luedtke, R.J. & M.A. Brusven. 1876. Effects of sand sedimentation on colonization of stream insects. J. Fish. Res. Bd. Can. 33: 1881-1886.

Luedtke, R.J., M.A. Brusven & F.J. Watts. 1976. Benthic insect community changes in relation to in-stream alterations of a sediment-polluted stream. Melanderia 23:21-39.

Lyford, J.H., Jr. & S.V. Gregory. 1975. The dynamics and structure of periphyton communities in three Cascade Mountain streams. Verh. Internat. Verein. Limnol. 19: 1610-1616.

Maloney, D.C. & G.A. Lamberti. 1995. Rapid decomposition of summer-input leaves in a northern Michigan stream. Am. Midl. Nat. 133: 184-195.

Maser, C. & J.R. Sedell, J.R. 1994. From the forest to the sea: the ecology of wood in streams, rivers, estuaries, and the ocean. St. Lucie Press, Delray Beach, FL.

McIntire, C.D. & H.K. Phinney. 1965. Laboratory studies of periphyton production and community metabolism in lotic environments. Ecol. Monogr. 35: 237-258.

Meehan, W.R. (ed.) 1991. Influences of forest and range management on salmonid fishes and their habitats. American Fisheries Society, Special Publication 19.

Merritt, R.W. & K.W. Cummins. 1984 & 1996. An introduction to the aquatic insects of North America (2nd & 3nd ed.). Kendall/Hunt Publ. Co., Dubuque, Iowa. 862pp.

Meyer, J.L. & C. Johnson. 1983. The influence of elevated nitrate concentration on rate of leaf decomposition in a stream. Freshwat. Biol. 13: 177-183.

Meyer, J.L. & C.M. Tate. 1983. The effects of watershed disturbance on dissolved organic carbon dynamics of a stream. Ecology 64: 33-44.

Meyer, J.L., C.M. Tate, R.T. Edwards & M.T. Crocker. 1988. The trophic significance of dissolved organic carbon in streams (Chap. 20, pp. 269-278). *In:* W.T. Swank & D.A. Crossley, Jr. (eds.), Forest hydrology and ecology at Coweeta. Ecological Studies 66, Springer-Verlag, New York.

Meyer, J., T. Crocker, D. D'Angelo, W. Dodds, S. Findlay, M. Oswood, D. Repert & D. Toetz (eds.). 1993. Stream research in the Long-Term Ecological Research Network. Long-Term Ecological Research (LTER) Network Office, Seattle, WA.

Minshall, G.W. 1978. Autotrophy in stream ecosystems. BioScience 28: 767-771.

Murphy, M.L. & J.D. Hall. 1981. Varied effects of clear-cut logging on predators and their habitat in small streams of the Cascade Mountains, Oregon. Can. J. Fish. Aquat. Sci. 38: 137-145.

Murphy, M.L. C.P. Hawkins & N.H. Anderson. 1981. Effects of canopy modification and accumulated sediment on stream communities. Trans. Am. Fish. Soc. 110: 469-478.

Murphy, M.L., J. Heifetz, S.W. Johnson, K.V. Koski & J.F. Thedinga. 1986. Effect of clear-cut logging with and without buffer strips on juvenile salmonids in Alaskan streams. Can. J. Fish. Aquat. Sci. 43: 1521-1533.

Naiman, R.J. (ed.). 1992. Watershed management: balancing sustainability and environmental change. Springer-Verlag, New York.

Naiman, R.J. & J.R. Sedell. 1979a. Benthic organic matter as a function of stream order in Oregon. Arch. Hydrobiol. 87: 404-422.

Naiman, R.J. & J.R. Sedell. 1979b. Characterization of particulate organic matter transported by some Cascade mountain streams. J. Fish. Res. Bd. Can. 36: 17-31.

Naiman, R.J. & J.R. Sedell. 1980. Relationships between metabolic parameters and stream order in Oregon. Can. J. Fish. Aquat. Sci. 37: 834-847.

Naiman, R.J., T.J. Beechie, L.E. Benda, D.R. Berg, P.A. Bisson, L.H. MacDonald, M.D. O'Connor, P.L. Olson & E.A. Steel. 1992. Fundamental elements of ecologically healthy watersheds in the Pacific Northwest Coastal Ecoregion (Chapter 6, pp. 127-188). *In:* Naiman, R.J. (ed.), Watershed management: balancing sustainability and environmental change. Springer-Verlag, New York.

Newbold, J.D., D.C. Erman & K.B. Roby. 1980. Effects of logging on macroinvertebrates in streams with and without buffer strips. Can. J. Fish. Aquat. Sci. 37: 1076-1085.

Newcombe, C.P. & D.D. MacDonald. 1991. Effects of suspended sediments on aquatic ecosystems. N. Am. J. Fish. Manage. 11: 72-85.

Niemi, G.J., P. DeVore, N. Detenbeck, D. Taylor, A. Lima, J. Pastor, J.D. Yount & R.J. Naiman. 1990. Overview of case studies on recovery of aquatic systems from disturbance. Environ. Manage. 14: 571-587.

Noel, D.S., C.W. Martin & C.A. Federer. 1986. Effects of forest clearcutting in New England on stream macroinvertebrates and periphyton. Environ. Manage. 10: 661-670.

O'Connor, M.D. & R.R. Ziemer. 1989. Coarse woody debris ecology in a second-growth *Sequoia sempervirens* forest stream (pp. 165-171). *In:* D.L. Abell (ed.), Proceedings of the California Riparian Systems Conference: protection, management, and restoration for the 1990's; September 22-24, 1988, Davis, CA. Gen. Tech. Rep. PSW-110. Berkeley, CA: Pacific Southwest Forest and Range Experiment Station, Forest Service, US Dept. Agric. 544 pp.

Patten, D.R. 1973. A literature review of timber-harvesting effects of stream temperature: research needs for the southwest. U.S. Forest Service, Research Note RM - 249. 4pp.

Pearce, R.B. 1987. Caspar Creek: discovering how watersheds respond to logging. Forestry Research West.

Pennak, R. W. 1989. Fresh-water invertebrates of the United States: Protozoa to Mollusca (3rd ed.). John Wiley & Sons, New York. 628 pp.

Petersen, R.C. & K.W. Cummins. 1974. Leaf processing in a woodland stream. Freshwat. Biol. 4: 343-368.

Platts, W.S., R.J. Torquemada, M.L. McHenry & C.K. Graham. 1989. Changes in salmon spawning and rearing habitat from increased delivery of fine sediment to the South Fork Salmon River, Idaho. Trans. Am. Fish. Soc. 118: 274-283.

Pozo, J. 1993. Leaf litter processing of alder and eucalyptus in the Aguera stream system (North Spain). I. Chemical changes. Arch. Hydrobiol. 127: 299-317.

Rabeni, C.F. & G.W. Minshall. 1977. Factors affecting microdistribution of stream benthic insects. Oikos 29: 33-43.

Rice, R.M. & J. Lewis. 1991. Estimating erosion risks associated with logging and forest roads in northwestern California. Water Resour. Res. 27: 809-818.

Rice, R.M. 1992. The science and politics of BMPs in forestry: California experiences (pp. 385-). *In:* Naiman, R.J. (ed.), Watershed management: balancing sustainability and environmental change. Springer-Verlag, New York.

Rice, R.M., F.B. Tilley & P.A. Datzman. 1979. A watershed's response to logging and roads: South Fork of Caspar Creek, California, 1967-1976. US Forest Service Research Paper PSW-146. 12 pp.

Rishel, G.B. J.A. Lynch & E.S. Corbett. 1982. Seasonal stream temperature changes following forest harvest. J. Environ. Qual. 11: 112-116.

Rosenberg, D.M. & V.H. Resh. 1982. The use of artificial substrates in the study of freshwater benthic macroinvertebrates (pp. 175-235). *In:* J. Cairns, Jr. (ed.), Artificial substrates. Ann Arbor Sci. Publ., Michigan.

Rosenberg, D.M. & A.P. Wiens. 1975. Experimental sediment addition studies on the Harris River, N.W.T., Canada: the effect on macroinvertebrate drift. Verh. Internat. Verein. Limnol. 19: 1568-1574.

Rosenberg, D.M. & A.P. Wiens. 1978. Effects of sediment addition on macrobenthic invertebrates in a northern Canadian river. Water Research 12: 753-763.

Rutherford, J.E. & R.J. Mackay. 1986. Patterns of pupal mortality in field populations of *Hydropsyche* and *Cheumatopsyche* (Trichoptera, Hydropsychidae). Freshwat. Biol. 16: 337-350.

Ryan, P.A. 1991. Environmental effects of sediment on New Zealand streams: a review. New Zealand J. Mar. Freshwat. Res. 25: 207-221.

Salo, E.O. & T.W. Cundy (eds.). 1987. Streamside management: forestry and fishery interactions. Univ. Washington, Institute of Forest Resources, Contribution 57.

Sedell, J.R., F.J. Triska & N.S. Triska. 1975. The processing of conifer and hardwood leaves in two coniferous forest streams: I. Weight loss and associated invertebrates. Verh. Internat. Verein. Limnol. 19: 1617-1627.

Short, R.A. & J.V. Ward. 1980. Leaf litter processing in a regulated Rocky Mountain stream. Can. J. Fish. Aquat. Sci. 37: 123-127.

Short, R.A., S.P. Canton & J.V. Ward. 1980. Detrital processing and associated macroinvertebrates in a Colorado mountain stream. Ecology 61: 727-732.

Shortreed, K.S. & J.G. Stockner. 1983. Periphyton biomass and species composition in a coastal rainforest stream in British Columbia: effects of environmental changes caused by logging. Can. J. Fish. Aquat. Sci. 40: 1887-1895.

Silsbee, D.G. & G.L. Larson. 1983. A comparison of streams in logged and unlogged areas of Great Smoky Mountains National Park. Hydrobiol. 102: 99-111.

Steinblums, I.J., H.A. Froehlich & J.K. Lyons. 1984. Designing stable buffer strips for stream protection. J. Forestry 82: 49-52.

Steinman, A.D. & C.D. McIntire. 1990. Recovery of lotic periphyton communities after disturbance. Environ. Manage. 14: 589-604.

Stockner, J.G. & K.S. Shortreed. 1976. Autotrophic production in Carnation Creek, a coastal rainforest stream on Vancouver Island, British Columbia. J. Fish. Res. Bd. Can. 33: 1553-1563.

Stockner, J.G. & K.S. Shortreed. 1978. Enhancement of autotrophic production by nutrient addition in a coastal rainforest stream on Vancouver Island. J. Fish. Res. Bd. Can. 35: 28-34.

Stout, R.J., W.H. Taft & R.W. Merritt. 1985. Patterns of macroinvertebrate colonization on fresh and senescent alder leaves in two Michigan streams. Freshwat. Biol. 15: 573-580.

Strickland, J.D.H. & T.R. Parsons. 1968. A practical handbook of seawater analysis. Bull. Fish. Res. Bd. Can. No. 167. 311 pp.

Swank, W.T. 1988. Stream chemistry responses to disturbance (Chap. 25, pp. 339-357). *In:* W.T. Swank & D.A. Crossley, Jr. (eds.), Forest hydrology and ecology at Coweeta. Ecological Studies 66, Springer-Verlag, New York.

Swank, W.T., L.W. Swift, Jr. & J.E. Douglass. 1988. Streamflow changes associated with forest cutting, species conversions, and natural disturbances (Chap. 22, pp. 297-312). *In:* W.T. Swank & D.A. Crossley, Jr. (eds.), Forest hydrology and ecology at Coweeta. Ecological Studies 66, Springer-Verlag, New York.

Swank, W.T. & D.A. Crossley, Jr. (eds.). 1988. Forest hydrology and ecology at Coweeta. Ecological Studies 66, Springer-Verlag, New York.

Swift, L.W. & J.B. Messer. 1971. Forest cuttings raise temperatures of small streams in the southern Appalachians. J. Soil Water Cons. 26: 111-116.

Tebo, L.B., Jr. 1955. Effects of siltation, resulting from improper logging, on the bottom fauna of a small trout stream in the southern Appalachians. Progressive Fish-Culturist 17: 64-70.

Thomas, R.B. 1990. Problems in determining the return of a watershed to pretreatment conditions: techniques applied to a study at Caspar Creek, California. Water Resour. Res. 26: 2079-2087.

Tilley, F.B. & R.M. Rice. 1977. Caspar Creek watershed study: a current status report. California Dept. Forestry, State Forest Notes No. 66. 15 pp.

Trayler, K. 1996. Comparison of the vertical distibution and abundance of invertebrates in streams of logged and unlogged catchments (abstract). Bull. N. Am. Benthological Soc. 13: 211.

Triska, F.J., J.R. Sedell & B. Buckley. 1975. The processing of conifer and hardwood leaves in two coniferous forest streams: II. Biochemical and nutrient changes. Verh. Internat. Verein. Limnol. 19: 1628-1639.

Triska, F.J. & J.R. Sedell. 1976. Decomposition of four species of leaf litter in response to nitrate manipulation. Ecology 57: 783-792.

Triska, F.J., V.C. Kennedy, R.J. Avanzino & B.N. Reilly. 1983. Effect of simulated canopy cover on regulation of nitrate uptake and primary production by natural periphyton assemblages (Chapter 7, pp. 129-159). *In:* T.D. Fontaine, III, & S.M. Bartell (eds), Dynamics of lotic ecosystems, Ann Arbor Science, Ann Arbor, MI.

Ulrich, K.E., T.M. Burton & M.P. Oemke. 1993. Effects of whole-tree harvest on epilithic algal communities in headwater streams. J. Freshwat. Ecol. 8: 83-92.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell & C.E. Cushing. 1980. The River Continuum Concept. Can J. Fish. Aquat. Sci. 37: 130-137.

Vitousek, P.M. & J.M. Melillo. 1979. Nitrate losses from disturbed forests: patterns and mechanisms. Forest Sci. 25: 605-619.

Wagener, S.M. & J.D. LaPerriere. 1985. Effects of placer mining on the invertebrate communities of interior Alaska streams. Freshwat. Invert. Biol. 4: 208-214.

Wallace, J.B. & R.W. Merritt. 1980. Filter-feeding ecology of aquatic insects. Ann. Rev. Entomol. 25: 103-132.

Wallace, J.B. 1988. Aquatic invertebrate research (Chapter 19, pp. 257-268). *In:* Swank, W.T. & D.A. Crossley, Jr. (eds.), Forest hydrology and ecology at Coweeta. Ecological Studies 66, Springer-Verlag, New York.

Wallace, J.B. 1990. Recovery of lotic macroinvertebrate communities from disturbance. Environ. Manage. 14: 605-620.

Wallace, J.B., W.R. Woodall & F.F. Sherberger. 1970. Breakdown of leaves by feeding of *Peltoperla maria* nymphs (Plecoptera: Peltoperlidae). Ann. Entomol. Soc. 63: 562-567.

Wallace, J.B. & M.E. Gurtz. 1986. Response of *Baetis* mayflies (Ephemeroptera) to catchment logging. Am. Midl. Nat. 115: 25-41.

Waters, T.F. 1995. Sediment in streams: sources, biological effects, and control. Am. Fish. Soc. Monogr. 7.

Webster, J.R. & E.F. Benfield. 1986. Vascular plant breakdown in freshwater ecosystems. Ann. Rev. Ecol. Syst. 17: 567-594.

Webster, J.R. & J.B. Waide. 1982. Effects of forest clearcutting on leaf breakdown in a southern Appalachian stream. Freshwat. Biol. 12: 331-344.

Webster, J.R., M.E. Gurtz, J.J. Hains, J.L. Meyer, W.T. Swank, J.B. Waide & J.B. Wallace. 1983. Stability of stream ecosystems (pp. 355-395). *In:* J.R. Barnes & G.W. Minshall (eds.), Stream ecology: application and testing of general ecological theory, Plenum Press, New York. 399 pp.

Webster, J.R., E.F. Benfield, S.W. Golladay, R.F. Kazmierczak, Jr., W.B. Perry & G.T. Peters. 1988. Effects of watershed disturbance on stream seston characteristics (Chap. 21., pp. 279-294). *In:* W.T. Swank & D.A. Crossley, Jr. (eds.), Forest hydrology and ecology at Coweeta. Ecological Studies 66, Springer-Verlag, New York.

Williams, D.D. 1984. The hyporheic zone as a habitat for aquatic insects and associated arthropods (Chapter 14, pp. 430-455). *In:* V.H. Resh & D.M. Rosenberg (eds.), The ecology of aquatic insects, Praeger Publ., NY. 625 pp.

Winterbourn, M.J. 1986. Forestry practices and stream communities with particular reference to New Zealand (pp. 57-73). *In:* I.C. Campbell (ed.), Stream protection: the management of rivers for instream uses. Water Studies Centre, Chisholm Institute of Technology, East Caulfield, Australia.

Woodall, W.R., Jr. & J.B. Wallace. 1972. The benthic fauna in four small southern Appalachian streams. Am. Midl. Nat. 88: 393-407.

Wotton, R.S. 1994. Particulate and dissolved organic matter as food (pp. 235-288). *In:* R.S. Wotton (ed.), The biology of particles in aquatic systems (2nd. ed.), Lewis Publ., Boca Raton, FL. 325 pp.

Wright, K.A., K.H. Sendek, R.M. Rice & R.B. Thomas. 1990. Logging effects on streamflow: storm runoff at Caspar Creek in northwestern California. Water Resour. Res. 26: 1657-1667.

Yount, J.D. & G.J. Niemi. 1990. Recovery of lotic communities and ecosystems from disturbance -- A narrative review of case studies. Environ. Manage. 14: 547-569.

Ziemer, R.R. 1981. Storm flow response to road building and partial cutting in small steams of northern California. Water Resour. Res. 17: 907-917.

Table 6. Number of macroinvertebrates collected from rock packs (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

Number of Macroinvertebrates Counted from Rock Packs (1987-1994)

	SP 87	F 87	SP 88	F 88	SP 89	F 89	SP 90	F 90	SP 91	F 91	SP 92	F 92	SP 93	F 93	SP 94	TOTAL
EPHEMEROPTERA																
Ameletus	4	19	15	149	33	28						8	14	6	63	339
Baetis	3394	386	6828	978	2992	564		2598	4210	637	13465	769	3969	2550	5833	49173
Centroptilum	4				3							15	10	7	80	119
Cinygma	274	53	298	280	346	235		25	1191	135	284	3	18	33	423	3598
Cinygmula	213	616	905	1309	263	960		549	238	580	614	427	159	790	515	8138
Drunella	4		8		5				1							18
Epeorus		2	7	2	20	217		7	13		1			1		270
Ephemerella									1		9			52		62
Ironodes	21	158	16	163	35	1596		2075	147	601	377	220	56	1183	493	7141
Nixe		9	1		5	249	Washed	353	271	41	367	167	208	174	678	2523
Paraleptophlebia	100	78	1230	505	940	550	Out	167	1233	188	392	334	1045	446	3538	10746
Rhithrogena		1		3								10	3	27		44
Serratella	155	194	445	90	107	40		30	93	98	673	333	109	240	496	3103
Timpanoga	2	2	5										5		10	24
PLECOPTERA																
Calineuria	123	133	710	301	201	395		263	307	207	282	176	242	218	352	3910
Capniidae		1				8						7		56		72
Hesperoperla	390	50	839	157	403	163		159	560	259	694	100	102	62	758	4696
Kathroperla			4	1	1	1										7
Leuctridae	4	54	109	286	121	324		51	84	37	30	18	264	77	352	1811
Malenka	270	56	1734	670	455	325		1189	507	128	752	92	744	211	1618	8751
Perlodidae															3	3
Pteronarcys	1		1		1						2					5
Soliperla	1		5		3			1	3		3				3	19
Soyedina		17	3	135	11	91		122	1	268	4	15	87	26	279	1059
Sweltsa	6	51	63	82	121	455		86	56	14	21	21	107	17	94	1194
Zapada		4			19	30		882	70	40	228	27	424	180	738	2642
	SP 87	F 87	SP 88	F 88	SP 89	F 89	SP 90	F 90	SP 91	F 91	SP 92	F 92	SP 93	F 93	SP 94	TOTAL
ODONATA																
Cordulegaster	1		1		4	8							2	2	3	21
Octogomphus		4		4	1	6		6	4	2	11	1	2	2	4	47
TRICHOPTERA																
Apatania	1	25	9	32	25	65		2		2	48	7	24	4	66	310
Cryptochia				1	1					2			1	2		7
Ecclisomyia		2		4	1	2						1	1	1	8	20
Farula	1	20	8	222	307	243		304	415	497	470	97	123	14	70	2791
Glossosoma	437	931	555	317	245	354		435	436	2290	2398	1457	494	847	1302	12498
Goeracea			4	1		5	Washed			1						11
Heteroplectron	9	292	27	752	71	458	Out	71	21	530	133	519	66	660	937	4546
Hydatophylax				4	6	2		1				19	4	70	45	151
Hydropsyche	26	41	4	21	1	294		1069	21	541	2574	44	1066	319	3281	9302
Hydroptila	137	4	124	3	107	1			1	1	4			4	11	397
Lepidostoma	172	108	1449	534	739	627		72	172	422	594	164	178	319	1323	6873
Micrasema		19	25	65	1	79		163	13	227	575	111	46	31	338	1693
Namamyia						3					4		3		4	14
Neophylax	32	1	51		84				10	1	23		1		37	240
Nerophilus				1		1		1				1		4	2	10
Palaeagapetus		4			1	1				2	1				1	10
Parapsyche	17	6	32	2	74	131		10	21	6	33		13		26	371
Parthina		24	5	117	77	229		26	3	14	15	3	51	19	185	768
Polycentropus		2	1									4		8	5	20
Psychoglypha	1	4	7	2	23			3	32	12	16		7	1	18	126
Rhyacophila	25	55	163	121	130	226		225	86	81	183	46	108	14	483	1946
Wormaldia	7		27		6	36						7	13	6	35	137
Yphria						2										2
Trichoptera (pupae)						1						4	2	5	37	49

Table 13. Summary of analysis of variance (ANOVA) results for 8 macroinvertebrate and algal variables, N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. This table shows significantly higher ($P < 0.05$) or not significant (ns) results for Treatment (pre- vs. post-), Site Position (Above vs. Below), and Season (Spring vs. Fall)*.

	Data Set	MI #	MI Taxa	EPT #	EPT Taxa	Chiron #	*Baetis* #	Chl-a	Al Biom
R^2	All	0.34	0.42	0.35	0.41	0.25	0.53	0.40	0.17
	B	0.49	0.50	0.47	0.43	0.42	0.61	0.45	0.15
	C	0.39	0.52	0.35	0.47	0.39	0.55	0.51	0.50
	G	0.31	0.42	0.35	0.46	0.20	0.42	0.62	0.37
	H	0.45	0.52	0.41	0.49	0.41	0.51	0.57	0.48
	I	0.45	0.52	0.40	0.51	0.41	0.57	0.57	0.31
Treatment	All	post	post	post	post	post	pre	post	post
	B	post	post	post	post	post	post	post	post
	C	post	post	ns	post	post	ns	post	post
	G	ns	post	ns	post	ns	pre	post	post
	H	ns	post	ns	post	ns	pre	post	post
	I	post	post	post	post	ns	ns	post	post
Site Position	All	Ab	Ab	ns	Ab	Ab	ns	ns	ns
	B	ns	ns	ns	ns	ns	ns	ns	ns
	C	Ab	Ab	Ab	Ab	Ab	ns	ns	ns
	G	Be	ns	ns	ns	Be	ns	ns	ns
	H	ns	ns	ns	ns	Ab	ns	Be	ns
	I	Ab	Ab	Ab	Ab	Ab	ns	Be	Be
Season	All	S	ns	S	S	S	S	F	S
	B	S	S	S	S	S	S	ns	ns
	C	S	ns	S	S	S	S	F	ns
	G	ns	F	ns	ns	ns	S	F	S
	H	S	ns	S	ns	S	S	F	ns
	I	S	S	S	S	S	S	F	ns

*(MI = macroinvertebrate; EPT = Ephemeroptera, Plecoptera, Trichoptera; Chiron = Chironomidae; Chl-a = Chlorophyll-a; Al Biom = Algal Biomass; # = density; post = post-treatment; pre = pre-treatment; Ab = Above; Be = Below; S = Spring; F = Fall).

Table 14. Summary of station pairwise comparison statistics for 8 macroinvertebrate and algal variables, N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. This table shows significantly different (* P < 0.05) or not significantly different (ns) comparisons.

