THE HAYWARD BROOK WATERSHED STUDY: HYDROGEOCHEMISTRY AND RESPONSES TO FOREST OPERATIONS

by

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ABSTRACT

Eight streams in the Hayward Brook Watershed, in southeastern New Brunswick, were monitored for stream discharge, stream chemistry and water quality since 1993. Watersheds ranged in size from less than 200 ha to over 900 ha, containing the full spectrum of stand types found in the Acadian forest. Monitoring was done on a continuous basis for water quality parameters such as pH, conductivity, turbidity, and through weekly grab sampling of streamwater. Grab samples were subsequently analyzed in the laboratory for a comprehensive list of physical and chemical parameters.

Continuous monitoring revealed a strong weather dependency of pH, conductivity in streamwater. Weekly grab sampling essentially missed this weather dependency, because variations in these streamwater parameters occured at the scale of hours. Nevertheless, the weekly grab samples were essential for data quality checking of the continuous records. Often, through problems with maintaining the automatically recording probes, data records where systematically above or below the weekly data, indicating systematic drifts and errors from one probe calibration date to another.

For the adjusted continuous records and from the weekly grab samples, it was found that:

1. *Streams very close to each other, within the identical ecophysical region had significant differences in streamwater chemistry and water quality parameters;*
2. *Harvesting less than 20% of the treatment watersheds had no significant impact on stream discharge or water quality;*

3. *The continuous recording of readily measured water quality parameters could be used to predict other, less easily measured water quality parameters;*
4. *Streamwater chemistry and water quality parameters, as they varied over time within each basin, were strongly affected by weather and season, in accordance with rate of stream discharge.*

In particular, select base cation concentrations in streamwater differed by an order of magnitude between adjacent watersheds, especially in summer during low flow conditions. The main cause for this was seepage from soil substrate types with highly different rates of soil weathering. Other ions generally showed minor differences from basin-to-basin. Also, most ion concentrations were relatively similar during spring snowmelt.

Differences in pre- and post-harvest stream discharge could not be attributed to forest harvesting. Also, differences in water chemistry and water quality parameters could not be attributed to forest harvesting, except for zinc. Zinc levels increased to four times above the analytical detection limit in one stream immediately following the installation of a galvanized steel culvert.

Conductivity and discharge were the most effective parameters in predicting stream chemistry, and proved most effective as predictors of other water quality parameters in the larger watersheds where relative differences between high and low flow conditions were less than in the small-sized watershed.

TABLE OF CONTENTS

Page

Page

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LIST OF TABLES

LIST OF FIGURES

CHAPTER 1

INTRODUCTION

Harvesting in forested watersheds has been proven to have significant and measurable impacts on the chemical and physical properties of the streams draining these watersheds (Bari et al 1996; Beaty 1994; Bell et al 1976; Hetherington 1982). Concerns regarding the detrimental environmental repercussions of these impacts have therefore been growing over the past number of decades. Over the past 100 years, numerous watershed projects have helped to develop some general principles to explain these responses, and even guide management activities, but the information from these projects is, by nature, entirely basin-specific, and therefore only applicable for the particular vegetation/geological/management combinations that exist in that basin.

The development of region specific guidelines is an essential step towards watershed management, especially when water quality issues are important. Today's forest managers need to respect water quality not only from the first order stream level upward, and also at the subcatchment level, and there needs to be information on which region-specific guidelines for streamwater conservation can be based. In particular, knowledge of how much of any given watershed can be harvested before a detrimental impacts on streams and water quality can be noticed is essential. In all of this, it is commonly assumed that small streams in the same general area respond similarly to weather and management actions, and can therefore be managed similarly, but this has not been verified.

How streams respond to weather and forest management actions in terms of stream discharge and water quality is not well understood in detail, partly because such investigations are expensive and thus number only few for any particular region. For example, there have only been a few forest watershed studies in Atlantic Canada to date, and the data derived from these studies have not yet been subject to systematic and comparative evaluations.

This thesis is about the changes in water quality and water quantity for the watersheds of the Hayward Brook Watershed Project (HBWP). This study involves eight basins, all located near each other, and within the same ecoregion. The Project provides extensive pre- and post-harvest stream discharge and water quality data, taken at daily and weekly sampling rates. The following hypotheses are being examined as part of this thesis.

HYPOTHESES

1. *Streams very close to each other, within the identical ecophysical region can have significant differences in streamwater chemistry and water quality parameters;*
2. *Harvesting less than 20% of Acadian forests has no significant impact on stream discharge or water quality;*
3. *The continuous recording of readily measured Water quality parameters can be used to predict other, less easily measured water quality parameters;*
4. *Streamwater chemistry and water quality parameters, as they vary over time within each basin, are strongly affected by weather and season, in accordance with rate of stream discharge.*

OBJECTIVES

There were four main objectives for this project:

1. To quantify differences in streamwater quality with respect to biophysical watershed characteristics.
2. To evaluate any effects that the 1995 harvesting activities may have had on streamwater quality and discharge.
3. To learn how weather and soil conditions affect streamwater quality data over the course of days, weeks, seasons and years.
4. To develop empirical relationships between weekly measurements of stream ion concentrations such as Ca^{2+}, Mg^{2+}, and the corresponding daily averages of the continuous hourly measurements of stream pH, temperature, and conductivity.

The following general steps were taken to complete this project:

1. Review of literature concerning impacts of forest harvesting on stream discharge and water quality.
2. Gather all relevant precipitation (rain and snow) and temperature data from Environment Canada weather stations in the region of interest.
3. Fill in missing weather station data points through regression analysis with neighboring weather stations.
4. Gather all collected probe and streamwater chemistry data for all watersheds.

5. Analyze water quality patterns with respect to watershed characteristics.
6. Evaluate harvesting impacts on water quality and discharge.
7. Perform all necessary statistical analysis to characterize relationships between ion loadings and physical parameters measured by the Hydrolab probes.
8. Calculate ion fluxes with this new information and make comparisons to published techniques.

OUTLINE

Each chapter contributes to the above objectives. Chapter 2 describes the study area and history/scope of the Hayward Brook Watershed study. Chapters 3 and 4 present literature reviews. Chapter 5 summarizes and discusses the data from grab samples collected for the HBWS. Chapter 6 discusses the calibration process concerning the continuously operating water quality probes, and then summarizes and discusses the resulting data. Chapter 7 evaluates extent of harvesting effects on water quality and discharge within the HBWP. Chapter 8 presents a number of regressions between the weekly stream ion concentrations and continuously monitored water quality parameters. The final chapter summarizes the main conclusions and recommendations.

CHAPTER 2

BACKGROUND: THE HAYWARD BROOK WATERSHED STUDY

The Hayward Brook Watershed Study (HBWS) is part of the Fundy Model Forest's ongoing effort to characterize ecosystem level responses to forest operations (Figure 2.1). Started in 1993, the HBWS incorporates six ongoing research projects with the overall objective to investigate physical and biological ecosystem responses to different harvesting techniques (Pomeroy *et al* 1998). The different projects include harvesting impacts on bird, bryophyte, and plant communities, as well as fish habitat and streamwater quality.

The Hayward Brook Watershed Study takes place on both the Hayward and Holmes brook watersheds. Both are located roughly five kilometers south of Petitcodiac New Brunswick, mostly on J.D. Irving freehold, with a small portion being Crown land. Both watersheds drain into the Petitcodiac river, and are covered by a mixed 80 year old Acadian forest (Figure 2.2).

The part of the project that is described within this Thesis deals with the data derived from the streamwater monitoring study. Involved were eight forest streams, each part of the main tributaries of Hayward Brook and Holmes Book.

OBJECTIVES

The objective of this chapter is to provide an overview of the water quality monitoring study of the HBWS.

STREAM MONITORING

Five of the eight streams were monitored continuously at automatic recording water quality stations; all eight streams were visited weekly to obtain weekly streamwater grab samples (Figure 2.1, Table 1.). The automatic stations consisted of a solar powered Valcom Vedas data logger, a Hydrolab water quality sensor, and a stage height transducer. The Hydrolab probes took hourly readings of pH, conductivity, dissolved oxygen, turbidity, and temperature (herein called continuously recorded streamwater quality parameters). The stage height transducer took a measurement of stage (water) height every half-hour, and these measurements were converted into stream discharge rates. All these data were either retrieved from site by Environment Canada personnel, through electronic transmission from the recording devices into a lap top computer, or were transmitted to Environment Canada headquarters in Moncton via GOES (Geostationary Orbiting Environmental Satellite, Pomeroy *et al 1998*).

Grab samples, taken roughly every week or after major rain events by Environment Canada personnel, consisted of three separate samples: a 1L bottle for pH, conductivity, color, turbidity, phosphorus, nitrogen, carbon, and major ions; a 500 ml bottle for extractable metals; and a suspended sediment sample. Grab sampling began in July of 1993, and monitoring with the automatic stations began in March of 1994, with both continuing until 1998. Forest harvesting started in May of 1995, and finished in September of the same year (Pomeroy *et al* 1998, see Table 2.1)

Table 2.1 Data monitored, treatment, and general cover at each watershed the HBWS.

Watershed (ha)	Type of Data	Monitoring Years	Treatment (% of area)	Treatment Date	Cover Type
1 (508)	Stream Discharge	March 1994 to 1998	Selective (15.5)	June 1995	Pine/SW
	Streamwater Chemistry	July 1993 to 1998			
	Hydrolab	March 1994 to 1998			
2 (214)	Streamwater Chemistry	July 1993 to 1998	Selective(17.3)	June 1995	Pine/SW/IHW
3 (268)	Streamwater Chemistry	July 1993 to 1998	no harvest	-	IHW/SW
4 (181)	Stream Discharge	March 1994 to 1998	no harvest	-	SW/IHW
	Streamwater Chemistry	July 1993 to 1998			
	Hydrolab	March 1994 to 1998			
5 (924)	Stream Discharge	March 1994 to 1998	Clearcut(4.8)	May 1995	IHW/SW
	Streamwater Chemistry	July 1993 to 1998			
	Hydrolab	March 1994 to 1998			
6 (356)	Stream Discharge	March 1994 to 1998	Clearcut(11.4)	May 1995	IHW/SW
	Streamwater Chemistry	July 1993 to 1998			
	Hydrolab	March 1994 to 1998			
9 (834)	Stream Discharge	April 1995 to 1998	Clearcut(10.4)/ Selective(22)	June 1995	Pine/IHW/SW
	Streamwater Chemistry	May 1994 to 1998			
	Hydrolab	April 1995 to 1998			
10 (573)	Streamwater Chemistry	June 1994 to 1998	Selective(20.9)	June 1995	Pine/SW/IHW

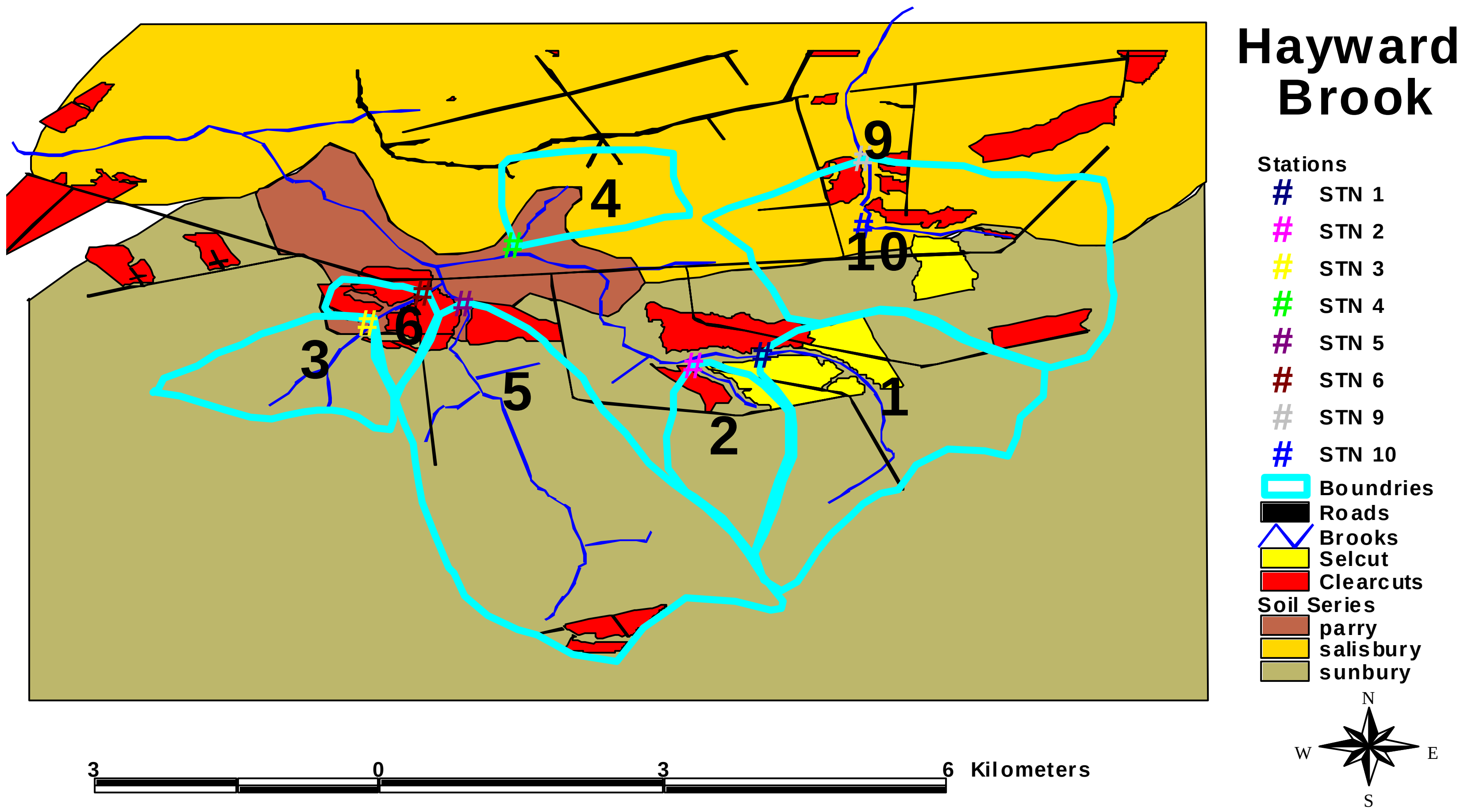

Figure 2.1 Study area, Hayward and Holmes Brooks

PHYSIOGRAPHIC FEATURES

Geology

The HBWS lies in the Caledonian Highlands geomorphologic region. Holmstrom (1986) identifies the bedrock of the area as Pennsylvanian era, belonging to the Petitcodiac Group consisting of gray and red sandstone, shales and quartz-pebble conglomerates. More specifically, the HBWS is located on three different forest soil units (FSU), the Salisbury, Parry, and Sunbury associations (see Figure 2.1). Typically the Salisbury and Parry soils have similar properties, derived from non to slightly calcareous red polymictic conglomerates, feldspathic to lithic sandstones and mudstones (Colpits *et al.* 1995). The two have developed on well to imperfectly drained lodgement till and are separated based on the texture of the parent material. The Salisbury contains more of the mudstones in its parent material, giving it a finer subsurface texture, where the Parry soils have a higher proportion of the sandstones, and more coarse fragments. The parent material for the Sunbury soils consists mainly of non-metamorphosed grey lithic and feldspathic sandstones, which lack calcium. It is high in coarse fragments and thus rapidly permeable.

Vegetation

The vegetation is typical of an eighty-year-old Acadian mixedwood forest, with intolerant hardwoods dominating to the south, white pine to the east, and fir-spruce and pine to the north (see Figure 2.2 and Table 2.1). During the summer of 1995, limited selection and clearcutting occurred in some of the watersheds leaving a thirty to sixty meter buffer (Figure 2.1). Thirty percent of the basal area was harvested in the buffer associated with watershed 1, according to Dr. Krause's Buffer Zone Management case study (Krause 1996).

Topography

With the HBWS lying in the Caledonian highlands, the terrain is gently rolling, with rounded moraines reaching 250 meters above sea level (a.s.l.). Many of the streams are cut deep into the land, with slopes adjacent to them approaching 100%. All streams in this study converge outside of the study area, and converge with the Petitcodiac river at an elevation of 50 meters a.s.l. Due to the hilly terrain, the study area contains very little wetland, except for some beaver activity creating dams in watershed 4.

Climate

The HBWS lies in the southern portion of climate region 2, which is characterized by having warmer than average annual temperatures, between 5.2 and 5.6 ^{O}C for the study period, and 1600 to 1800 growing degree-days. Precipitation averaged roughly 1200 mm per year, with roughly 400 mm occurring during the May to September growing season (Figure 2.3). All weather station data required for analysis (see below) was obtained from Environment Canada.

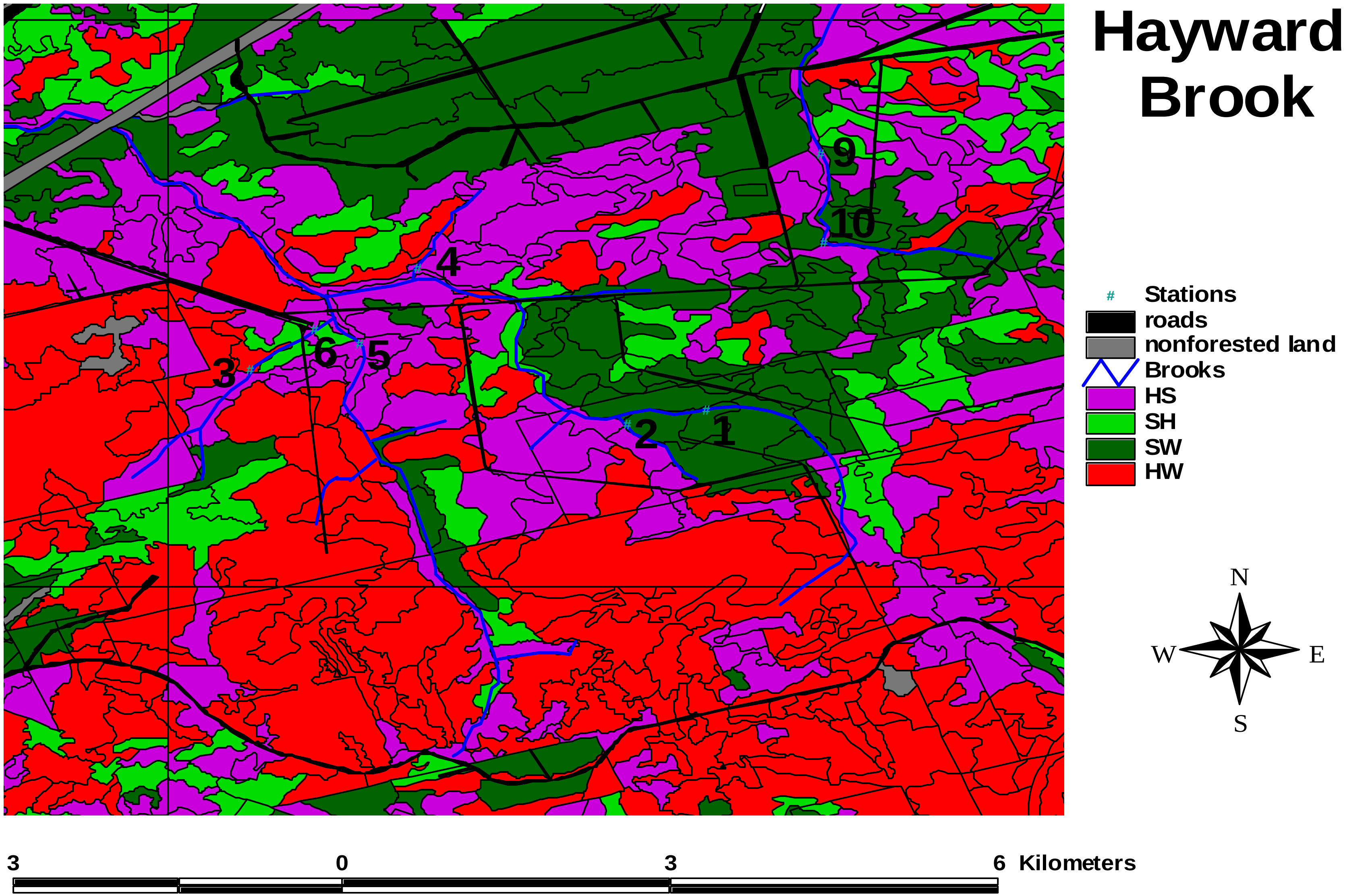

Figure 2.2 Map showing study area and general vegetation types.

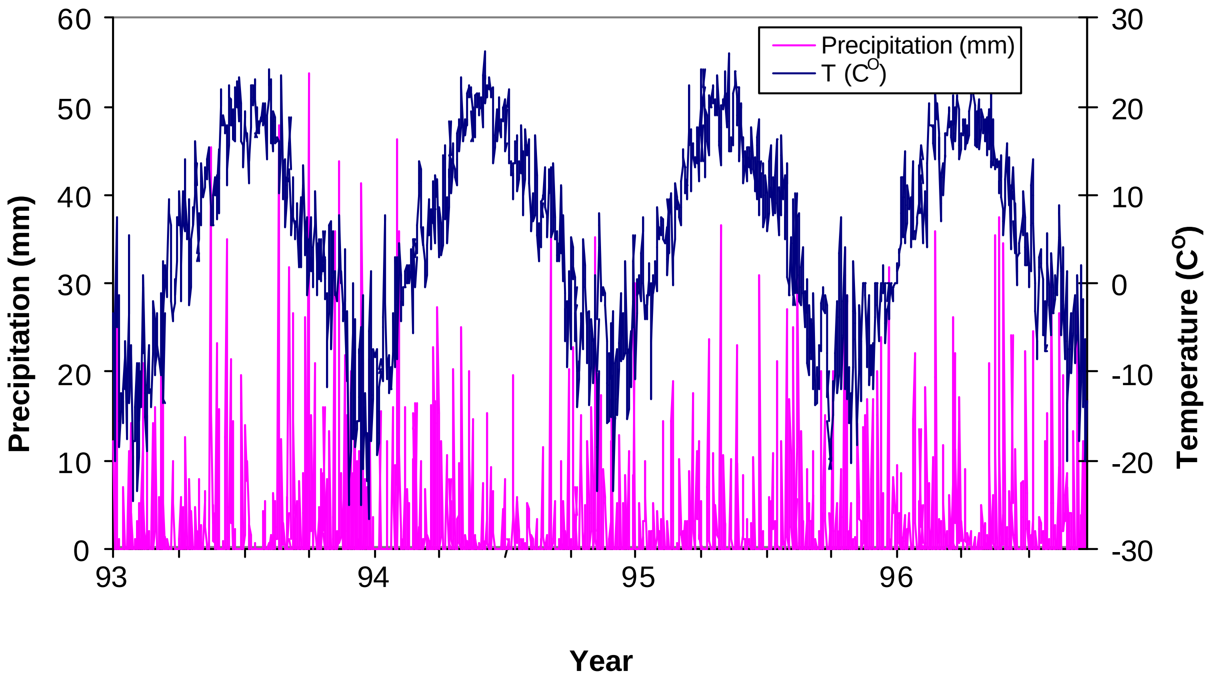

Figure 2.3 Precipitation and temperature at the Parkindale station near Hayward Brook.

CHAPTER 3

EFFECTS OF FOREST HARVESTING ON BASIN-WIDE WATER YIELD: A REVIEW OF LITERATURE

INTRODUCTION

The use of watersheds as geographical boundaries for forest management purposes has been popular for many years. For example, two experimental watersheds were established in Czechoslovakia in 1867, for the purpose of investigating the importance of forests in moderating surface runoff. To settle debate over the importance of forests with regards to floods, and peak flow volumes, paired watersheds were used in Switzerland, in 1902. As concern about dwindling timber supply and soil erosion grew, National Forests were created in the United States, and the Wagon Wheel Gap project started in 1909 (Swank and Johnson 1994). This was the first of many subsequent efforts made to quantify streamflow before and after harvesting. When the worst flood on record occurred in the Mississippi delta in 1927, pressure on the government to investigate the impacts of forest management on water resources increased. This initiated a boom in watershed research across the United States and elsewhere, of which the Coweeta Hydrologic Laboratory, the Hubbard Brook Experimental Forest, the Jonkershoek Research Station in South Africa, and the Nashwaak Experimental Watershed in New Brunswick, Canada, are notable examples. As a result of much of this work, it was found that post-harvest streamflow tends to be larger than pre-harvest streamflow. Clarke, (1994) therefore examined the role of mathematical models in finding optimal harvesting scenarios for the enhancement of streamflow. As well, Anderson was interested in learning how to increase snow capture and hence water yield through varying forest management practices (Anderson 1956; 1960; 1962; 1963; 1967; Anderson and Gleason 1960; Anderson and Hobba 1959).

Regardless of the focus of forest watershed research, related projects are used to effectively characterize the various hydrological properties of the watersheds, as they differ by region, geological substrate, topography, climate, forest cover, and responses to various surface (forest) treatments. As such, most watershed projects are very site specific, and lack a process that permits extrapolation of results to other watersheds (Clayton and Kennedy 1985). While the impacts of specific watershed management on water yield and quality tend to be well documented, watershed-specific treatments have not been oriented towards testing refined hypotheses. Thus, portable concepts about process are lacking somewhat.

Many literature reviews of watershed studies are readily available (Bosch and Hewlett 1982; Hewlett et al. 1970; Hibbert 1967; Hornbeck et al. 1993; MacGregor 1994; Bell et al. 1974). Some in fact concentrate on issues of water yield and water quality in relation to portion or percentage of the watershed cut, which is the topic of this review. There are fewer examples, where the importance of the spatial arrangement of the cuts, their size, and their positional relationship within the watershed are addressed.

Today's realities present forest management with a new challenge that was not encountered in many parts of Canada in preceding decades, namely the increasing fragmentation of the forested land into a patchwork (mosaic) of cut, and uncut, planted and not planted regenerating forest stands. There have also been increasing pressures from the public to limit opening sizes, and to preserve old-growth forest, wildlife habitat, rare forest types, and forest streams as much as possible. This now provides another impetus regarding the refining of forest hydrology at a scale that can take into account small cut sized in relation to topography, and position within a watershed. The objectives for this chapter are to:

1. Review current literature regarding the effects of forest harvesting on discharge;
2. In doing so, examine the effects of cutting (complete, partial, thinning) on the watershed wide water input/output balance for the purpose of modelling;
3. Examine a number of processes that affect the input/output balance, especially evapotranspiration and snow capture.

LITERATURE REVIEWED

Hibbert (1967) reviewed 39 catchment experiments and came to three main conclusions:

1. Reduction of forest cover increases water yield
2. Establishment of forest cover on sparsely vegetated land decreases water yield.
3. Response to treatment is highly variable and, for the most part, unpredictable.

With the addition of 55 new experiments, Bosch and Hewlett (1982) supported the first two conclusions made by Hibbert, but refined the third conclusion, by stating that there are observable differences in response to harvesting, by vegetation classes.

Through regression analysis, coniferous and eucalyptus forests were found to increase annual water yield by roughly 40 mm for every 10% reduction in cover (Figure 3.1). This number was roughly 25 mm for hardwoods, and roughly 10 mm for shrub land. Bosch and Hewlett admitted that the confidence levels that are associated with these numbers are low however Evans and Patric (1983) cited similar numbers.

The variability of response within these forest types and associated watersheds was extensive, with 100% removal of hardwood inducing both a 31 mm increase (Ursic 1970), and a 414 mm increase (Swank and Miner 1968) in streamflow in the first year following harvest. A 100% removal of softwood overstory induced a 226 mm (Rogerson 1979) and an 840 mm increase in streamflow (Pace and Fogel 1968).

In another review, Sahin and Hall (1994) used a fuzzy linear regression technique to analyze the results of 145 experimental catchments, and found that a 10% reduction of overstory in coniferous watersheds tends to increase annual streamflow rates by 20 to 25 mm. For deciduous forest watersheds, the numbers amount to 17 to 19 mm increases in annual streamflow.

The review by MacGregor (1994) summarizes some of the same projects, generates the same conclusions and generalizations, but does not address the issue of cut size, or the contribution of cut size to streamflow. Maximum increases in mean annual streamflow for the first year following harvesting, however, were found to be 4.5 mm for every percent removal of the cover.

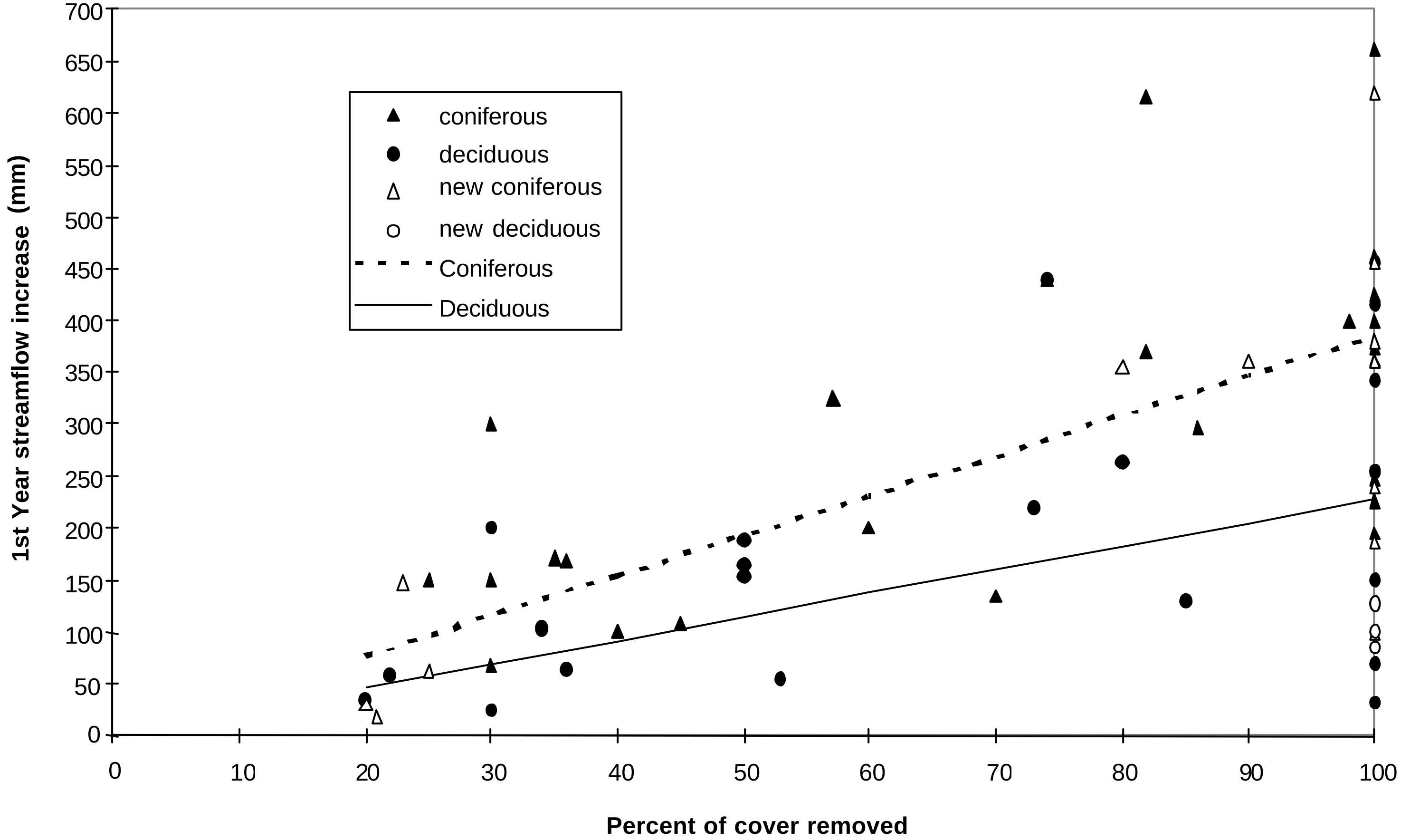

Figure 3.1 First year streamflow increases (mm) following different degrees of harvesting with regression lines from Bosch and Hewlett (1982) and new data points (open symbols).