Station

MI Density

	B	C	G	H
C	*			
G	*	*		
H	*	*	ns	
I	*	*	*	*

MI Taxa

	B	C	G	H
C	ns			
G	*	*		
H	*	*	ns	
I	*	*	ns	ns

EPT Density

	B	C	G	H
C	ns			
G	*	*		
H	*	*	ns	
I	*	*	*	*

EPT Taxa

	B	C	G	H
C	*			
G	*	*		
H	*	*	ns	
I	*	*	ns	ns

Chironomidae Density

	B	C	G	H
C	*			
G	*	ns		
H	*	ns	ns	
I	*	ns	*	*

***Baetis* Density**

	B	C	G	H
C	ns			
G	*	*		
H	*	*	*	
I	*	*	*	*

Chlorophyll-a

	B	C	G	H
C	ns			
G	*	*		
H	*	*	ns	
I	*	*	ns	ns

Algal Biomass

	B	C	G	H
C	ns			
G	*	ns		
H	*	ns	ns	
I	ns	ns	*	*

Table 15. Results of statistical comparisons of macroinvertebrate densities between Above and Below sites at each station (t-test of log transformed densities). Probabilities are given for significant differences. (ns = not significant, a = significantly greater at Above site, b = significantly greater at Below site).

Macroinvertebrate Number / Rock Pack

	SP 87	F 87	SP 88	F 88	SP 89	F 89	SP 90	F 90	SP 91	F 91	SP 92	F 92	SP 93	F 93	SP 94
Bb vs Ba	0.01 b	ns	ns	ns	ns	0.03 b		ns	ns	ns	ns	ns	ns	ns	ns
Cb vs Ca	ns	0.01 a	ns	0.04 a	ns	ns		<0.01 a	<0.01 a	ns	ns	ns	ns	<0.01 a	<0.01 a
Gb vs Ga	ns	ns	ns	ns	ns	ns		<0.01 b	ns	0.02 b	ns	ns	ns	ns	ns
Hb vs Ha	ns	ns	ns	ns	ns	ns		<0.01 a	ns	ns	ns	ns	ns	0.02 b	ns
Ib vs Ia	0.01 b		ns	<0.01 a	ns	ns		<0.01 a	ns	ns	ns	ns	ns	ns	ns

Table 16. Summary of analysis of variance (ANOVA) results for 10 macroinvertebrate variables, N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. This table shows significantly higher ($P < 0.05$) or not significant (ns) results for Treatment (pre- vs. post-), Site Position (Above vs. Below), and Season (Spring vs. Fall)*.

	Data Set	Sc #	Sc %	Sh #	Sh %	Cf #	Cf %	Cg #	Cg %	P #	P %
R^2	All	0.32	0.39	0.33	0.19	0.23	0.20	0.36	0.33	0.38	0.18
	B	0.38	0.46	0.45	0.38	0.32	0.25	0.53	0.51	0.42	0.29
	C	0.26	0.47	0.42	0.30	0.25	0.22	0.46	0.42	0.54	0.29
	G	0.38	0.45	0.46	0.28	0.26	0.27	0.26	0.40	0.41	0.21
	H	0.37	0.38	0.34	0.17	0.35	0.34	0.48	0.41	0.43	0.16
	I	0.30	0.44	0.36	0.24	0.40	0.35	0.47	0.44	0.40	0.32
Treatment	All	post	post	post	ns	post	post	ns	pre	ns	pre
	B	post	post	post	pre	post	ns	post	ns	post	pre
	C	post	post	ns	pre	post	post	post	ns	ns	pre
	G	post	post	ns	ns	post	post	ns	pre	ns	pre
	H	post	post	post	post	post	post	ns	pre	ns	pre
	I	post	post	ns	ns	post	post	ns	pre	ns	pre
Site Position	All	Ab	ns	Ab	Ab	ns	ns	ns	ns	Ab	ns
	B	Ab	Ab	ns	Ab	Be	Be	ns	Be	ns	ns
	C	Ab	ns	Ab	ns	ns	ns	Ab	Ab	Ab	ns
	G	ns	ns	ns	Ab	ns	ns	Be	Be	ns	ns
	H	ns	ns	ns	ns	Ab	Ab	ns	ns	ns	ns
	I	ns	ns	Ab	ns	ns	ns	Ab	ns	Ab	ns
Season	All	F	F	ns	F	S	S	S	S	S	ns
	B	ns	F	ns	F	S	S	S	S	S	ns
	C	ns	F	ns	F	S	S	S	S	S	ns
	G	F	F	F	F	S	S	S	S	ns	ns
	H	F	F	ns	F	S	S	S	S	S	ns
	I	F	F	S	ns	S	S	S	S	S	ns

*(Sc = Scrapper; Sh = Shredder; Cf = Collector - filterer; Cg = Collector - gatherer; P = Predator;
= density; % = relative abundance; post = post-treatment; pre = pre-treatment; Ab = Above; Be = Below;

Table 17. Summary of station pairwise comparison statistics for 10 macroinvertebrate variables, N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. This table shows significantly different (* P < 0.05) or not significantly different (ns) comparisons. (# = density; % = relative abundance).

	Station								
	B	C	G	H		B	C	G	H
Scraper #					**Scraper %**				
C	ns				C	*			
G	*	*			G	ns	*		
H	*	*	ns		H	*	*	ns	
I	*	*	*	*	I	*	*	ns	ns
Shredder #					**Shredder %**				
C	ns				C	ns			
G	*	*			G	*	*		
H	*	ns	*		H	*	*	*	
I	ns	ns	*	ns	I	*	*	ns	*
Collector - filterer #					**Collector - filterer %**				
C	*				C	*			
G	*	*			G	ns	ns		
H	*	*	ns		H	ns	*	ns	
I	*	*	*	*	I	ns	ns	ns	ns
Collector - gatherer #					**Collector - gatherer %**				
C	*				C	*			
G	*	*			G	*	ns		
H	*	*	ns		H	*	ns	*	
I	*	ns	*	*	I	ns	*	ns	*
Predator #					**Predator %**				
C	ns				C	ns			
G	*	*			G	*	ns		
H	*	*	*		H	*	*	*	
I	*	*	*	ns	I	ns	ns	ns	*

Table 19. Alder leaf decay rates (k) for the autumn of six years (1988-1993) from six sites in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (k-values determined from equation, $W_t = W_0e^{-kt}$)

Leaf Decay Rates

Year	Bb	Ba	Gb	Ga	Ib	Ia
1988	0.0275	0.0750	0.0369	0.0220	0.0293	0.0291
1989	0.0225	0.0310	0.0294	0.0205	0.0604	0.0282
1990	0.0133	0.0184	0.0146	0.0156	0.0218	0.0157
1991	0.0567	0.0456	0.0677	0.0912	0.0621	0.0453
1992	0.0459	0.0605	0.0560	0.0636	0.0549	0.0413
1993	0.0278	0.0202	0.0226	0.0266	0.0254	0.0326

Table 23. Variation in numbers of diatoms counted (1989-1994) from N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

Number of Diatoms Counted (1989-1994)

	SP 89	F 89	SP 90	F 90	SP 91	F 91	SP 92	F 92	SP 93	F 93	SP 94		TOTAL
Achnanthes deflexa								1					1
Achnanthes exigua	94	209		58	23	94	18	108	2	21	7		634
Achnanthes lanceolata	7120	4383		5583	5704	3604	4101	3139	4280	4003	2634		44551
Achnanthes laterostrata	6	1		13	11	58	4	60	1	18	8		180
Achnanthes microcephala	1	3			18	46	42	7	10		18		145
Achnanthes minutissima	9	26	Washed	3	5	18		453	9	40	17		580
Achnanthes saxonica	901	1603	Out	474	144	593	284	578	53	351	270		5251
Amphora perpusilla	6	115		30	9	668	94	1054	19	182	49		2226
Cocconeis pedicularus		1											1
Cocconeis placentula	1561	2828		3348	3477	4292	4476	3946	4022	4248	4267		36465
Cymbella minuta		1									1		2
Cymbella tumida								3			30		33
Diploneis elliptica				2		4	2	7	1	7	7		30
Epithemia adnata	6	2		10	4	10	14	34	2	3	21		106
Eunotia maior								14	1	47	19		81
Eunotia sp. VIII-12	13	1		34	24	40	20	3					135
Gomphonema acuminatum						4		3		1			8
Gomphonema clevei								20		18	59		97
Gomphonema constrictum								17	1				18
Gomphonema cf. *subclavatum*							1	2		51	6		60
Gomphonema sp. *	9	1		8	14	42	59	11			2		146
Gyrosigma sp. XII-11								1					1
Melosira sp. XIV-1					2	2	123	17	12	279	231		666
Navicula cf. *cryptocephala*				6	16	23	65	32	4	47	41		234
Navicula elginensis				1				1		1			3
Navicula sublitissima	2	1		5	8	9	25	23	6	19	50		148
Navicula sp. XIV-11								8	8	18	26		60
Navicula sp. XIX-33							3				8		11
Navicula sp. XX-1							3						3
Navicula sp. XXIII-26										1			1
Nedium sp. XVIII-1						1	1	1					3
	SP 89	F 89	SP 90	F 90	SP 91	F 91	SP 92	F 92	SP 93	F 93	SP 94		TOTAL
Nitzschia frustulum		5					2						7
Nitzschia inconspicua	232	45		1	39	9	7	23	134	18	146		654
Nitzschia sigmoidea											1		1
Nitzschia sp. I-21	5		Washed	6	27	8	48	35	1	37	62		229
Nitzschia sp. VI-4	2		Out	1	1	4	19	19		83	37		166
Nitzschia sp. XII-3				6	2		12						20
Pinnularia sp. I-33								1					1
Rhoicosphenia curvata	32	175		11	56	269	278	155	17	50	460		1503
Rhopalodia gibberula							4	17					21
Surirella linearis					4	1	64						69
Surirella sp. XVI-33									1		2		3
Surirella sp. XVII-24								1		1	4		6
Surirella sp. XIX-35							9		2	1	3		15
Surirella sp. XXII-19									1				1
Synedra ulna	1				12	1	22	6	13	55	113		223
Tabellaria sp. XXV-12											1		1
TOTAL NUMBERS	10000	9400		9600	9600	9800	9800	9800	8600	9600	8600		94800
TOTAL SPECIES	17	17		19	21	23	28	33	23	26	30		46

*Unknown *Gomphonema* present in girdle view; probably included in above 4 species.

Table 24. List of the most common diatoms in the quantitative algal samples collected 1989-1994 from Stations B, C, G, H, & I in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

	Total # Cells Counted	% of Total Cells Counted
Achnanthes lanceolata	44,551	47.0
Cocconeis placentula	36,465	38.5
Achnanthes saxonica	5,251	5.5
Amphora perpusilla	2,226	2.3
Rhoicosphenia curvata	1,503	1.6
Melosira sp.	666	0.7
Nitzschia inconspicua	654	0.7
Achnanthes exigua	634	0.7
Achnanthes minutissima	566	0.6
	92,516	97.6

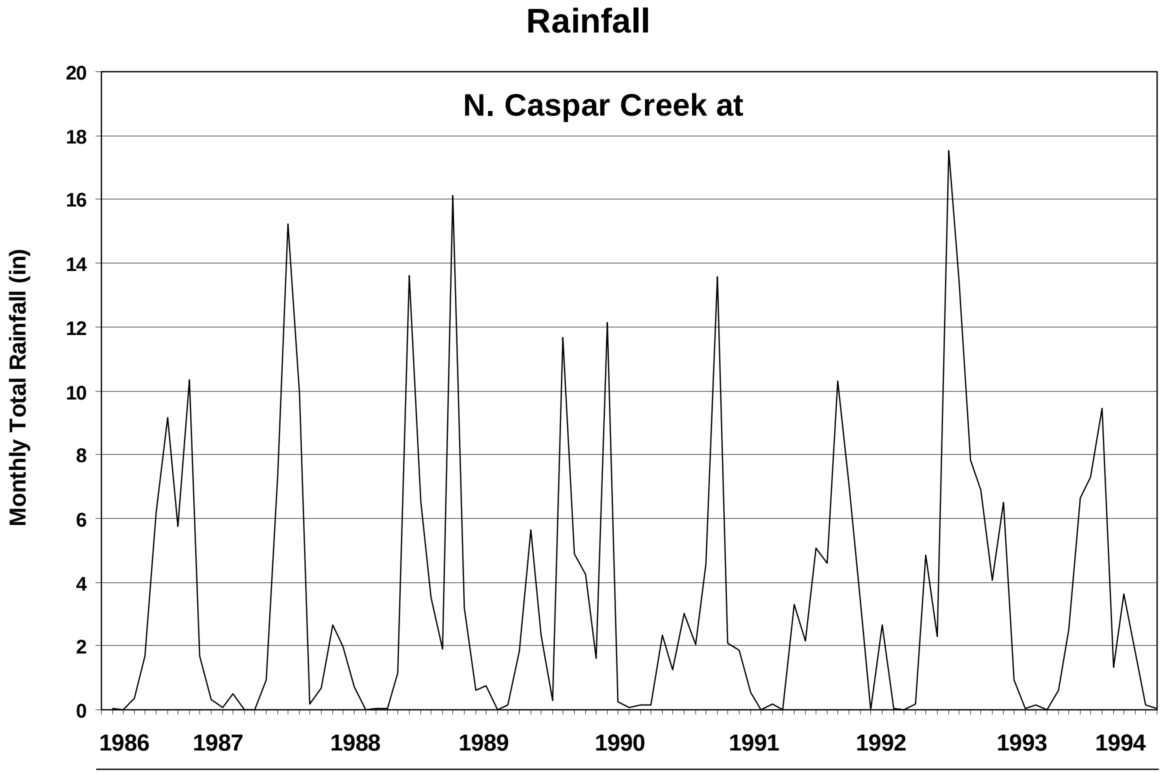


Fig. 3. Variation in monthly total rainfall (1986-1994) at Station NFC in N. Caspar Creek basin, Jackson Demonstration State Forest, Fort Bragg, CA. Mean annual rainfall (1985-1994) = 41.7 in (1060 mm). Data provided by the U.S. Forest Service, Redwood Sciences Laboratory, Arcata, CA.

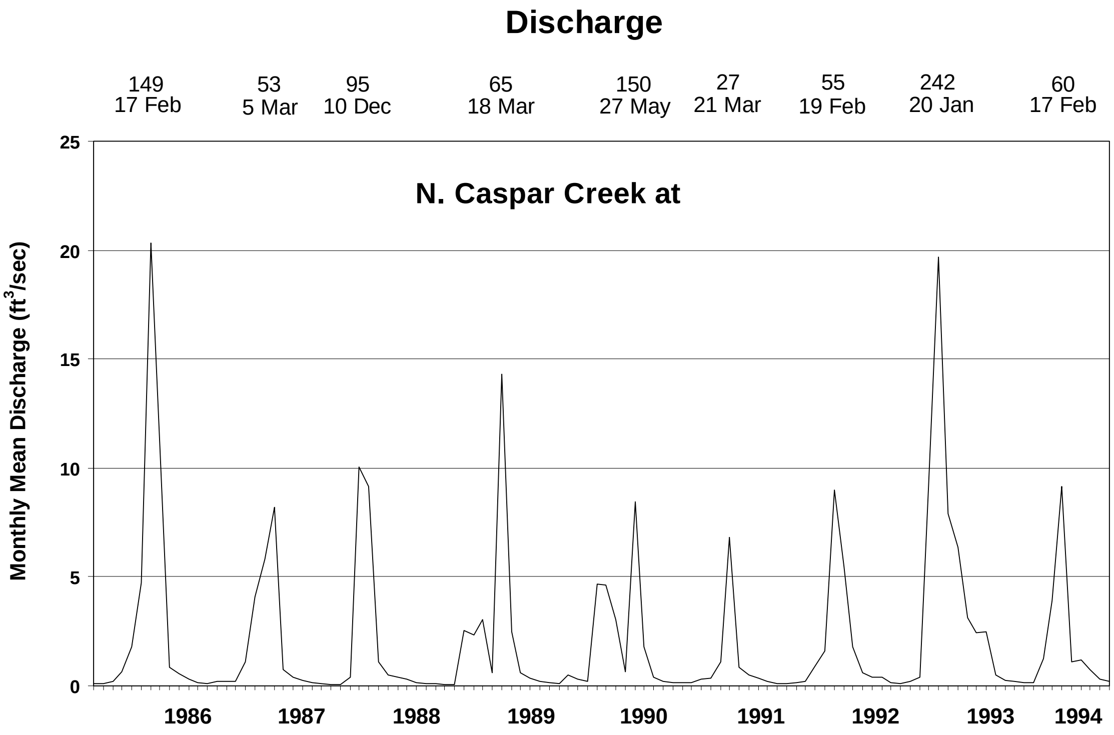


Fig. 4. Variation in monthly mean discharge (1985-1994) at Station NFC in N. Caspar Creek basin, Jackson Demonstration State Forest, Fort Bragg, CA. Values along top of graph are peak discharges (cfs) and dates of peak discharge for each water year. Data provided by the U.S. Forest Service, Redwood Sciences Laboratory, Arcata, CA.

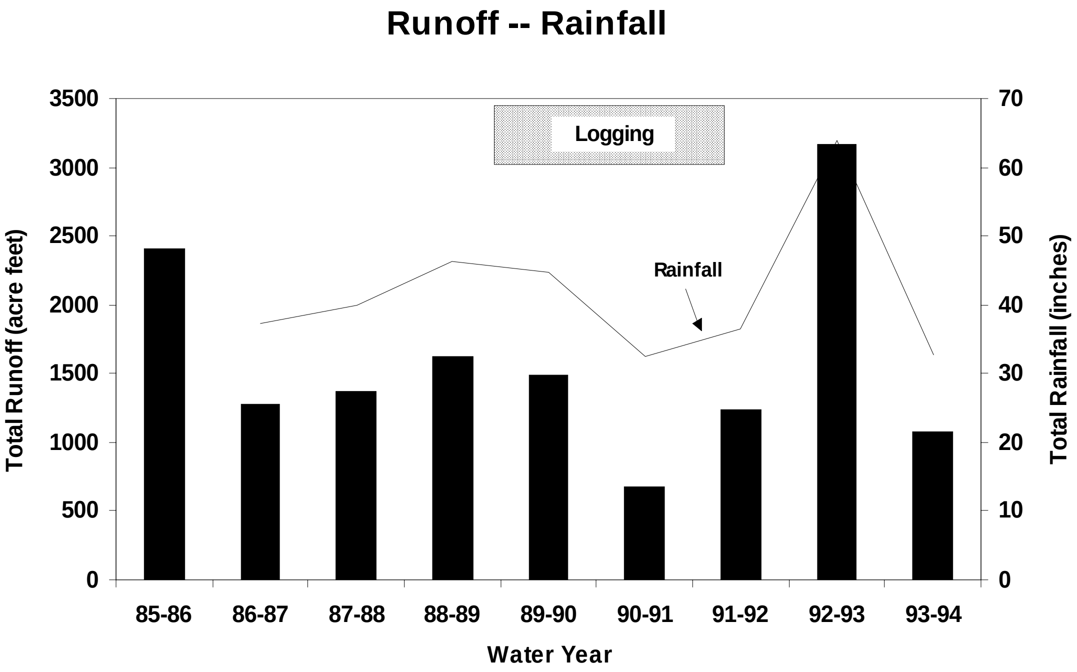


Fig. 5. Total runoff and rainfall for water years 1986-1994 at Station NFC in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. Data provided by U.S. Forest Service, Redwood Sciences Laboratory, Arcata, CA.

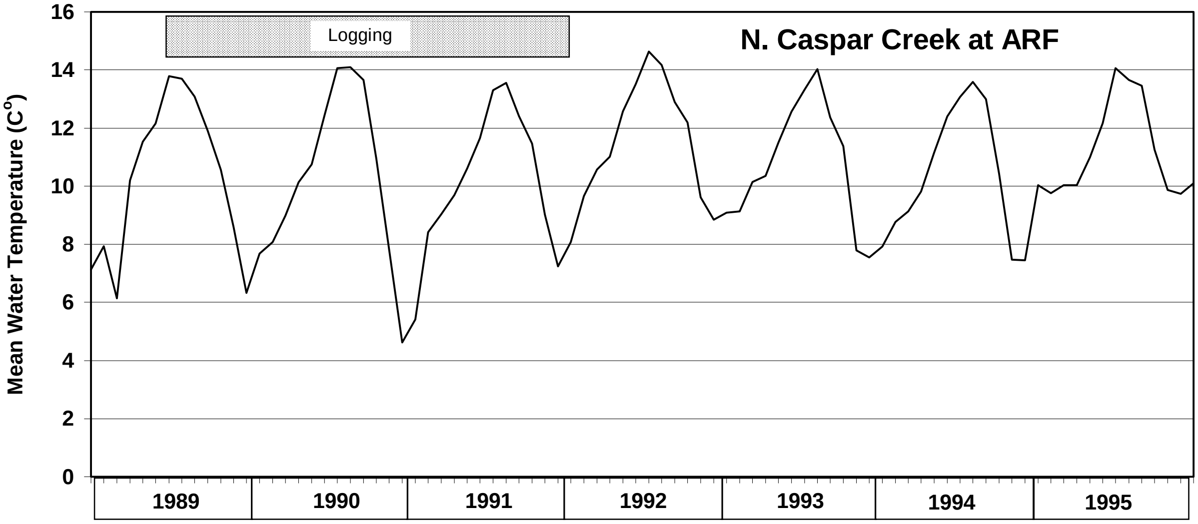


Fig. 6. Variation in monthly mean water temperature (1989-1995) at Station ARF in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. Data provided by the U.S. Forest Service, Redwood Sciences Laboratory, Arcata, CA.

Fig. 7. Map of Station B in N. Caspar Creek, at confluence with Side Tributary B.

Fig. 8. Map of Station C in N. Caspar Creek, at confluence with Side Tributary C.

Fig. 9. Map of Station G in N. Caspar Creek, at confluence with Side Tributary G.

Fig. 10. Map of Station H in N. Caspar Creek, at confluence with Side Tributary H.

Fig. 11. Map of Station I in N. Caspar Creek, at confluence with Side Tributary I.

			Mean Monthly		Monthly Total		Mean Monthly
		Water Temperature			Rainfall at NFC		Discharge at N
			at ARF				
			(C)		(in)		(cu ft/sec)
		mean	min	max			
1985	J						
	F						
	M						
	A						
	M						
	J						
	J						
	A						0.09
	S						0.09
	O						0.18
	N						0.67
	D						1.81
1986	J						4.72
	F						20.32
	M						11.32
	A						0.86
	M						0.57
	J						0.31
	J						0.17
	A				0.03		0.12
	S				0		0.22
	O				0.36		0.20
	N				1.70		0.20
	D				6.18		1.07
1987	J				9.14		4.07
	F				5.75		5.80
	M				10.34		8.18
	A				1.70		0.77
	M				0.33		0.42
	J				0.07		0.24
	J				0.51		0.15
	A				0		0.09
	S				0		0.06
	O				0.93		0.05
	N				7.37		0.40
	D				15.21		10.04
1988	J				9.91		9.15
	F				0.19		1.07
	M				0.68		0.50
	A				2.64		0.38
	M				1.97		0.28
	J				0.72		0.17
	J				0.01		0.10
	A				0.05		0.08
	S				0.05		0.06
	O				1.16		0.05
	N				13.60		2.52
	D	7.12	5.1	9.3	6.52		2.33
1989	J	7.94	6.0	9.2	3.52		3.03
	F	6.14	3.1	8.7	1.92		0.60
	M	10.19	7.1	12.3	16.13		14.33
	A	11.53	9.5	14.0	3.18		2.50
	M	12.16	10.2	16.5	0.60		0.59
	J	13.78	11.3	20.3	0.76		0.37
	J	13.69	10.6	20.1	0		0.21
	A	13.08	10.8	18.7	0.16		0.13
	S	11.91	9.9	15.6	1.84		0.10
	O	10.57	8.2	13.1	5.62		0.52
	N	8.58	6.3	10.6	2.32		0.32
	D	6.32	3.0	10.3	0.30		0.21
1990	J	7.68	2.7	11.0	11.67		4.66
	F	8.08	5.3	9.3	4.87		4.64
	M	8.99	7.3	11.2	4.22		3.03

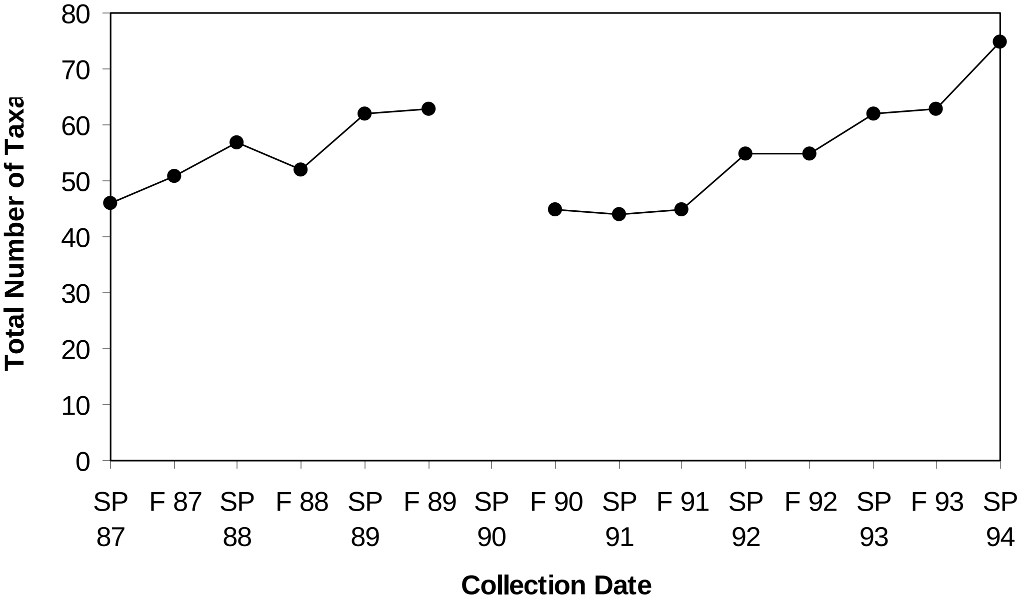


Fig. 12. Variation in total number of macroinvertebrate taxa for 15 collection dates (1987-1994) and for all stations in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

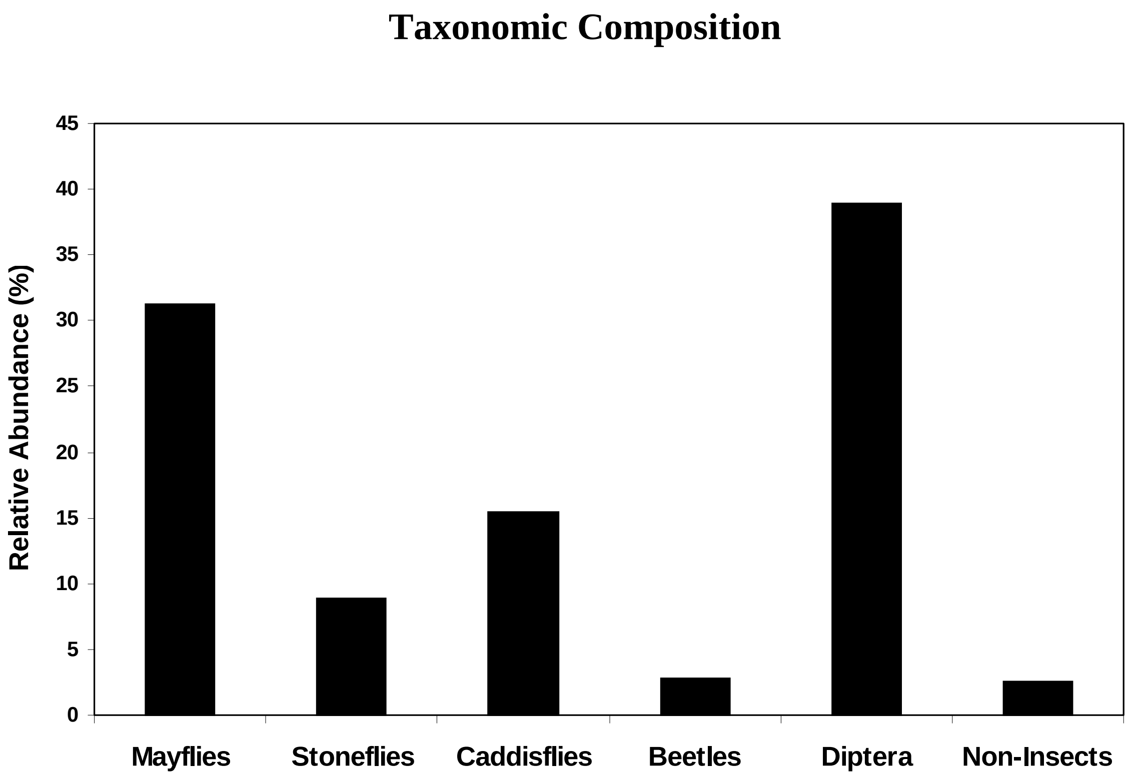


Fig. 13. Macroinvertebrate relative abundance in rock pack samples for entire study period (1987-1994) and for all stations in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

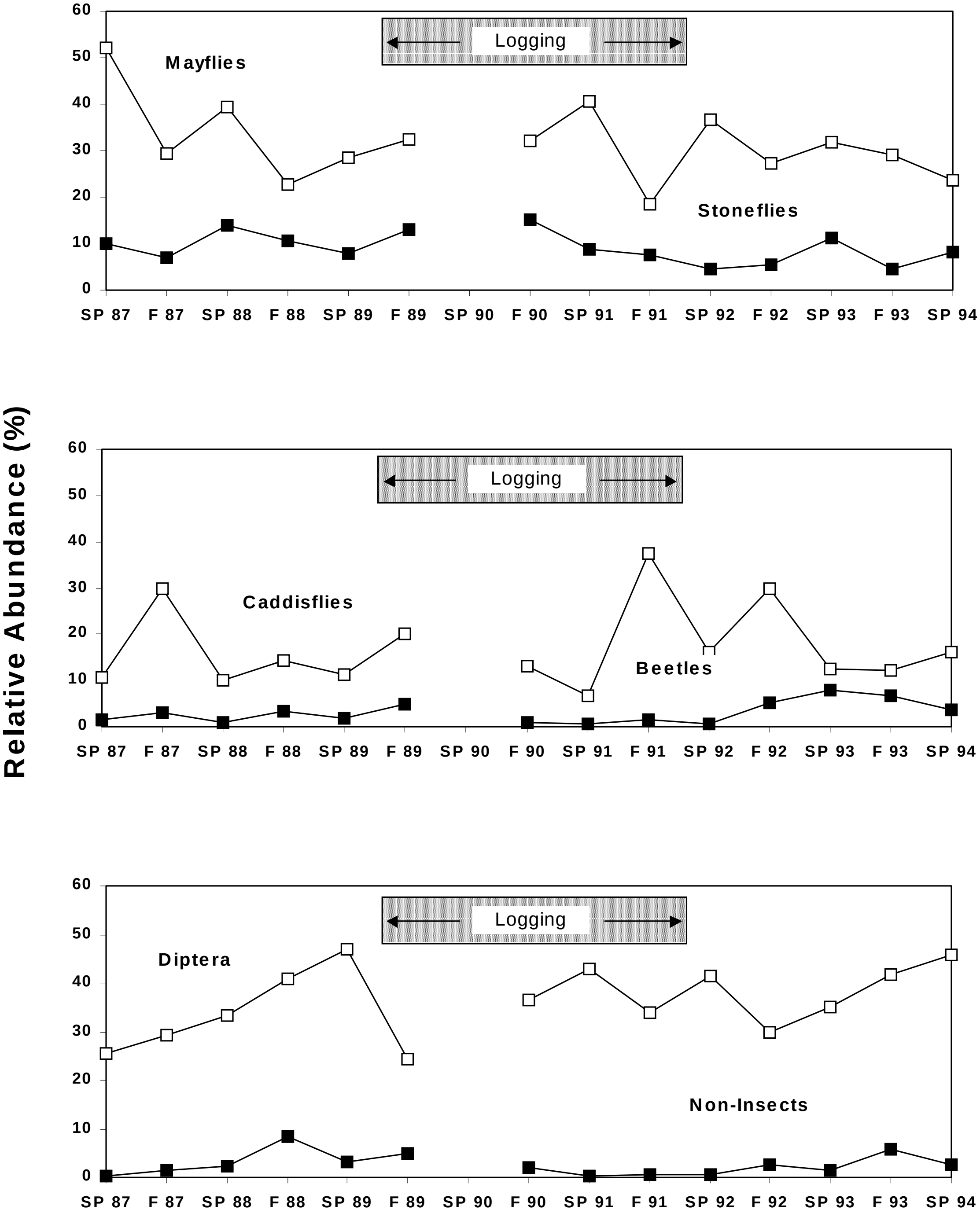


Fig. 14. Relative abundance of six major macroinvertebrate taxa for 15 sample periods (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

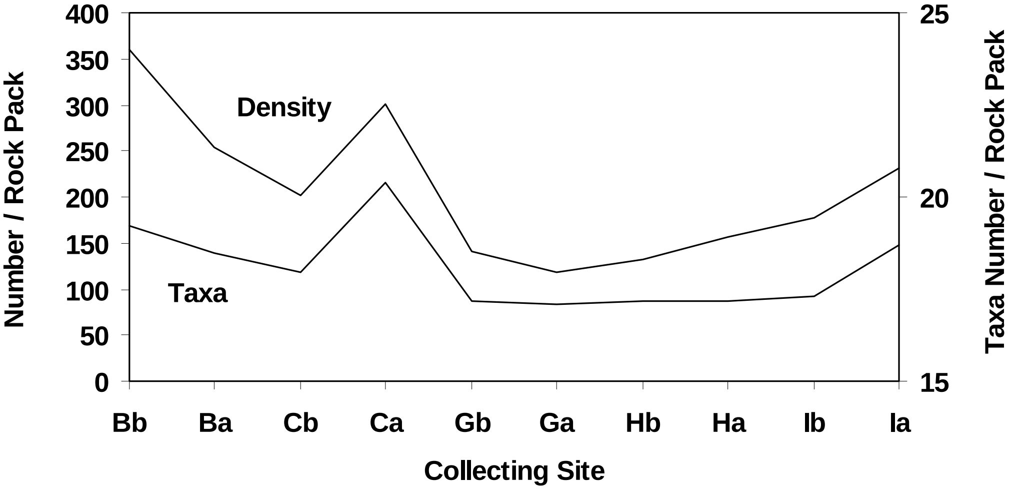


Fig. 15. Overall macroinvertebrate mean density and number of taxa at 10 collecting sites (1987-1994), N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

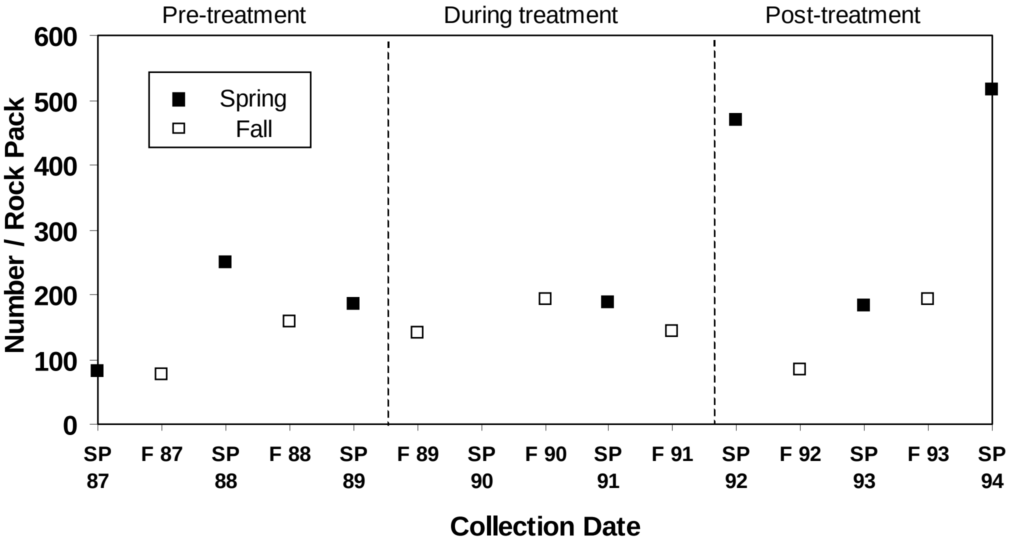


Fig. 16. Overall mean density of macroinvertebrates for the 15 samplecollecting sites periods (1987-1994) and combining all 10 collecting sites in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

Fig. 17. Variation in macroinvertebrate mean densities for 15 collection dates (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.
MI densities > 600 are shown at top of graph. Significantly different ($P < 0.05$) Above and Below densities are indicated with an askerisk (*)

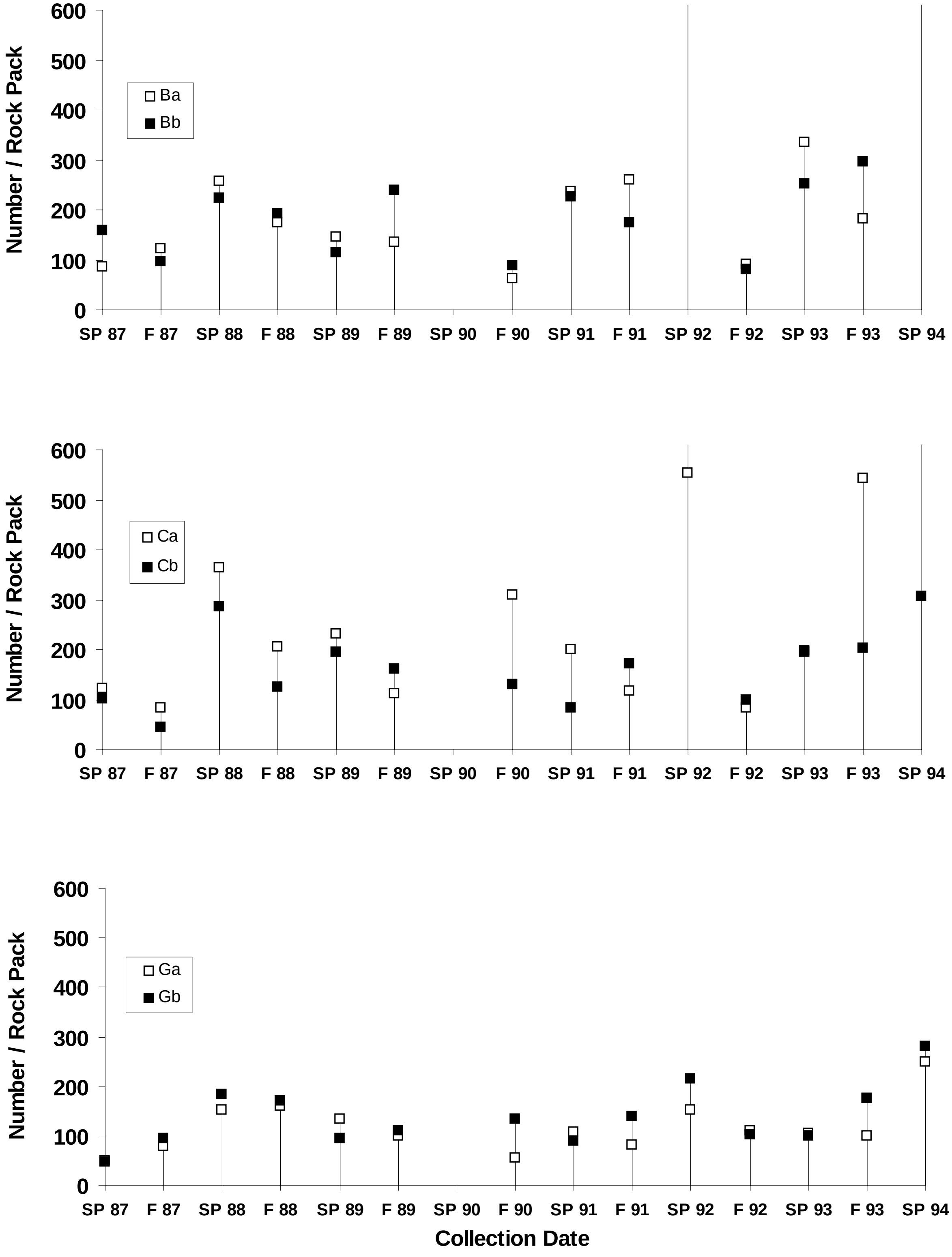
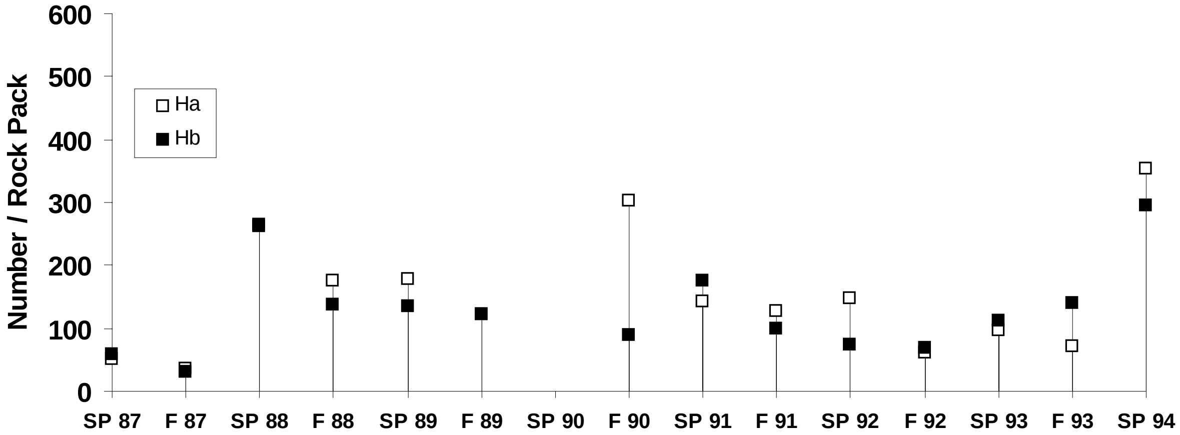


Fig. 17. Variation in macroinvertebrate mean densities for 15 collection dates (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. Significantly different ($P < 0.05$) Above and Below densities are indicated with an askerisk (*).



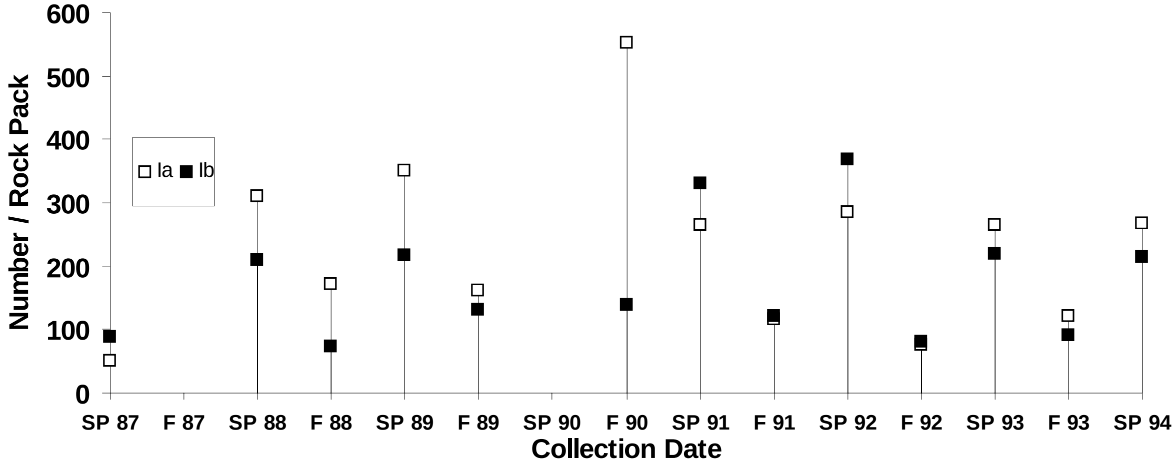
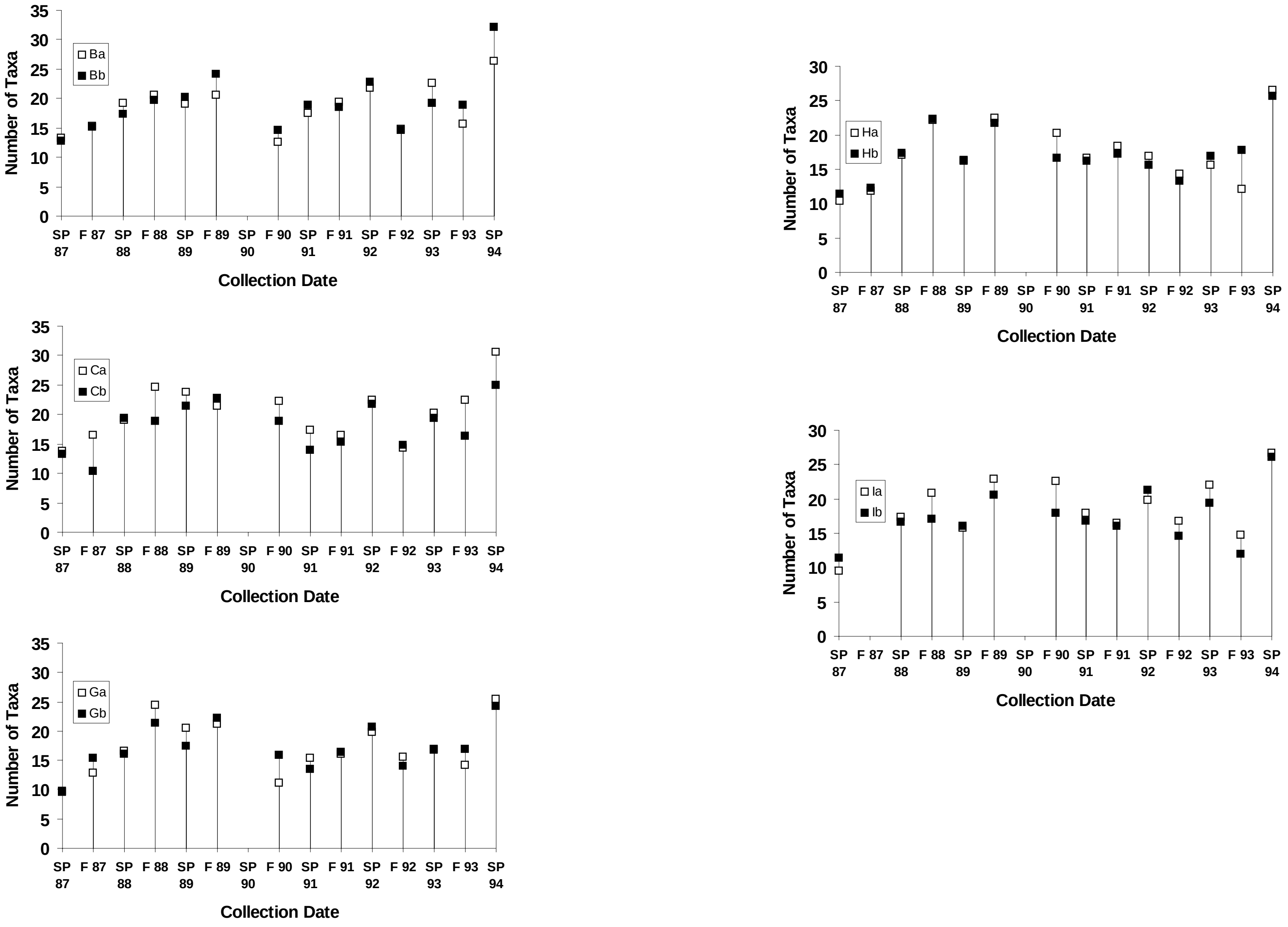


Fig. 18. Variation in mean number of macroinvertebrate taxa for 15 collection dates (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.