Bosch and Hewlett (1982) reported that any reduction in forest cover of less than 20% was not detectable by measuring streamflow. Schroder (1996), who reviewed many of the same watersheds, organized them by climatic regions, and found that as little as 15% removal of vegetation in mountainous regions can result in an increase in streamflow. In contrast, a 50% removal was required to yield demonstrable changes in streamflow in the central plains of the USA.

Here are further citations:

1. A 21% removal of softwood cover in the Marmot creek watershed induced a 17 mm increase in annual streamflow (Swanson et al. 1986).
2. Deforesting 50% of two mountainous watersheds in West Virginia resulted in a yearly increase of 282 and 300 mm in total streamflow (Patric and Reinhart 1971).
3. Megahan et al. (1995) reported the results of a paired catchment study in Idaho, where 23% of one watershed was clearcut with heli-logging and burned; there was no significant increase in streamflow.
4. When roughly the same portion of a larger lodgepole pine watershed was harvested, it resulted in a 147 mm increase in annual water yield (Burton 1997).
5. After clearcutting and mechanical site preparation of a 28.6 ha watershed in the central Appalachians, Koehenderfer and Helvey (1989) reported a 99 mm, 71 mm, and 31 mm increase in streamflow for the first three years after harvest.
6. Croft and Monninger (1953) reported a 102 mm increase in water available for streamflow after removing dominant aspen and leaving the understory, and another 100+ mm after the removal of the understory.

7. Johnston (1970) used a neutron meter to measure soil moisture in the upper 9 feet of the soil profile in an aspen stand, and found clearcutting to increase soil moisture by about 152 mm per year.
8. Converting an oak forest to grassland resulted in a mean annual increase in streamflow over six years of 114 mm (Lewis 1968).
9. In the Carnation Creek study, one watershed was clearcut on 90% of its area, with a resultant 360 mm increase; another watershed was evenly cut over 7 years on 40% of its area with no significant increase (Hetherington 1982).
10. When 80% of the Jamieson Creek watershed in the foothills of the Rocky Mountains, British Columbia, was harvested, the resultant increase in streamflow was 356 mm.
11. Fahey (1994) reported a 60-80% increase in water yield following clearcutting for hardwood watersheds in New Zealand, with recovery in 6-8 years; a 30 to 50% decrease in streamflow would be expected after converting grasslands to plantations.
12. Reforesting grassland with conifers in central New York reduced November to April peak flows by 40%, while reducing the overall annual runoff by 26% (Ayer 1968).
13. Clearing 50% of a watershed in Japan with 50 m wide contour strips resulted in a 21 to 35% increase in summer streamflow.
14. Insect infestation and fire also can have similar effects on the hydrological cycle (Hillman 1971). For example, Love (1955) reported that after insects killed 30% of the forested area of the White River Basin in Colorado, annual water level increased by 22%.
15. Beaty (1994) reported a 60% increase in streamflow following a fire that burned all of a watershed in western Ontario.

16. In the Nashwaak Experimental Watershed Study, 100% clearcutting produced a post harvest increase of less than 10% over twelve post harvest years (Jewett et al. 1996).

An explanation of the variation of streamflow responses to clearcutting within vegetation classes has been attempted using differences in the mean annual precipitation (MAP). Regardless of the vegetation class, high precipitation areas usually induce rapid regrowth, and thus rapid return of streamflow to its original state, with the opposite applying to low MAP areas. First post-harvest year effects, however, are usually pronounced in high rainfall areas. For example, Rothacher (1970) reported an increase in water yield of 457 mm after clearcutting a 237 acre watershed in a high precipitation region of the Oregon Cascades.

In "Opportunities to increase water yield in the southwest by vegetation management", Hibbert (1981) refers to the Rich and Thompsons (1974) method for estimating post-treatment streamflow increases as a function of pre-treatment streamflow (Figure 3.2).

Evapotranspiration

Evapotranspiration, defined as evaporation from all water, soil, snow, ice, vegetation and other surfaces, plus transpiration (Chow 1964), is generally recognized as the most pronounced direct way by which watershed hydrology is changed (Biswell 1969; Croft and Hoover 1951; Goodell 1965; McGinnies et al. 1963; Rich 1952; Sinclair 1960; Woods 1966; Zon 1912; DeByle et al. 1969; Rosenzweig 1969). Grelle et al. (1997),

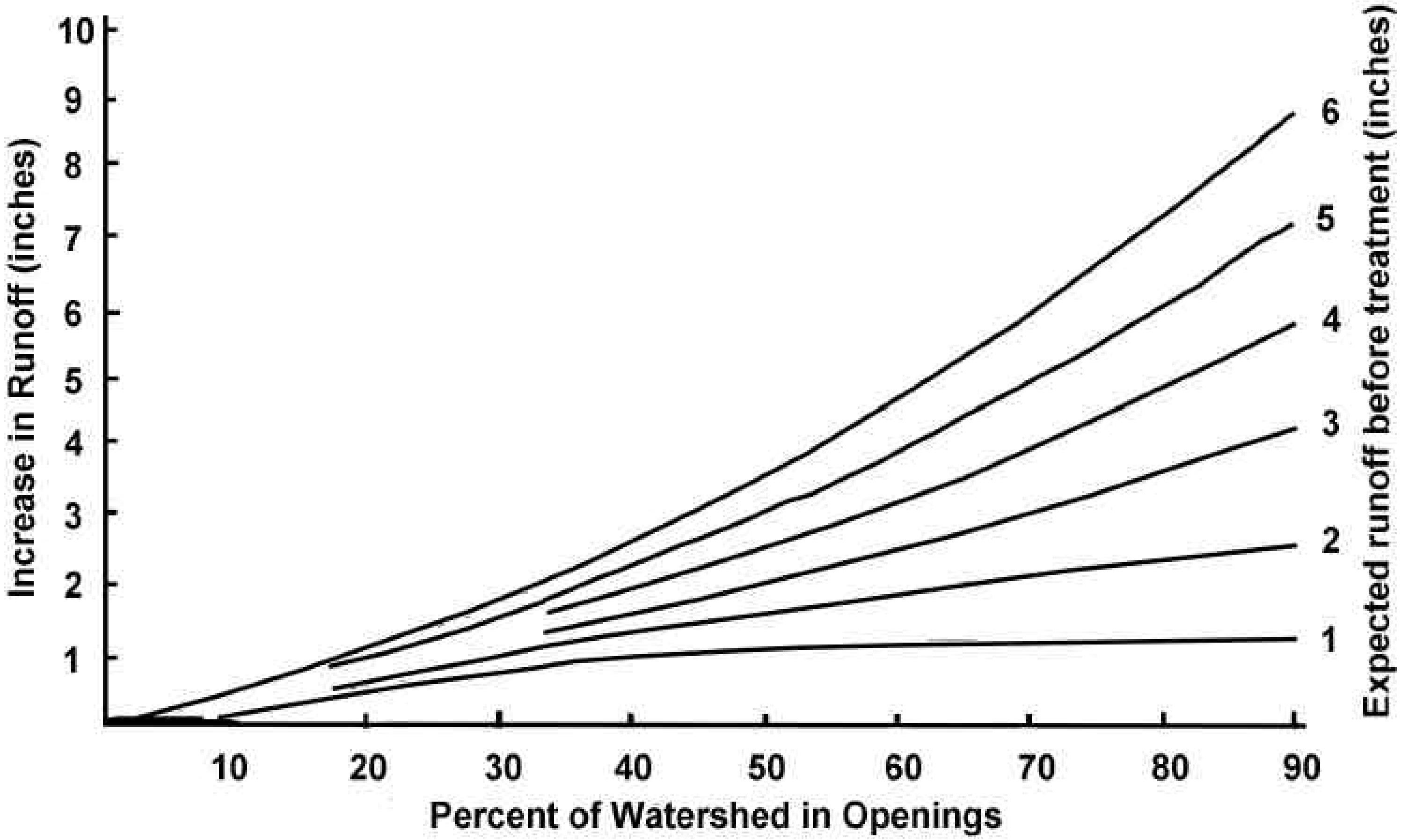

Figure 3.2 Expected increase in runoff (streamflow) as it relates to pre-treatment runoff verses the percentage of the watershed in openings (Rich and Thompson 1974).

therefore, evaluated the three major components of the evapotranspiration budget for a boreal forest:

1. Evaporation from the forest floor (56 mm),
2. Interception evaporation (74 mm), and
3. Transpiration (243 mm).

Baier (1967) estimated that evapotranspiration is responsible for returning 70% of annual precipitation to the atmosphere, and suggested that evapotranspiration is determined primarily by soil moisture availability, meteorological factors, and plant physiological characteristics. Through modeling, Federer and Lash (1978) estimated that a four week change in the timing of leaf development would cause a 10 to 60 mm change in simulated streamflow, and that a 20% variation in daily transpiration would result in 120 mm variation in simulated streamflow. Johnston et al. (1969) compared water usage by different vegetation types at different ages. Aspen sprouts, for example, used 114 mm less water than mature aspen, and oak sprouts used 30 mm less than mature oak.

Transpiration from understory vegetation is usually not differentiated from overstory vegetation. However Roberts and Rosier (1994) estimated that transpiration from understory vegetation in a beach and ash stand at a chalk site in southern Britain contributed to 45% of the annual water loss from the watershed. Johnson and Kovner (1956) reported an average increase in annual streamflow of about 51 mm for six years after removing the laurel and rhododendron understory from a hardwood stand in the southern Appalachian Mountains. Croft and Monninger (1953) reported a 100+ mm increase in water available for streamflow after removing the understory vegetation from a clearcut aspen stand.

Actual evapotranspiration (AET) is often less than potential evapotranspiration (PET), as soil water levels are often limiting for much of the year. The relationship between evapotranspiration and soil water, however, is still not well understood. There is some evidence that evapotranspiration rates are independent of soil moisture until the water levels drop below the permanent wilting point (PWP) (Vihmeyer 1956; Van Bavel 1960; Lowry 1959). Other studies have shown that AET typically remains at 90% of PET until about 65% of the total available water has left the soil, after which AET decreases steadily (Pierce 1958; Gardner 1960; Shaw 1968). The third predominant hypothesis is that of a linear relationship between AET and soil moisture, with AET decreasing with decreasing soil moisture until the PWP is reached (Thornthwaite and Mather 1955).

Calculating evapotranspiration from climatic data is complicated and has been approached in many different ways (Stricker 1982). Mihan (1986) cites seven methods for calculating evapotranspiration, all of which vary considerably in their data requirements and have had numerous modifications made since their initial conception. Yin and Brook (1992) compared some of the temperature-based PET calculations in a swamp watershed where they state that AET should be equal to PET due to the lack of a water deficiency at any time of the year. When AET was estimated with a steady state water balance model, it was found to be most highly correlated with the PET estimate calculated by the Thornthwaite equation (r^2=0.817), followed by the Blaney-Criddle equation (r^2=0.781), followed by the method used by Holdridge (r^2=0.768). Another study took the Penman equation as correct, and compared this equation with others by means of regression analysis to provide a means for correcting the other equations (Mohan 1991).

Further support for the Penman equation was found in a report by Essery and Wilcock (1990) where PET estimates from the Penman equation were compared with the evaporative losses from irrigated grass lysimeters, from open water tanks based on rainfall data, from streamflow data and from groundwater data. Using 12 years of data showed the Penman equation to be the most accurate. Other comparisons of calculations and estimations are also available (Stricker 1982). Abbaspour (1991) compared eight different methods of estimating daily evapotranspiration, and compared them to measurements of AET from the Peace River region of British Columbia, Canada.

Some alternative methods for quantifying evapotranspiration also exist. For example, Claassen and Halm (1996) estimated evapotranspiration in a mountainous watershed from chloride ion concentrations in stream baseflow. DeByle et al. (1969) consided measured soil moisture depletion plus summer precipitation to be an estimate of actual evapotranspiration, and showed how AET varied from 130 mm for grasslands, to 613 mm for an aspen stand. Walker and Brunel (1990) examined evapotranspiration through daily variations of isotopic compositions in foliage.

Attempts have been made by researchers to quantify the spatial variations of evapotranspiration across a watershed (Famiglietti and Wood 1993; Flerchinger et al. 1996; Sabur 1991). For example, Dunn and Mackay (1995) illustrated how land-use changes may have significant effects on the hydrology of lowland areas, but not upland areas. Moelders and Raabe (1995) discussed that simply increasing the temporal resolution of hydro-meteorological modeling is insufficient for adequate hydrological assessments at the catchment scale, because meteorological conditions, are also affected spatially, especially by topography and related flow accumulation patterns. Ambrose and Najjar (1982) found that evapotranspiration on a mountainous watershed could be calculated accurately by using the Brochet and Gerbier formula, which is derived from the Penman equation.

Fog

Most research does not account for the potential effects fog has on the water budget in watersheds. Fog cannot only add water to a watershed through fog drip from foliage, but it can also essentially halt evapotranspiration processes. Yin and Arp (1994) estimated fog water input to be as high as 106 mm for a coastal watershed in Nova Scotia, and Ingraham and Matthews (1995) noted a similar importance of fog water inputs for the coastal watersheds in California.

Snow Capture

Snow capture in a forested watershed is governed by a number of factors, the major ones being meteorological conditions (wind, snow density and quantity), and type and condition of the vegetation. Schmidt and Gluns (1991) examined three conifer species: pine (*Pinus contorta* var. *latifolia* Engelm.), fir (*Abies lasiocarpa* (Hook.) Nutt.) and spruce (*Picea engelmannii* Parry), to characterize any differences in snow catch efficiency by foliage type. These authors found that spruce branches intercepted 43% of the total snow water equivalent; pine 38% and fir 37%. Satterlund and Haupt (1967) showed snowfall capture to follow a sigmoidal curve given by:

$$S = S_O/[1+e^{-k(P-Po)}]$$

where S_O is the interception storage capacity, P is the total storm precipitation in inches, and Po is the amount of snow accumulated at the time of most rapid accumulation.

For Douglas fir and western white pine, roughly 5% of intercepted snow is lost through evaporation, while the rest falls to the ground in liquid or solid form (Satterlund and Haupt 1970). Guttenberger (1994) measured evaporation of intercepted snow, and found that this made up 60% of the evapotranspiration budget for a spruce stand. Lundberg and Halldin (1994) found a snow-evaporative loss of 3.3 mm/day from 6 m spruce trees in Sweden. These authors also concluded that one needs to measure relative humidity, aerodynamic resistance, wind speed, and total intercepted snow mass accurately in order to model the rate of evaporation from intercepted snow in an accurate manner.

The difference in snow capture between harvested and non-harvested watersheds is a primary cause of increased spring peak flow levels. Even in unharvested watersheds, spring peak flow has been calculated to contribute up to 68% of the total year's flow for a watershed in Utah (Glasser 1969), and 80% for a watershed in southern British Columbia (Shiau 1975). In a study in Lodgepole pine in Wyoming (Berndt 1965) with three opening sizes (5, 10, and 20 acres) located on four aspects (N, S, E, W), snowpack was always greater (405 mm average) in the openings than in the undisturbed stand, with no significant differences between opening sizes. Aspect had a small impact on snowpack, with the eastern aspect having the most snow, followed by the southern, western, and northern aspects, with about 100 mm of snow water equivalent between the lowest and highest snowpack accumulations. Snow in the uncut areas, however, persisted for 10 days longer than in the openings.

In a study of snow catch in circular openings of nine opening sizes varying from 0.25 to 6 tree heights (H) in diameter in Alberta (Golding and Swanson 1978), 2H and 3H opening sizes accumulated the most snow, followed by 1H and 0.75H openings. Ablation rates were smallest in the 0.75H and 1H diameter openings, and steadily increased up to the largest opening sizes.

Evaporation from snowpack is counteracted by increased capture of snow in openings (Meng et al. 1995). Stegman (1996) states that geographic location, latitude, orientation to prevailing wind, and aspect are the primary factors effecting snowpack evaporation, but that a 5H by 1H sized opening maximized water yield, and minimized evaporation. In a Lodgepole pine study in Wyoming (Gary 1974), an opening 1H in width, and 5H in length perpendicular to the prevailing wind did not induce a watershed-wide increase in snowpack. However, within the clearing itself, snowpack was greater at peak times than in adjacent stands, with the upwind adjacent forest capturing slightly more snow than the downwind one.

When two Colorado watersheds of the same size (40 ha) had 40% of their areas harvested, one by 12 1.2 ha circular cuts, and one by a single continuous clearcut, each induced significant increases in streamflow comparable to each other (Troendle and King 1987). This was probably due to the large sizes of the circular cuts (roughly 125 m in diameter). Troendle and King (1985) also noted a 9% increase in snow water equivalent over the watershed when 50% of the area was harvested by strip cutting. Due to the apparent increase in snowmelt rate, peak streamflows increased by 20%, and were advanced by seven days over the average of the previous years.

In a study in Lodgepole pine by Wilm and Dunford (1948), twenty 5-acre plots were arranged in a randomized pattern, with sixteen harvested with selective cutting. Snow disappeared at roughly the same time, however accumulations (262 mm in unharvested areas versus 343 mm in harvested areas) and associated melt rates were higher in harvested blocks. Similar results have been noted for ponderosa pine (Berndt and Swank 1970), red pine (Hansen 1969), black spruce (Bay 1958), and mixed conifers (Anderson 1967).

The presence of slash has been noted to affect snowmelt. Anderson and Gleason (1960) measured snow depth on May 4, 1959, and found the average water equivalent to be 107 mm in areas where slash had been burned; areas containing slash contained an average water equivalent of 23 mm. Obviously, this difference is due to radiation capture of exposed slash, which helps to warm and melt the snow that is in immediate contact with the exposed slash.

It is important to note that clearcutting is only one way to modify the snow catch efficiency of watersheds. Snowpack after thinning a stand of lodgepole pine increased 30% over pre-thinned values (Gary and Watkins 1985). Hansen (1969) reported on a study where portions of a red pine stand were thinned to 30, 60, 90, 120, and 150 square feet per acre. The snowpack after snow events increased roughly 2% for every 10 square foot decrease in basal area within the 180 to 60 square foot range. Lesch (1977) reported on two separate thinnings of Radiata pine in Australia. Here, removing roughly 1/3, and 2/3 of the stems resulted in 19 and 99 mm increased annual streamflow. Afforesting of grasslands in Malawi with pine induced significant changes in minimum low flows, but not in maximum peak flows (Mwendera 1994).

Roughly 2/3 of the 20% increase in annual streamflow from an herbicided grassland watershed in Wyoming came from increased snowmelt (Sturges 1994).

Models

Mathematical models are often used to simulate the hydrological behavior of forested watersheds before and after harvesting, and in relation to other factors such as climate change, acid precipitation, expected water yield, flood forecasting, etc. MacGregor (1994) provides citations and abstracts for 17 different hydrological models, some designed for year-round predictions of streamflow (Bernath et al. 1982; Bernier and Hewlett 1982; Croley 1982; Dickinson 1982; Leaf and Alexander 1975; Tsykin et al. 1982), with others predicting other components of watershed hydrology as well. Arp and Yin (1992,1993) cited 22 models that deal with the flow of water and/or the flow of energy (heat) through forest soils and forest watersheds. These authors then proceeded to formulated a new series of forest hydrological models that emphasize:

1. Integration of heat and water flow based on mass and energy balances;
2. Ready yet reliable application to forest watersheds for which only limited data for model initialization is available;
3. Portability of model calibrations from one watershed to another, across climatic regions, across forest cover types, and for specific forest disturbance regimes (e.g., harvesting);

4. Emphasis on year-round model verification by testing model output with information about water and heat flow observations (throughfall, forest floor and soil percolates, streamflow, heat flux), soil moisture, water table, snowpack depth or snow water equivalents, for specific watersheds;
5. Adopting new high-level modeling platforms such as STELLA and ModelMaker to facilitate the use of models not only by modelers but also by field practitioners;
6. Availability of dynamic linkages to other modeling tools, e.g., spreadsheets, mapping programs, to expand model use within, e.g., the Windows modeling environment;
7. Expansion of the hydrological modeling process to include the spatial distribution of forest cover, topography and soils within the watershed context, in order to obtain day-by-day updates on changes in soil moisture distributions, by model simulated soil moisture regimes.

Models have been applied to predict spatial variations in soil moisture, and thus soil properties for different situations (Keys and Arp 1996a; 1996b; Meng and Arp 1997; Meng et al. 1997; 1996). They have also been used as essential subcomponents of larger projects (Oja et al. 1995; Yin et al. 1994; Yanni et al. 1994).

Still, much work needs to be done. For example, many models do not account for, or incorrectly account for water losses due to deep seepage through permeable bedrock, or excessively deep soils. Miller et al. (1988) noted no significant differences between pre- and post-harvest water yields for both clearcut and selection harvesting of watersheds with permeable bedrock in Arkansas. Wallach (1997) noted errors in surface runoff predictions because of inappropriate formulations for infiltration and subsurface lateral flow. Most importantly, work has only begun to predict hydrological behavior of individual watersheds based on the basic physiography of the watersheds, such as size, orientation, soil and bedrock substrate, vegetation type, extent of disturbance to the vegetation type.

CONCLUSIONS

According to the literature, the reduction of vegetated cover reduces transpirational losses and interception losses, increases soil moisture, and the amount of soil water available for streamflow. Opening the forest canopy can either reduce or increase snow catch efficiency, depending on gap size, though snow in openings typically melts sooner than under a canopy. Opening sizes with dimensions of two to three tree heights generally produce the greatest snow accumulation. Also, elongated cuts oriented parallel to the prevailing wind capture more snow than those oriented perpendicular. Thinning stands also increases snowpack depth beneath the remaining canopy.

Due to the greater decrease in leaf area and other vegetative surfaces, the reduction in transpiration and interception is greater with softwood species than with hardwood species. Since hardwoods typically occupy richer sites, regrowth following cutting is usually more rapid, and hydrological impacts are more short-lived on hardwood sites

The spatial arrangement of cuts can be important, especially in northern latitudes with areas of moderate to strong relief, where the solar radiation balance can be strongly affected by the sunlight angle of incidence. As well, removal of vegetation from wet areas of a watershed can affect the groundwater table more than removing it from areas where water is often growth limiting.

Forest hydrology models are starting to show promise as management tools. Caution should be used to ensure that models are not applied without understanding model input requirements, and modeling results in general. The aim is to predict the hydrological repercussions of harvesting scenarios, in advance of implementation, such that the repercussions do not conflict with other management objectives, especially those pertaining to sustainability.

CHAPTER 4

EFFECTS OF TIMBER HARVESTING ON WATER QUALITY A REVIEW OF LITERATURE

INTRODUCTION

The purpose of this chapter is to:

1. Review current literature regarding the effects of forest harvesting on select water quality parameters;
2. In doing so, examine the effects of cutting (complete, partial, thinning) on the watershed wide water ion balance, and related water quality parameters for the purpose of modeling.

The parameters to be addressed refer to streamwater temperature, sedimentation loads, electrolyte loads, acidity, alkalinity, and nutrient concentrations (N, S, P, Ca, Mg, K) and a few other elements. The overall intent of this chapter is to provide a number of entry points into the current literature on this subject, from a pragmatic, practical perspective. These entry points address streamwater quality effects arising from:

1. Clearcutting without slash burning,
2. Clearcutting with slash burning,
3. Forest fires,
4. Vegetation management at and near the stream.

In each case, this review is not meant to be exhaustive. The results referred to in this report, and its documentation, are especially useful for modelling the potential effects of partial and complete forest watershed harvesting on streamwater chemistry.

In reviewing the literature, there were a number of difficulties that needed to be negotiated:

1. The accumulated literature on the subjects was vast,
2. Specific findings tended to be anecdotal for specific watersheds, thereby limiting the potential for generalization without further investigations,
3. While water quality parameters have already been quantified, it remains difficult to relate these numbers to specific watershed processes as they are affected by geo-chemical substrate, forest cover type, topography, climate, soils, and prevailing surface disturbance regimes.

In completing this review, focus was on re-iterating the findings that tend to be common across the various reports. A table is provided that briefly summarizes the main findings of a select number of recent key reviews.

LITERATURE REVIEWED

Clearcutting without slash burning

Water chemistry is affected by clearcutting in a number of ways. Clearcutting practically eliminates nutrient uptake by vegetation, thereby allowing for more nutrients to be moved by the increased levels of post-harvest soil moisture/water movement. Clearcutting also modifies the energy balance at the soil surface and within the soil. Increased soil temperatures, in turn, help to speed the rate of bio-geo-chemical reaction processes, soil respiration, organic matter decomposition, and organic matter mineralization. Increased amounts and mobility of surface water also leads to increased soil erosion. Erosion, in turn, enhances the sedimentation loadings of streams and lakes. Eroded sediment may carry significant quantities of nutrients as well.

Responses vary from place to place, but some generalizations can be made. White and Krause (1993) noted much higher nutrient losses, especially nitrate, following harvesting of hardwood forests versus softwood. Softwoods, generally occupying nutrient-poor sites, typically show a less pronounced impact on water quality upon harvesting.

Natural weathering of parent material is generally responsible for the chemical characteristics of streams during base flow periods. Such characteristics vary with:

1. Type of parent material (Hornbeck et al. 1997; Mulder 1995; Shultz 1993; Phillips 1988; Feger et al. 1990; Kalkhoff 1993; Kalkhoff 1995; Peters 1994),
2. Time of day (Kobayashi et al. 1990), and season (Rice and Bricker 1995; Neal 1992).

3. Forest overstory composition and modifications (Adams and Boyle 1979).
4. Anthropogenic influences (Christophersen et al. 1990; Hooper et al. 1990; Rustad et al. 1986; Billett and Cresser 1992; Beck and Reda 1994; Kuhn 1991; Forti et al. 1996; Rump et al. 1976.
5. Water biology factors (Haury et al. 1996).

The Hubbard Brook case study provided many examples regarding streamwater chemical responses to clearcutting in a tolerant hardwood watershed (Hornbeck et al. 1986). All major cations (H^+, Ca^{2+}, Mg^{2+}, Na^+, K^+, Si^{2+}, NH_4^+, Al^{3+}), and all measured anions (NO_3^-, Cl^-) except sulphate increased in concentration immediately after cutting, and slowly decreased again to reference levels within four years after cutting. Sulfate concentrations, in contrast, decreased, only to return to normal four years later. Other hardwood sites have shown similar characteristics (Plamondon et al. 1982; Martin 1980; Martin et al. 1986; Lawson et al. 1985).

With soil being generally richer in the hardwood sites, conditions following harvesting are generally better for decomposition of residual organic matter, including slash. Subsequent releases of nitrogen, mainly ammonium and nitrate-N, are consequently higher than on non-hardwood sites. The nitrification of organic-N and ammonium-N to nitrate-N has a post-harvest lag time. Presumably, naturally occurring nitrification inhibitors in the undisturbed forest are slowly degraded on the post-harvest site (Rice and Pancholy 1973). The highly water-soluble nitrate-N can either be taken up by the new vegetation, or is leached downwards to eventually end up in the stream

(Krause 1982). Peak nitrate concentrations in streamwater have been shown to coincide with maximum nitrification rates in the soil during summer (Stottlemyer 1992).

The Nashwaak Experimental Watershed Project involving a hardwood dominated watershed showed elevated post-harvest H, Na, Ca, K, Mg, P, NO_3^-, and NH_4^+ concentrations in soil and stream solutions concentrations (Jewett et al. 1995).

In general, phosphorus is in relatively insoluble forms in forest soils and is therefore very effectively retained within the soils of a watershed. Only a few studies involving slash burning actually showed small increases in streamwater phosphorus (Brown et al. 1973; McColl and Grigal 1975; Tiedemann et al. 1988; Nicolson 1977).

Base cations such as Ca^{++}, Mg^{++}, K^+ are released from the soil matrix as a result of increased post-harvest decay of organic matter. Such releases tend to neutralize some of the soil acidity. Consequently, post-harvest streamwater pH may also rise (Nykvist and Rosen 1985; Cosby et al. 1990; Kraske and Fernandez 1993; Nicolson et al. 1982).

Many base cations are naturally occurring in streams of undisturbed watersheds. Most of these cations are added to the stream on account of soil chemical weathering (Christophersen et al. 1994). Such weathering typically produces soluble forms of Na, K, Ca, Mg, and SiO_2, depending on the weatherability of the soil minerals. Since mineral composition of soils and soil parent materials can be highly variable, it follows that the stream chemical composition tends to follow the overall geo-chemical variations within the watershed substrates (Billett et al. 1997; Phillips and Stewart 1997).

Dissolved organic carbon (DOC) in streams is an important water quality parameter. This parameter can affect if not determine the acid-base balance, the degree of metal complexation, and water color (Christophersen et al. 1994; Dosskey and Bertsch 1994; McAvoy 1988).

A paired watershed study in Wales, UK, showed how clearcutting one watershed did not necessarily increase the overall carbon flux, but how summer highs were significantly greater in the cut watershed, presumably due to the increases in surface temperature and subsequent release of water-soluble carbon.

Wetlands are known to be a source of both mobile organic carbon (DOC), and stationary organic carbon. The stationary component of soil or wetland organic matter provides a strong retention filter for nitrogen, phosphorus, aluminum, and iron (Emmett et al. 1994; Butturini and Sabater 1996; Johnston et al. 1997).

In a lysimeter study in Mississippi, McClurken et al. (1985) noted a rapid decrease in forest floor thickness following clearcutting. These authors also noted a shift in the nitrogen and phosphorus input/output balance, to one where both nutrients were accumulating in the forest floor to higher amounts than in control plots, in both clear-cut and selection cut treatments. Potassium, however, seemed to be leaving the sites at a higher rate than entering. Also in this study, sediment concentrations in runoff were proportional to harvest intensity. This is concurrent with the findings of other researchers (Swank and Johnson 1994).

Singh and Kalra (1975) discussed the results of clearcutting six of thirteen watersheds in three different areas in Alberta. Numerous water chemistry parameters were compared between clear-cut and control watersheds. Analysis of variance was used to compare between-watershed variations during both the spring snowmelt period, and the mid-summer recession. The stream associated with one watershed area showed no significant changes in the concentration of any elements, and only a significant change in the yield of sulfur during July to August. Another watershed area showed increases in the yields of bicarbonate (HCO_3^-) and chloride (Cl^-) ions at the 90% confidence level, as well as soluble forms of Ca, Mg, K, and S at the 75% confidence level, all during spring snowmelt. A third watershed area showed increases in yield of Na^{++} during spring snowmelt, and increases in Na^{++} concentration during the low flow period.

The Carnation Creek watershed showed a five-fold increase in nitrates, and a 75% increase in phosphates after a 12% clear-cut followed by a burn. In an adjacent sub-catchment with a 60% clear-cut, nitrates increased 10 fold, and phosphates increased 5 fold (Symons 1977). However, clearcutting 13% of a trembling aspen stand in Utah showed no significant impacts on water chemistry.

Clearcutting three black spruce watersheds in southern Ontario caused an increase in the concentration of many ions (see Table 4.1), but a decrease in bicarbonate, nitrate, sodium, magnesium, and calcium levels (Marek et al. 1986). Another study involving black spruce also showed a decrease in nitrate as well as a decrease in sulfate, but a significant increase in alkalinity and bicarbonate concentrations, along with calcium, magnesium, potassium, and sodium. In this study, three different cutting designs were used: large block cutting, patch cutting, and strip cutting, with no significant differences between them noted. In a paired watershed study in Georgia, where forests were harvested right to the stream banks, base flow concentrations of all nutrients measured showed no changes due to harvesting. However, storm-flow values of nitrate were often 4 to 8 times higher, and values of P, Na, Ca, and Mg were often increased as well. Since storm water and spring snowmelt water often make up the majority of the annual water flux, it follows then that the overall flux of these nutrients should be higher in the clear-cut watershed.

Clearcutting a Loblolly Pine plantation in Tennessee resulted in increases in sediment concentration for five years after harvesting, as well as increases in storm-flow concentrations of potassium, but no changes in nitrogen or phosphorus (McClurkin et al. 1985). Emmett et al. (1994), and Stevens et al. (1990) investigated nitrate leaching from numerous plantation watersheds in Wales. Emmett *et al.* found that nitrate quantities in throughfall increased with stand age. Nitrate losses from the watersheds were less than 5kg/ha for plantations less than 30 years old regardless of age, but increased with increasing input after age 30. Reynolds et al. (1992) also noted a similar trend, i.e. a decrease in nitrogen retention with increasing age. There was also an increase in streamwater nitrate, which corresponded with an increase in streamwater hardness. The author suggested that mineralization and release of nitrate may increase with increasing base content of the soil, with soil of higher base content leading to water with higher hardness values.

Specific conductance is often used as a parameter to evaluate water chemistry, as it is considered a good estimate of total dissolved solids (TDS) in the water. Figures 4.1 and 4.2 from unpublished data on the Hayward Brook Stream Hydro-geo-chemical Study show relationships between daily mean conductivity (µS/cm) and daily stream discharge (m^3).

Table 4. 1. Summary of various watershed studies.

Cover	Treatment	NO_3^-	NH_4^+	Ca	Mg	Na	P	S	K	HCO_3^-	Cl	pH	T(^{O}C)
SW	CC (44%)							inc					
SW	CC (38%)			inc	inc			inc	inc	inc	inc		
SW	CC (100%)												+2
SW	PB (100%)												+7
SW	CC (100%)												+8
SW	B (100%)	nc	nc	+3.8	+.12	+2.8			nc			nc	
SW	CC(23%)	+123	+38	+660	+77	+377	+12	+143	+810		+280	dec	
SW	CC(40%)	+11	+2	+270	+47	+380	+6.7	+440	+383		+300	dec	
SW	CCB(12%)	X5					+75%						
SW	CC (60%)	X10					X5						
SW	CC (100%)	-0.02	+0.05	-0.47	-0.13	-0.18	+0.02	+0.65	+1.47	-3.72	+0.48	-0.77	+3.3
MW	CC (100%)	X4-8		inc	inc	inc	inc		nc			nc	inc
HW	CC (13%)	nc		nc	nc	nc	nc	nc	nc		nc	nc	
PINE	CC (100%)	nc	nc				nc		inc				
SW	CC (100%)			+7	+9	+6			+48				
HW	CC (91%)	inc		inc	inc	inc		inc	inc	inc		nc	
SW	CC(100%)	inc	inc	inc	dec	dec	inc	inc	inc		inc	+0.6	
HW	CC(100%)	inc	inc	inc	inc	inc	inc	dec	inc		inc	inc	
HW	CC(100%)	+59		+40					+48				
HW	CC	+56.7	+0.2	+60.9					+14.7				

Notes: * = ppm values in streamwater
** = mean of three year increase in total yield over background levels (Kg/km^2)
*** = mg/l in excess of reference watersheds, significant at p=.05
$ = ppm in overland flow in excess of control
$$ = 10 year increase over pre harvest means
$$$ = cumulative differences over 4 years between treated and reference watersheds

Pagenkopf (1978) sites an empirical relationship between conductivity and TDS, where the conductivity value (in µS/cm) is multiplied by 0.5 to 1.3 to get an estimate for TDS. Kobayashi et al. (1990) observed diurnal variations in specific conductance and streamflow, corresponding to evapotranspiration. During the day, streamflow decreased along with total ion concentration. This was followed by an increase in late evening. Others have noticed similar relationships between conductivity and stream-flow, but also between individual ions and stream-flow (Scrivener 1982).

Clearcutting with slash burning

The burning of slash essentially does in a matter of hours what biological decomposers do in years. For this reason, the rate of release of nutrients in soluble forms is usually in excess of both usage by remaining and new vegetation, and the nutrient storing capacity of the soil. It is for this reason that slash burning usually has a greater impact on water quality than clearcutting alone (Feller and Kimmins 1984).

There are, however, studies that have shown an enhancement of nutrient retention after burning. Johnson et al. (1997) for example show how burning changed the relative proportions of different types of organic matter, such that the total cation exchange capacity (CEC) per organic matter content increased following harvesting. Even though nutrient release increased during and as a result of burning, harvesting did not deplete the overall nutrient quantities in the soil. For more details, see McColl and Grigal (1979) and Wells and Jorgensen (1979).

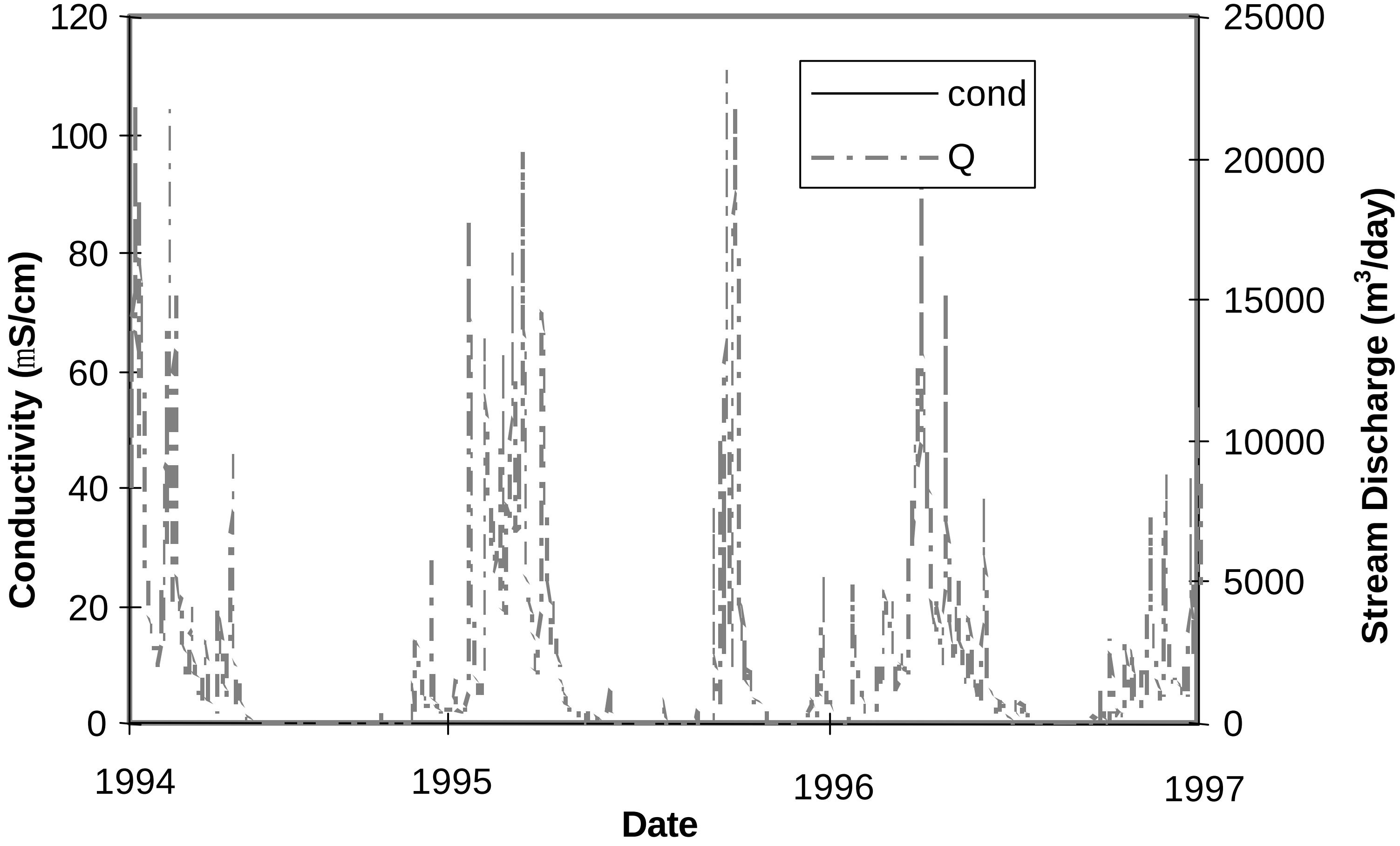

Figure 4.1 Conductivity (µS/cm) and stream daily discharge (m^3) for three years at Hayward Brook (Watershed 1).

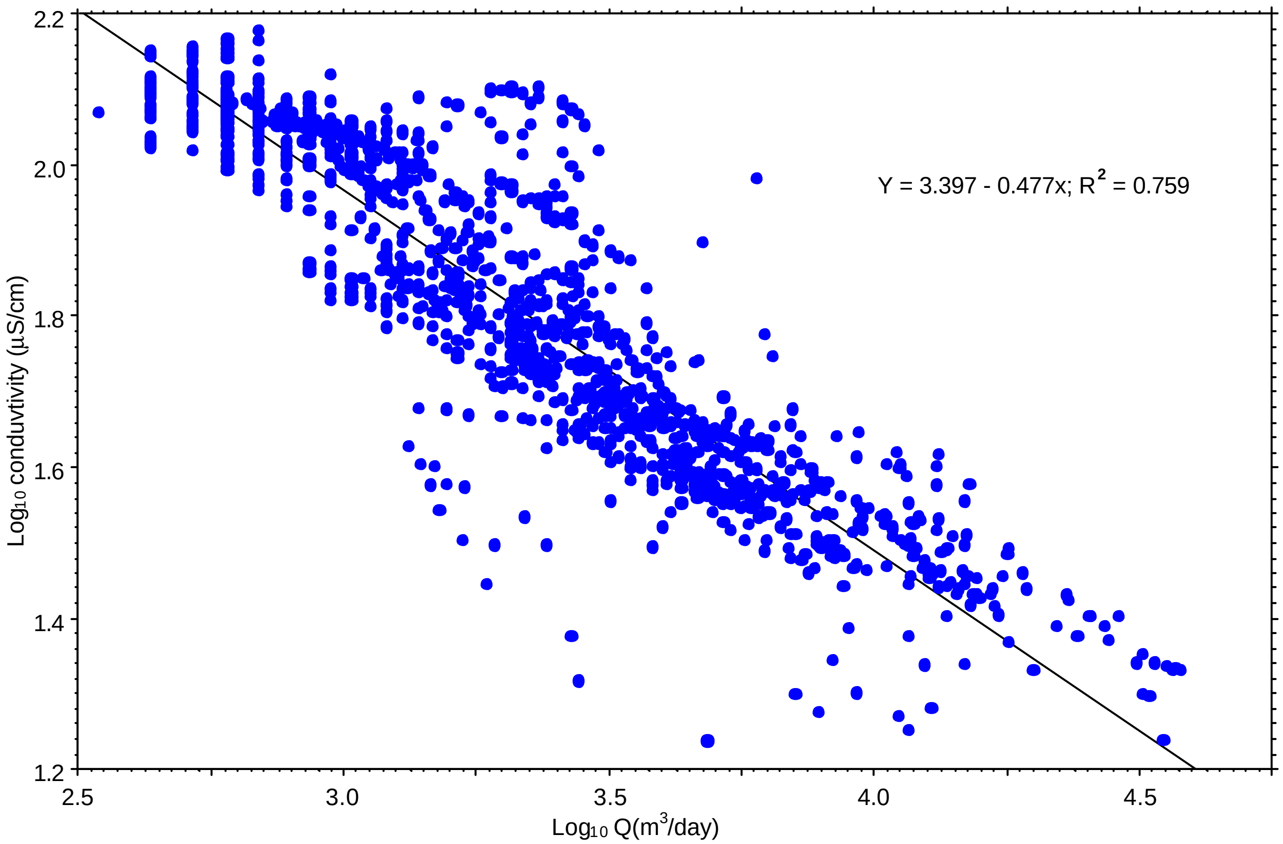

Figure 4.2 Graph illustrating a linear regression line through log transformations of conductivity (µS/cm) and daily discharge (m^3) data from Hayward brook.

DeByle and Packer (1972) reported the results of an extensive study of water quality and soil physical changes following clearcutting in Montana. On two different watersheds, each with an associated control, there were two very different responses to clearcutting. One control was completely surrounded by clear-cut and burned areas. This control showed a substantial increase in the concentration of many nutrients in the overland flow, to levels far in excess of the clear-cut watershed. The authors suspect that the tree canopy in the control watershed captured large amounts of blown dust and ash from the surrounding area, which eventually showed up in the surface water quality determinations. The other pair of watersheds showed typical responses, with concentrations of K, Ca, Mg, Na, and P in surface water being far in excess of the control watershed, and returning to very similar concentrations (± 1ppm) 2 to 4 years later.

The design of a watershed project in northern Idaho, where a Cedar-Hemlock-Grand Fir ecosystem was clear-cut and then slash burned, provided an interesting look at the change of stream ion concentrations along the same stream from above a cut, to the center of a cut, to below a cut (Snyder et al. 1975). In this study, three different catchments were used, two of which had similar results, with the other having seemingly the opposite results. For the most part, differences were statistically significant between above and on sampling stations for pH, electrical conductivity, turbidity, filterable solids, bicarbonate, nitrate, sulfate, potassium, calcium, and magnesium. On two sites, differences were generally greater between above and on sites, than above and below sites. A comparison of the above and below sites showed generally fewer differences, and some were barely significant. The streams in these two watersheds were both ephemeral and converged with larger streams, each flowing through the cut areas. These streams obtained most of their water from the uncut portion of the watershed, and thus diluted the effects of the cutting. The watershed with some exceptional results was the one that had relatively low discharge for the majority of the year, did not converge with another high-order stream, and showed steadily increasing ion concentrations through the watershed.

Following burning of residue from a clear-cut Douglas fir watershed in Oregon, nitrate levels in streamwater were 0.047 ppm higher than the control watershed, and ortho-phosphate values were 0.023 ppm higher. Both these values are in excess of the response to clearcutting alone.

Two clear-cut softwood watersheds in Kloten, Sweden, showed increased yields of many elements (see Table 4.1) after 23 and 40% clear-cuts (Grip 1982).

Forest Fire

Following the burning of a large softwood-dominated watershed in northern Washington, organic nitrogen levels were consistently higher in the streamwater of the burnt watershed than the control watershed. In contrast, nitrate levels increased only slightly above background levels in one sample, over the two years of measurement (Tiedemann 1973). Urea and ammonia nitrogen concentrations were at the same level as the control watersheds throughout the sampling period. Ca, Mg, Na, and K were all higher in the burned watershed than in the control (see Table 4.1). Differences in alkalinity, conductivity and pH were generally less straightforward, although alkalinity followed similar trends as the base cations, probably due to the association of the bicarbonate ion with the mobile cations in the soil. Conductivity was very erratic, and showed the influence of dilution quite strongly in the spring with a sudden drop in April. This was followed closely by the fluctuations in cation concentrations, probably due to the direct relationship between ionic solute concentration and electrical conductivity (Hem 1985; Pagenkopf 1978). Water pH showed no significant differences between watersheds. Other studies show similar results (Mackay and Robinson 1987).

Water Temperature

The conservation of water temperature is vital for a healthy stream environment. According to van Groenewoud (1977), streamwater should be adequately conserved with a buffer strip of 15 m on either side of the stream. Cool temperatures are required for proper metabolism of fish and other aquatic organisms, ensuring high solubility of oxygen, and limiting numbers of fish pathogens (Hewlett and Nutter 1969).

Anderson (1973) reviewed the effects of clearcutting on stream temperature, and found that in the eastern United States, clearcutting right to the stream edge increased the average stream temperature up to 4.5^{O}C, and increased the maximum stream temperature by almost 7^{O}C. In the Pacific Northwest, stream temperatures as high as 28^{O}C have been recorded in clear-cut watersheds. In general, however, removing shade from a watercourse will cause a temperature increase of 5-6^{O}C. He suggested that water temperature increases following shade removal are inversely proportional to discharge rates, with smaller streams being more susceptible to heating.

Numerous models and equations have been generated to predict stream temperatures in response to harvesting. Brown (1970), e.g., provided a formula that predicts temperature changes in streams in clear-cut watersheds based on:

1. the heat absorption rates by the water (BTU/sq. ft./min),
2. the surface area of the stream exposed by the cutting, and
3. the minimum discharge rates in the summer.

Brown (1969, 1972) also accounted for conductive heat transfers by bedrock stream bottoms for disturbed and undisturbed watersheds. Dewalle and Kappel (1974) provided an equation to calculate the rate of temperature change over time following clearcutting based on:

1. the net heat exchange rate (cal/cm^2/min),
2. the density of water,
3. the specific heat of water, and
4. the depth of water.

They also modified this formula to account for short-term changes in temperature, and calculated diurnal variations.

In 1976, DeWalle described a method of accounting for atmospheric stability in predicting water temperatures. DeWalle et al. (1977) described a computer model that considered not only incoming solar radiation, but also the long wave radiation balance, the processes of evaporation/condensation, sensible heat exchange, and channel conduction.

Levno and Rothacher (1969; 1967) investigated the effects of clearcutting and slash burning on water temperature in Oregon, and found that the stream in the logged watershed showed a $2^{O}C$ increase after logging. After burning, there was a $6\text{-}8^{O}C$ increase during the summer months.

Swift and Messer (1971) investigated the effects of various harvesting techniques on streamwater temperatures with watersheds of different aspects at the Coweeta Hydrologic Laboratory. When an entire south facing hardwood watershed was cut and converted to a mountain farm, daily maximum water temperatures rose to over $29^{O}C$ during the summer after the first year of treatment, i.e. almost 12^{O} more than the untreated control watershed. Even after 8 years, the maximum temperature was still 11 degrees higher than the control watershed. On another south facing watershed, a similar treatment induced only a $3.5^{O}C$ increase in maximum water temperature. Chemically killing all streamside vegetation on another watershed of the same aspect raised water temperatures by less than $3^{O}C$. Streams on north or northwest facing watersheds showed minimal increases in temperature. A coppice forest after eight years actually showed a decrease in mean stream temperature, due to increased shading. As a general rule, south-facing watersheds showed warmer temperatures only in the wintertime, with minimal differences during the summer months.

Brown and Krygier (1967; 1970) showed how commercial clearcutting not only changed mean temperatures and maximum temperatures, but also changed the range of diurnal fluctuations. Brown et al. (1971) investigated the relative importance of various tributaries to the water temperature of a higher order stream in the Oregon Cascades.

Sedimentation

Erosion and subsequent sedimentation, regardless of its source, have well known consequences on the stream environment (Morgan 1997). Larger particles of organic matter are retained on the stream-bottom, where the decomposition of these particles consumes water-dissolved oxygen (Kemmer 1979; Hillman 1971). Sediments that settle out in spawning beds cause mortality to eggs and small fry (Berg 1982).

The translucent optical qualities of the water are impaired by suspended sediments, which reduce the extent of light penetration, and thus change the growing conditions for photosynthetic organisms. Sediment carrying waters also absorbs sunlight energy, which in turn changes the thermal properties of the water, and alters the temperature change rates.

Hewlett (1979) supplied a modification of the universal soil loss equation, used typically for cultivated agricultural fields, for use in post-harvest situations, where soil exposure typically occurs in small patches. With this equation, total coarse plus fine sediment is calculated as follows:

$$\mathbf{E = 400\ R\ K\ S^{1/2}\ W}$$

where:

R is Wischmeier’s annual rainfall intensity index,

K is the Soil Conservation Service’s soil erodibility index,

S is the mean basin slope and

W is the proportional fraction of the basin that is bare soil, weighted by distance.

This last factor quantifies the varying probability that exposed patches at different distances from the watercourse have different relative sediment contributions to the sediment load of the watercourse. The authors concluded that, on average, roughly 5 percent of soil detached during normal forestry operations reaches the stream, with the rest being captured by organic debris, and by soil depressions. This was noted by others as well (Solebenet 1997). These authors further concluded that sediment from actual logging practices accounted for about 10% of the total excess suspended sediment load, with the rest due to road construction and channel damage. Values for the sediment yield from the un-harvested forest were roughly 92 kg/ha/yr. This value increased by 84 kg/ha/yr due to harvesting, and increased again due to road construction by 816 kg/ha/yr. When separated from road sedimentation, sediments due to harvesting were composed of ~73% inorganic materials, and ~27% organic materials.

The problems with road construction were also identified by Fredriksen (1972), who expressed concern about how the number of landslides and other mass wasting phenomena increased with road construction. Even in areas of relatively low relief, roads that required cutting into a bank often destabilized the upper soil layers. This destabilization was further enhanced through the harvesting-induced increases in soil moisture content.

In an attempt to decide how far forest roads should be located from streams, Trimble and Sartz (1957) measured how far sediments were carried across the forest floor from cross draining culverts. These authors found a linear relationship between the percent slope of the land, and the distance required to filter out sediment. The actual filtering ability of the forest floor was found to vary with thickness and porosity. For the hardwood site studied, a 50% grade required roughly 38 meters, and a 20% grade roughly 20 m of forest floor before visible evidence for down slope sediment accumulation disappeared.

Rothwell (1977) showed how proper road construction and location in a watershed could result in no depletion of water quality, and developed guidelines for watershed management purposes (Rothwell 1978; 1983).

Performing a shelterwood cut on a 0.5 ha softwood watershed in the Ozark highlands in Arkansas increased sediment yields in the stream, from a pretreatment average of 11.9 kg/ha, to 31.3 kg/ha (Lawson 1985). A subsequent pre-commercial thinning and removal of the remaining over-story showed no additional effects on turbidity. A study in a watershed with only 43% vegetated cover and a 10% slope showed that removal of vegetation in this sparsely vegetated watershed induced a rapid rise in erosion. The effects were maximized when only 15% of the vegetation remained, with subsequent removal having minimal additional effects (Rogers and Schumm 1991).

Clearcutting larch-Douglas fir watersheds significantly decreased the bulk density of the top 2.5 cm of soil, and increased soil porosity. A prescribed burn of logging residue, however, increased the bulk density slightly. Both bulk density and porosity returned to pre-harvest conditions four years later (DeByle and Packer 1972). Soil erosion reached a maximum two years after the burn at 183 kg/ha, compared to negligible amounts for the control watershed. Maximum sediment yields from summer storms were almost as high as spring storms. The majority of sediment loss from the burned watershed therefore occurred during peak runoff periods with similar results noted by Rab (1996). In an investigation of burn intensity affecting sediment yield, Robichaud and Waldrop (1994) noted that dry areas that burn intensely produce almost six times as much sediment as moist areas.

Miller et al. (1985) reported the results of clearcutting in two paired watershed studies, one in Oklahoma and one in Arkansas. The study in Oklahoma, which involved broadcast burning and contour trenching, produced sediment yield ratios between clear-cut and control treatments of 8:1, 4:1, 3:1, and 2:1 for the first four years after harvesting respectively. The study in Arkansas, which involved both clear-cut harvesting and selection harvesting, along with broadcast burning of clear-cut areas, produced sediment yield ratios between clear-cut and control treatments of 20:1, 6:1, and 3:1. They also found that, during most years, total sediment yields are generally the result of a few but important high precipitation events.

SUMMARIES OF LITERATURE REVIEWS

Krause (1982)

Krause reviewed the available literature concerning the effects of forest management practices on water quality in Canada, and ranked the streams in the British Columbia Coastal mountains and the Alberta foothills as being those most susceptible to sedimentation following harvesting, and those on the Canadian shield as least susceptible. He identified the primary cause of soil disturbance, erosion, and stream sedimentation to be road construction, with actual logging rarely contributing to overall sediment loading of streams.

Maximum daily water temperature following clearcutting and complete stream exposure were found to increase by 10^0C or more, but such increases were mostly confined to 5 and 10^0C. It was suggested that such increases could be minimized if not eliminated by providing an adequate streamside buffer zone.

Krause identified two main causes for increases in dissolved solids following clearcutting:

1. After harvesting, there would be a dramatic decrease in evapotranspiration. Subsequently, upper soil horizons would contribute more water to streamflow than before.
2. Loads of soluble organic matter and minerals are typically found to be higher in topsoil water solutions than in sub soil water solutions.

Post-harvest induced changes in surface and soil microclimate significantly were found to alter many of soil biological processes, especially soil organic matter decomposition. For example, increased soil temperatures would lead to increased soil respiration, and thus increased decomposition rates of soil organic matter. The accompanying increases in organic matter mineralization would add electrolyte and mineralized nutrients to the soil percolate, especially Ca and Mg. In areas with acidic forest floors, increased decomposition after clearcutting may release sulphate ions (SO_4^{--}) instead of bicarbonate ions (HCO_3^-), thereby reducing the pH of the percolating water. At the same time, the leaching of K and Na and organic anions may increase as well. In areas favoring nitrification, nitrate ions (NO_3^-) replace the bicarbonate ions in the soil solution. This also leads to soil water acidification.

Significant increases in stream nitrate following clearcutting were found to be associated with tolerant hardwoods. Overall nitrification rates tend to be affected the total nitrogen content of the forest floor, its C/N ratio, soil pH and soil base saturation.

White and Krause (1993)

These authors suggested that the water quality parameters that are most prone to change or increase after a forest disturbance are temperature and turbidity. Increased stream temperature changes are mostly due to direct stream exposure to solar radiation. This effect is moderated when the streams are deep and run fast. The quality limit for potable water is 15^0C. This value can be exceeded when the post-harvest buffer zone that is left next to the stream is inadequate.

Stream sediment loading is most often induced by mineral soil exposure caused by: road construction/maintenance, off-road transportation, and stream crossings. Proper road construction and maintenance, along with well-constructed stream crossings reduce such impacts. Special attention must also be given to: the type of machinery used for harvesting and wood extraction, the timing of these operations, and location of skid trails and landings.

The use of filter strips adjacent to streams (areas where the forest floor and ground vegetation are left intact) is to be encouraged. However, such filter strips may not always prevent sedimentation from reaching the stream, because sediment can travel some distance through underground channel flow as well in areas where such channels exists.

The predominant change in stream chemistry following harvesting is an increase in nitrate nitrogen. The magnitude of such increases depends on forest cover type, and post-harvest speed of vegetative re-growth. Post-harvest increases in nitrification rates are greatest in hardwood watersheds. Such increases, however, rarely exceed the maximum limit for potable water of 10 mg/L.

In summary, buffer zones are important for preventing stream bank erosion and streamwater heating. The filtering properties associated with the forest floor in these buffer zones serve to trap overland sediment flow, due to shallow runoff.

MacGregor (1994)

MacGregor also referred to the heightened nitrate concentrations in post-harvest soils and streams, and also cited areas where post-harvest H, Na, K, Ca, and Mg levels were slightly elevated. Some of the studies reviewed noted increases in specific conductance, phosphorous, no significant increases in chloride and ammonium concentrations, and a drop in sulfate. The return time of water quality to pre harvest conditions seemed to vary, with values ranging from 3 to 20 years.

In other articles, no increases in specific conductance, or increases in dissolved solids during non- storm flow were noted. Sometimes, only minor changes in some of the water quality parameters occur during storm flow following clearcutting. Clearcutting during the dormant season affects streamwater chemistry less than clearcutting during the growing season.

Sedimentation of streamwater following logging was found to be associated with improper road construction/maintenance, with areas of steep slopes being most prone to erosion. Increases in sediment loading were found to persist longer than ion loading. In some cases this impact is essentially permanent, i.e. when the recovery time exceeds the harvest rotation time.

MacGregor also referred to forest fertilization, and how fertilizer application may temporarily enrich streamwater, but generally not in excess of the set limits for potable water. Applications of pesticides and fertilizers was also reviewed: neither pesticide or herbicide concentrations were found to exceed 1ppm in streamwater.

MacGregor reviewed water temperature effects as well. Specifically, clearcutting tends to enhance both maximum and minimum streamwater temperatures.

Other reviews

Other reviews in this area exist (Corbett et al. 1978; Ellefson 1985; Hornbeck 1979; Hornbeck and Ursic 1979; Packer 1967; Sopper 1975; Osborne and Kovacic 1993; Kunkle 1974).

CONCLUSIONS

In New Brunswick, where significant buffer zones along streams are mandatory, water quality concerns following harvesting are reduced, but are by no means eliminated. Even if there are no road crossings, the effect of harvesting on groundwater chemistry and thus stream chemistry can be significant, and must be considered in management decisions.