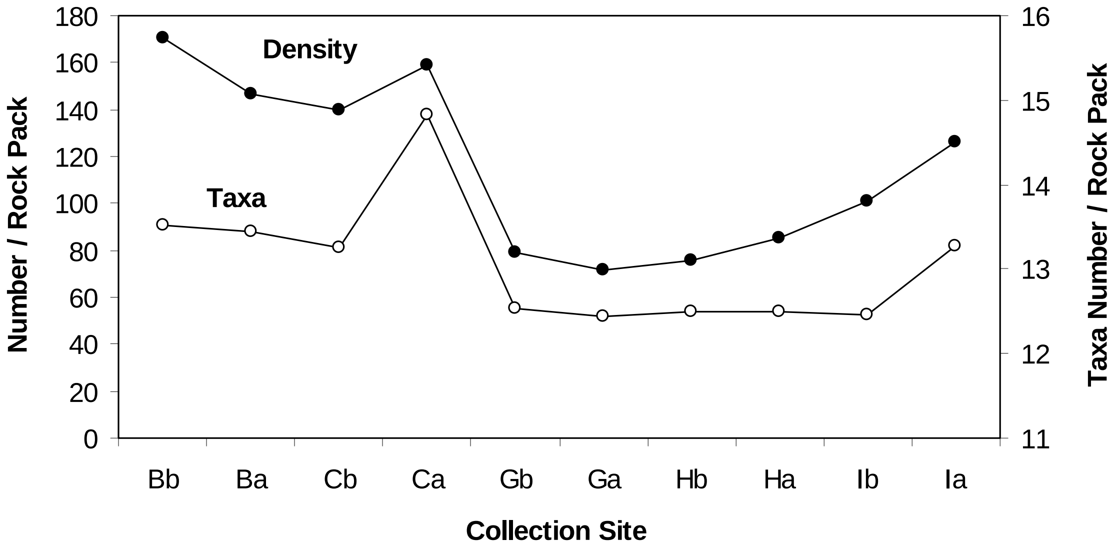


Fig. 19. Variation in EPT (Ephemeroptera, Plecoptera, Trichoptera) mean density and taxa number at 10 sites for entire study (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

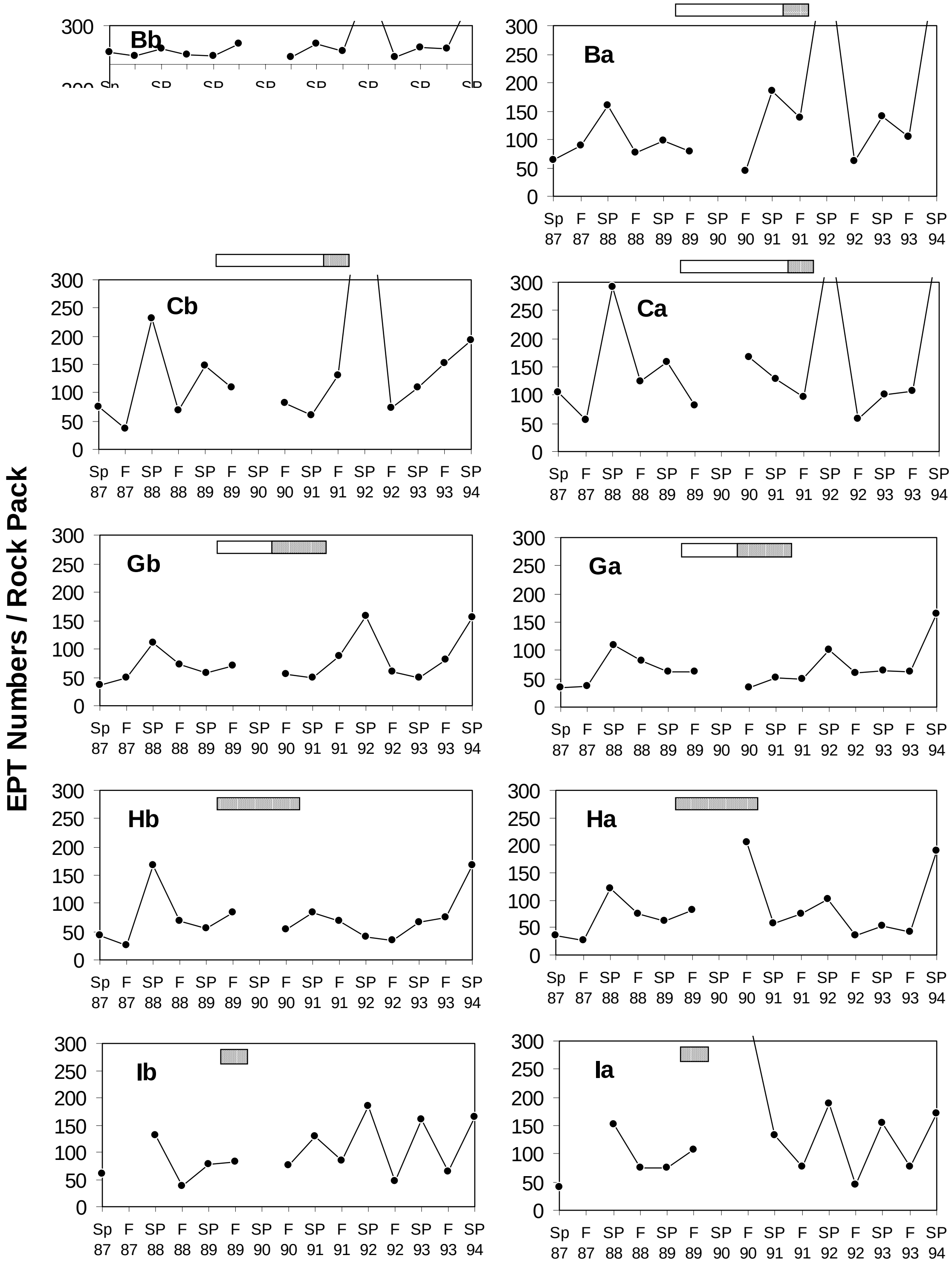


Fig. 20. Variation in EPT (Ephemeroptera, Plecoptera, Trichoptera) mean densities for 15 collection dates (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (top horizontal bar: light bar = upstream logging; dark bar = local logging)

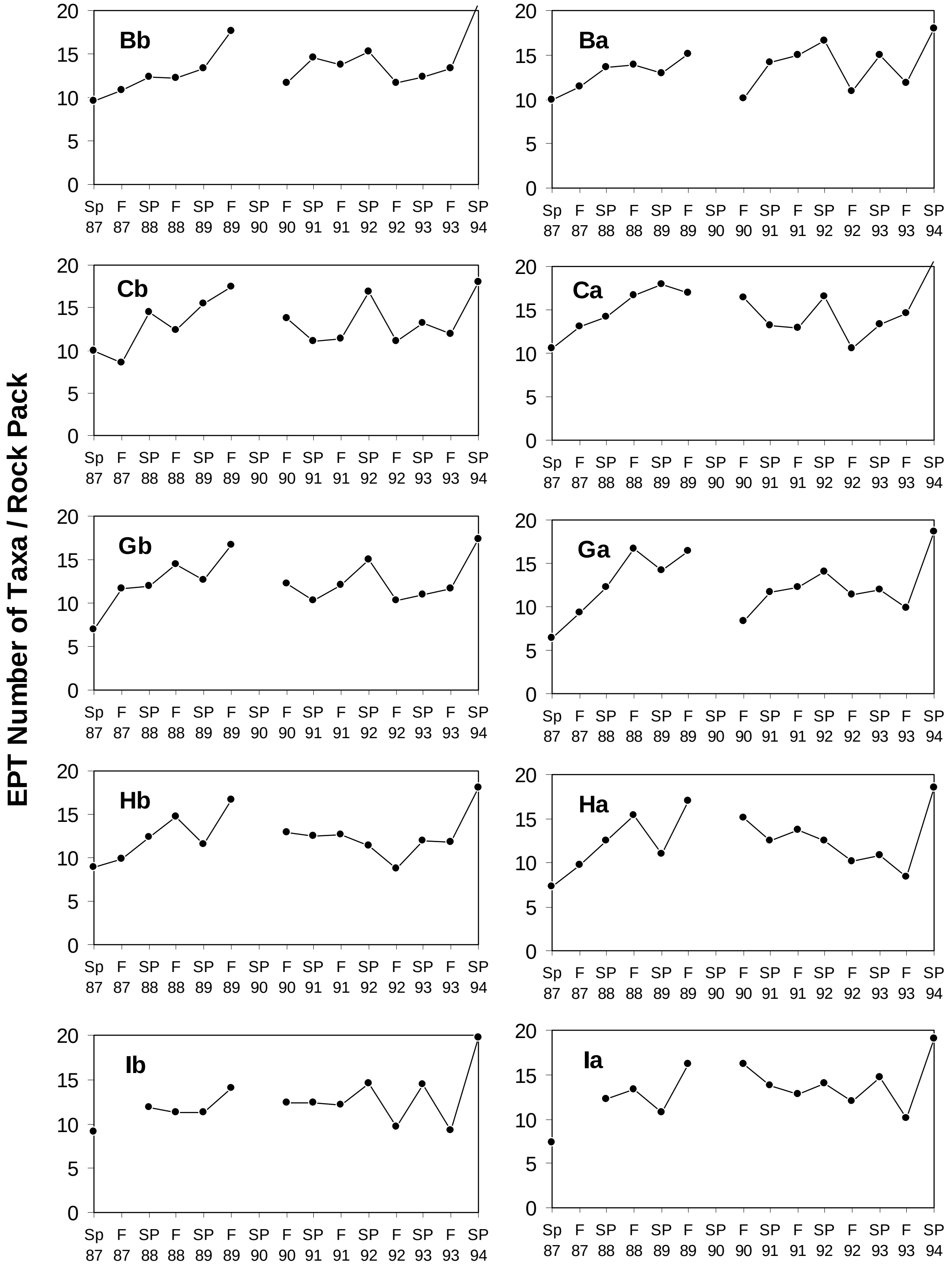


Fig. 21. Variation in EPT (Ephemeroptera, Plecoptera, Trichoptera) mean number of taxa for 15 collection dates (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (top horizontal bar: light bar = upstream logging; dark bar = local logging)

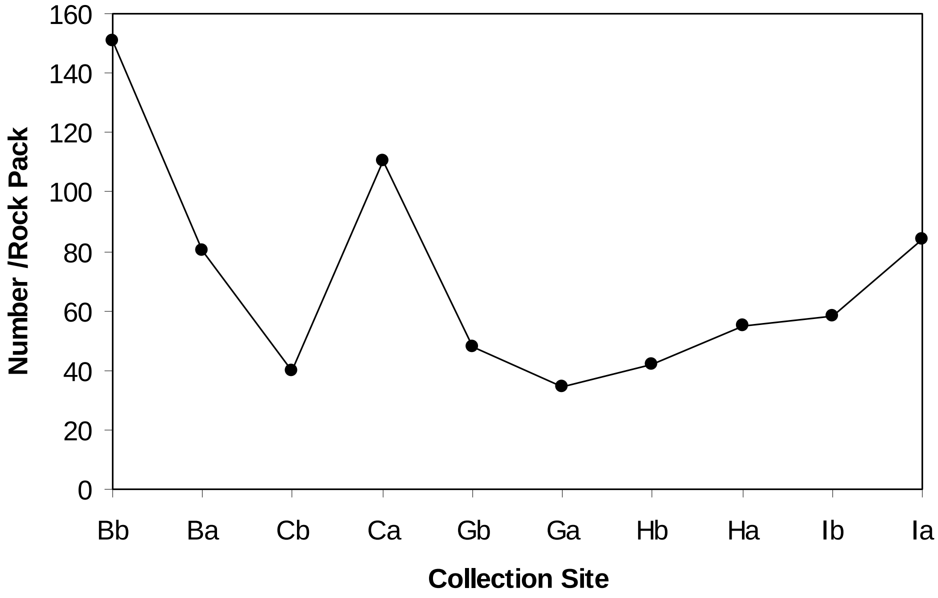


Fig. 22. Variation in Chironomidae mean density at 10 sites for entire study (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

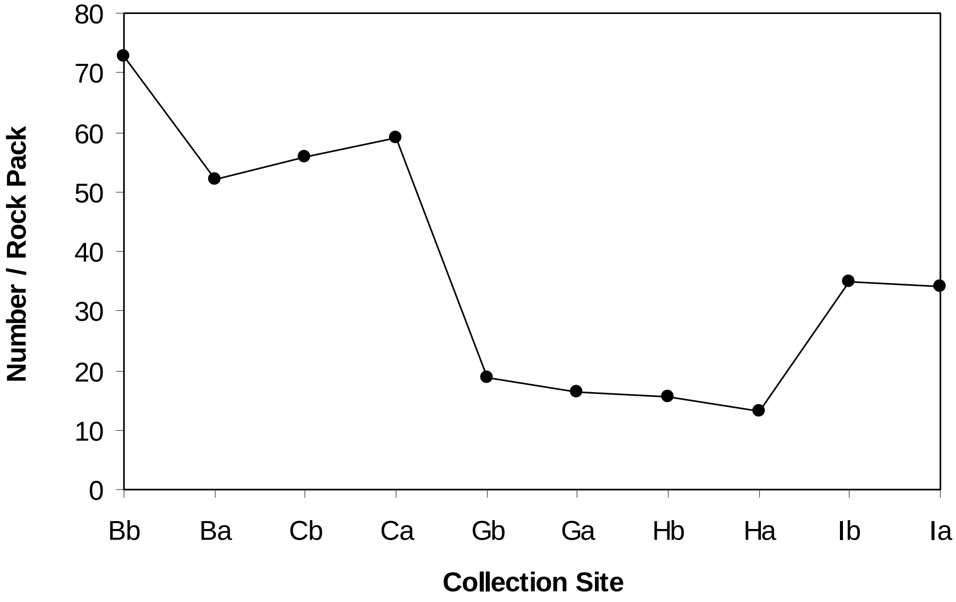


Fig. 23. Variation in Baetis mean density at 10 sites for entire study (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

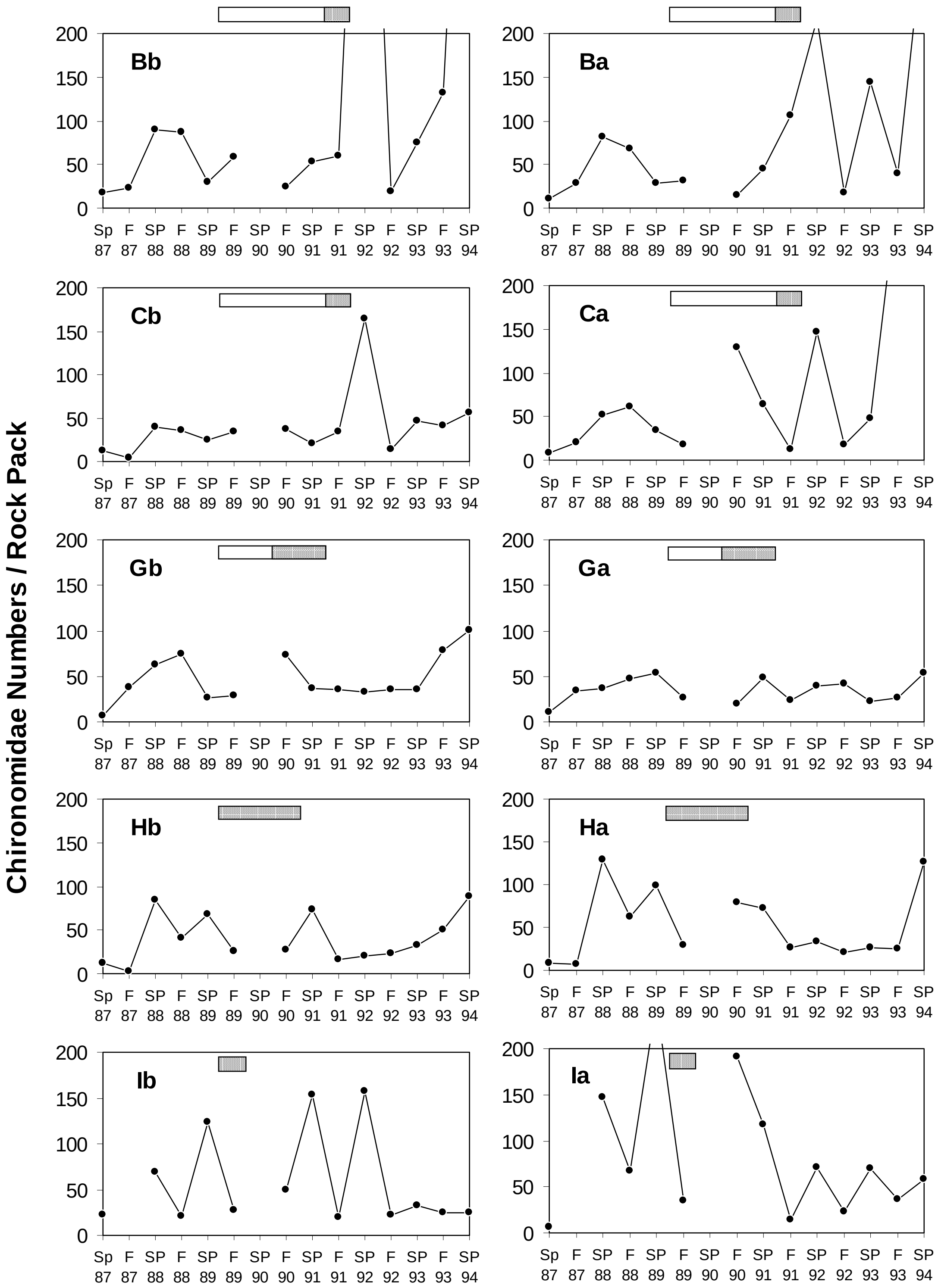


Fig. 24. Variation in Chironomidae mean density for 15 collection dates (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.
(top horizontal bar: light bar = upstream logging; dark bar = local logging)

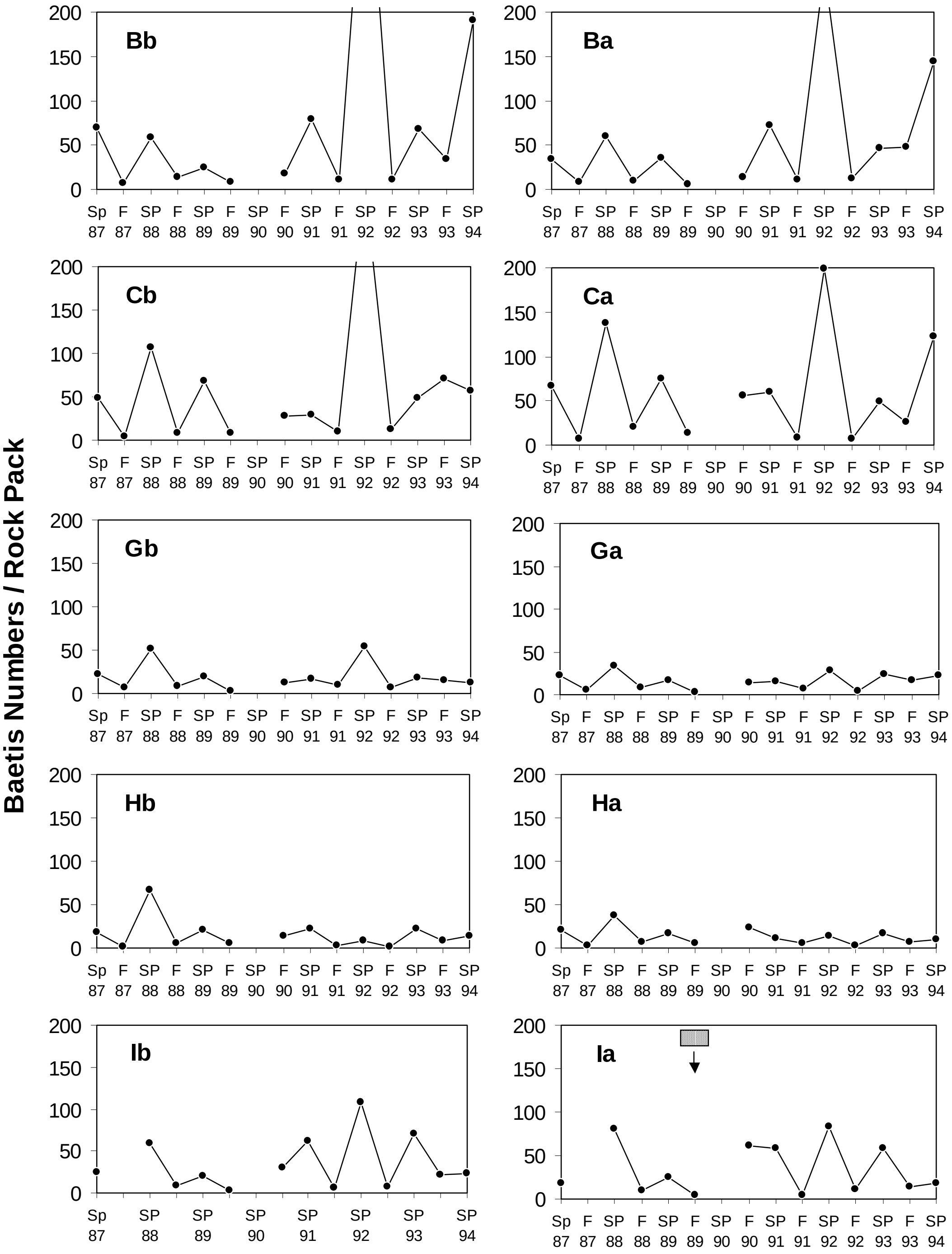


Fig. 25. Variation in Baetis mean density for 15 collection dates (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (arrow = increased solar radiation; top horizontal bar: light bar = upstream logging; dark bar = local logging)

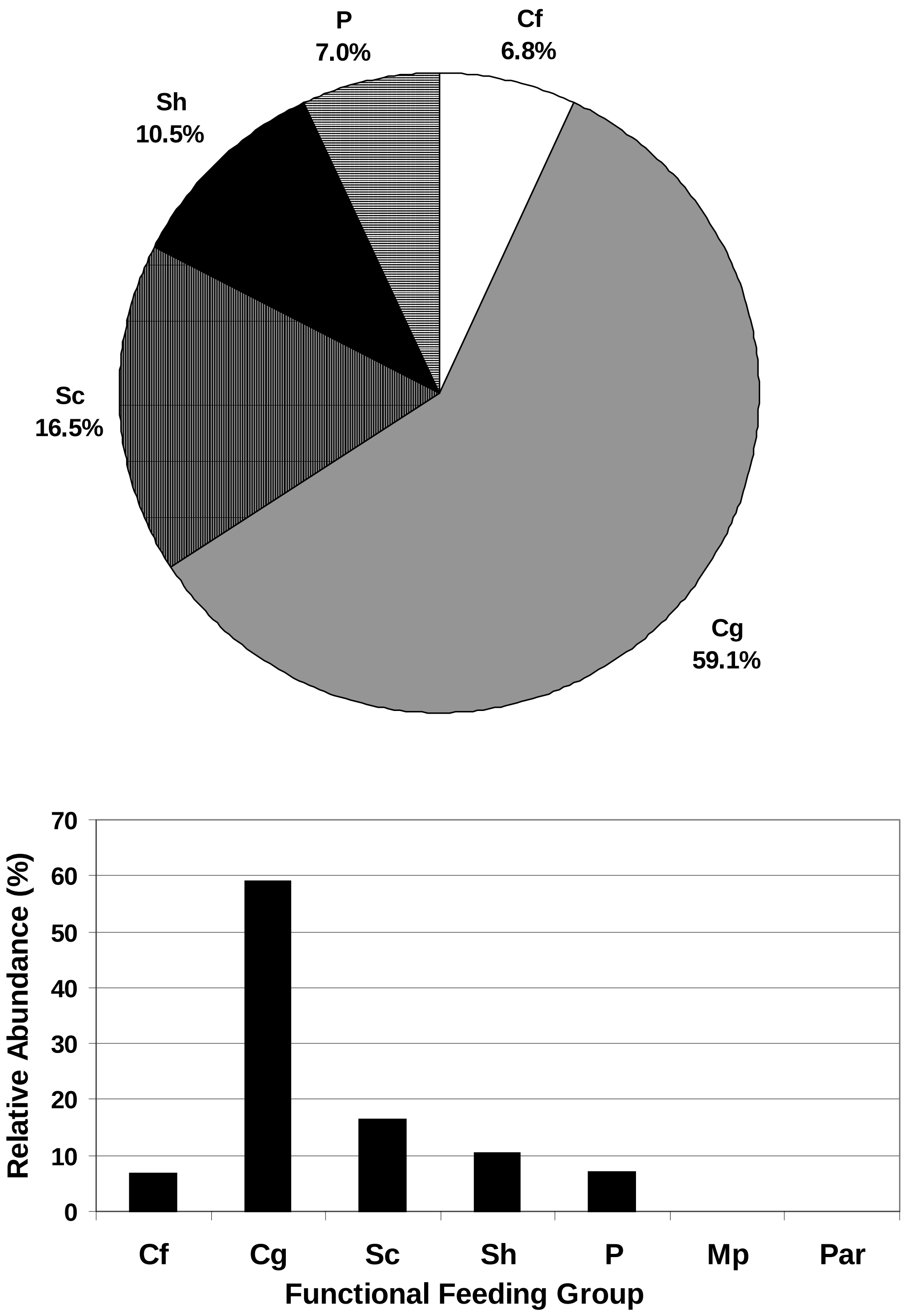


Fig. 26. Relative abundance (%) of Functional Feeding Groups for the entire study period (1987-1994) and for all stations in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (Cf = Collector-filterers, Cg = Collector-gatherers, Sc = Scrapers, Sh = Shredders, Mp = Macrophyte Piercers, and Par = Parasites)