Increasing the rate of decomposition of residual organic matter on the forest floor following harvesting provides measurable quantities of soluble nutrients which can be transported to the stream by increased quantities of soil moisture flow, and increased heights of the groundwater table. This may or may not result in an increased concentration in the streamwater, but will result in an increased net fluxes from the watershed. These effects differ between watersheds, with harvesting of hardwood watersheds generally having a greater effect on stream chemistry than those of softwood watersheds.

Nitrogen is generally the element most affected by harvesting, especially in areas where post-harvest N transformations including nitrification are enhanced. The post-harvest recovery time for most watersheds is highly variable, but seems to depend on nutrient availability for revegetation, quantity of advanced regeneration and related seed arrival rates and sprout production, and site preparation techniques.

With buffer strips intact, the majority of stream sedimentation is due to improper road construction, and stream crossings. Ditches cannot be allowed to drain surface water from roads to streams, but must be diverted onto areas with intact forest vegetation. In areas where forest floor disturbance is extensive, buffer strips need be widened, to ensure that overland flow is filtered to avoid sediment from reaching the stream. If the increase in water flow following harvesting is in excess of the existing stream bank capacity, then excessive stream bank erosion can occur during high flow periods, e.g., times of rapid snowmelt, and high precipitation events during summer.

CHAPTER 5

STREAMWATER QUALITY:

VARIATIONS BY BASIN, WEATHER AND SEASON

INTRODUCTION

The main objective of this chapter is to summarize and discuss the water quality data from the streams of the HBWS with respect to biological and geophysical basin characteristics. Watersheds 4, 5, 6 and 9 of this study contained only two soil types, with the headwaters being one type, and the lower reaches being another. Watershed 1 had only one soil type (see Figure 1.1). Other basin differences refer to watershed size (watershed 4 is the smallest at 181 ha, and watershed 5 is the largest with 924 ha). Watershed 4 had the fastest hydrological response, while watershed 9 had the least fastest response. Watershed 5 had the largest hardwood component, and watershed 1 had the largest softwood component. Watersheds 4 and 3 served as controls, with no harvesting. The watershed with the largest area cut was watershed 9 at 32.9% (Table 2.1).

From the review of literature presented in Chapters three and four, and the small areas harvested in each watershed, it was not expected that any significant harvest repercussion on water quality would be noted in the HBWS. For this reason, all pre- and post-harvest data were pooled to have as much data as possible for inter-basin comparisons.

The water quality data are grouped as follows:

Water quality parameters that are mainly affected by soil weathering make up group A; the members of this group are Ca, Mg, Na, SiO_2, K, Al, Mn, Zn, Fe, HCO_3^-, and TIC. Group B consists of parameters whose concentrations are generally affected by biological processes in the top portions of forest soils; the members of this group are the TOC, NO_3^--N, total N, and P in streamwater. Concentrations of SO_4^{2-} and Cl^- in the streams are generally controlled by precipitation (Vitousek 1977); these are members of group C. Within each group, each item will be discussed individually.

The chemical characteristics of streamwater during baseflow periods are generally related to the chemical and physical characteristics of the watershed substrate. Weathering of the substrate contributes base cations and silica first to the groundwater, and then to the streamwater. In general, one expects that the faster the rate of soil weathering, the higher will be the concentrations of the group A members, and the higher will be the pH as well. Soil weathering refers to the gradual dissolution of soil minerals. Dissolution of soil minerals such as feldspars produces silicic acids, gibbsite, and highly soluble base cations. With sufficient CO_2 in the soil to form carbonic acid, feldspars can also decompose into clays and soluble bicarbonate compounds.

Among igneous substrates, plutonic rocks are the least weatherable, metamorphic rocks weather slightly faster, and basins with substrates composed of calcareous sedimentary rock have streamwater with the highest concentrations of base cations, silica and bicarbonate. Latitude and elevation affect soil weathering rates indirectly by way of soil temperature, with weathering reactions occurring at a higher rate in warmer soils than in cooler soils (Hudson and Golding 1997; Kimmins 1987). Soil weathering occurs extensively in tropical soils, but also in humid temperate soils.

Mineral oxidation and reduction are other important factors in soil mineral weathering, and the oxidation state of the mineralized elements often determines the solubility of that element. For example, when iron is found in the reduced or ferrous form (Fe^{2+}) in a mineral, it is much more easily weathered than when it is in the ferric form (Fe^{3+}). Organic matter often facilitates the reduction reaction, by providing electrons for the reduction reactions. As well, organic matter forms many complexes with some of the mineralized elements, which tends to increase the overall mobility of heavy metals through soils and soil substrates.

Since weathering reactions only occur where water or air contact the soil minerals, it is important to note that the fracturing of bedrock, and the overall texture of the soil substrate also affects the overall rate of soil weathering, and therefore the release of mineral substances from the soils to the streams. For example, watersheds overlying fractured bedrock, or bedrock composed of materials with numerous cracks and fissures, has a much greater weathering surface than watersheds on smooth, non-fractured bedrock. Soil texture affects groundwater and in turn streamwater in two main ways. Finer textured soils have higher surface area to volume ratios, thereby increasing the weathering surface. The same soils, however, have a slower rate of water movement to the stream after rain or snowmelt events. This means that water in such soils has a longer residence time, and thus has a longer time to react chemically with the soil matrix.

For similar reasons, streamflow during baseflow periods with minimal rainfall consists of water that has been in the system for long periods of time, and typically has elevated solute concentrations. Researchers have shown how some groundwater solute concentrations can be predicted by the hydraulic conductivity of the soils. Hudson and

Golding (1997) showed how the concentrations of numerous elements could be predicted through linear correlations, with hydraulic conductivity as main predictor, and with groundwater concentrations increasing with decreasing hydraulic conductivity.

In the HBWS, streams draining larger watersheds typically had higher solute concentrations than the smaller watersheds of similar substrate compositions. From this one can conclude that as the order of watersheds increases the proportion of water that enters the streams through deep groundwater flow increases. Hence, solute concentrations of the stream should increase as well.

During dry periods, weathering of minerals can still occur within the thin film of water that adheres to the surface of the soil minerals. Because the moisture content of the soil is generally below field capacity during dry weather, this means that the water does not move downward towards the water table or the stream until the next rain or snowmelt event. At that time, however, one can expect temporary peaks in solute concentrations, followed by gradual dilution (Laudon and Slaymaer 1997).

Numerous articles within the past decade have focused on how to differentiate between different points in the hydrograph (i.e. baseflow, baseflow recession, interflow, direct runoff etc.) and the chemical properties of the water during these flow periods. These methods all take advantage of the fact that rainwater chemistry can easily be measured, and that the water in the stream after a rain event is a result of mixing of the soil/ground/rain water components within the soil matrix.

RESULTS AND DISCUSSION

Streamwater concentrations for each stream in the HBWS are summarized by group in Figures 5.1 to 5.17, from mid 1993 through 1997 inclusively.

Group A

<u>Calcium</u>

In all watersheds, calcium showed obvious seasonal trends with peak levels in late summer and fall, and lowest levels in spring during snow melt (Figure 5.1). In general, late fall soil and subsoil temperatures and moisture levels are slightly higher than summer, providing good conditions for weathering, and microbial decomposition of that summers litter. These processes combined with the issue of residence time discussed above were the primary factors driving the late summer and fall peaks seen in all watersheds.

Watershed 6 contained roughly the same area underlain by the Parry soil series as did watershed 5, but was smaller and thus had a higher percentage of the watershed within this richer soil, resulting in slightly higher low flow concentrations. During the summer, the majority of the relatively poor Sunbury soil series soils dried up, leaving the majority of the flow to originate from the richer Parry soils. It is speculated that this property, unique to watershed 6, explains why it had the greatest range of calcium concentration of all of the watersheds (2 to 21mg/L). Watershed 3, a subcatchment of watershed 6 was at the transition zone between the two soil types, and showed temporal dynamics of calcium concentration more consistent with soils of the Sunbury type. The maximum calcium concentration recorded was 4.6 mg/L, more one-quarter of the maximum recorded at watershed 6.

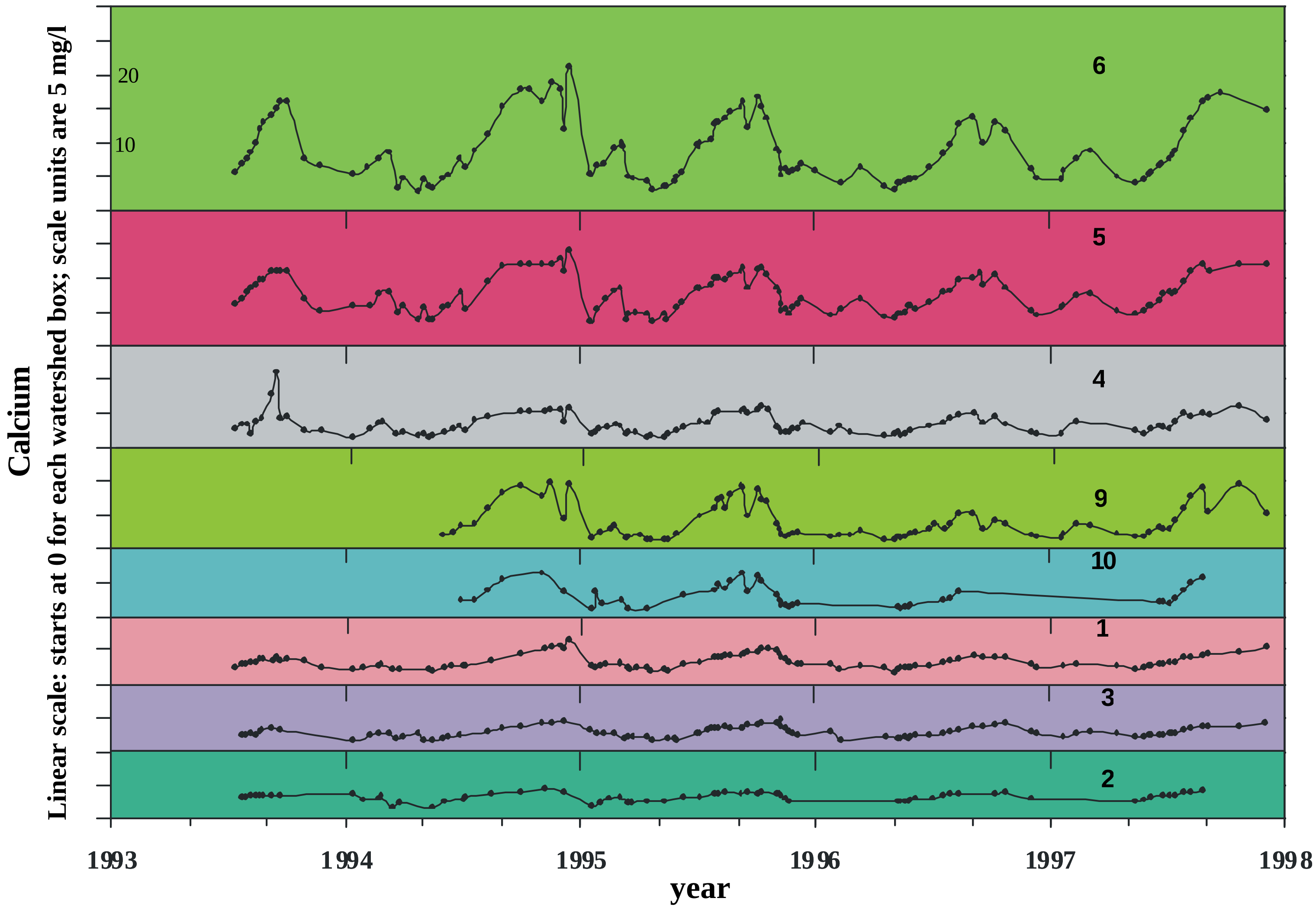

Figure 5.1 Graph illustrating calcium concentrations (mg/L) in streamwater by watershed for the study period.

Watershed 5, with only slightly lower concentrations than watershed 6, was mainly in the Sunbury soil series with only minimal area in the Parry series. Its overall mean concentrations were relatively high, partly due to the Parry soil, but also because the watershed was one of the largest in the study. Even when the watershed was at its driest, there would still be significant drainage from the Sunbury soils, explaining differences in peak calcium concentrations. Watershed 9 was primarily a softwood-dominated site, half in the Salisbury soil series and half in the relatively poor Sunbury soil series. It was one of the largest watersheds in the study and thus had deep groundwater flow, giving it slightly higher concentrations of calcium than the smaller watershed (4), located in richer soils. Watersheds 9 and 10 (on the same stream) showed similar relationships when compared to each other as did watersheds 3 and 6 mentioned above. The differences in fertility between Sunbury and Salisbury soils is not as great as between the Sunbury and Parry, and not surprisingly the differences in peak calcium levels were also not as great. This was probably due to the fact that watershed 10 (573 ha) never completely dried out. Watersheds 1, 2, 3, and 10 were located in the Sunbury soil series and all had the lowest calcium concentrations. Their slightly different calcium levels were primarily due to size, with the largest (10) having the greatest concentration, and the smallest (2) having the lowest.

Sodium

The primary source of sodium is the weathering of sodium containing rocks such as halite and feldspars with some being deposited in precipitation. Garrels and Mackenzie (1971) report sodium concentrations in rainwater reaching a maximum of 4.6

mg/L in coastal regions, with more typical concentrations for the northeastern United States (the nearest in the study to Hayward brook) being less than 1 mg/L (Berner and Berner 1996). Dry deposition is also a significant input to watersheds, as trees are an excellent interceptor of particulate matter. These dry deposits can be washed off the leaves during rain events in significant quantities after dry periods. This combined with increased weathering rates and residence time can create large peaks during late fall.

Maximum sodium concentrations at watersheds 4 and 9 were both an order of magnitude higher than any of the other watersheds with peaks as high as 17.9 and 16.6 mg/L respectively. Since all the watersheds were close to each other and would receive similar amounts of dry deposition and the patterns were not mimicked by chloride, it would seem that there is no atmospheric salt (NaCl) or halite sources unique to these watersheds. These two watersheds were also the only ones within the Salisbury soil series and would indicate that there may be some source of sodium in this bedrock, perhaps sodium feldspar.

Watersheds 5 and 6 (Parry and Sunbury soils) had almost identical sodium concentrations throughout the study period, and were roughly double the values for watersheds found exclusively on Sunbury soils (watersheds 1, 2, 3). The station for watershed 10 was at the transition zone between the Sunbury soils and Salisbury soils and thus recorded generally higher sodium levels than watersheds 1, 2 and 3.

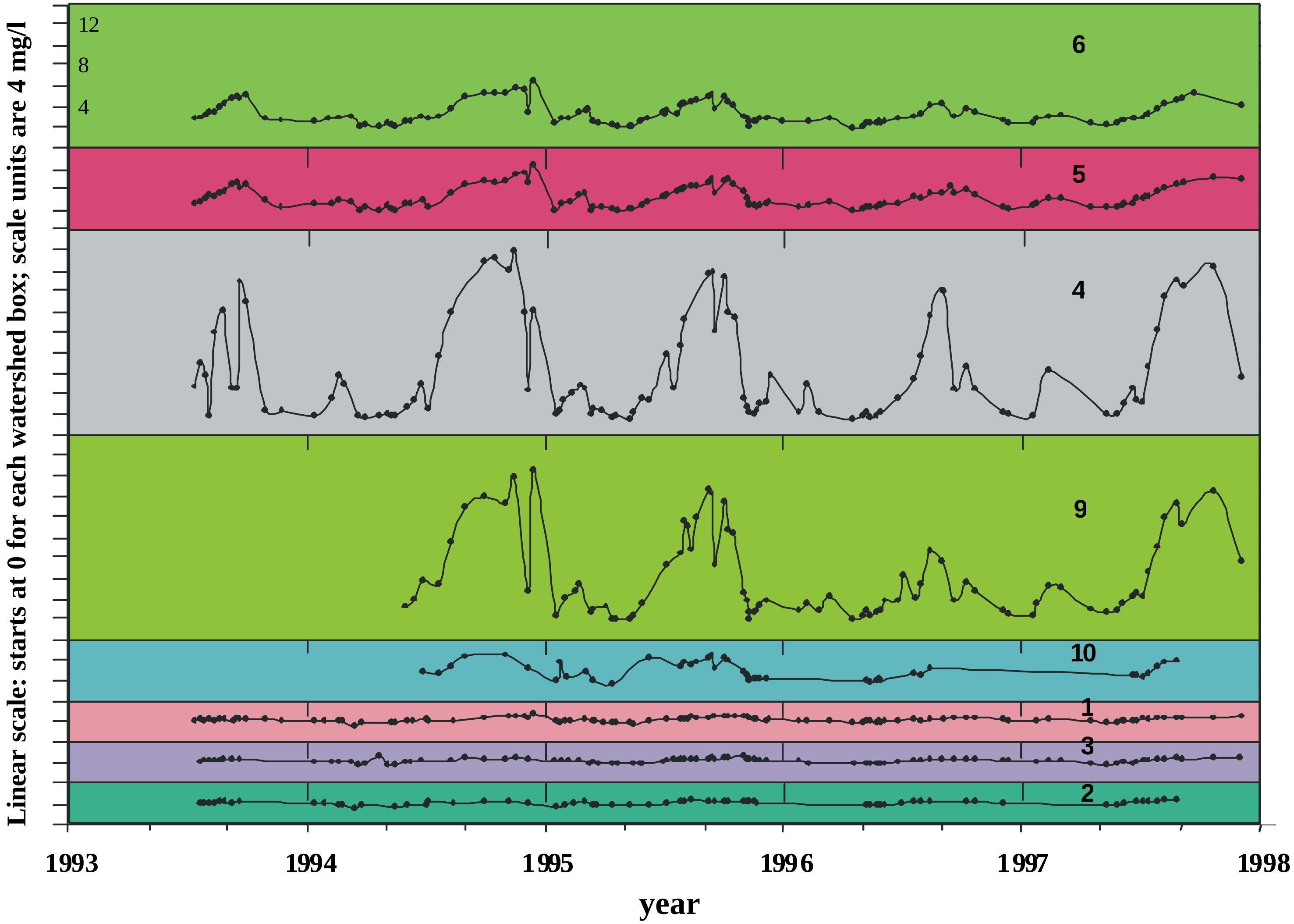

Figure 5.2 Graph illustrating sodium concentrations (mg/L) in streamwater by watershed for the study period.

Magnesium

Magnesium levels in all watersheds were relatively low, with no measurements above 2.4 mg/L. Magnesium concentrations in rainwater are generally very low with levels in the northeastern United States averaging 0.048mg/L (Berner and Berner 1996). The majority of the magnesium found in the different watersheds would seem to originate from weathering of parent materials. All watersheds have the same pattern of higher concentrations in summer and fall, typical of previous elements.

Silica

Dissolved silica is generally an indication and a result of weathering reactions. Weathering of quartz, or quartz containing rocks can result in soluble silica, which can then be transported to streams. For this reason, and the fact that its concentration in rainwater is usually below the analytical detection limit, silica has been used as a tool for separating hydrographs into different components. According to Langmuir (1997), silica in streams, rivers and lakes typically ranges from 14 to 70 mg/L.

Peak silica concentrations in the studied brooks ranged from 9 to 19.4 mg/L during baseflow periods, and as low as 1.4 mg/L during snow melt. The range of values was less extreme than other elements, and thus the annual cycle was less obvious, but there were still definite periods of higher concentrations during dry months, and lower during wet times of the year. Silica concentrations in groundwater have been shown to be buffered in soil systems through reactions with silica on the surfaces of fine textured soil materials.

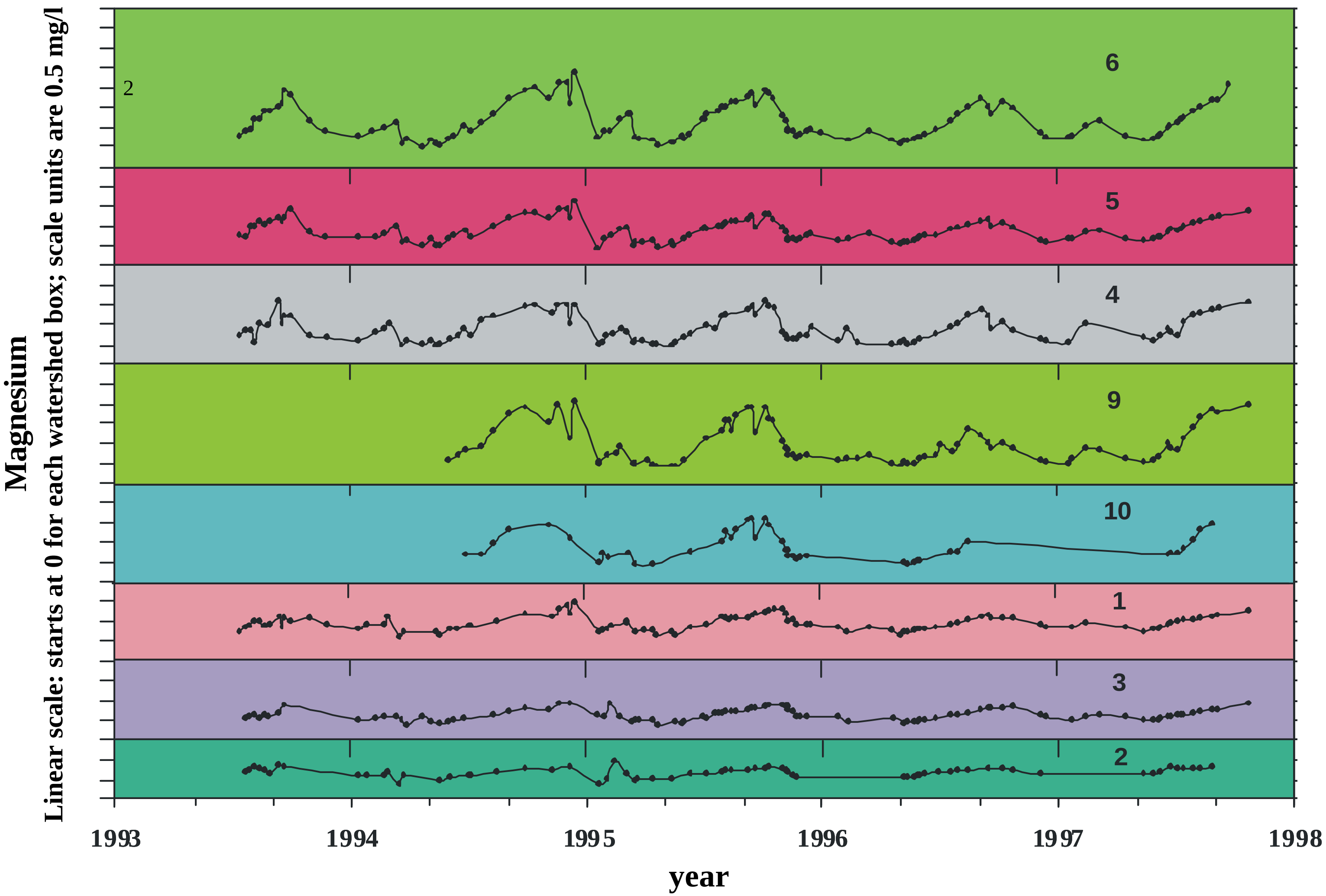

Figure 5.3 Graph illustrating magnesium concentrations (mg/L) in streamwater by watershed for the study period.

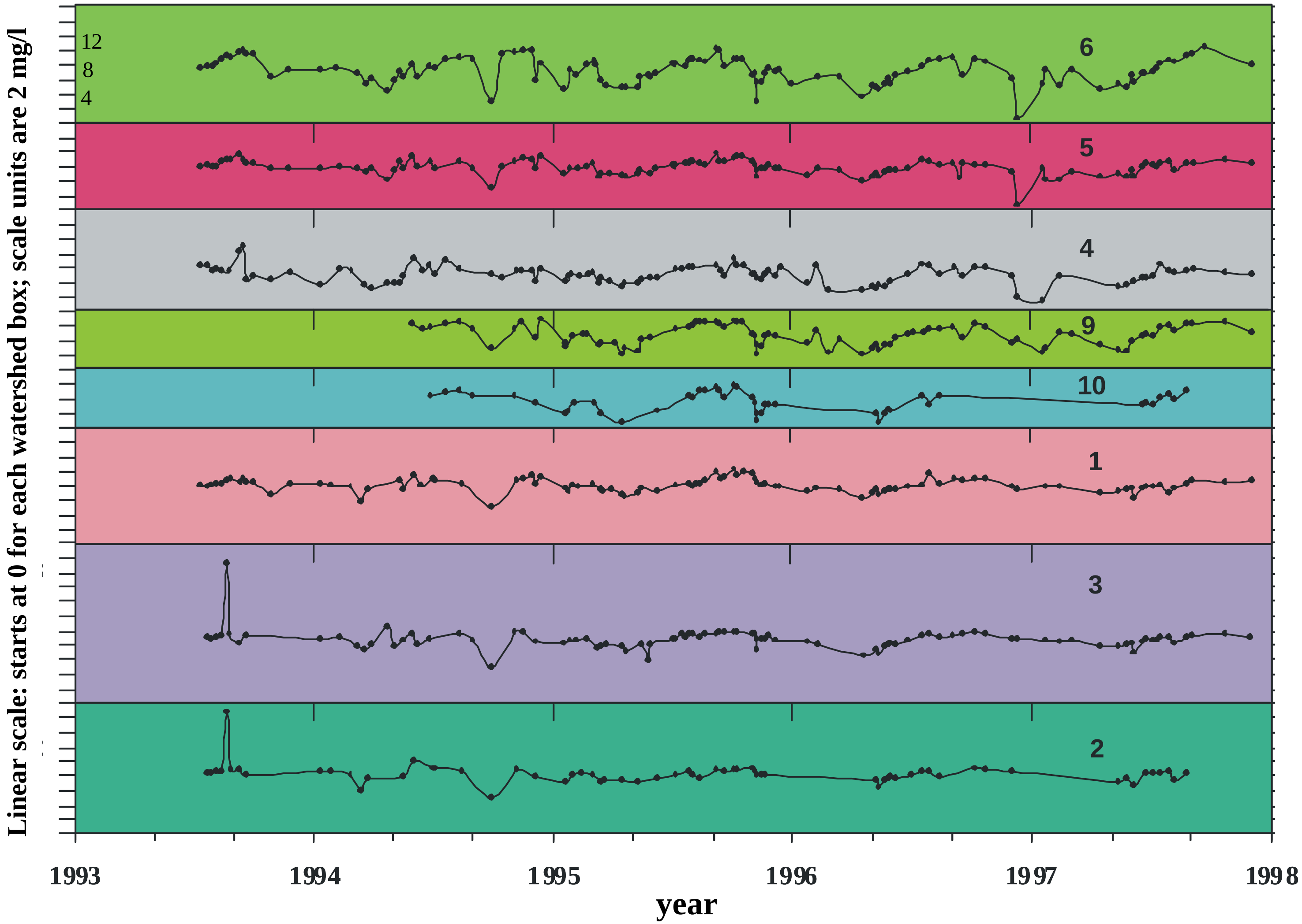

Figure 5.4 Graph illustrating silica concentrations (mg/L) in streamwater by watershed for the study period.

Langmuir (1997) cites an example from Bricker et al. (1968) where water containing no silica and 50 ppm silica was added to the top of two separate columns containing identical soil mediums. After the water filtered out the bottom, silica was measured at 9 ppm in both situations. This characteristic may explain the mild annual cycle of silica compared to other elements measured.

The mean silica value for all watersheds is 7.7mg/L, with the means for individual watersheds varying by only $\pm$ 2 mg/L. Not surprisingly, in light of other weathering byproducts discussed, watershed 6 had the highest mean silica values as well as the highest mean concentrations. Smaller watersheds entirely in the Sunbury soil type (1, 2, and 3) also had high maximum concentrations followed by larger watersheds. Other watersheds with portions in the Salisbury and Parry soils as well (5, and 10) showed lower levels, with the lowest being watershed 4, with no Sunbury soils.

Silica is often considered colloidal, due to its characteristics when complexed with metallic oxides and hydroxides, and thus often increases during times of elevated sediment loadings. Periods of higher discharge, where elements are often diluted, are also periods with higher levels of suspended solids, which may contribute to the relatively mild annual cycle of silica.

Aluminum

Aluminum levels were consistently low throughout the study period, with only a vague annual cycle of increased concentrations during base flow periods. All watersheds averaged very similar amounts with the range being only 0.48mg/L. The annual cycles are partly due to the dilution effect discussed earlier, but also the fact that aluminum as found in forest soils is most mobile in acidic conditions. Accelerated rates of decomposition in summer and fall produces organic acids, which effect the mobility of aluminum in two ways. Organic chelates produced by this decomposition can facilitate the hydrolysis of aluminum (Pritchett 1979), which further decreases the pH of the soil. This acidic environment increases the solubility of many forms of aluminum while these chelates can form stable and much more soluble organic/metal ion complexes with aluminum.

Manganese

Manganese behaves very similarly to aluminum, but is generally in lower concentrations in soils, averaging 1g/kg (Brady 1990). It is considered a micronutrient and is rarely deficient in forest soils (Pritchett 1979). Precipitation contributes minimally to the manganese budget, with concentrations less than .5mg/L being typical. Manganese solubility is dependent on the pH environment of the soil, but also the oxidation state within the mineral. It is more often found in solution in deep groundwater where oxygen levels are minimal and manganese stays in its reduced (Mn^{2+}) form. Organic matter concentrations are also lower in deep groundwater, reducing the immobilization of manganese by complexation with organic chelates.

Most watersheds in the HBWS recorded very low levels of manganese, with all but three measurements below 0.15mg/L. For watersheds 1 to 5 and 10, the majority of measurements were at the analytical detection limit or lower (0.01mg/L), with few (<10 over the entire study period) sporadic measurements slightly higher. Watersheds 6 and 9 showed mild seasonal trends, with slightly higher concentrations during late summer as soil pH decreased and organic chelates were created.