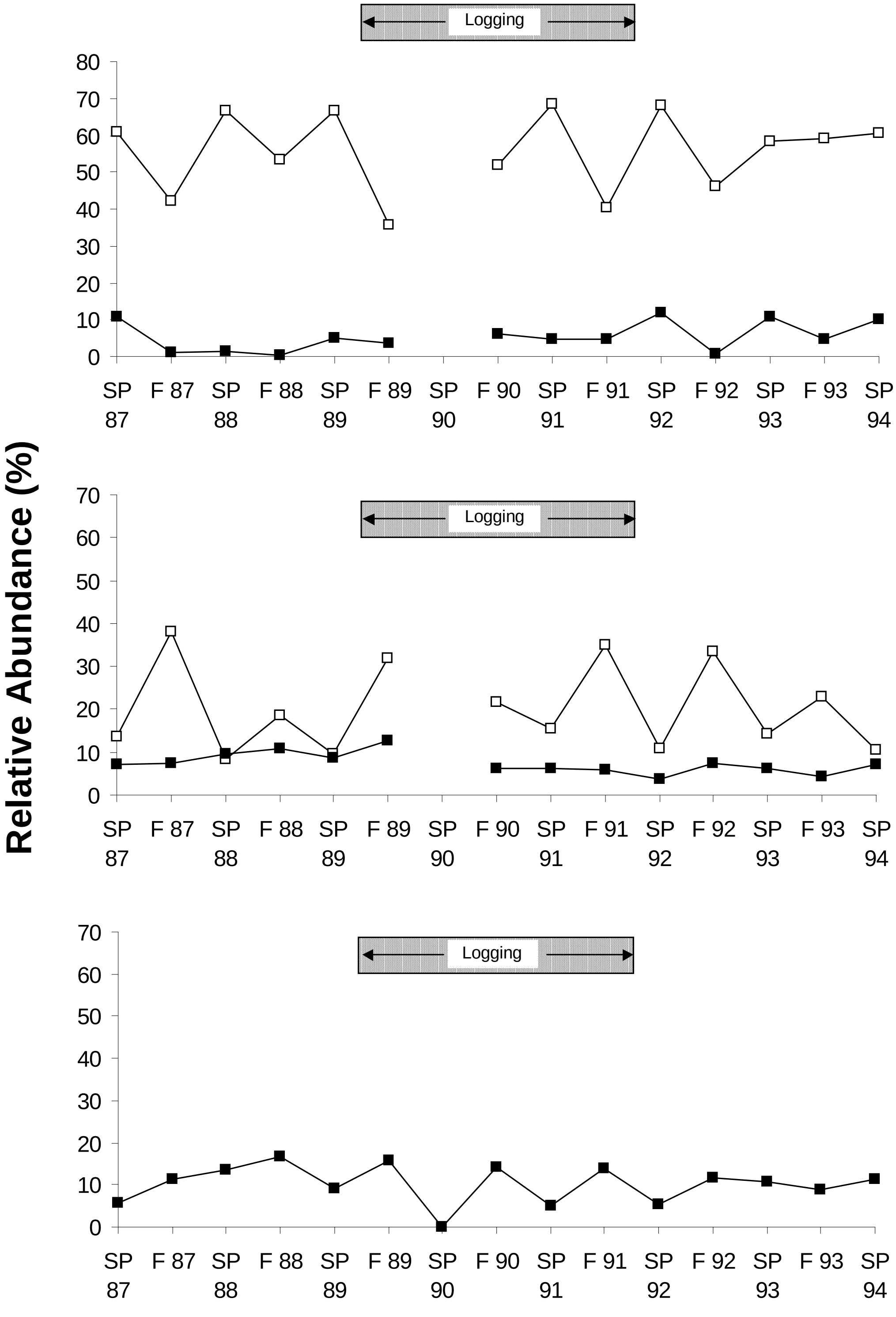


Fig. 27. Variation in relative abundance (mean) of Functional Feeding Groups for 15 sample periods (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

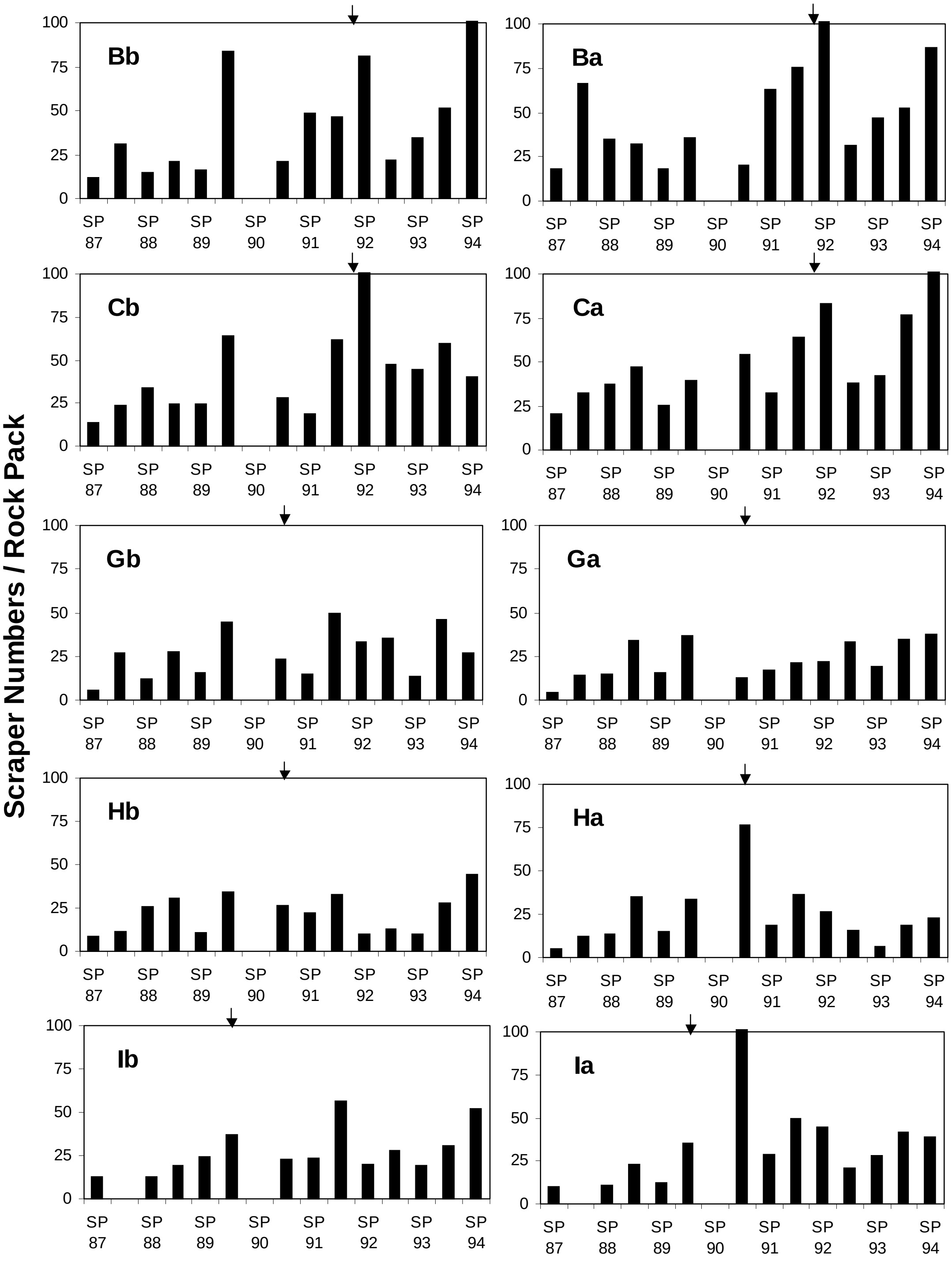


Fig. 28. Variation in Scraper mean density for 15 collection dates (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (arrow = increased solar radiation; horizontal bar: light = upstream logging; dark = local logging)

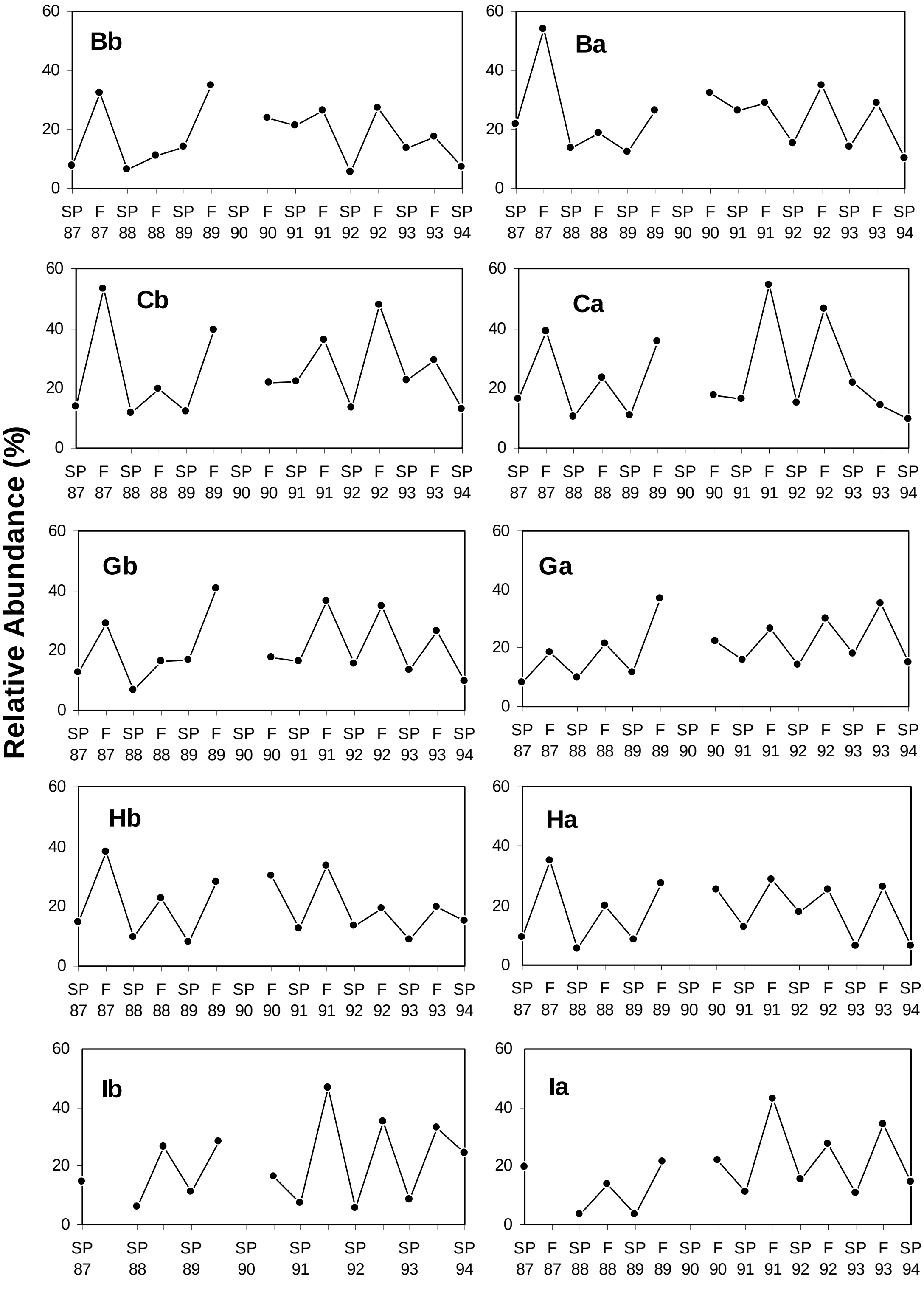


Fig. 29. Variation in Scraper relative abundance (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (arrow = increased solar radiation; top horizontal bar: light bar = upstream logging; dark bar = local logging)

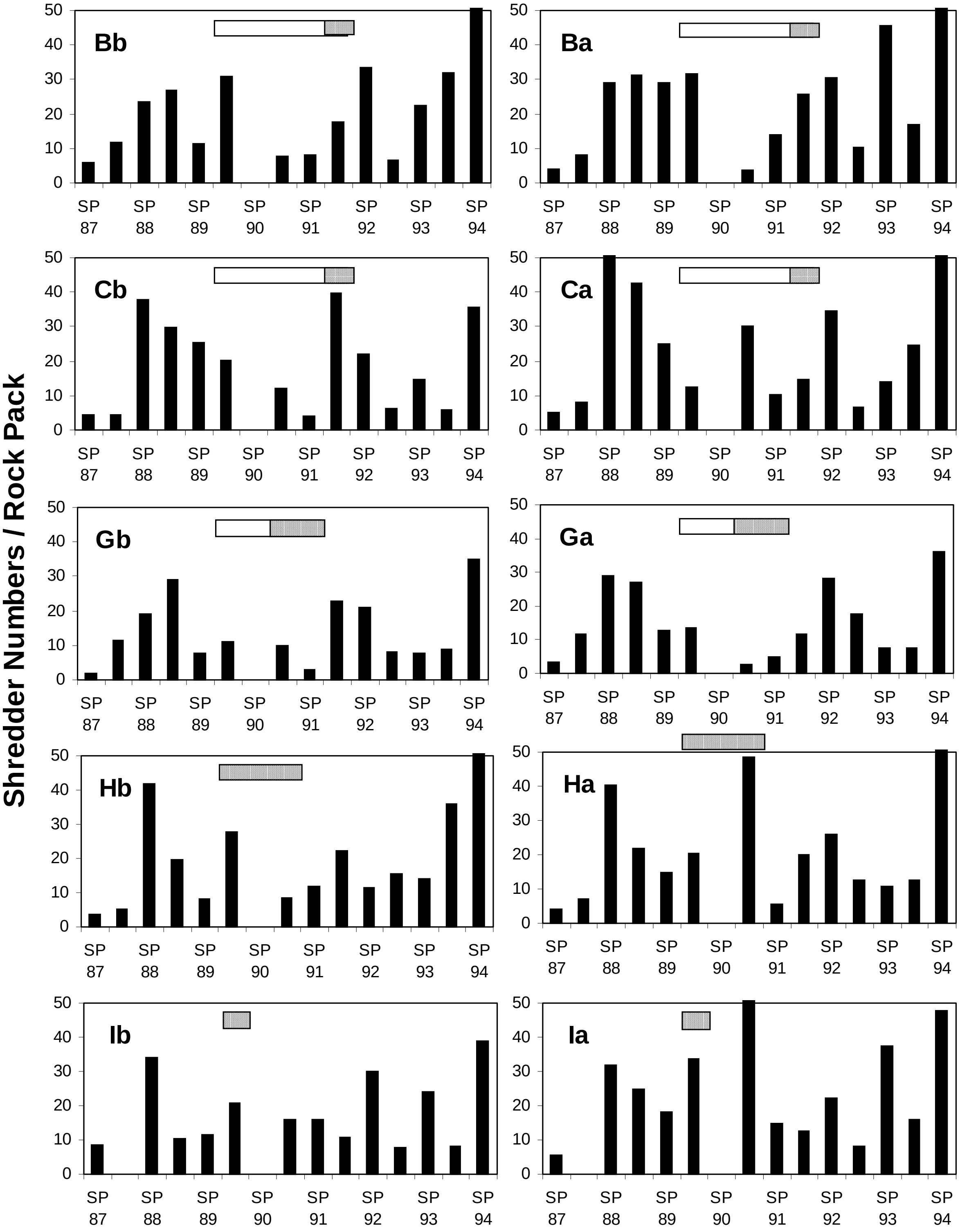


Fig. 30. Variation in Shredder mean density for 15 collection dates (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (top horizontal bar: light bar = upstream logging; dark bar = local logging)

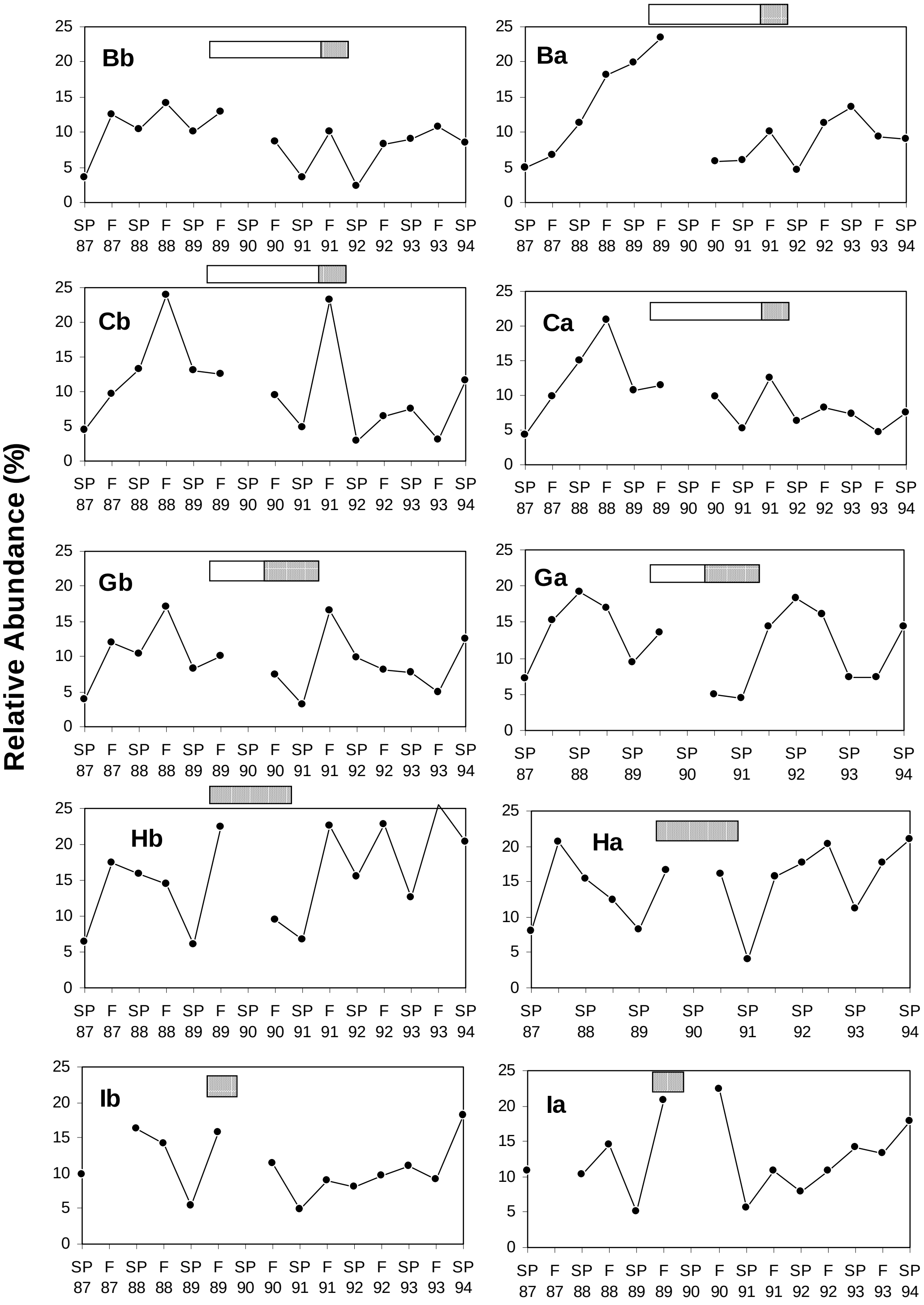


Fig. 31. Variation in Shredder relative abundance (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.
(top horizontal bar: light bar = upstream logging; dark bar = local logging)

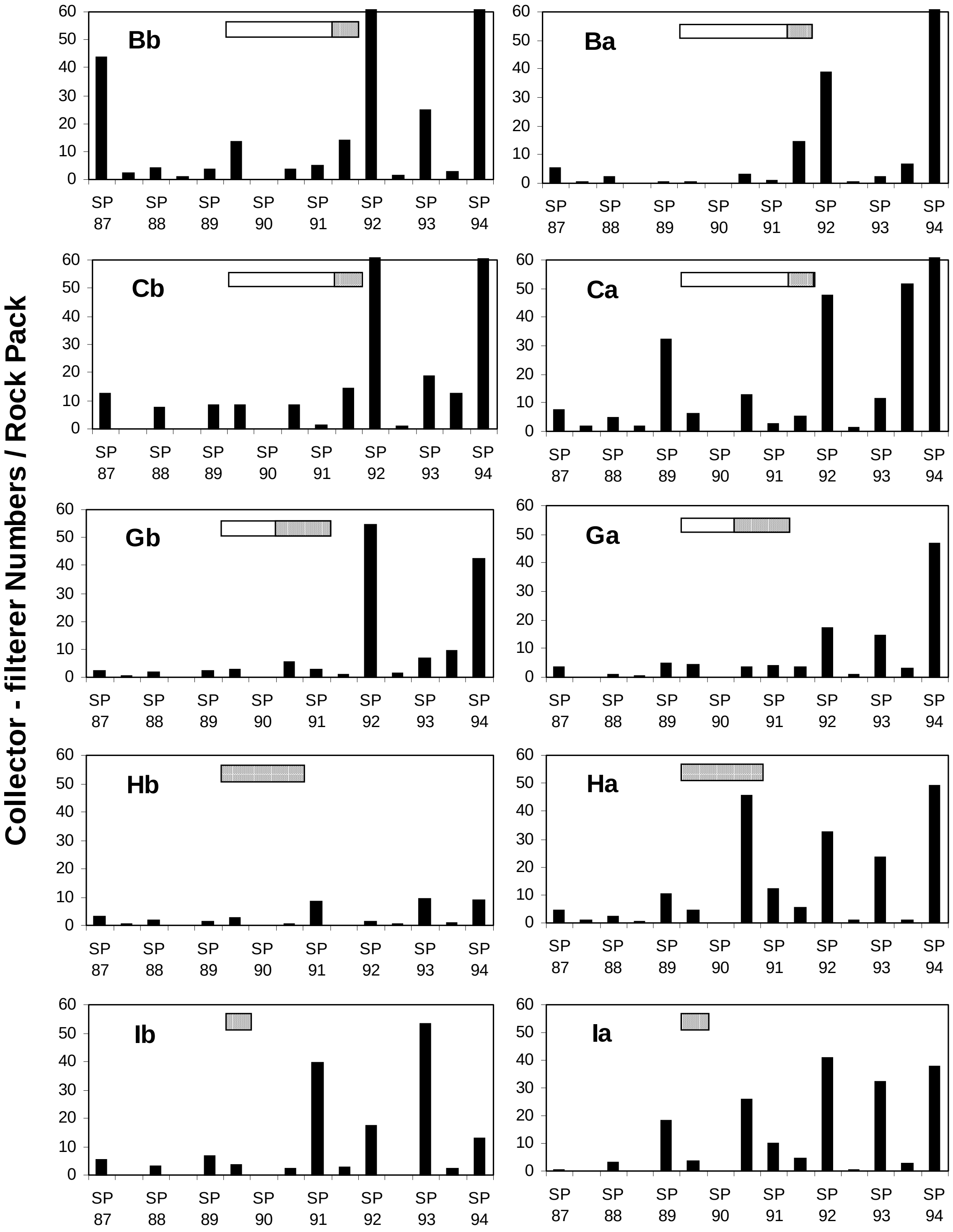


Fig. 32. Variation in Collector - filterer mean density for 15 collection dates (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (top horizontal bar: light bar = upstream logging; dark bar = local logging)

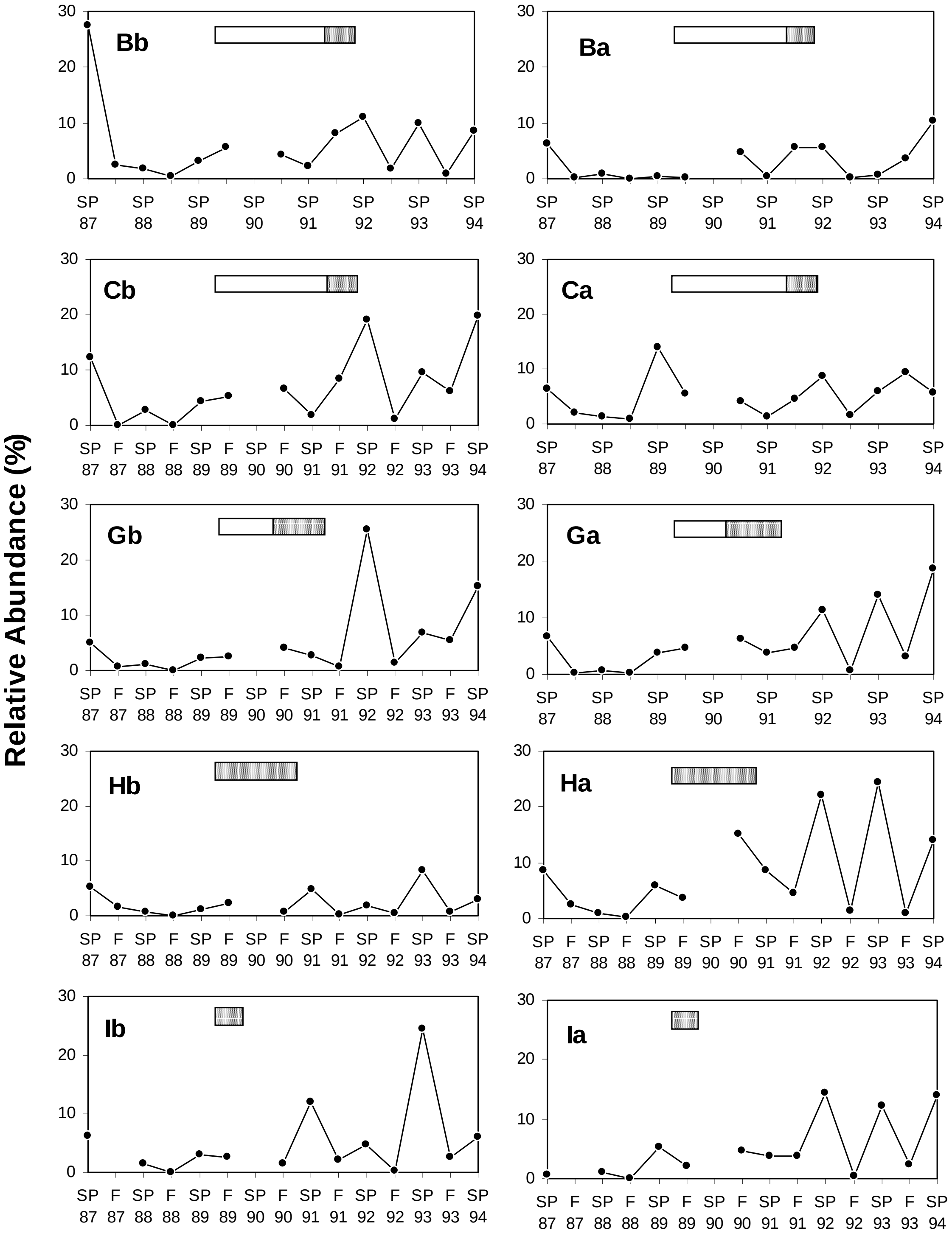


Fig. 33. Variation in Collector-filterer relative abundance (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.
(top horizontal bar: light bar = upstream logging; dark bar = local logging)

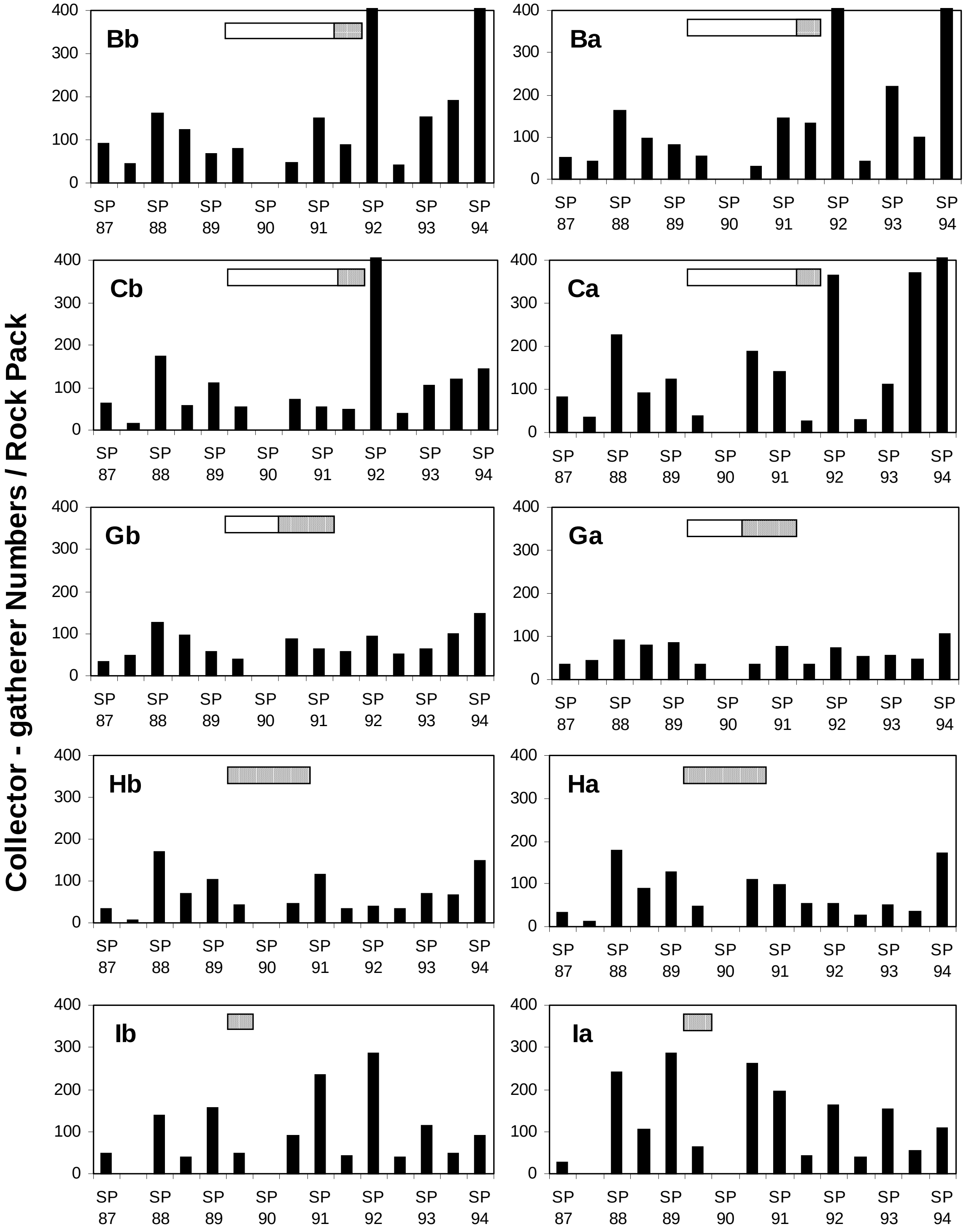


Fig. 34. Variation in Collector - gatherer mean density for 15 collection dates (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.
(top horizontal bar: light bar = upstream logging; dark bar = local logging)

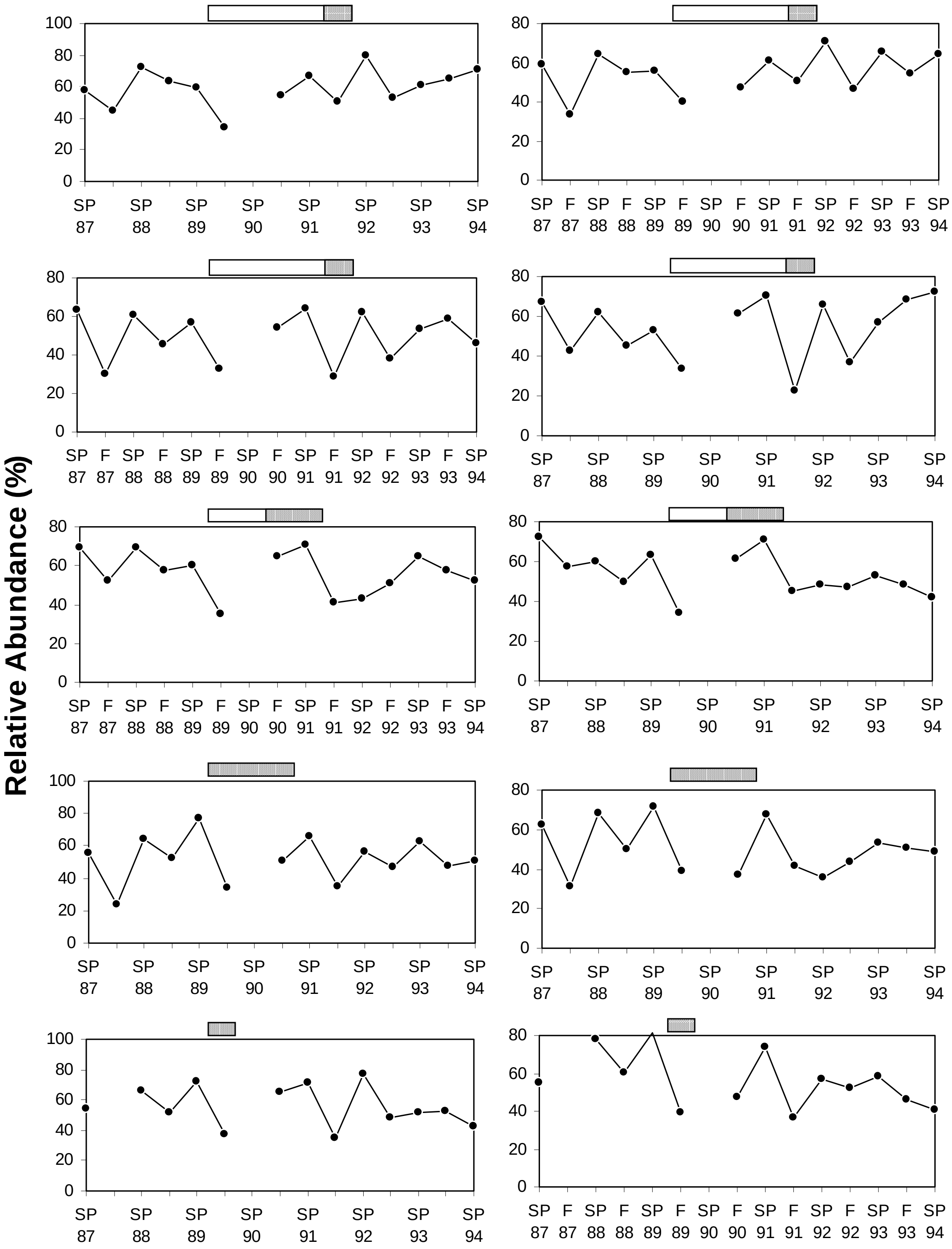


Fig. 35. Variation in Collector - gatherer relative abundance (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.
(top horizontal bar: light bar = upstream logging; dark bar = local logging)

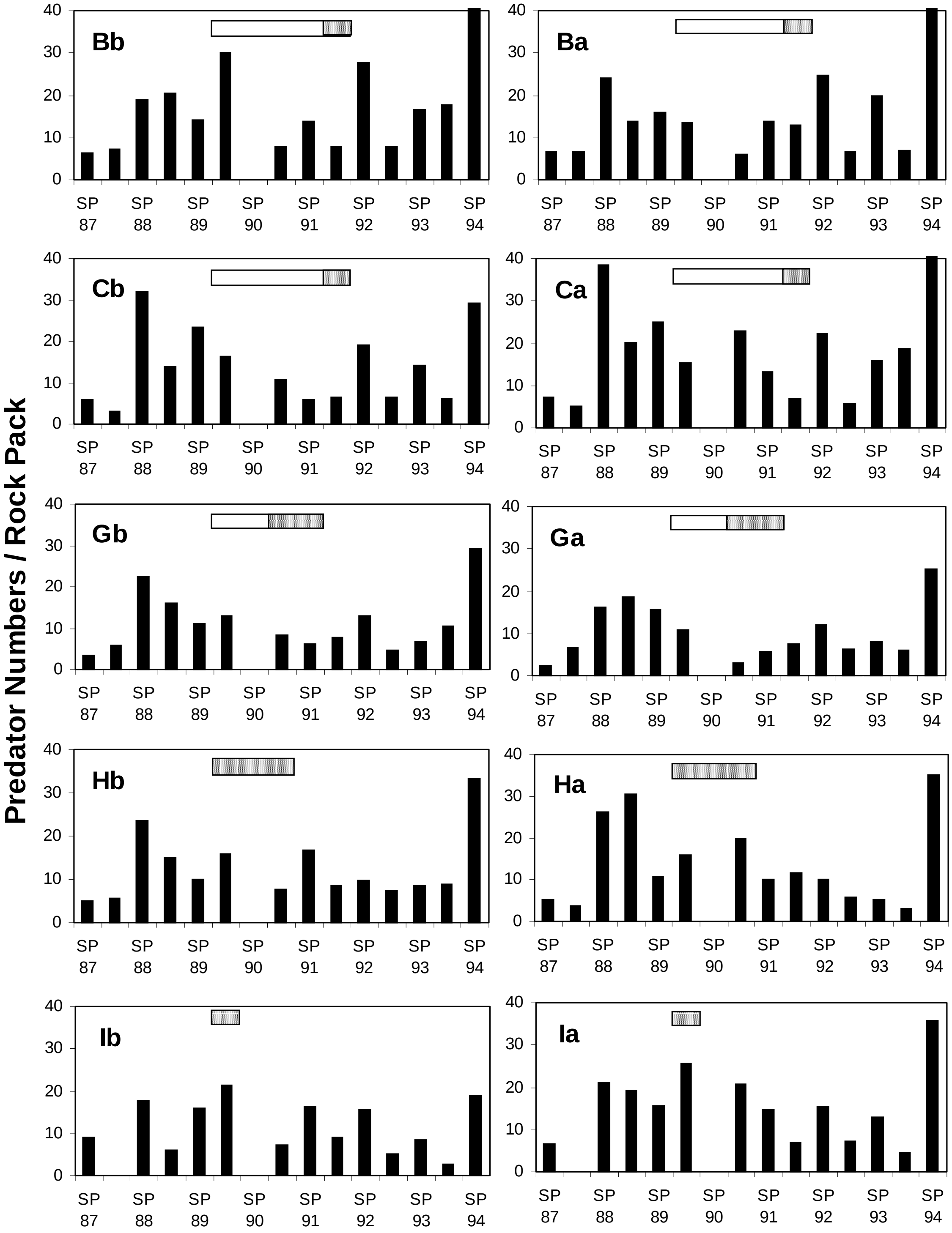


Fig. 36. Variation in Predator mean density for 15 collection dates (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.
(top horizontal bar: light bar = upstream logging; dark bar = local logging)

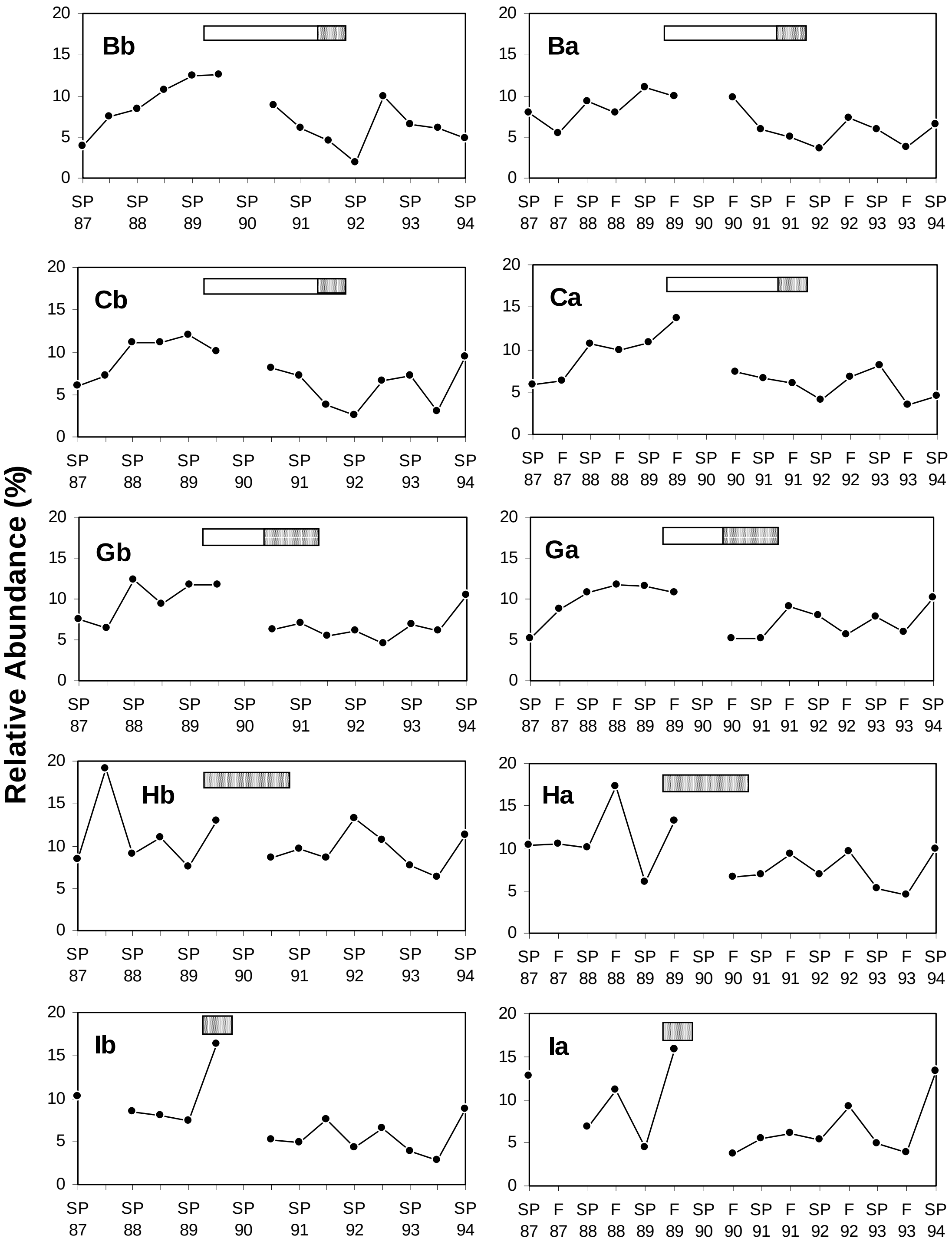


Fig. 37. Variation in Predator relative abundance (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.
(top horizontal bar: light bar = upstream logging; dark bar = local logging)

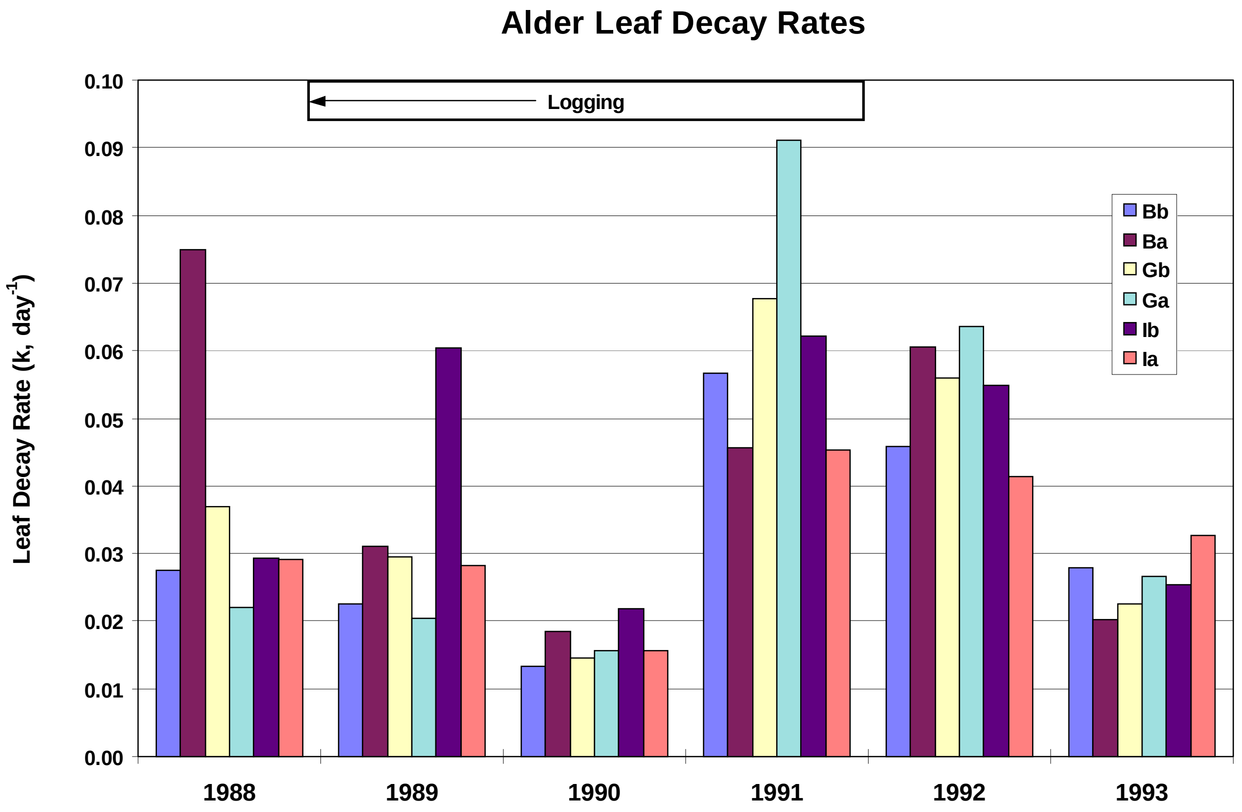


Fig. 38. Variation in alder leaf decay rates during the study period 1988-1993 at six sites in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

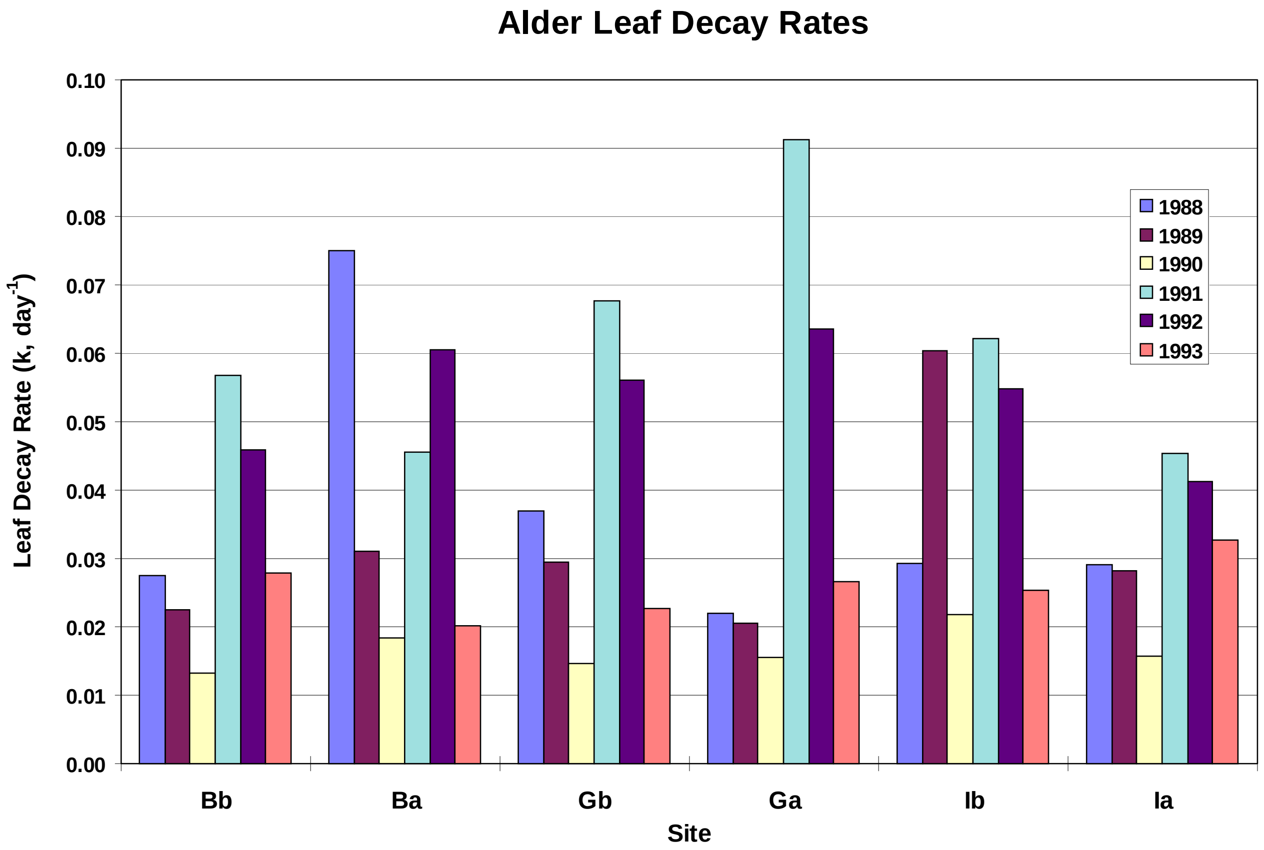


Fig. 39. Variation in alder leaf decay rates during the study period 1988-1993 at six sites in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

Water Temperature at Leaf Decay

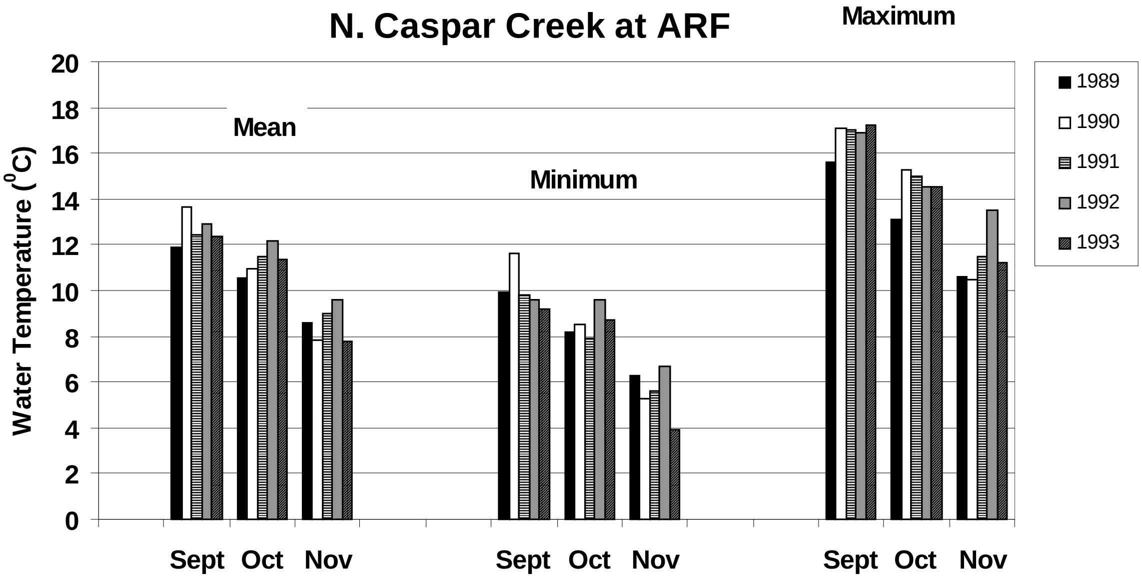


Fig. 40. Monthly mean, minimum and maximum water temperatures during the months of leaf decay (September, October, and November) for 1989-1993 in N. Caspar Creek, Jackson Demonstration Forest, Fort Bragg, CA.