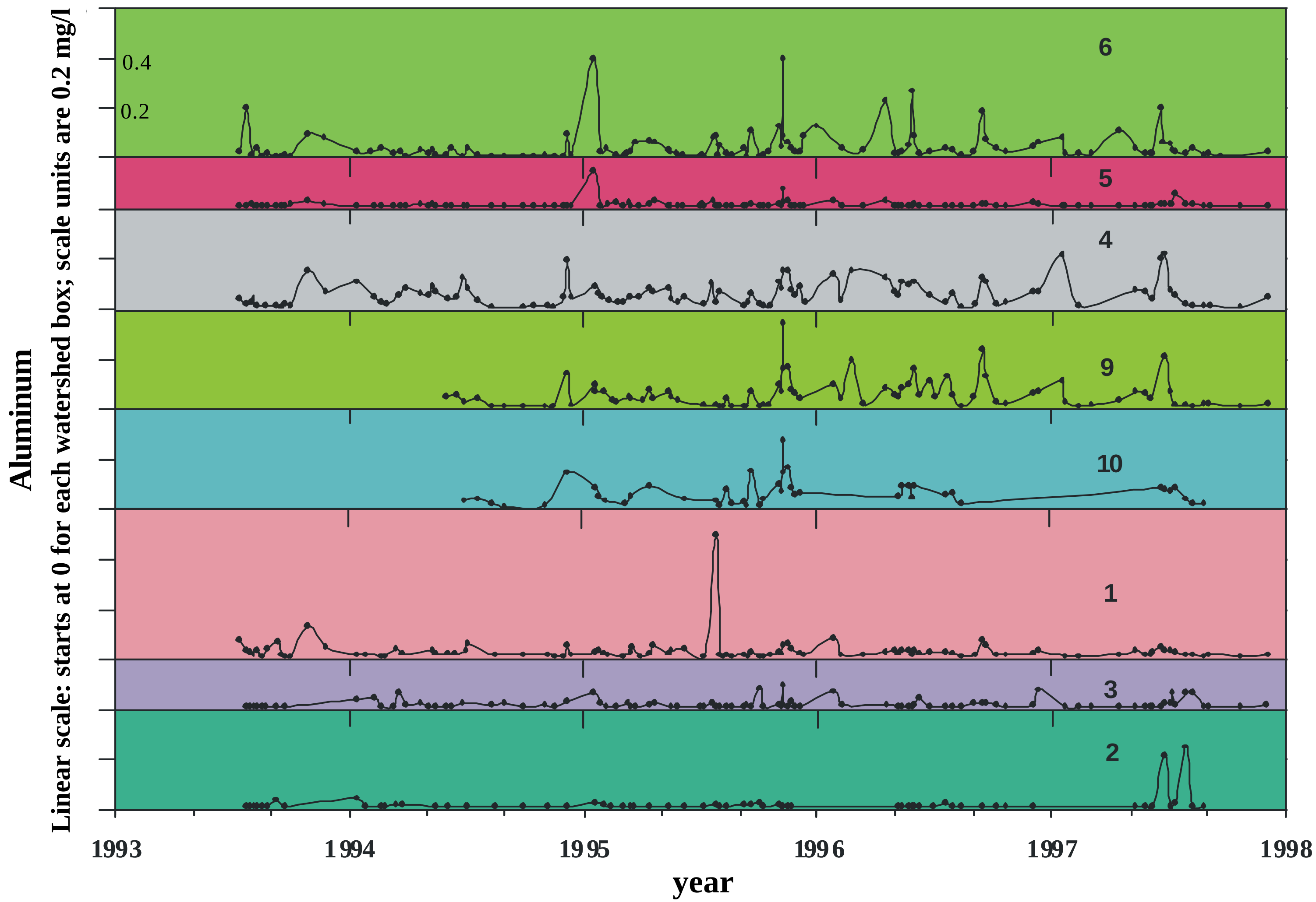

Figure 5.5 Graph illustrating aluminum concentrations (mg/L) in streamwater by watershed for the study period.

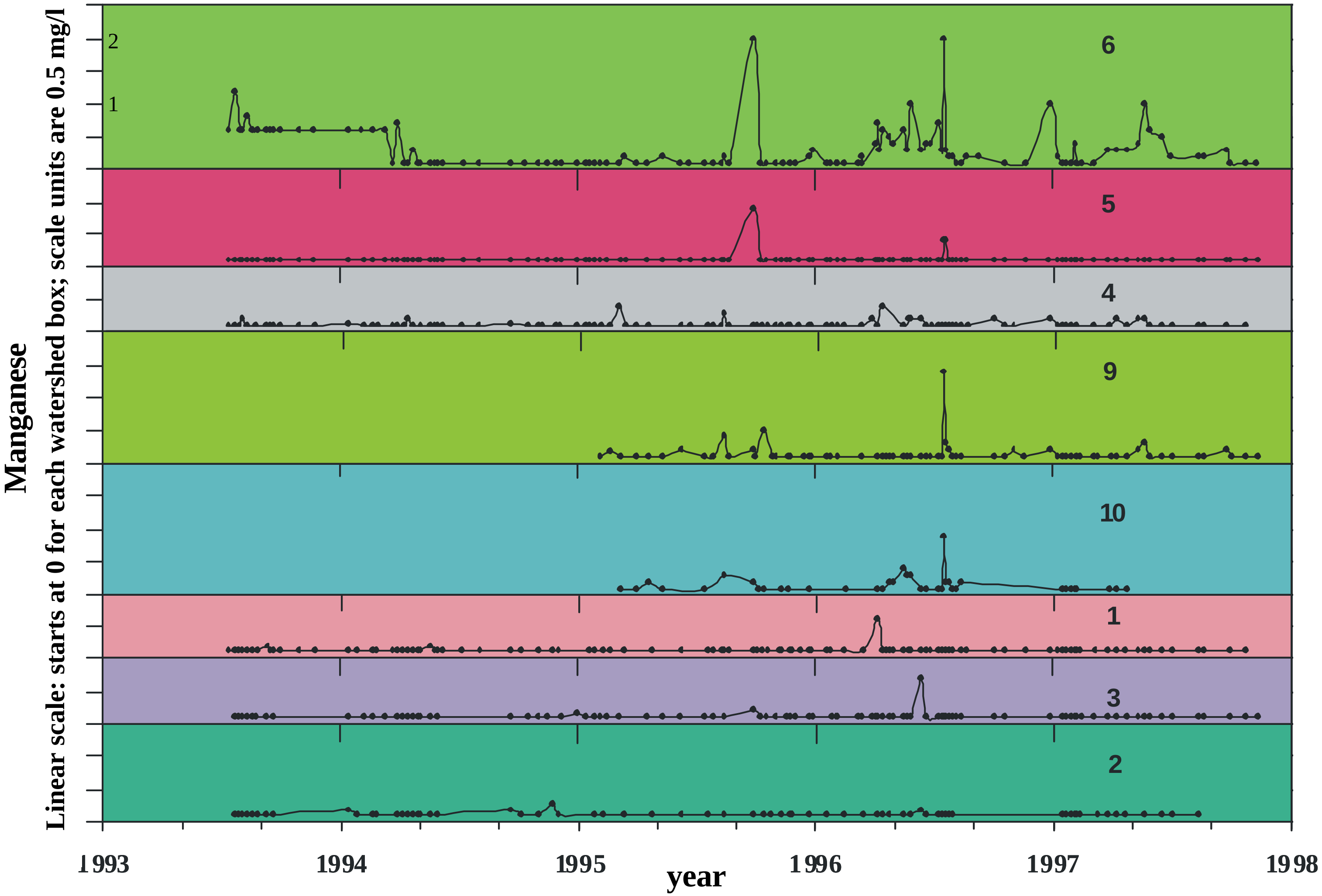

Figure 5.6 Graph illustrating manganese concentrations (mg/L) in streamwater by watershed for the study period.

Iron

Iron is one of the most abundant minerals in the earth's crust, and is rarely a limiting nutrient in higher order plants. It is most often found as oxides or hydroxides, or in complex with organic chelates. Most forms are typically insoluble, except at very low pH, or in its reduced form (Fe^{2+}) in anaerobic conditions, but is mobile when complexed with an organic chelate. A higher concentration of organic anions (DOC) generally coincides with elevated iron levels.

Iron levels were consistently low throughout the study period in all watersheds, with peak levels reaching more than 1mg/L on only three occasions. The average iron concentrations seemed slightly higher for watersheds in the Salisbury soil series where poorer drainage from the compacted subsoil would promote iron reduction, and thus mobility. Watershed 10, with generally flatter topography, is also above the average. All watersheds seemed to show mild seasonal trends, with higher concentrations during late summer and fall, for similar reasons to manganese and aluminum.

Potassium

In general, potassium is considered to be closely related to sodium, though is generally less widely spread and thus usually found in a lower concentration. Its primary source is mineral weathering, with the global mean concentration in rainwater being only 0.032mg/L, and 0.08mg/L in the north eastern United States (Berner and Berner 1996). Mineral weathering occurs at a higher rate in warmer seasons and combined with the concentrating effects discussed earlier, explains the annual cycle of potassium apparent in all watersheds. In general, concentrations in all streams were very similar, with the mean

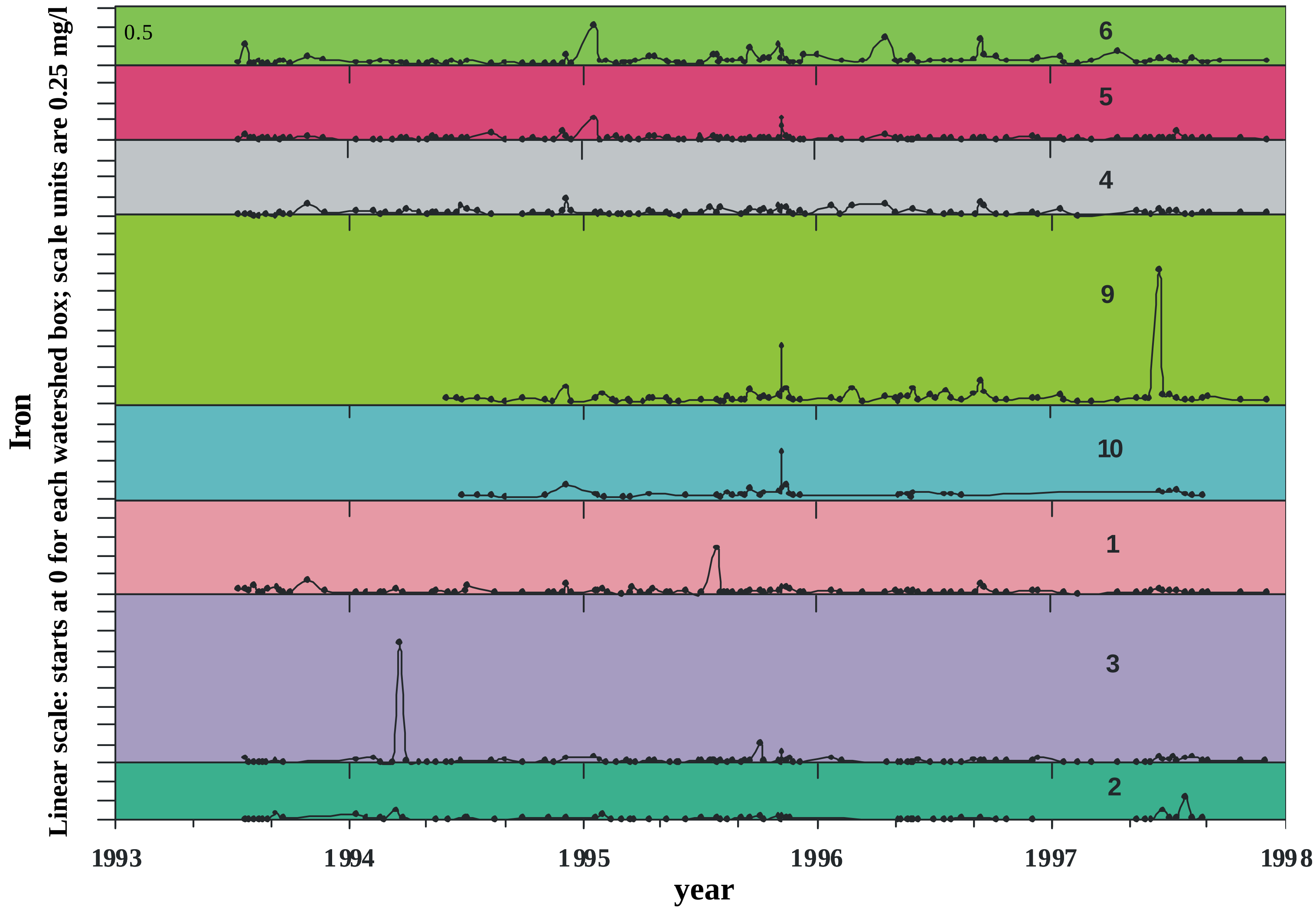

Figure 5.7 Graph illustrating iron concentrations (mg/L) in streamwater by watershed for the study period.

concentration in all watersheds being .5 mg/L, with a standard deviation of only .1mg/L. Concentrations in watersheds 4 and 9 were slightly higher, perhaps indicating a source in the Salisbury soils, though watershed 6 also had slightly higher levels.

Zinc

Zinc levels were constantly at or below the analytical detection limit, in all streams, except for a short period in the summer of 1995 when a galvanized steel culvert was installed where a road crossed the stream. Zinc dust was washed off the culvert, and downstream, where it may have settled out, only to be re-suspended during the next few storm events.

Alkalinity

The alkalinity of water is essentially its ability to accept protons, which is determined by the sum of all bases in the system that are titratable by a strong acid. The vast majority of alkalinity in streams below pH 8.3 is due to the bicarbonate ion (HCO_3^-), with very minor sources being carbonate, and some dissolved organic anions (Langmuir 1997). For the Hayward and Holmes brooks, with pH values being less than this, it is assumed that the mg/L values represent the levels of bicarbonate. As per convention, the mg/L values provided by the labs represent alkalinity as $CaCO_3$, or in other words, if the alkalinity was entirely due to dissolved $CaCO_3$, what would the $CaCO_3$ concentration be.

Bicarbonate ions have two main sources in forested watersheds, either the weathering of silicate or carbonate minerals by carbonic acid, or dissolution of carbonate minerals by carbonic acid.

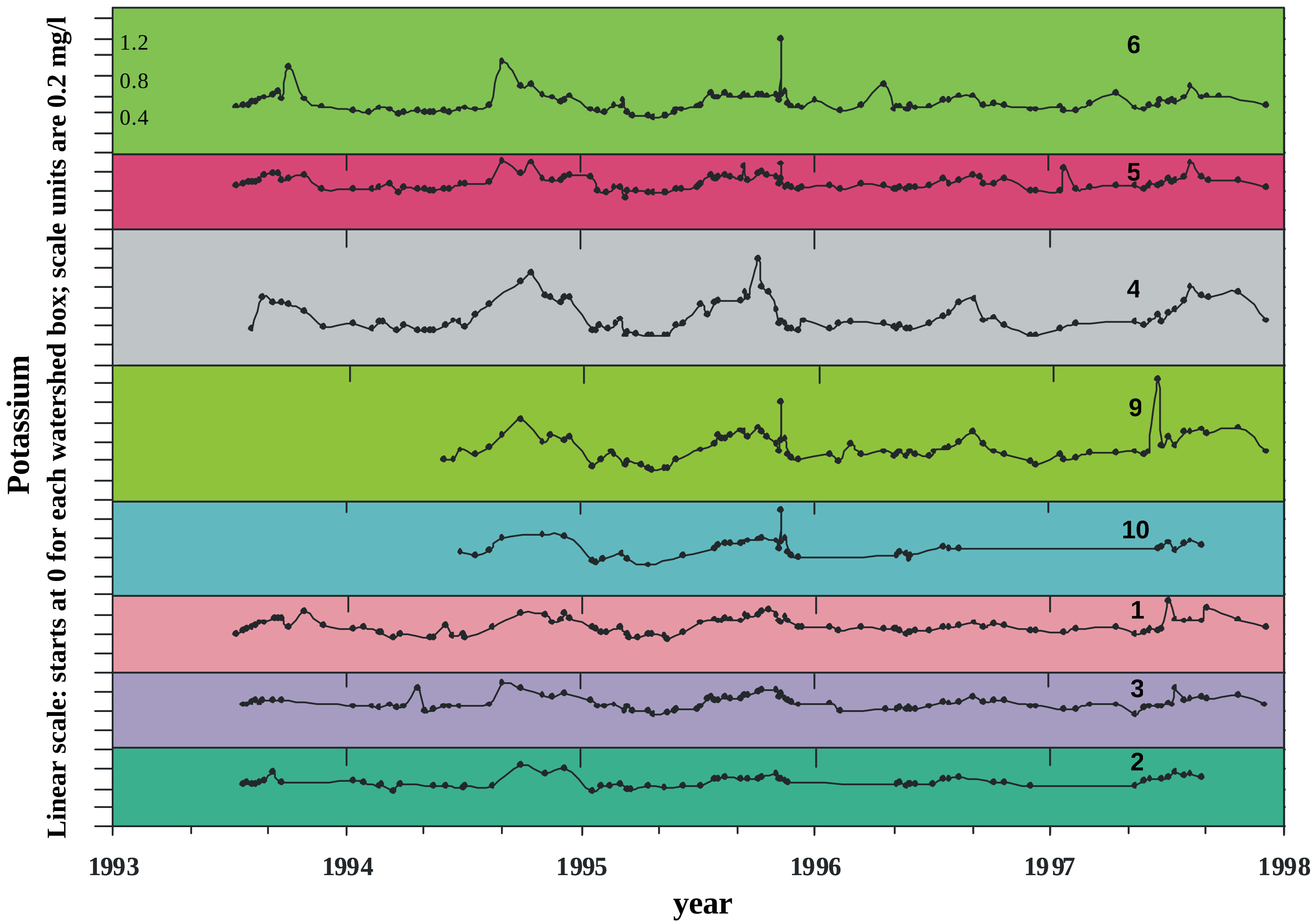

Figure 5.8 Graph illustrating potassium concentrations (mg/L) in streamwater by watershed for the study period.

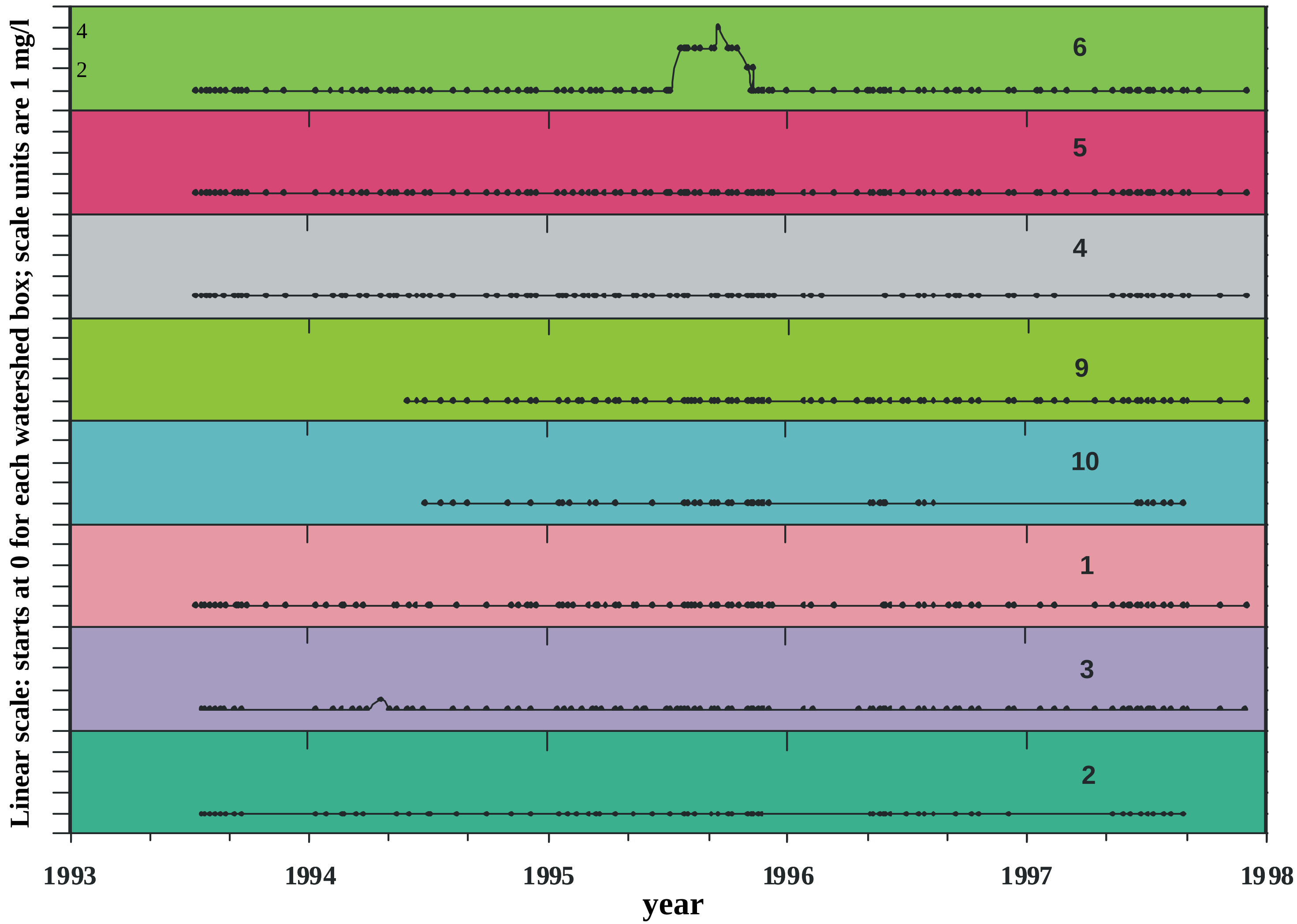

Figure 5.9 Graph illustrating zinc concentrations (mg/L) in streamwater by watershed for the study period.

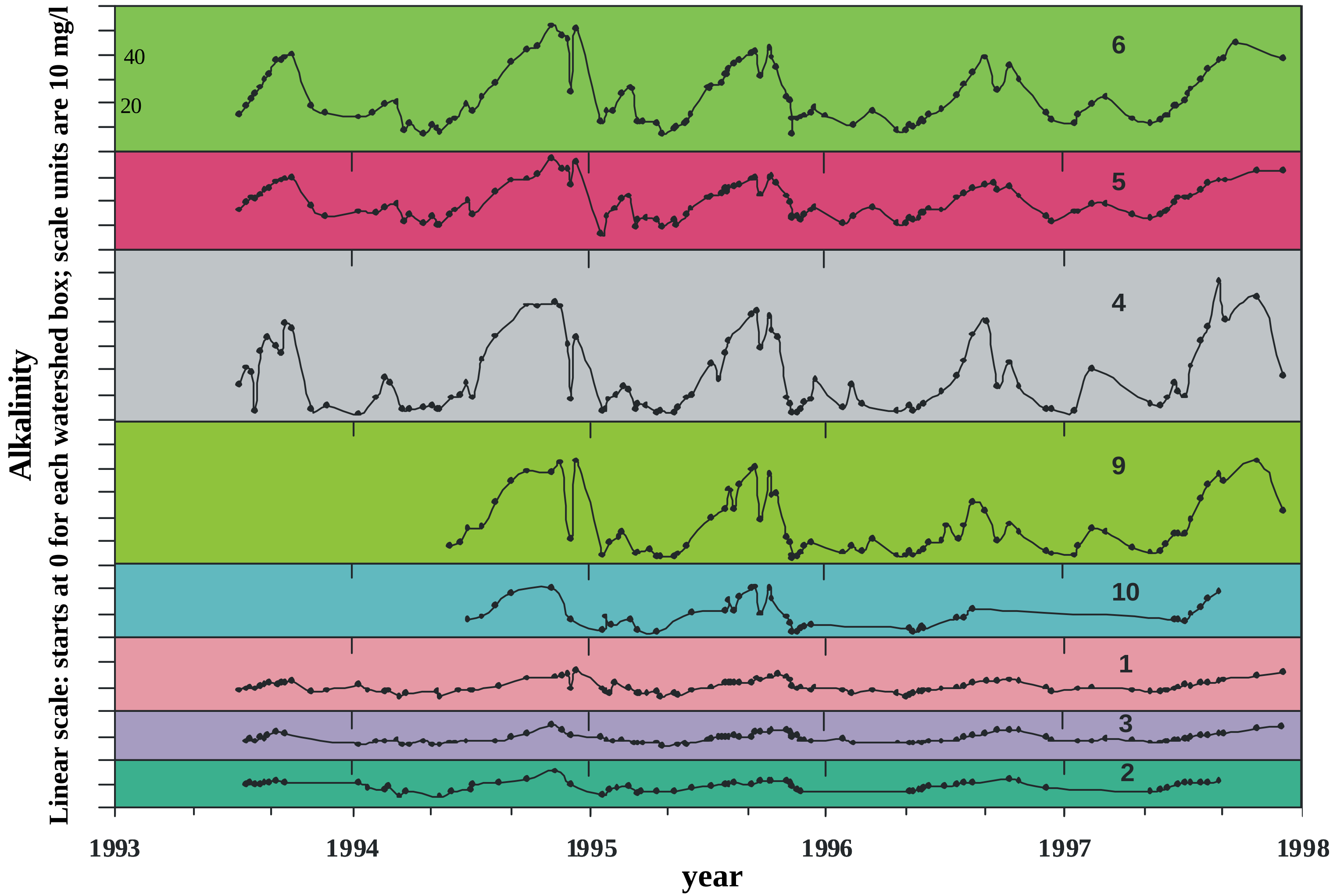

Figure 5.10 Graph illustrating alkalinity (mg/L HCO_3^-) in streamwater by watershed for the study period.

As is typical of most river systems, bicarbonate together with calcium make up the majority of the ionic composition of Hayward Brook streams. Bicarbonate has the highest average values for any ion measured, at over 15 mg/L. Since it increases in soil solution with weathering, it has annual trends typical of weathering byproducts mentioned earlier such as silica and calcium, and understandably increases with pH. During the summer, when root and microbial respiration (subterranean CO_2 production) are greatest, carbonic acid levels in the soils are highest, and thus bicarbonate levels are elevated. As well, soil temperatures are higher, favoring weathering reactions.

Total inorganic carbon

Dissolved inorganic carbon, abundant in forest soils, is primarily in three main forms (Mulder and Cresser 1994). $H_2CO_3(CO_2(aq)+ H_2CO_3)$, HCO_3^-, and CO_3^{2-}. Dissolution of CO_2 is dependant on the partial pressure of CO_2 either in the atmosphere, or in the soil matrix, and bicarbonate is also supplemented through acid weathering of parent material as mentioned above. It is curious to note that bicarbonate is indirectly measured twice in the chemical analysis of Hayward brook streamwater. Alkalinity, as mentioned, also evaluates bicarbonate and CO_3^{2-}, as well as other protophilic anions, and the difference between the two measurements gives an idea of the importance of these other acid buffering anions. However, as has been mentioned above, the alkalinity values given represent $CaCO_3$ equivalents, and if alkalinity is assumed to be due primarily to bicarbonate, the conversion factor 1.22 can be applied to convert to mg/L bicarbonate (see Kemmner 1979 for full explanation). Similarly, TIC values are given as mg/L of carbon and can be converted to concentrations of HCO_3^- (multiply by 4.583). When calculated bicarbonate concentrations from each analysis are compared in Figure 5.11.2,

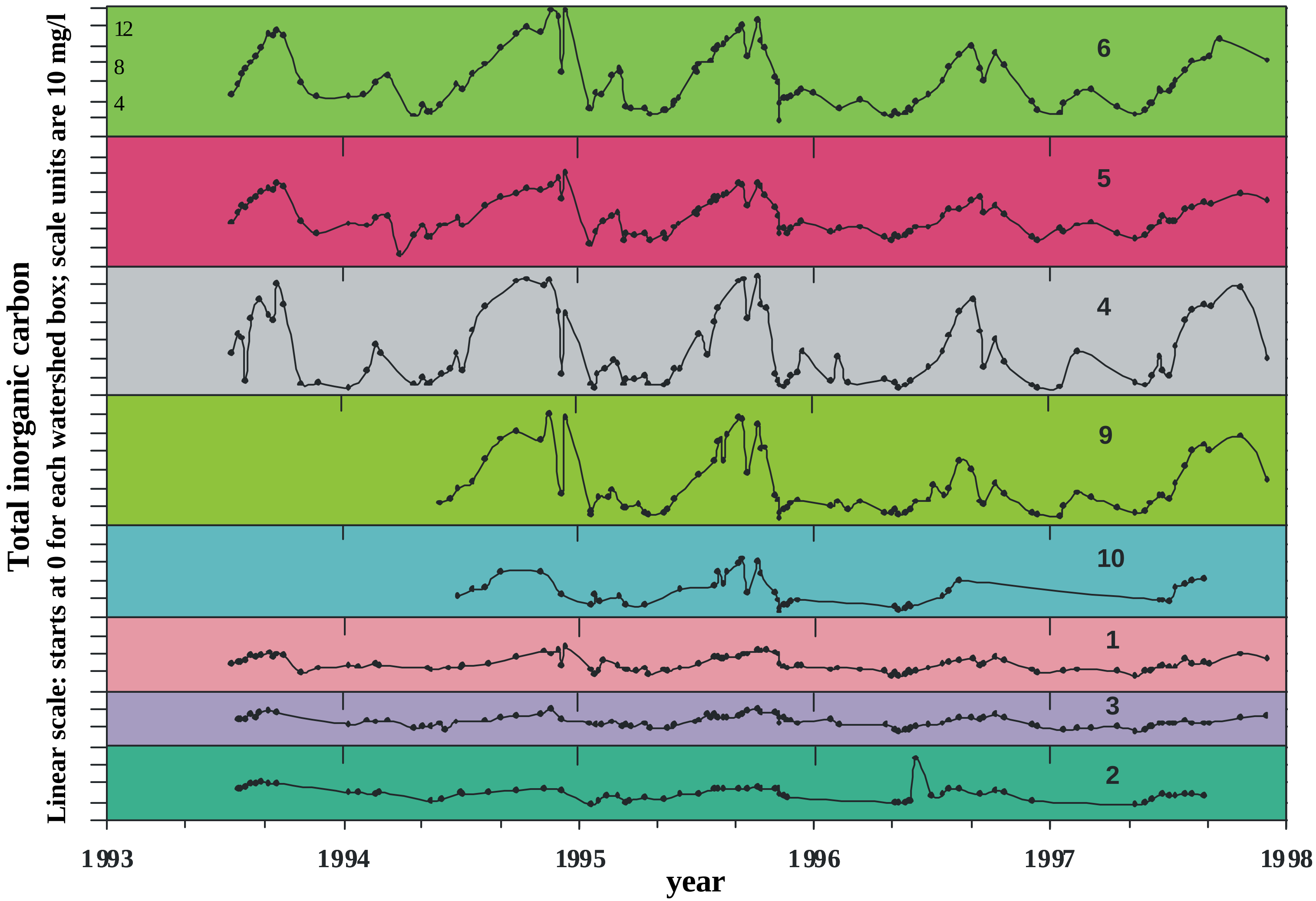

Figure 5.11.1 Graph illustrating total inorganic carbon concentrations (mg/L) in streamwater by watershed for the study period.