Fig. 41. Pre- and Post-Logging fisheye photographs of the forest canopy at Station G, N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

Fig. 42. Pre-Logging fisheye photographs of the forest canopy at Station B, N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

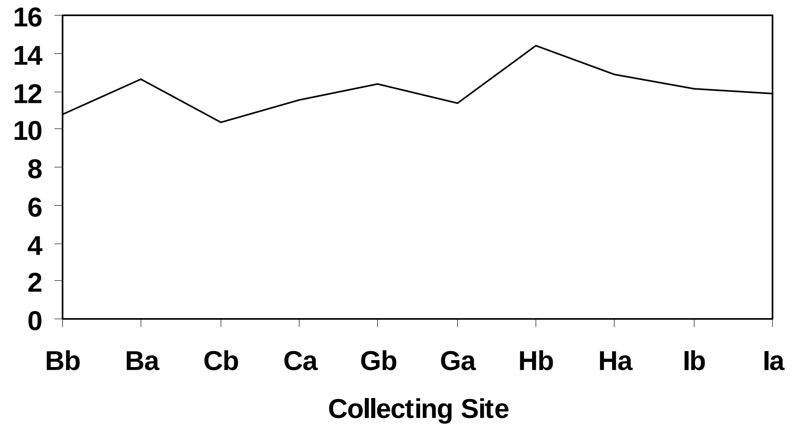


Fig. 43. Number (mean) of diatom species collected at 10 sampling sites (1989-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

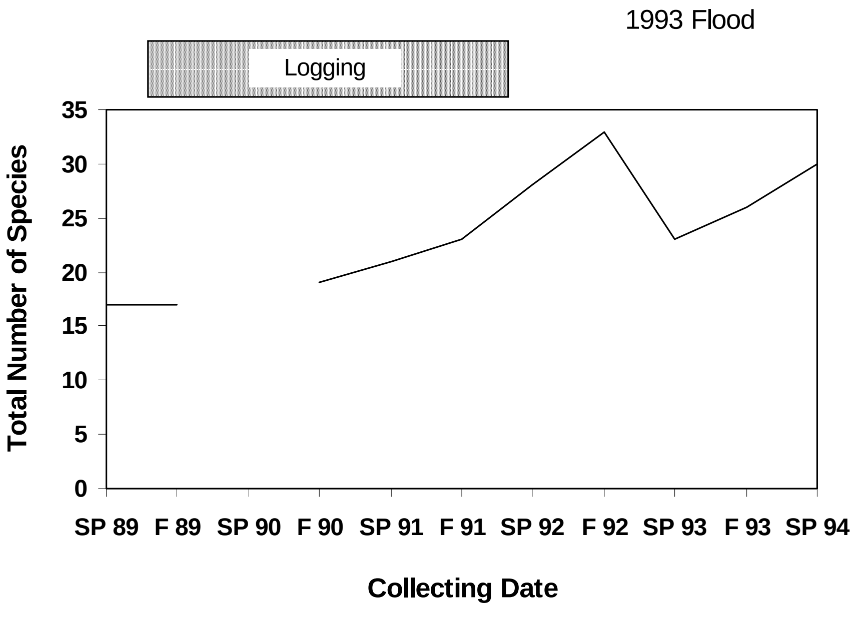


Fig. 44. Variation in total number of diatom species for all stations and 11 collecting dates (1989-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

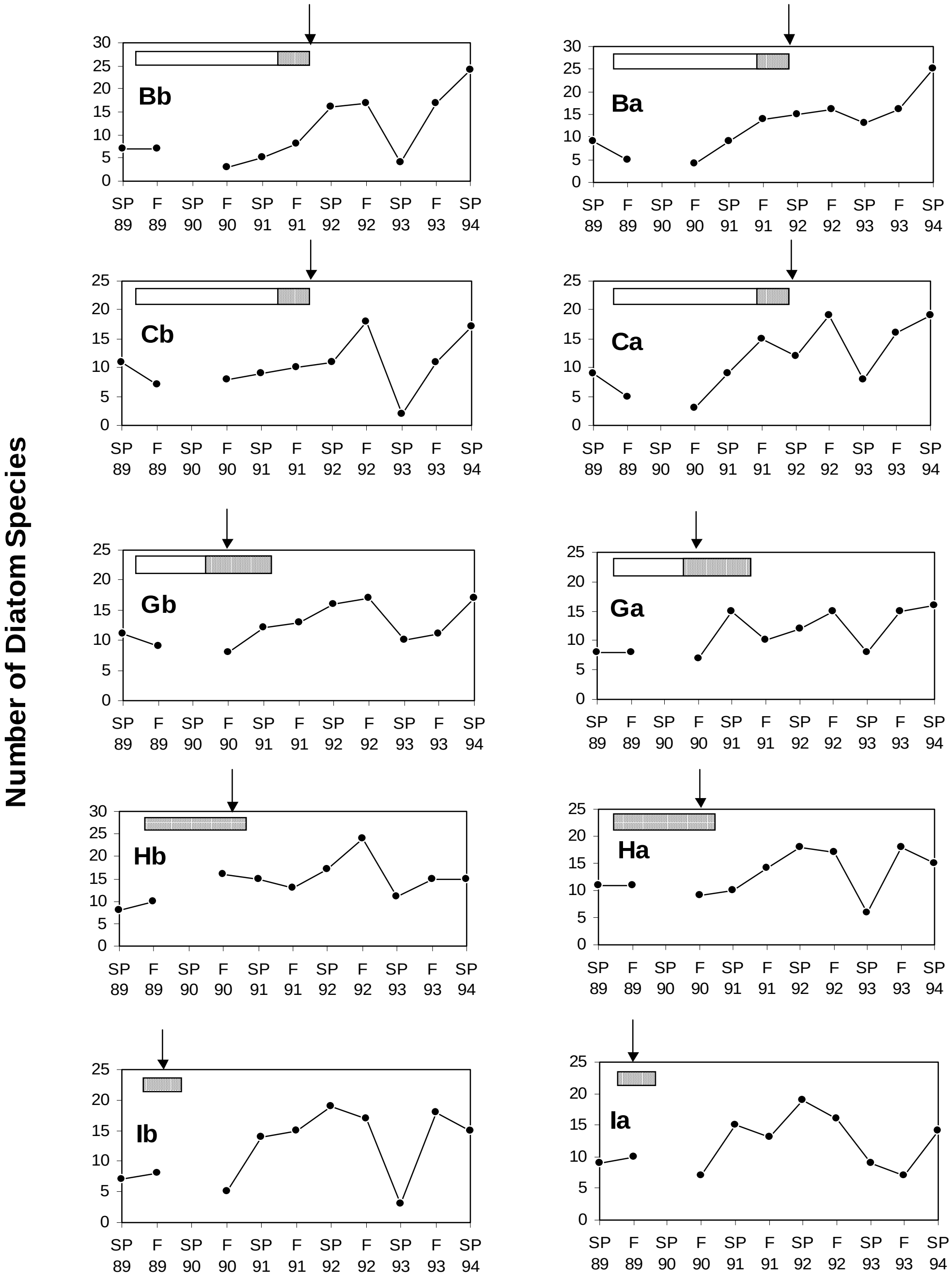


Fig. 45. Variation in number of diatom species collected (1989-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (arrow = increased solar radiation; top horizontal bar: light bar = upstream logging; dark bar = local logging)

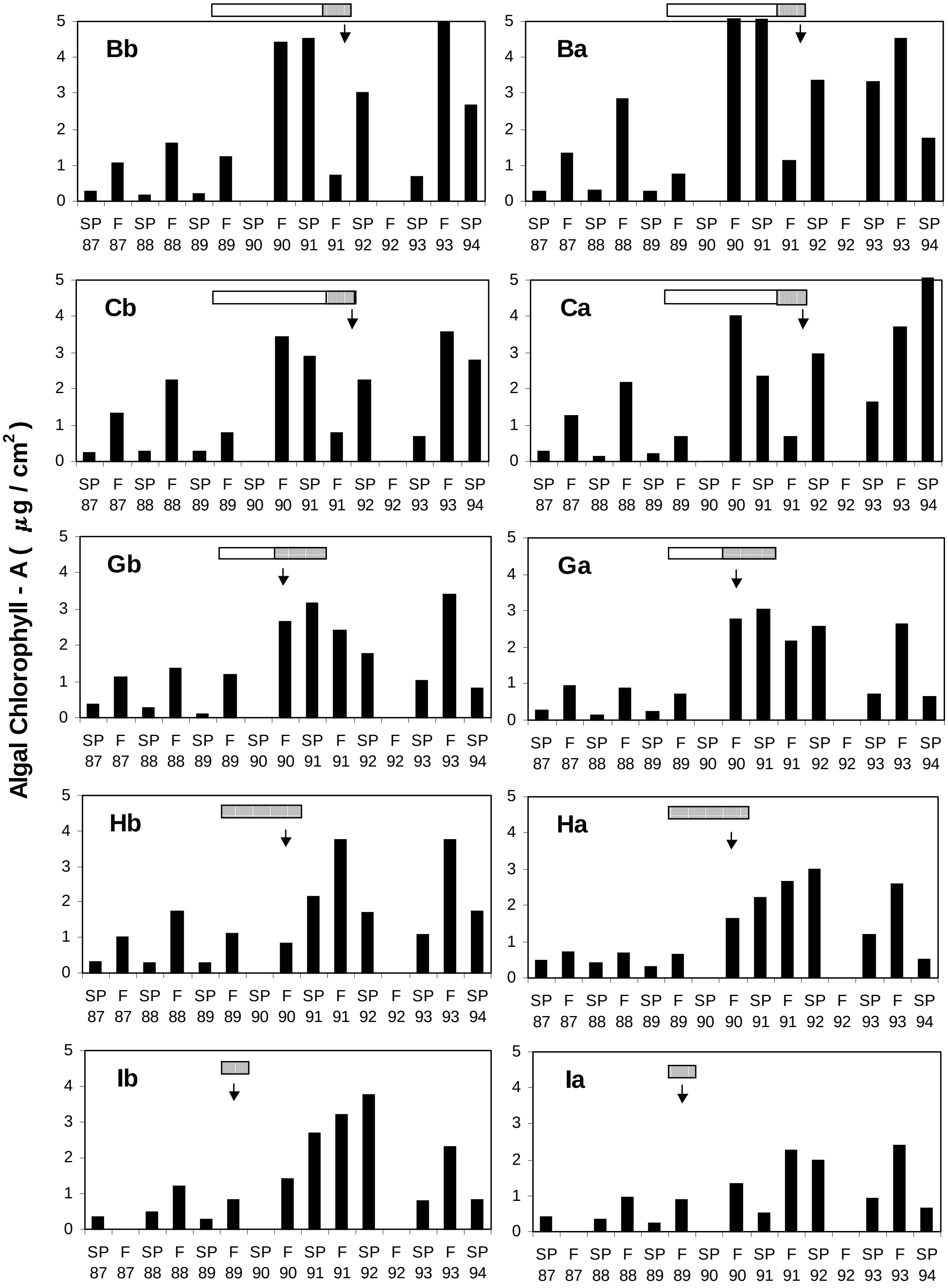


Fig. 46. Variation in algal Chlorophyll - a for entire study period (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (arrow = increased solar radiation; top horizontal bar: light bar = upstream logging, dark bar = local logging)

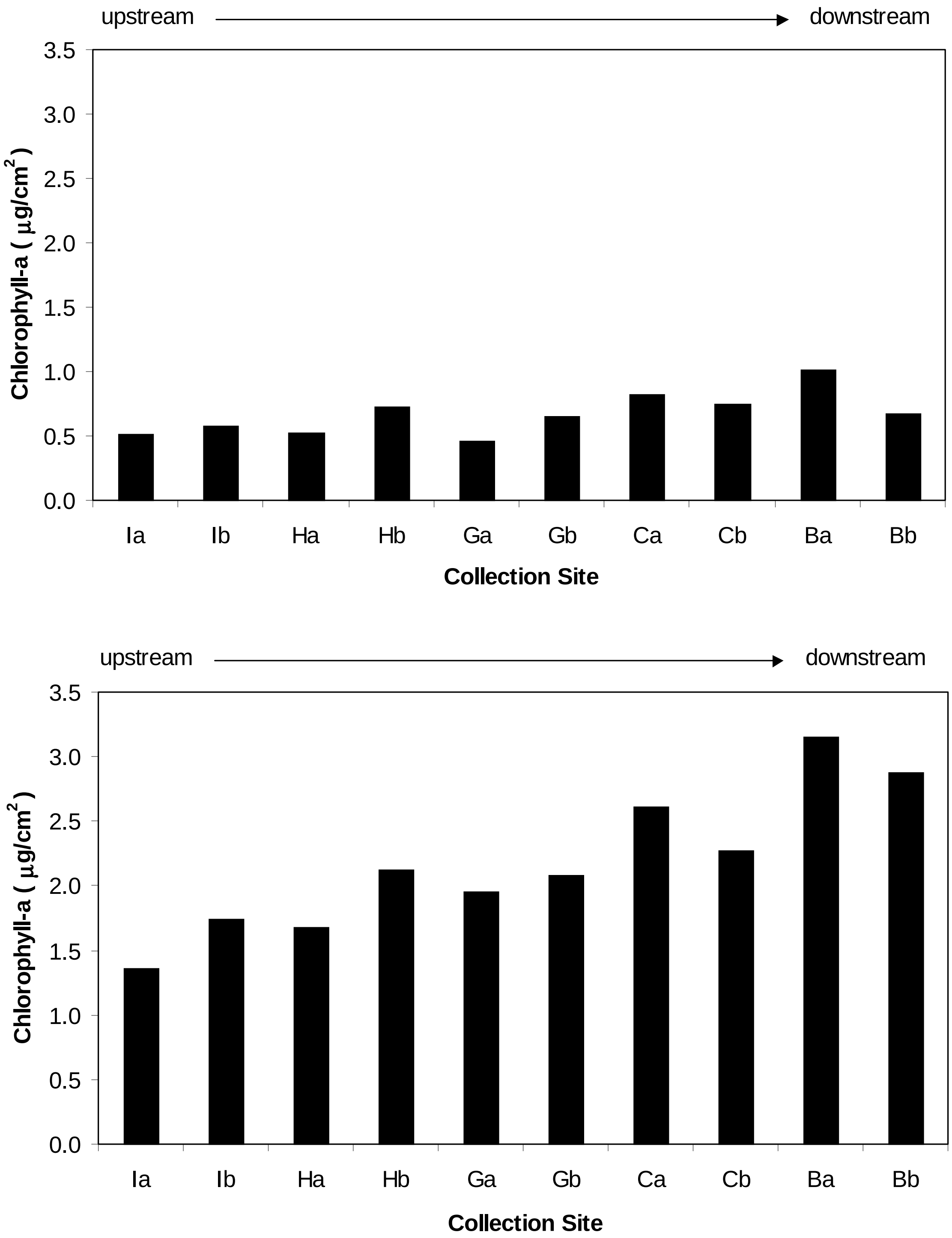


Fig. 47. Variation in algal chlorophyll-a (mean) for pre-treatment and post-treatment conditions at 10 sites in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

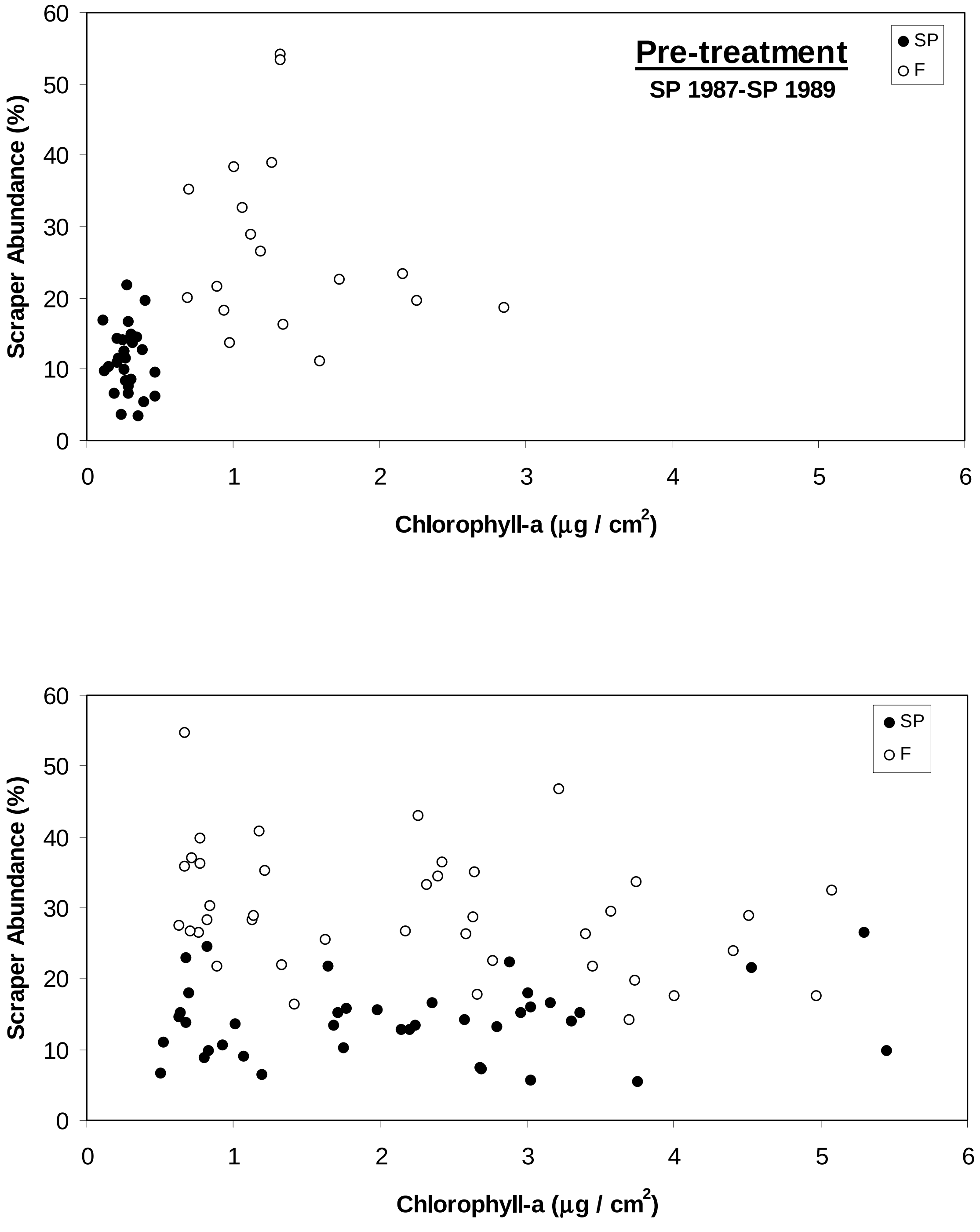


Fig. 48. Scatter plot of scraper relative abundance (%) and chlorophyll-a (mg / cm2) for pre-treatment and post-treatment periods, N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (SP = spring; F = fall)

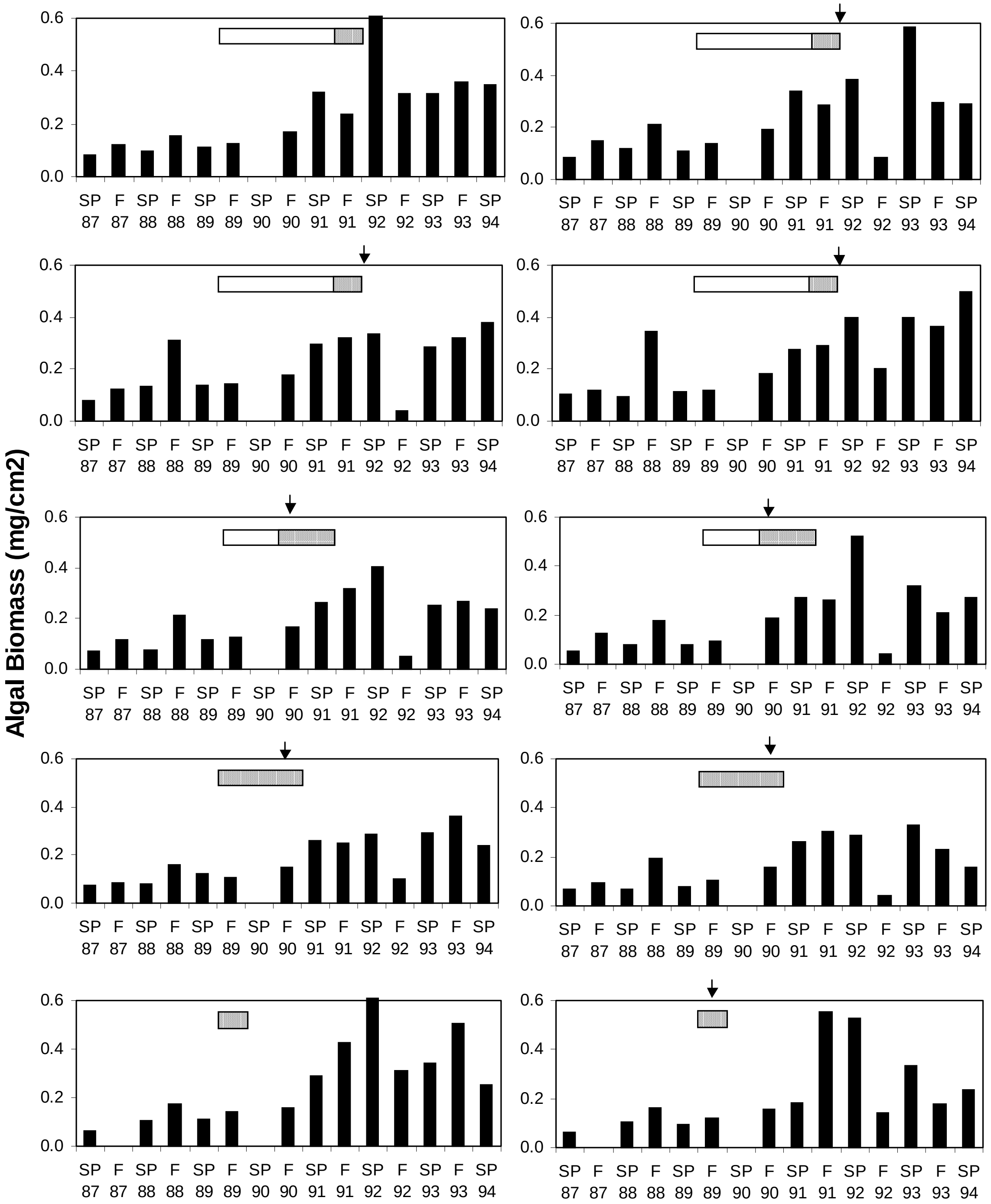


Fig. 49. Variation in algal biomass for entire study period (1987-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (arrow = increased light; top horizontal bar: light bar = upstream logging, dark bar = local logging)