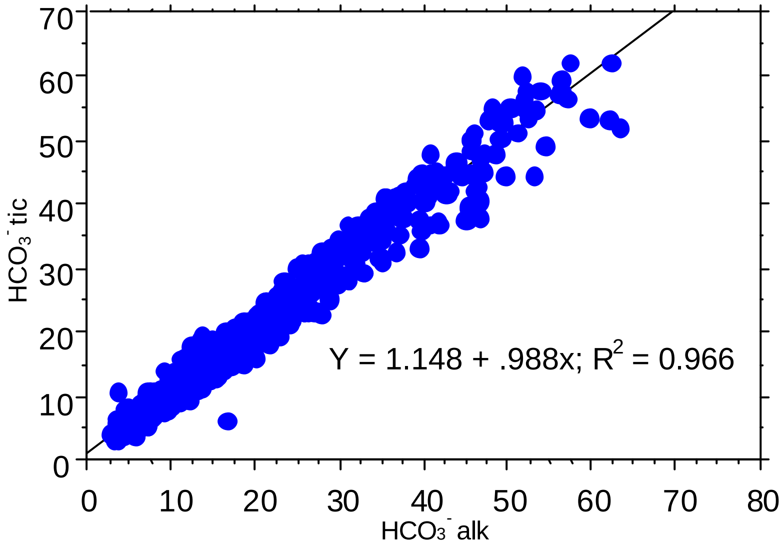

Figure 5.11.2 Graph illustrating the relationship between HCO_3^- (mg/L) as calculated from alkalinity (HCO_3^- alk) and total inorganic carbon (HCO_3^- tic).

as is expected, the values end up very similar with a slope very close to one, indicating that the majority of alkalinity and TIC are from the bicarbonate ion.

Generally TIC showed annual cycles, related to microbial decomposition (CO_2 production), increased dissolution rate during the warmer months as well as, increased weathering of parent materials, and subsequent release of bicarbonate. The differences in TIC values between watersheds were the same as for alkalinity.

Group B

Total organic carbon (TOC)

Total organic carbon in streamwater is a measurement (mg/L) of the total amount of carbon, either dissolved or suspended, in the given sample. The samples are filtered and incinerated in a combustion chamber, and the CO_2 produced is measured and assumed to be the result of organic carbon combustion. The mg/L value given with such measurements is that of carbon. The organic materials, some of which contribute to the yellow brown color seen in many streams, are most commonly found as humic substances, or more specifically, humic acid, fulvic acid, tannins and lignins (refractory compounds). Some studies have addressed these substances in even greater detail, including separation into different forms of carbon within the organic molecule (Clair *et al.* 1991), and analysis for carbohydrates, complex and simple organic acids (labile compounds), phenolic compounds, fatty acids and hydrocarbons (Wallis 1979). These are all considered dissolved organic carbon (DOC <0.45µ).

Some organic molecules are truly soluble, where much matter, like humin remain in suspension. Suspended or colloidal organic matter can originate from broken down particulate organic matter (POM> 0.45µ), such as leaves that fell directly into the stream, or matter that was actually transported through the soil matrix.

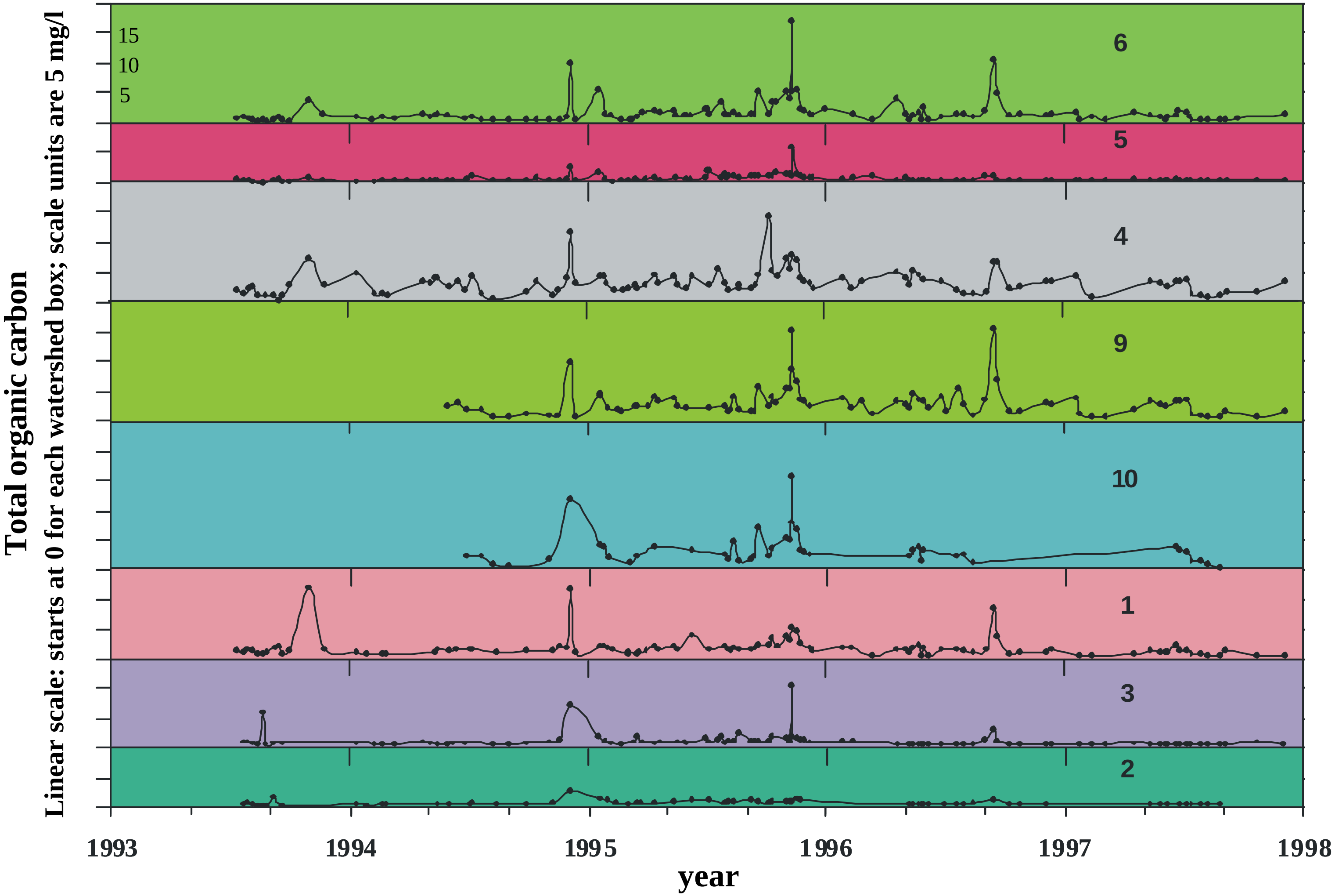

Figure 5.12 Graph illustrating total organic carbon concentrations (mg/L) in streamwater by watershed for the study period.

TOC levels in streams are related to a number of ecological processes, the most important of which are:

1. the rate of soil organic matter decomposition in the basin;
2. the rate of organic matter deposition directly into the stream;
3. the rate of POM breakdown into soluble or colloidal forms by benthic organisms.

When moisture levels are adequate, and soil temperatures are warm such as in August and September, the rate of organic matter decomposition and the production of soluble constituents is greatest. Provided there is adequate water movement to the stream at these times, TOC levels can be increased. One of the most significant aspects of soluble organics is their cation exchange properties. Organic ions are chelating agents for many metals, and are thought to be responsible for podzolization in humid regions. Many cations are typically insoluble in the pH environment of most soils and streams, but can be brought into solution when complexed with an organic chelate. For this reason, seasonal dynamics of metal concentrations in streams often reflect those of organic carbon, though it has been pointed out that color is not indicative of metal concentrations.

Organic matter can be deposited directly to the stream through litterfall, and throughfall. Generally litterfall from trees above the stream channel, or blown from a distance, is in a coarse form and not evaluated with TOC determinations. This litterfall however is the food for many different aquatic organisms, which subsequently break it down to soluble or suspendable form. The organic matter content of throughfall however is generally in a simpler and physically smaller form, and either immediately suspendable or soluble, or is easily broken down by aquatic organisms.

Organic matter in throughfall under Engelmann spruce has been found to have similar chemical properties to the original rain, (aside from dust washed off the surface) except for having very high proportions of tannins and lignins (Wallis 1979). In fact these concentrations were higher than usually found in surface waters. These would subsequently be diluted if directly deposited to the stream from overhanging vegetation, or further metabolized in the forest floor and mineral soil under trees in the rest of the watershed. Organic matter deposited via throughfall or litterfall is quickly attacked by bacteria and fungi, preparing it for use by higher invertebrates like insect larvae which are in turn food for fish. This breakdown also contributes to organic matter in streamwater. The rate of this breakdown, and in turn release to streamwater, is related to the number of organisms available for decomposition. When the stream is in a slightly warmer state while still having enough dissolved oxygen, decomposer populations are highest, and thus breakdown and organic matter release are highest.

To a certain degree, the overall amount of organic matter found in streams can be related to the amount of low lying or swampy areas in the watershed. Salisbury soils are less well drained than the others, and may help to explain the elevated TOC levels in watersheds 4 and 9, relative to the others. The headwaters of watershed 10 were also relatively flat, and the stream meanders slowly through wet areas.

Nitrate

Perhaps one of the most important, and often studied elements in watershed studies is nitrogen. One of the limiting nutrients to plant growth, and generally not found in parent materials, it lends itself well to studies concerning the repercussions of forest harvesting on streamwater quality. The main supplies of nitrogen to unfertilized soil are through the activities of nitrogen fixing bacteria, and decomposition of organic matter. Since it is usually in such low supply, relative to plant uptake potential, nitrogen absorption from the soil is usually very fast, keeping streamwater concentrations very low. On occasions during summer and fall when mineralization and fixation are greatest, a high precipitation event could wash soluble nitrogen out of the soil into the stream, but typically, absorption by terrestrial and aquatic flora and fauna will prevent this. In fact, studies have shown that nitrate-nitrogen levels in streams are typically at their lowest during mid summer, though at their highest in soil water (Jewett et al. 1995).

In the HBWS, nitrate levels were very low, with the highest recorded concentration being only 0.12mg/L, and most being at or below the analytical detection limit. Sporadic peaks showed elevated nitrate levels, usually in the summer, with watershed 4 most often showing heightened levels during summer. This may be part;y explained by the low permeability of the subsoils of the Salisbury soils, which can, as described by Krause and White (1993), promote rapid flow channels below the rooting zone, carrying nitrate to the stream.

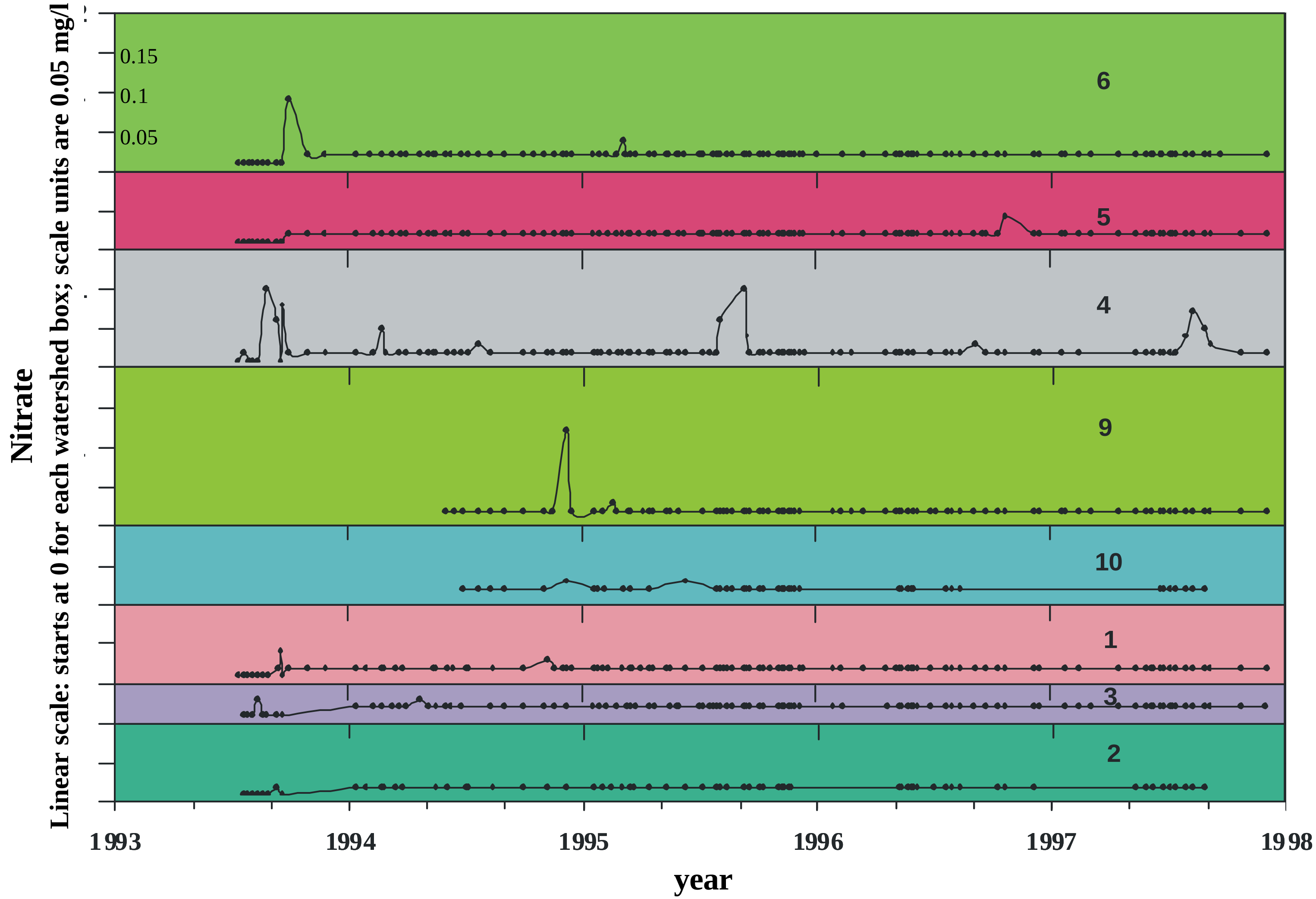

Figure 5.13 Graph illustrating nitrate concentrations (mg/L) in streamwater by watershed for the study period.

Total Nitrogen

This parameter includes most nitrogen within stream samples, including that within organic molecules. It is a broad, “catch all” parameter, typically evaluated with the Kjeldhal method, which evaluates all nitrogen except nitrate and nitrite and a few others of no importance in natural waters. Subtraction of nitrate nitrogen from total nitrogen gives an appreciation of the amount of nitrogen in organic form, as well as dissolved ammonia and ammonium. In general, 85% of nitrogen outputs in river water is organic nitrogen (Semkin et al. 1994), with the rest being nitrate, ammonia and ammonium. For this reason, total nitrogen levels follow the same general trends as TOC (Figure 5.14).

Phosphorus

In many terrestrial ecosystems, phosphorus is more limiting to plant growth than is nitrogen. The forms typically found in forest soils are very insoluble in the pH range of most forest soils (Brady 1989), and thus not readily available to most plants. Atmospheric input is minimal, and few parent materials contain significant quantities. Any phosphorus that is in a soluble form is immediately taken up my plants and as a result, phosphorus concentrations in most forest streams, including those in the HBWS are minimal. Nutrients that are intensively used by vegetation, such as nitrogen and phosphorus, often show heightened levels during winter when plants are dormant, and uptake negligible. Studies have shown that over 60% of phosphorus can leave the watershed adhered to the surface of suspended soil particles.

In some of streams monitored, a mild annual cycle was noted, with slight increases in phosphorus during summer and fall rain events.

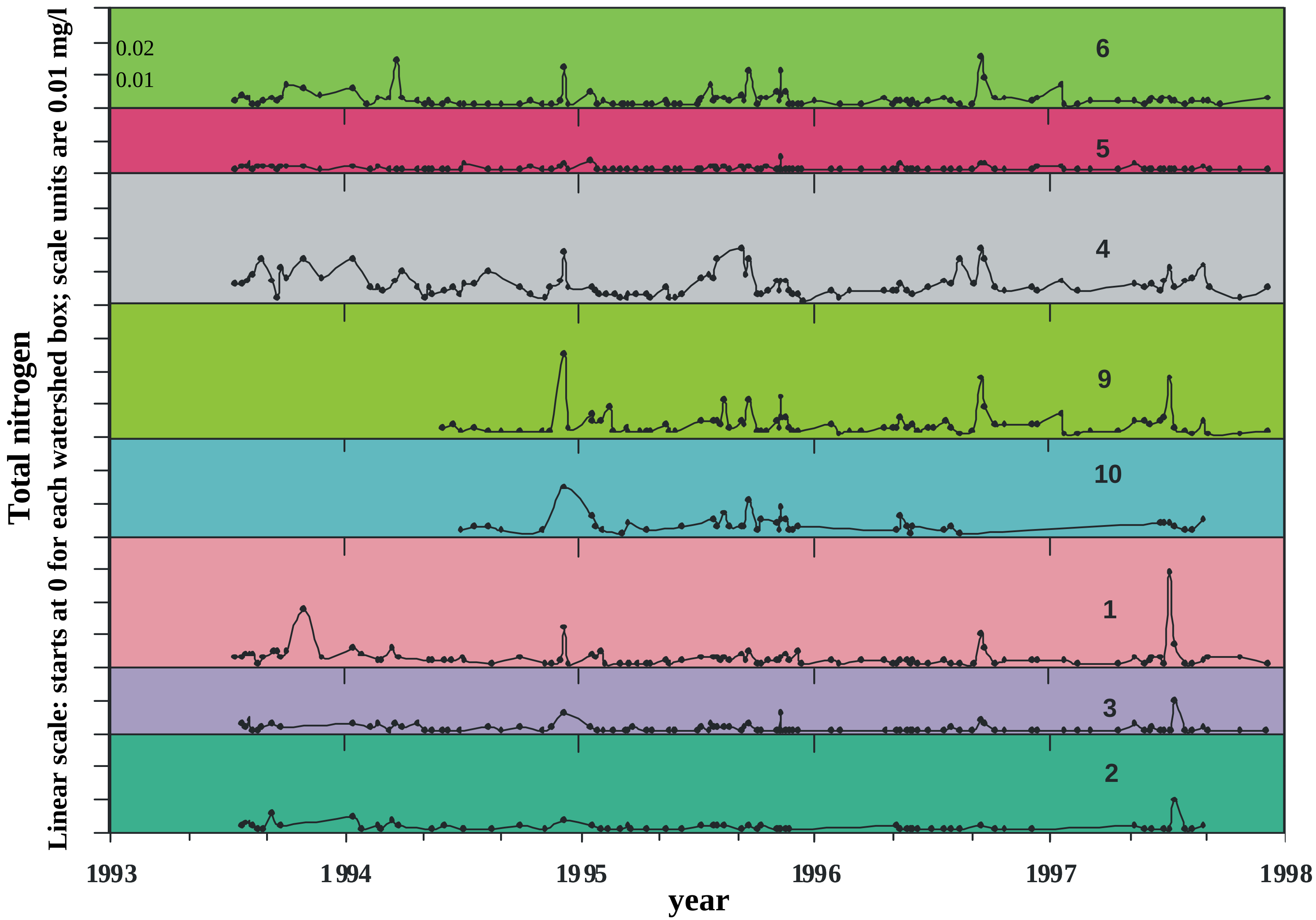

Figure 5.14 Graph illustrating total nitrogen concentrations (mg/L) in streamwater by watershed for the study period.

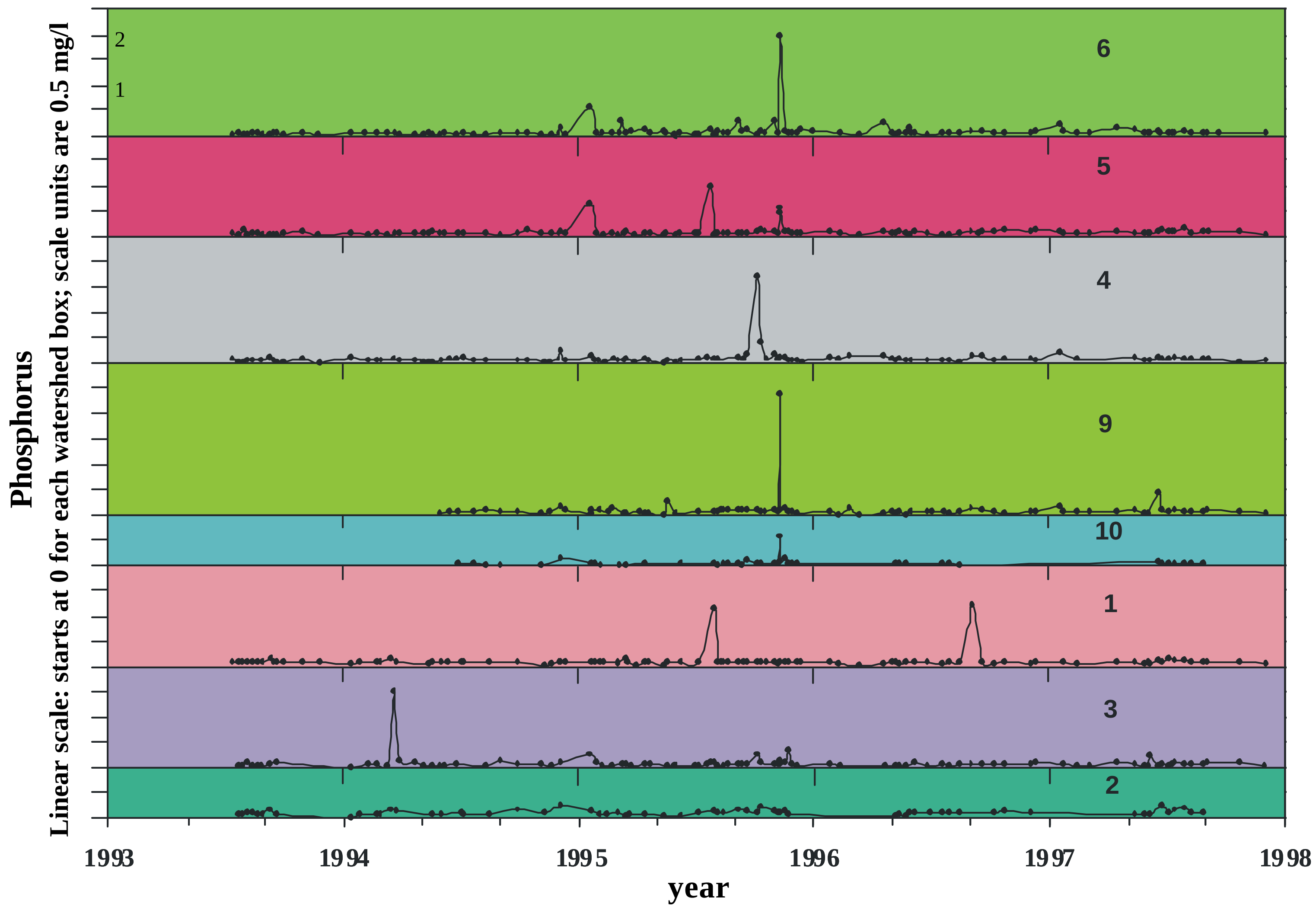

Figure 5.15 Graph illustrating phosphorus concentrations (mg/L) in streamwater by watershed for the study period.

Watersheds 4 and 9 showed a more obvious trend, where in others there was essentially no seasonal difference.

Group C

Chloride

Typically, the main source of chloride is dry and wet atmospheric deposition. Within a few hundred kilometers of the ocean, sodium chloride in sea spray can cause atmospheric Cl^- deposition to be more than an order of magnitude higher than further inland. It is used only minimally by plants, and not stored in our soils with much tenacity. For this reason, chloride trends tend to follow those of precipitation. In all HBWS streams there appeared a declining concentration of chloride in the winter and spring months, probably due to dilution and the lower ionic strength of snow meltwater. All but watershed 9 showed peak chloride concentrations in late fall and winter, when precipitation was typically highest, (see Figure 5.16) and dry deposition that had accumulated during the summer months was being washed away. Watershed 9, however, showed peaks over the entire study period that were far higher than other streams. As has been noted in the discussion of sodium above, the watersheds are all very close together, and it is doubtful there is any measurable difference in atmospheric deposition between watersheds. The peaks for chloride coincide temporally with peaks previously noted for sodium, however when considered on an equivalent basis, the gap between sodium and chloride levels is significant, indicating that the source may not be road salt, or a halite deposit.

Sulfate

In the absence of significant quantities of gypsum and pyrite, the primary source

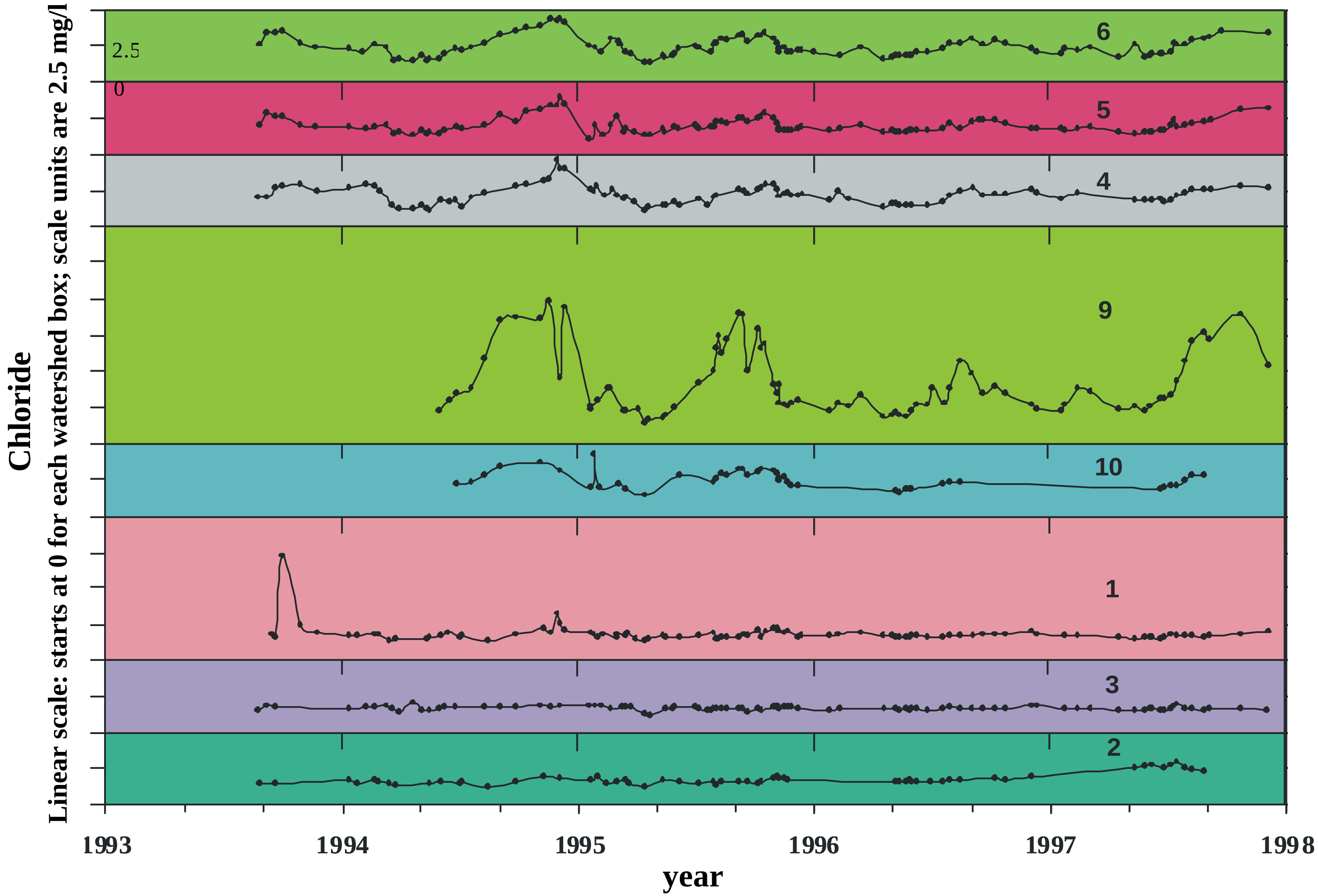

Figure 5.16 Graph illustrating chloride concentrations (mg/L) in streamwater by watershed for the study period.

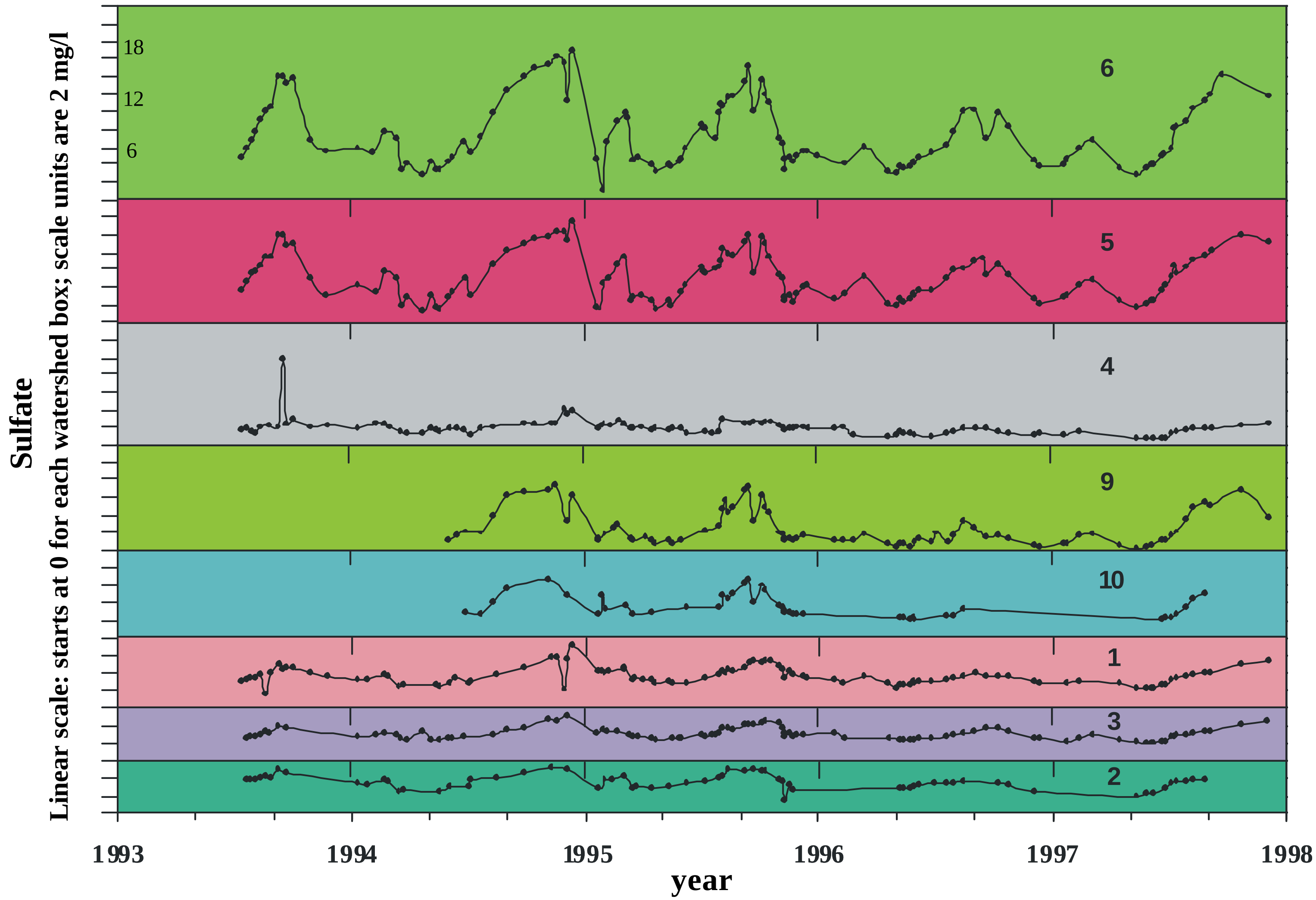

Figure 5.17 Graph illustrating sulfate concentrations (mg/L) in streamwater by watershed for the study period.

of sulfate is typically dry or wet atmospheric deposition. Atmospheric sulfate is produced through the burning of fossil fuels and organic matter, but also through volcanic activity, and ocean spray. It has received great attention recently due to its acidifying effect on forest soils and streams, and the resultant damage to aquatic habitat. Since the mid 80's, atmospheric quantities of sulfate have steadily dropped, due to stricter emission controls, and usage of higher quality, low sulfur fuels. Sulfur is generally found in the same quantities as phosphorus, though is much more mobile and readily available to plants. Microorganisms can readily transform different forms of sulfur into sulfates, which are used by plants, and highly soluble. Sulfur deposition depends on a number of factors, including topography, altitude, proximity to atmospheric sources (industry or oceans), and forest cover type. There has been noted, a positive correlation between the amount of softwood cover in a watershed, and the amount of annual sulfur export (Hultberg et al. 1994).

Sulfate exports for streams 1-4 in the HBWS were almost identical, with watershed 9 showing slightly higher levels, and 5 and 6 being significantly higher than the rest. This trend is similar to those noted for other elements that are byproducts of substrate weathering, and relates to the higher proportion of base rich, granitic bedrock containing pyrite in the Parry soils.

CONCLUSIONS

(i) The most important factors in determining the majority of stream chemical characteristics at the HBRW are soil type and characteristics, and watershed size.

(ii) Bicarbonate contributes to the majority of the alkalinity measured in the HBWS.

(iii) Water chemistry and water quality parameters can be different for similar watersheds within the same local region.

CHAPTER 6

AUTOMATED WATER QUALITY MONITORING AT THE HAYWARD BROOK WATERSHED STUDY

INTRODUCTION

The preceding chapter summarized the results of weekly monitoring of the watersheds of the HBWS. This chapter summarizes the results obtained from hourly automatic recording of stream discharge, temperature, electrical conductivity, pH, turbidity, and dissolved oxygen content. These data are explored in terms of the general synchronicity (or lack thereof) among these water quality parameters, and across the 5 watersheds that where monitored at this intensity. Also presented is the data quality protocol by which the field-recorded data were corrected. It was found that the original data were often not continuous as expected, but were discontinuous and fragmented, with the fragmentation occurring at the time of in-field probe calibration. Through numerical adjustments, and with the help of the data from the weekly sampling effort, it was possible to realign the various data fragments.

OBJECTIVES

(i) To present the process used to realign the data fragments that resulted from in-field calibrations of the automatically recording water quality sensors.

(ii) To present the corrected data in a way that facilitates visual comparisons.

(iii) To address the synchronicity (or lack thereof) among the data, by variable (stream discharge, temperature, electrical conductivity, pH, turbidity), and across watersheds.

(iv) To determine relationships between monitored variables, by watershed, and suggest how these relationship differ from one basin to another, according to readily determined basin characteristics, such as basin drainage area, soil substrate, substrate permeability, etc.

DATA CORRECTION PROCESS

The in-field calibrations for all Hydrolab probes were performed at regular intervals according to the manufacturers recommendations at all stations. The subsequent data quality evaluation process revealed how much the data, as recorded by each of the 5 sensors on each of the 5 Hydrolab probes, deviated from one another just before and after each in-field calibration step. The reading from each sensor of each probe was then numerically modified to account for this deviation, in order to produce seamless data records for each of the 25 individual data tracks. From this, the correct relative positioning of the various data fragments was re-established. The results for the 5 electrical conductivity and 5 pH data tracks were then compared with the corresponding weekly data tracks, to ensure absolute alignment for these parameters. With respect to stream temperature monitoring, there were only a few easily identified data fragments, which did not affect the absolute positioning of the entire data tracks. With dissolved oxygen, there were serious problems with the functioning of some of the dissolved oxygen sensors (notably watershed 4). These data were corrected for continuity only, as much as was feasible.

There were no problems of data fragmentation associated with stream discharge per se. Here, the main problem was the determination of the actual catchment area of each of the 5 basins, to establish discharge values in terms of mm/day. The original data were in term of m^3/sec. The correct watershed catchment areas were established by way digital elevation modeling using the flow accumulation function in ArcView, and by using a special spatial analysis technique that establishes catchment area for each point along a stream, including the exact location of the Hydrolab probes (Moore *et al.* 1991).

RESULTS

Discharge

Preliminary inspection of the corrected data tracks revealed that the stream discharge tracks have the strongest synchronicity among the 5 watersheds, from the hourly scale to the annual scale (Figure 6.1). In general, most watersheds in the study behaved similarly, having a yearly discharge peak during spring runoff, and typically showed a response to the heightened precipitation during the fall and winter months. There was very little stream discharge in summer, in general. In winter, stream discharge depended on the weather. During cold winters, stream

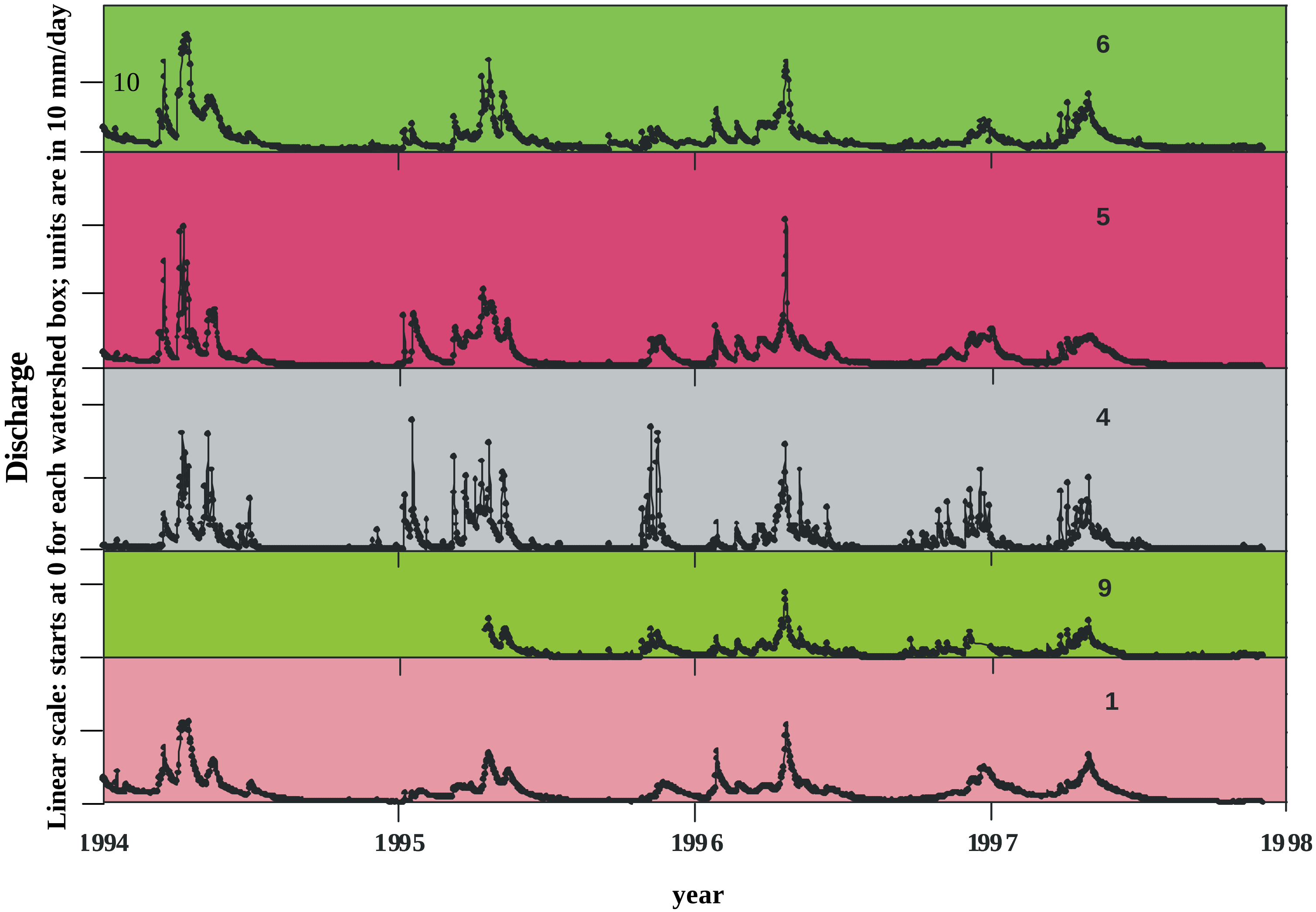

Figure 6.1 Graph illustrating discharge (mm/day) in streamwater by watershed for the study period.

discharge drops until the snow starts to melt in spring. During warm winters, stream discharge was highly erratic depending on the recurring sequence of snow, frost, thaw, and rain. As to be expected, the smaller-sized basin were more peaked, while the larger watersheds are more buffered in their stream discharge response to storm events, and to seasonal variations. Discharge per unit area (i.e. mm) was very similar for most watersheds, with differences being attributed to differences in topography, drainage, and vegetation types. Figure 6.2 illustrates the importance of soil type to discharge, with the proportion of the watershed in the generally well-drained Sunbury soils being the most important determining factor for total discharge.

Watershed 1 had the highest average discharge per unit area, probably due to the steeper terrain, and rapidly permeable substrate (Table 6.1). With a permeable substrate, incident precipitation can quickly be drained downward, and out of the rooting zone and end up in the stream. Watershed 5, the largest watershed in the study, recorded the highest peak discharge in the spring of 1996, and the second highest mean daily discharge. The watershed had some of the steepest slopes, and a higher proportion of hardwoods, as well as being partially on rapidly permeable subsoil. Watershed 6 had very similar characteristics, with slightly lower mean daily discharge, probably due to the shallower soils keeping water in the rooting zone, and available for uptake. The general discharge characteristics of the different basins are summarized in Table 6.1.

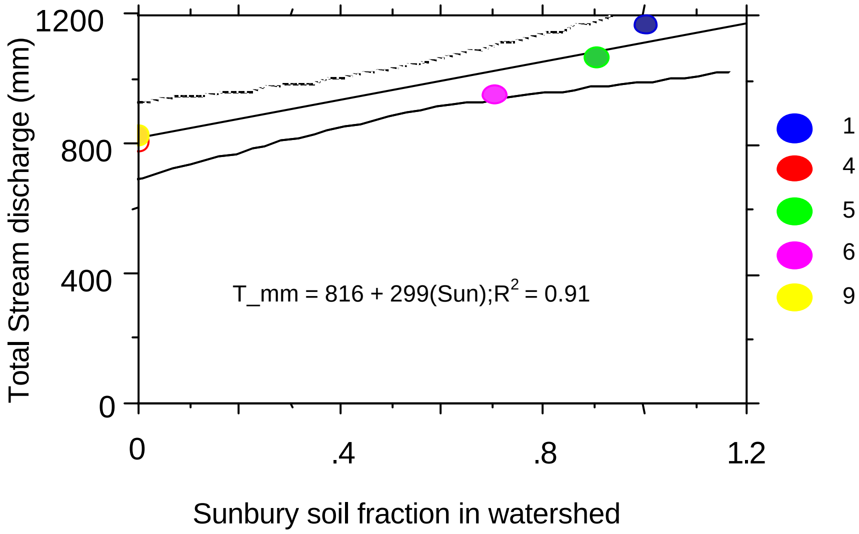

Figure 6.2 Graph illustrating relationship between total stream discharge for 1994 – 1997 inclusive and the fraction of the watershed containing Sunbury soils, for the five monitored watersheds; where T_mm = total stream discharge in mm, and Sun = the fraction of the watershed comprised of Sunbury soils.

Table 6.1 Table summarizing watershed area (ha), total, maximum, and minimum discharge values (mm/day) recorded for watersheds with automated probes.

Watershed	Area	Total	Maximum	Minimum
1	508	1172	9.18	0.02
4	181	816	12.79	0.05
5	924	1074	14.31	0.11
6	356	955	8.25	0.1
9	834	837	8.97	0.03

Conductivity

Very similar to stream discharge in terms of overall data reliability are the tracks for electrical conductivity (EC), where all the peaks and lows correspond to each other across the watersheds (Figure 6.3).

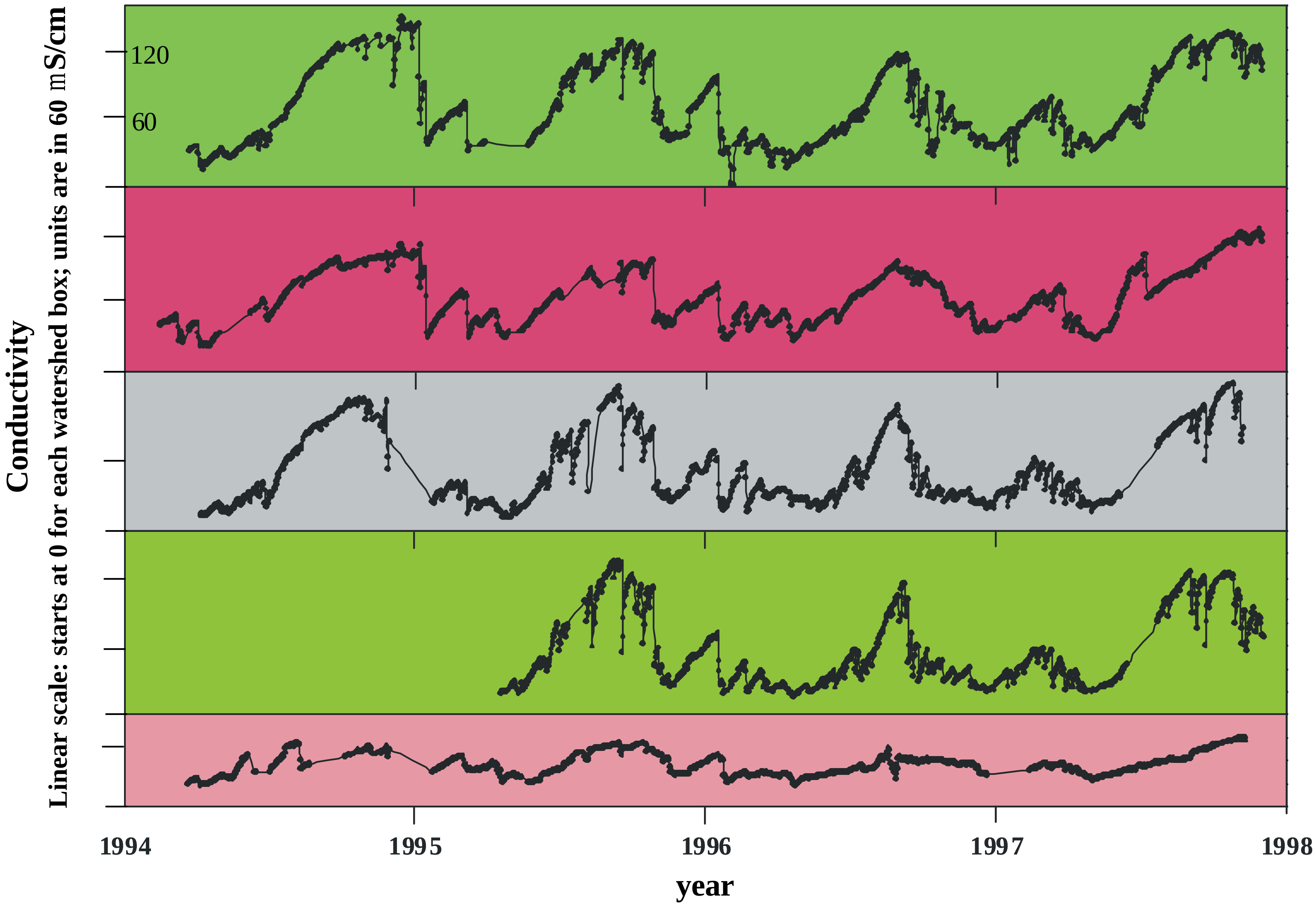

Figure 6.3 Graph illustrating conductivity (µS/cm) in streamwater by watershed for the study period.

Table 6.2 summarizes the general characteristics of the watershed in terms of conductivity. Indeed soil types and associated drainage characteristics seemed to explain the majority of the differences in maximum and minimum conductivity values, but upon closer examination, the mean conductivity values seemed more determined by a combination of discharge characteristics and soil types. Figure 6.4 illustrates the importance of these two variables to the mean conductivity.

Table 6.2 Table summarizing average, maximum, and minimum conductivity values (µS/cm), and standard deviations, for watersheds with automated probes.

Watershed	Mean	Maximum	Minimum	std
1	36	61	18.7	9.6
4	54.2	129.4	12.4	31.9
5	65.4	125.1	21.8	24.3
6	72.6	150.6	17.2	34.4
9	55.1	133.7	16	55.1

For watershed 1, in the Sunbury soil with the lowest inherent fertility, the average conductivity and the seasonal dynamics were the least. There was still a notable dilution effect during spring, but since the groundwater at these sites had only a minimal ionic load to begin with, there was not much to dilute. Watersheds 4 and 9 had intermediate average conductivity, for slightly different reasons. Watershed 9 was not only located in the richer Salisbury subsoils, but it was also larger, which gave it a relatively larger proportion of groundwater. Of notable importance, however, was the high sodium concentrations discussed in Chapter 5, which significantly increased the conductivity to levels in excess of what was

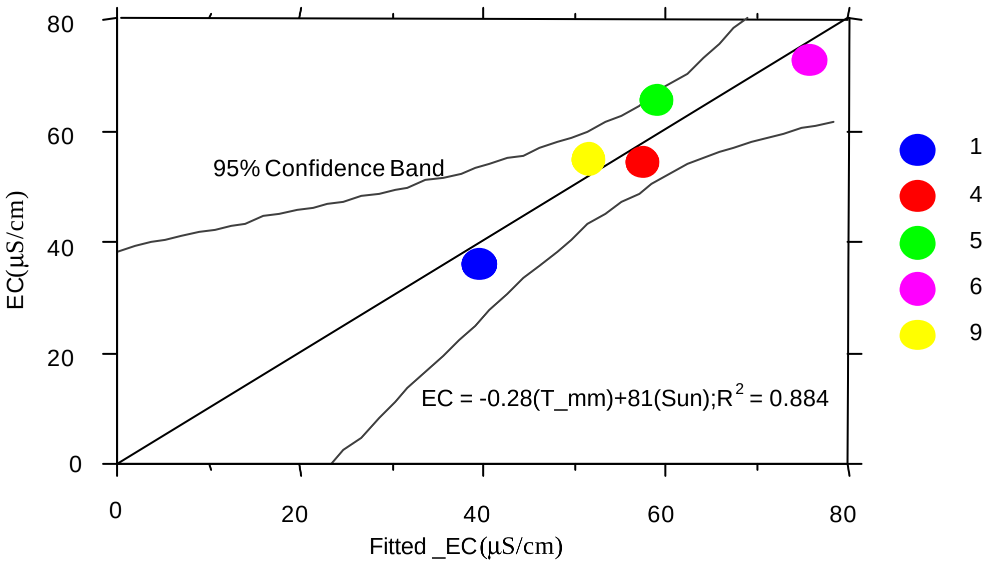

Figure 6.4 Graph comparing mean conductivity to fitted mean conductivity as predicted by the equation given, where EC = electrical conductivity (µS/cm), T_mm = total discharge (mm) and Sun = the fraction of the watershed comprised of Sunbury soils.

anticipated from the levels of other ions discussed. Watershed 4 was quite small, and with the compacted subsoils typical of the Salisbury soil series, it was very flashy or dynamic with respect to discharge. Due partially to its southern aspect, snowmelt seemed to happen very quickly, and thus spring snowmelt discharges were the most dilute in the study. The lower reaches of the watershed were dominated by the Parry soils, and when the headwaters dried up in the summer, the only groundwater supply of water was from this richer soil. For these reasons, watershed 4 had one of the greatest ranges in conductivity values of all the watersheds in the study. The remaining two watersheds, 5 and 6, had some of the highest recorded conductivity values, and had the highest average conductivity. They were both at least partially in the rich Parry soils with watershed 6 being almost half Parry soil, and had very similar seasonal trends. The headwaters of watershed 5 were still in the Sunbury soils, and though they would get dryer in the summer, they would never dry up, leaving a significant portion of the flow during the summer having originated from these low ionic strength subsoils. Watershed 6, which was significantly smaller, had the highest recorded conductivity measurement, at over 150 μS/cm. When the headwaters of this watershed, also in the Sunbury soil series, dried up, the majority of the flow would come from the rich Parry soils, leaving it with the highest conductivities. Also, there was a negative linear relationship between these two variables: when discharge is high, electrical conductivity is low. This relationship applies across the basins, although as shown in Table 6.3 by way of regression analysis using log EC versus log Q, the nature of the EC(μS/cm) versus Q(mm/day) correlations differs by basin. According to this table, those watersheds that produced the greatest seasonal and daily variations in ion concentrations also show the largest EC-Q correlation coefficients (a). For example, Watershed 1, with some of the lower average ion concentrations of the watersheds with automatic monitoring systems, showed the lowest coefficients. Watershed 4 is clearly the most erratic, and is also the smallest watershed (181 ha), with the greatest topographic variability.

Table 6.3 Summary of regression equation parameters relating discharge to conductivity for watersheds with automated probes. Regression equation follows this form: Log (Q) = $a\log_{10}(EC) + b$; standard errors are in brackets, where Q = mm/day, and EC = μS/cm. All P values < .0001, n=986.

Watershed	a	b	R^2
1	-.21 (.003)	1.52 (.002)	.82
4	-.268 (.005)	1.48 (.003)	.73
5	-.342 (.004)	1.73 (.002)	.86
6	-.468 (.006)	1.71 (.003)	.83
9	-.437 (005)	1.54 (.003)	.89

pH

A similar synchronous relationship existed between stream discharge and pH (Figure 6.5). In periods of low flow, when base rich weathering byproducts dominated the solution chemistry, pH was high. During high flow, with a higher proportion of the discharge coming from precipitation or snow melt, pH was low. In terms of acidity, the watersheds ranked as follows:

Table 6.4 Table summarizing mean, maximum, and minimum pH values recorded for watersheds with automated probes.

Watershed	Mean	Maximum	Minimum
1	6.97	7.42	6.43
4	7.12	8.04	5.89
5	7.35	7.95	6.55
6	7.39	7.88	6.39
9	7.05	7.91	5.88

Logically, pH dynamics for Hayward brook watersheds closely reflected those of alkalinity discussed in chapter 5, and hydrological characteristics. Watershed 1 on the Sunbury soils had one of the lowest average alkalinity values for the study period, as well as the lowest average pH. For many of the reasons discussed with conductivity, the range of values for the Sunbury watersheds was minimal as well. Weathering potential was low, and thus bicarbonate production was low as well, even during low flow periods. As well, the pH range (max-min) was found to be strongly correlated to total stream discharge over the study period (Figure 6.6). The watersheds with the greatest per hectare discharge over the study

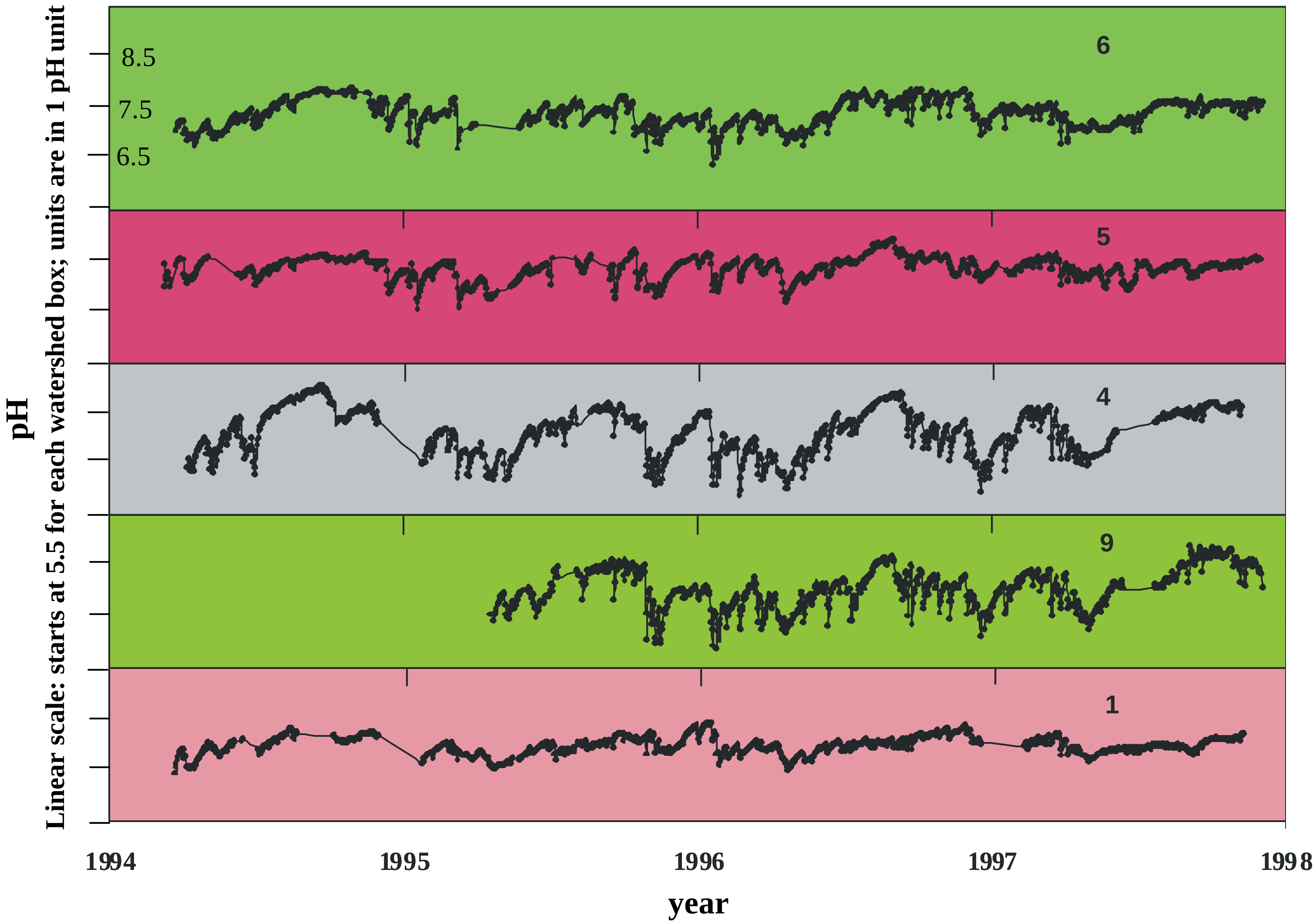

Figure 6.5 Graph illustrating pH in streamwater by watershed for the study period.

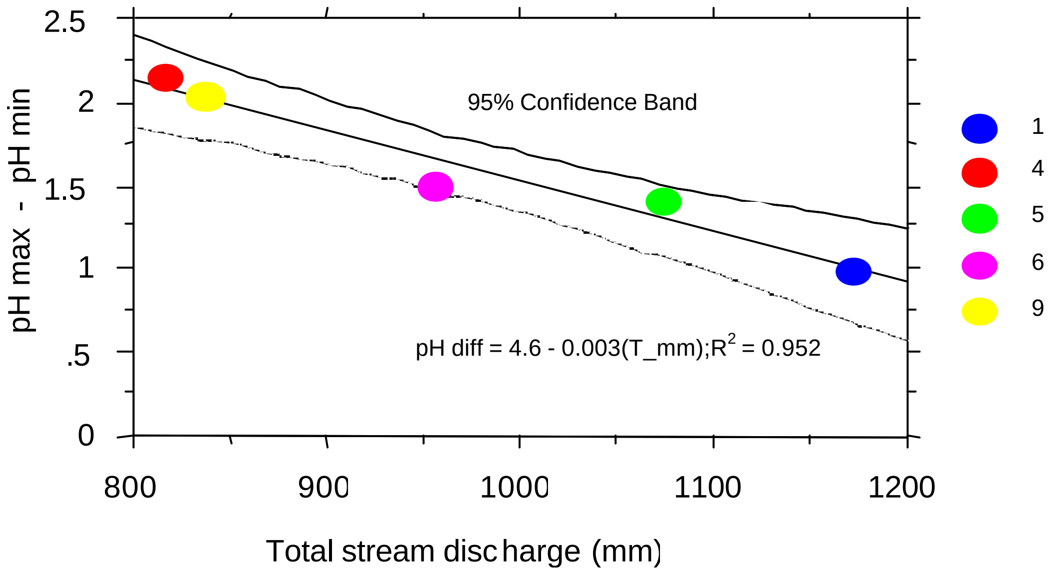

Figure 6.6 Graph illustrating the relationship between the pH range for the watersheds monitored and total discharge for the study period where pH diff = the range in pH, and T_mm = total discharge.

period showed the least range in pH and were the watersheds dominated by the highly permeable Sunbury soils. Watersheds 4 and 9 had intermediate pH's with respect to the other watersheds but had the highest range of values, for the same reasons discussed in conductivity. Watershed 4 was so flashy with respect to discharge, that in springtime, the pH was approaching that of snow (pH 5). In fact, watershed 4 has the highest recorded pH, at 8.04 during late summer 1994. Not surprisingly, watersheds 5 and 6 had the highest average pH's, with watershed 6 recording the highest in the study, along with the highest conductivity as discussed above.