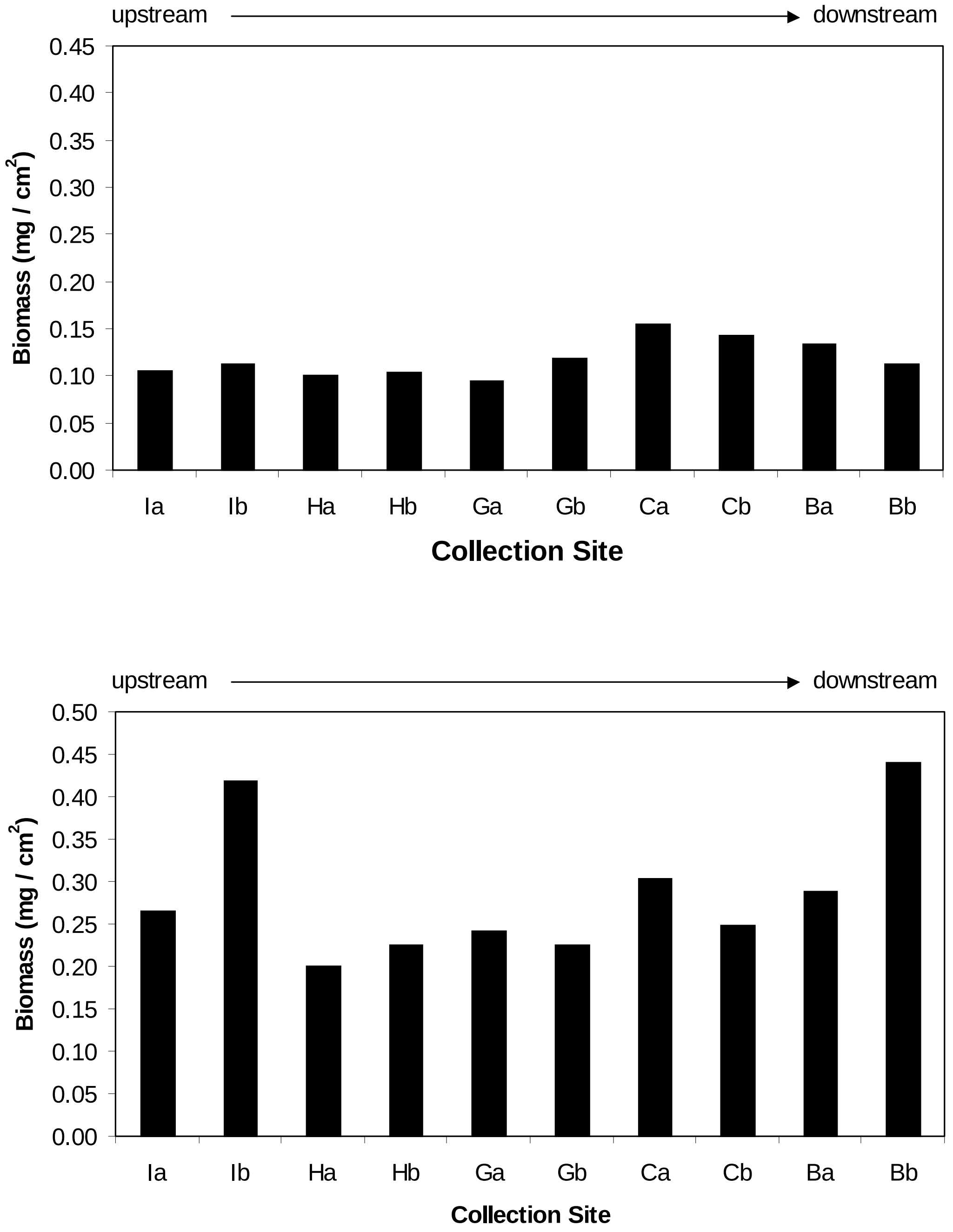


Fig. 50. Variation in algal biomass (mean) for pre-treatment and post-treatment conditions at 10 sites in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA.

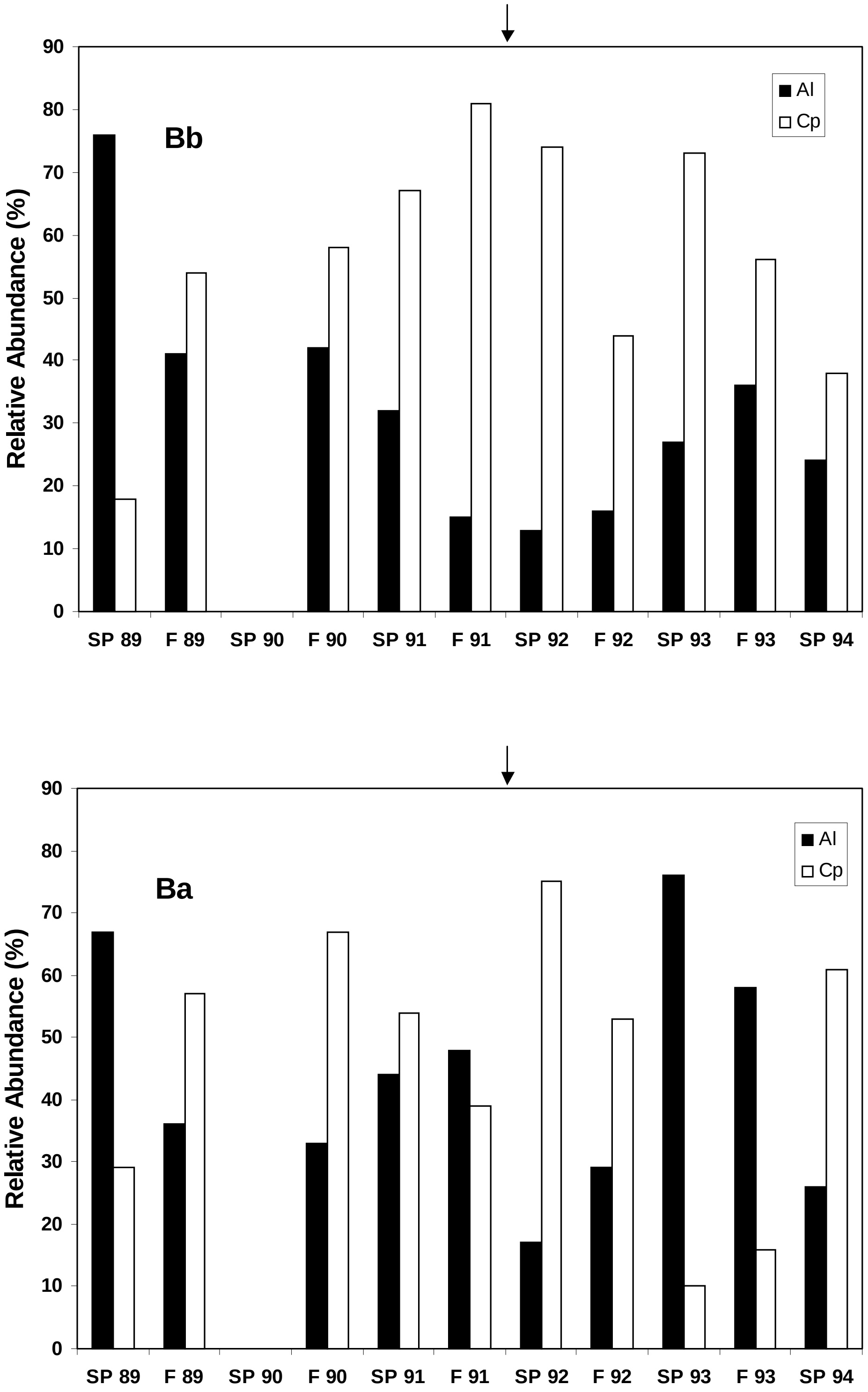


Fig. 51. Variation in relative abundance (%) of Achnanthes lanceolata (Al) and Cocconeis placentula (Cp) at Sites Bb and Ba (1989-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (arrow = increased solar radiation; horizontal bar: light = upstream logging, dark = local logging)

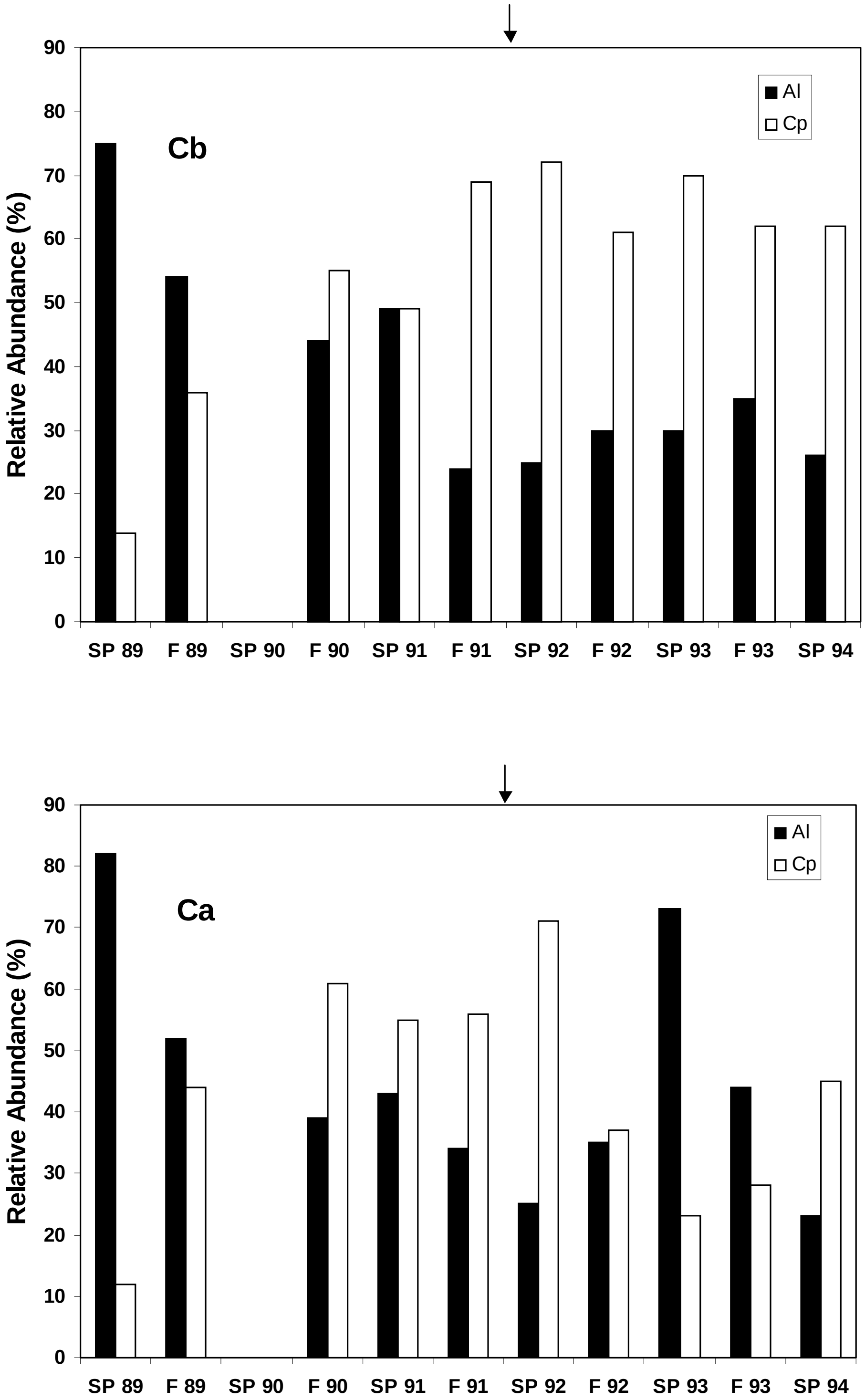


Fig. 52. Variation in relative abundance (%) of Achnanthes lanceolata (Al) and Cocconeis placentula (Cp) at Sites Cb and Ca (1989-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (arrow = increased solar radiation; horizontal bar: light = upstream logging, dark = local logging)

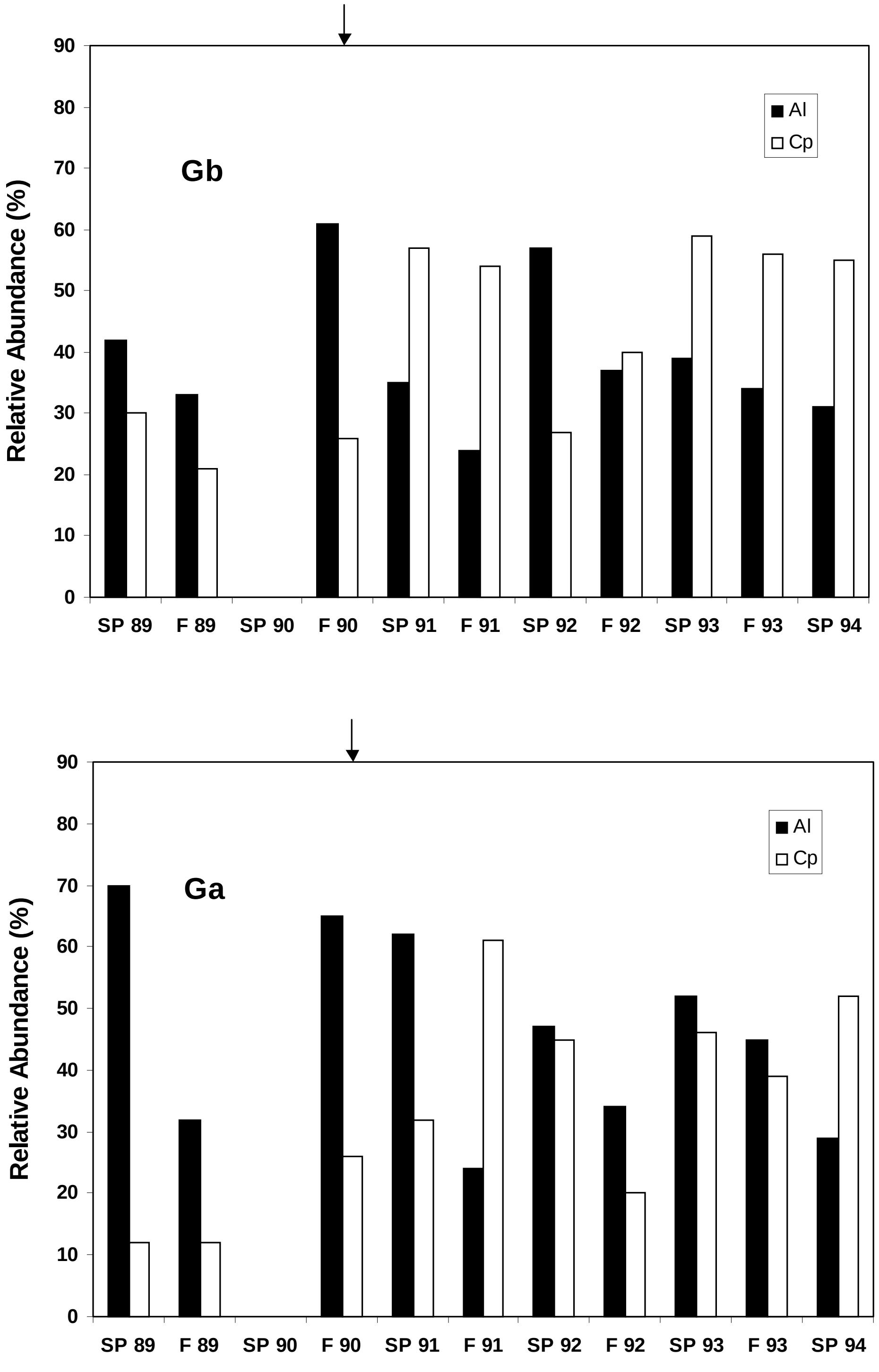


Fig. 53 . Variation in relative abundance (%) of Achnanthes lanceolata (Al) and Cocconeis placentula (Cp) at Sites Gb and Ga (1989-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (arrow = increased solar radiation; horizontal bar: light = upstream logging, dark = local logging)

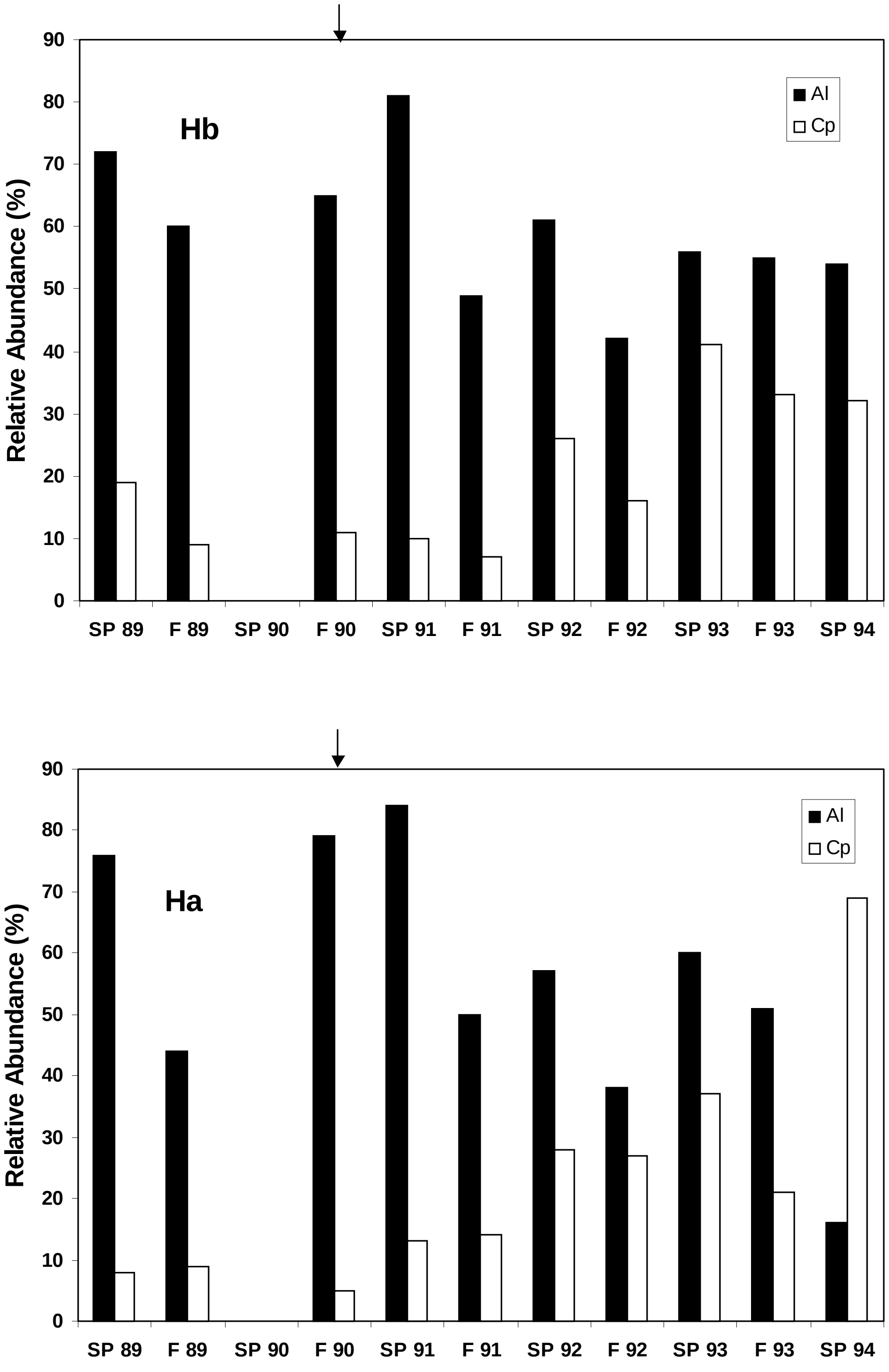


Fig. 54. Variation in relative abundance (%) of Achnanthes lanceolata (Al) and Cocconeis placentula (Cp) at Sites Hb and Ha (1989-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (arrow = increased solar radiation; horizontal bar: dark = local logging)

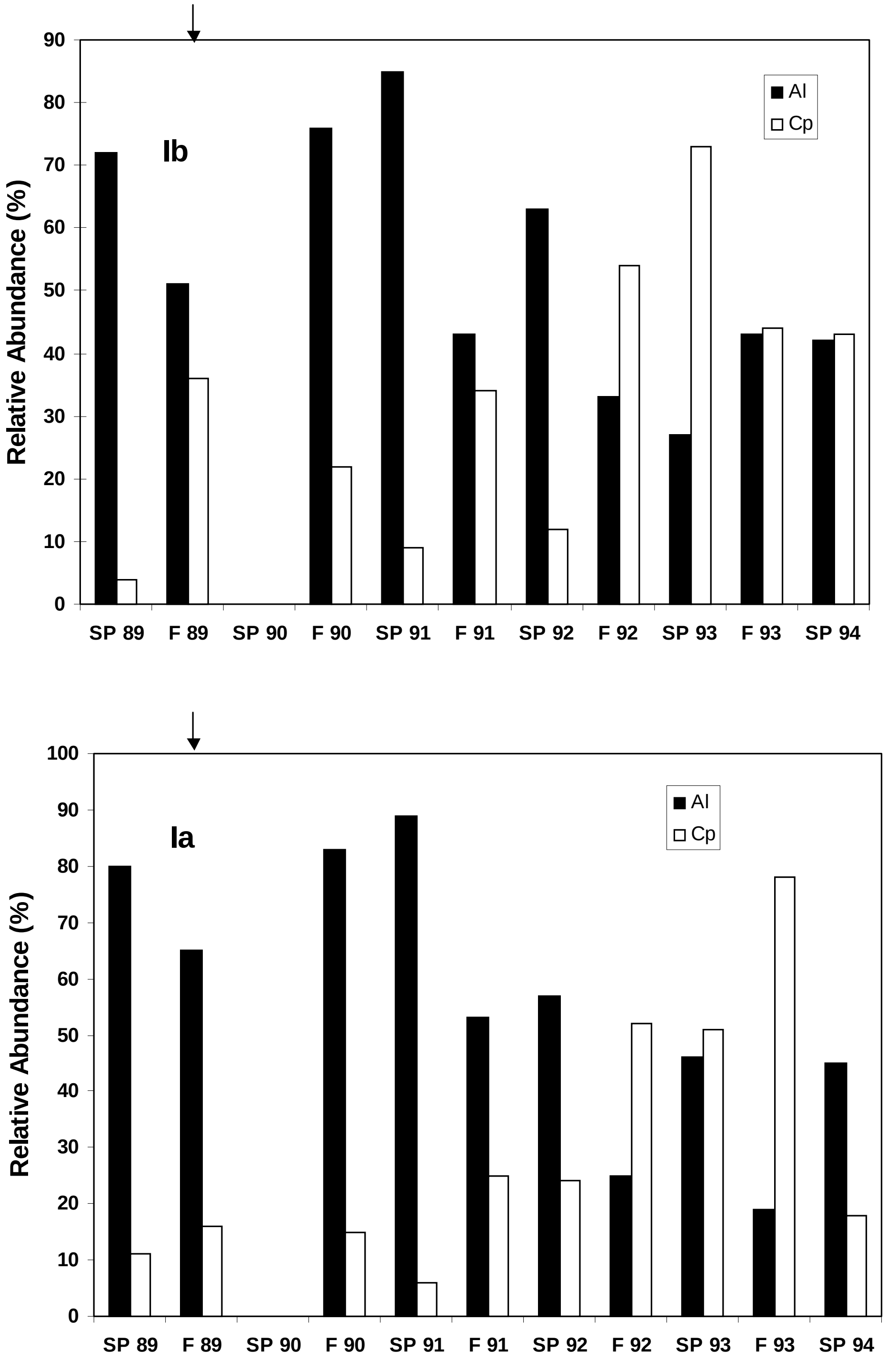


Fig. 55. Variation in relative abundance (%) of Achnanthes lanceolata (Al) and Cocconeis placentula (Cp) at Sites Ib and Ia (1989-1994) in N. Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA. (arrow = increased solar radiation; horizontal bar: dark = local logging)