Temperature

Stream temperature tracks were also highly synchronized with one another, but these data tracks were not synchronized to discharge, electrical conductivity or pH (Figure 6.7). Mean annual stream temperature differences only varied by 0.7 ^{O}C (Table 6.5) between the watersheds.

Table 6.5 Table summarizing mean, maximum, and minimum temperature values recorded for watersheds with automated probes.

Watershed	Mean	Maximum	Minimum
1	6.29	13.81	0
4	6.15	18.1	0
5	5.83	13.59	0
6	5.76	13.99	0
9	5.59	15.53	0

The watersheds with the largest temperature variations were those that were mostly fed by surface water. The watersheds with the least variations would be mostly fed by groundwater.

Turbidity

There was only limited synchronicity among the five turbidity tracks (Figure 6.8). Differences were primarily due to chronic sediment run-off from roads, occasional road construction activities, and within-stream erosion, depending on specific surface conditions and activities within each of these watersheds. Harvesting itself contributed little, except for the extra road construction and road traffic (Pomeroy, personal communication). The turbidity data is summarized in Table 6.6.

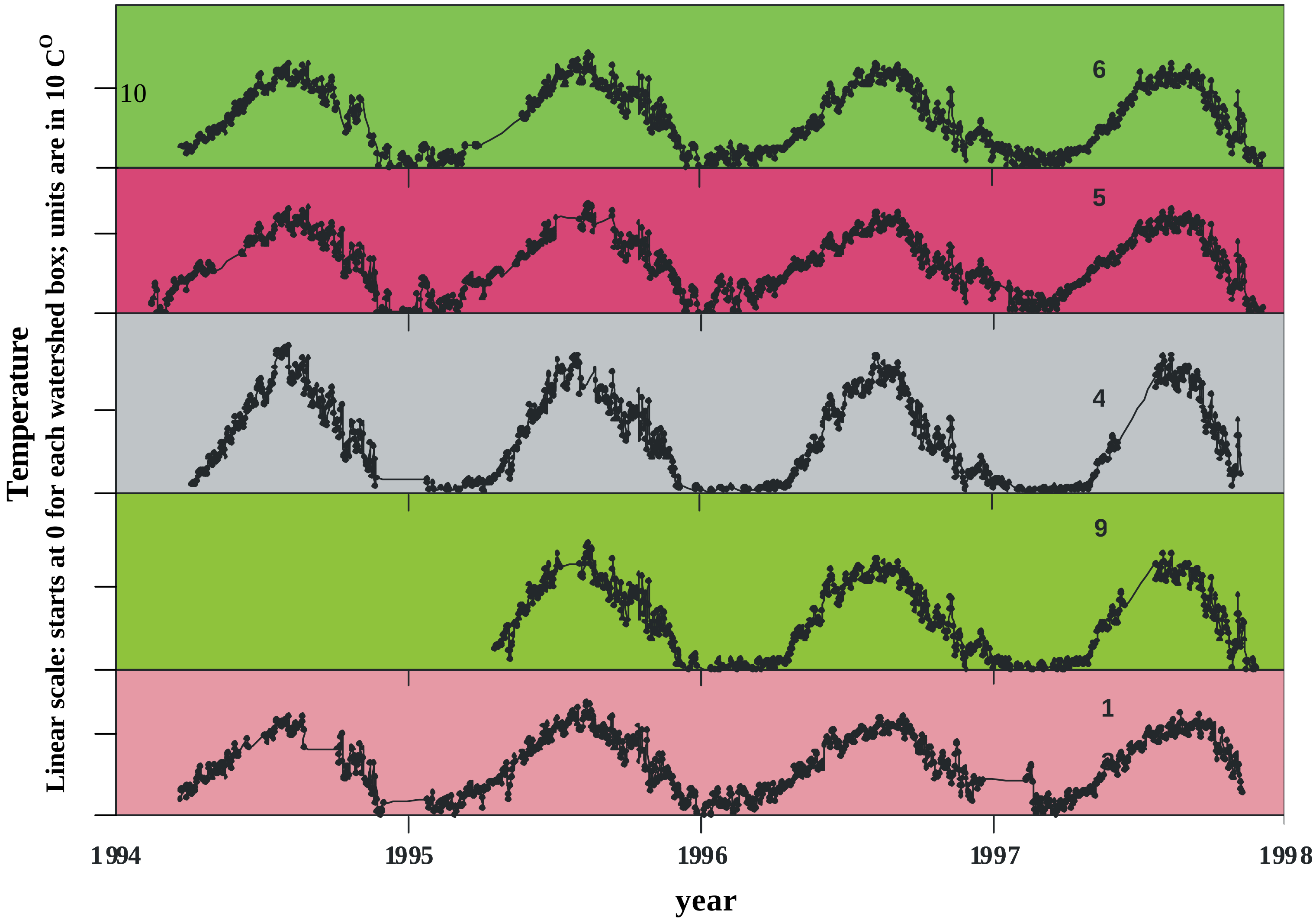

Figure 6.7 Graph illustrating temperature (^{O}C) in streamwater by watershed for the study period.

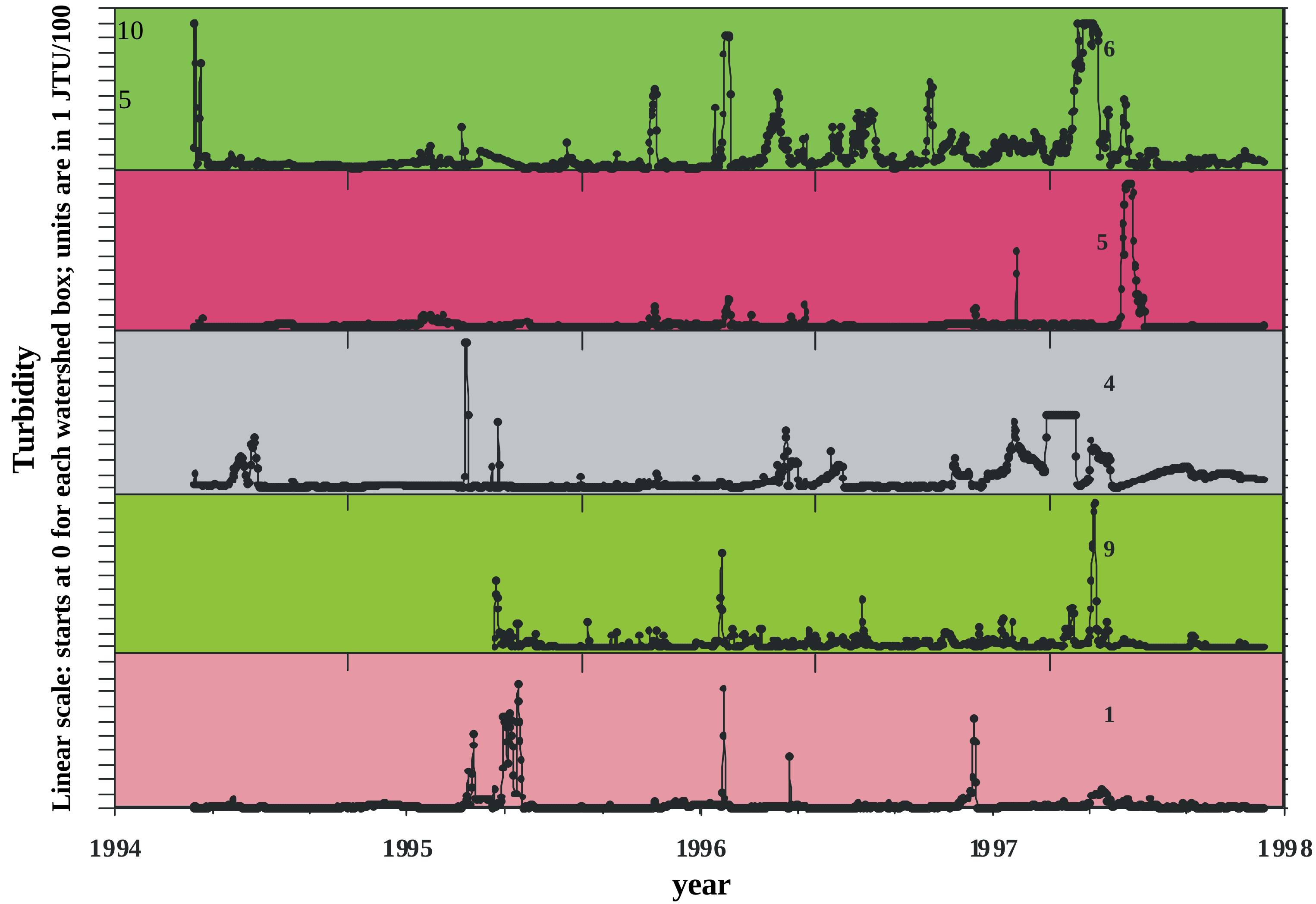

Figure 6.8 Graph illustrating turbidity (JTU/100) in streamwater by watershed for the study period.

Table 6.6 Table summarizing mean, maximum, and minimum turbidity values recorded for watersheds with automated probes.

Watershed	Mean	Maximum	Minimum
1	20	856	0.005
4	63	1000	0.005
5	42	1000	0.005
6	99	1000	0.005
9	45	1000	0.005

Dissolved Oxygen

The dissolved oxygen tracks were, in general, synchronized to stream temperature, both seasonally as well as episodically during sustained days of higher or lower stream temperature. Highest dissolved oxygen levels occurred in winter, when stream temperatures and biological oxygen consumption rates within the stream were lowest (Figure 6.9).

CONCLUSIONS

Continuous monitoring has revealed strong synchronicities between stream discharge, electrical conductivity and pH. Based on weekly sampling, synchronicities extended to the temporal stream variations for Ca, Mg, alkalinity, etc. Since taking all these measurements is an expensive undertaking, both in terms of equipment and personnel, it is suggested that routine measurements of stream discharge, electrical conductivity, pH, Ca, Mg, K, alkalinity can be replaced by empirically estimating many of these parameters based on a single continuous

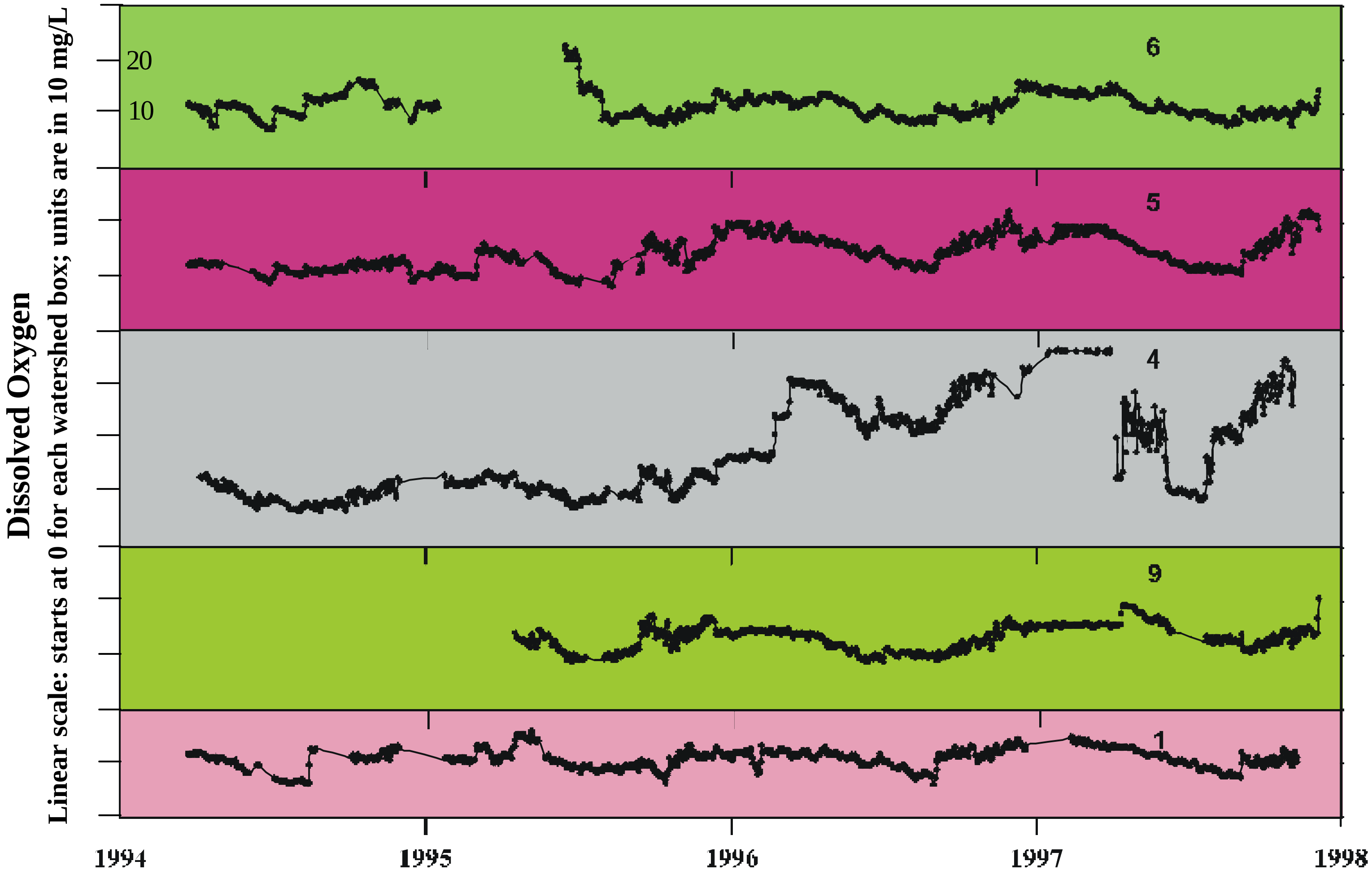

Figure 6.9 Graph illustrating dissolved oxygen (mg/L) in streamwater by watershed for the study period.

measurement such as stream discharge or electrical conductivity. The latter is likely the least expensive and also the physically most robust routine measurement. For stream discharge, matters of basin calibration and stream channel stability often add uncertainty to specific measurements (Dingman 2002). Specific correlations between stream discharge, electrical conductivity, and the other parameters listed above can be obtained through occasional short-term, high-flow, low- flow calibrations, done for each basin. Knowing the effect of basin characteristics on these calibrations would enable additional generalization of these calibrations to other watershed of similar type.

Stream temperature appears to be perhaps the most robust and easily determined streamwater parameter. Small temperature probes, each encased with a small memory device (Pomeroy *et al 1998*), can be left unattended in streams for over five years, depending on durability of the device-internal energy cell, the temperature sampling rate, and the amount of data that can be stored. The temperature record so stored can be retrieved later on, for analysis, and this temperature record can then also be used to estimate other streamwater parameters such as dissolved oxygen.

CHAPTER 7

PRE- AND POST-HARVEST DIFFERENCES IN WATER CHEMISTRY AND STREAM DISCHARGE FOR THE HAYWARD BROOK WATERSHED STUDY

INTRODUCTION

The HBWS provided a unique opportunity to compare a number of different catchments of different sizes and geomorphology that were literally adjacent to each other. It was assumed that any differences in either stream chemistry, or discharge was entirely due to features of the watershed, and not atmospheric deposition or spatial differences in precipitation. This provided an opportunity to compare between watersheds, and can be especially useful for evaluating the impacts of forest harvesting when some of the nested watersheds are left as controls. The general theme of this chapter has been discussed at depth in Chapters three and four, and will not be repeated here, however the data collected for the HBWS presented some unique problems that should be detailed.

OBJECTIVES

It is the purpose of this chapter to evaluate and discuss the effects of harvesting in some of the HBWS watersheds on stream discharge and water quality. The problems encountered during the evaluation of this that are unique to the HBWS will also be detailed.

METHODS

Water chemistry grab sampling began in the summer of 1993 and discharge/probe measurements in March of 1994. Since harvesting began in the summer of 1995, this leaves only minimal pre-harvest data to give an appreciation of how the watersheds behaved in the natural, undisturbed state. Typically, studies evaluate harvest repercussions through statistical analysis of pre- and post-harvest data, assuming the sample size of each to be sufficient to make results valid. Care must be taken when interpreting the results, as naturally changing conditions, and data characteristics can be mistaken for harvest effects. Many of the remote sites were not visited for extended periods, due to weather or manpower problems and thus data sets for some watersheds are minimal, and incomplete. This makes drawing firm conclusions regarding harvest effects inappropriate.

Even with the Nashwaak study with seven years of pre-harvest data, Jewett (1995) discussed the problem of yearly variability in precipitation making stream chemistry naturally different. This is especially relevant when attempting to evaluate the differences in pre- and post-harvest streamflow, and attributing it to forest harvesting. For stream chemistry, Jewett et al. (1996) addressed this issue by applying simple means tests to the monthly averages for ion concentrations for the pre- and post-harvest time periods. For discharge, which is more closely related to precipitation, a linear equation was developed with regression analysis to characterize the pre-harvest cumulative difference between the watersheds. This line was projected into the future as a means of predicting what the post-harvest cumulative differences would be should the watershed remain untreated. The difference between the predicted line and the actual line were attributed to harvest effects. Similar approaches for ion flux rates were used.

In the case of the HBWS, numerous attempts to generate reliable regression equations between the control watershed (watershed 4) and the treated watersheds failed. This is partly due to the minimal data set (March 1994 to June 1995), but also the erratic nature of the control, watershed 4.

RESULTS AND DISCUSSION

Results of the above analysis are displayed in Figures 7.1.1-7.6.3. Each harvested watershed will be discussed individually, with comparisons to unharvested watersheds made as needed. When probe data is available (see Table 1.1) it is also used in this analysis.

The majority of ions or physical properties showed no significant differences between the pre- and post-harvest periods, while there were exceptions.

Watershed 1

This watershed was selectively cut on 15.5% of its area, with a 60 meter buffer, resulting in roughly 5% canopy removal. The simple means test showed a significant increase in the concentration of sodium, magnesium and potassium in the streams after harvest (see Figures 7.1.1 to 7.1.3). This is discounted as a harvest effect for a number of different reasons. As has been discussed, concentrations of weathering byproducts generally increase with decreasing discharge. Figures 7.1.4 and 7.1.3 show how both precipitation and discharge are significantly less after harvest, probably inducing the

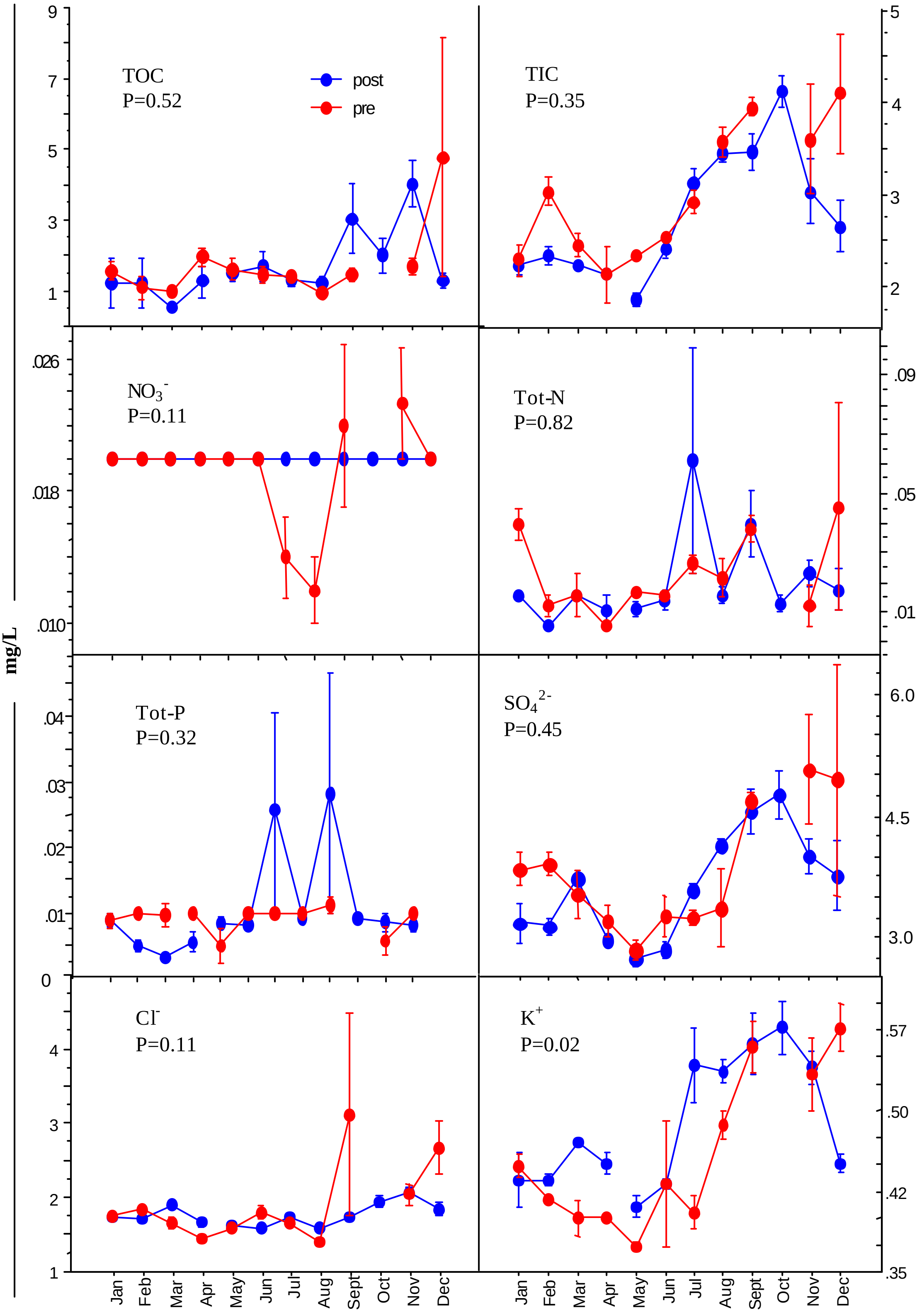

Figure 7.1.1 Mean monthly streamwater concentrations (mg/L) (Cl^-, K^+, Tot-P, SO_4^{2-}, NO_3^-, TOT-N, TOC, TIC) for watershed 1. Pre - versus post-harvest significance levels (P-values) are given, with error bars for monthly means.

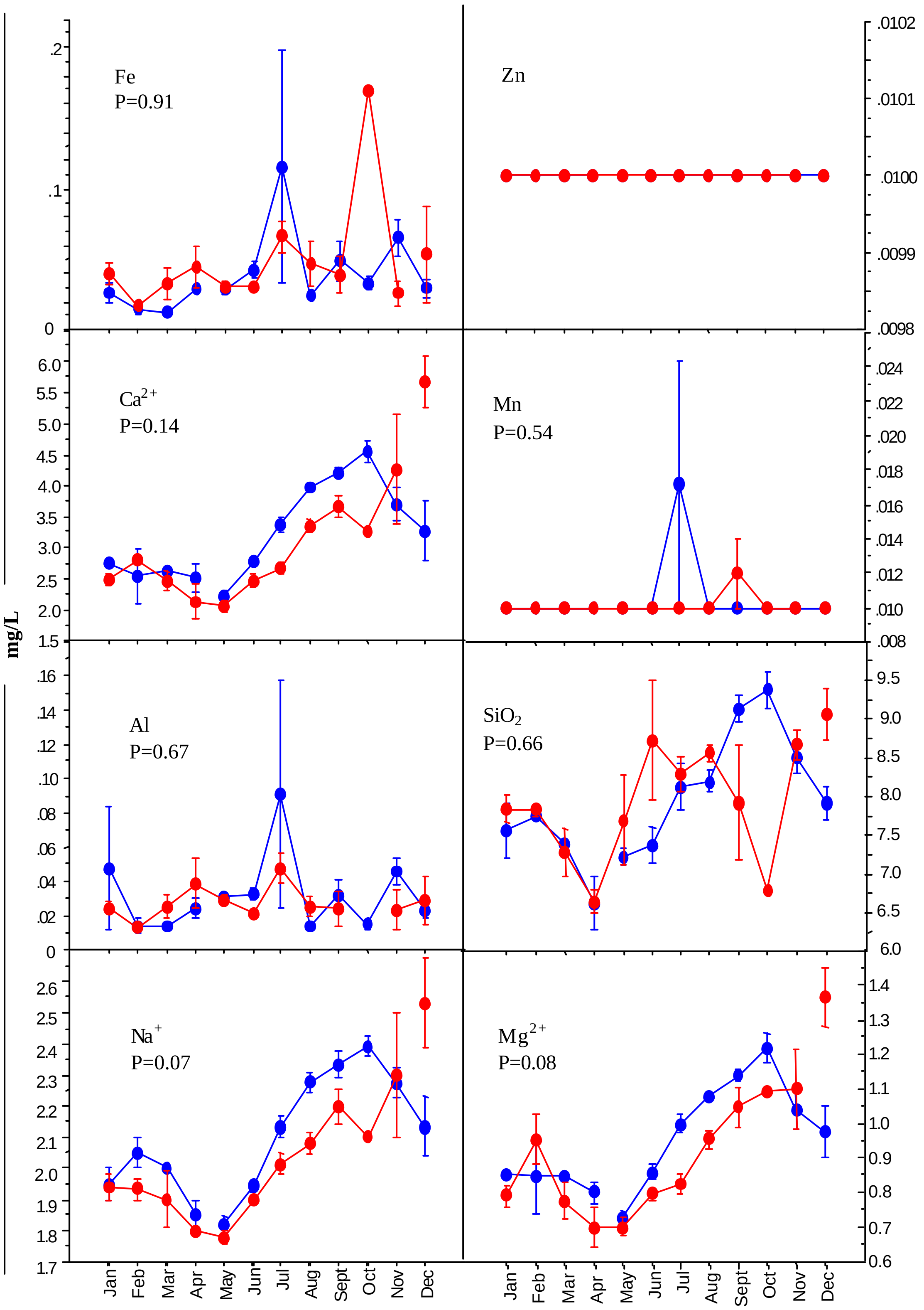

Figure 7.1.2 Mean monthly streamwater concentrations (mg/L) (Na^+, Mg^{2+}, Al, SiO_2, Ca^{2+}, Mn, Fe, Zn) for watershed 1. Pre- versus post-harvest significance levels (P-values) are given, with error bars for monthly means.

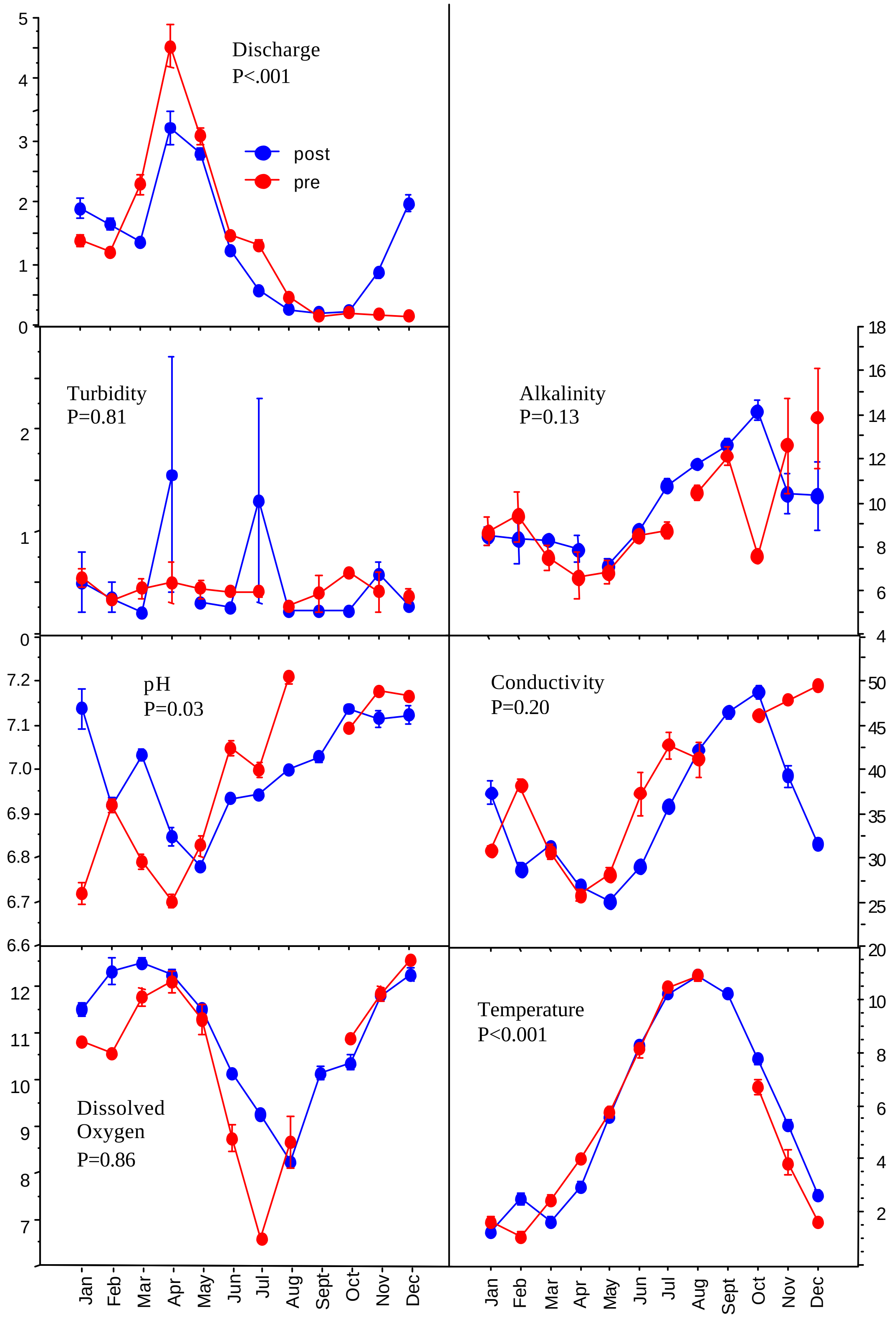

Figure 7.1.3 Mean monthly values for dissolved oxygen (mg/L), temperature (^{O}C), pH, conductivity (sie/cm), turbidity (JTU), alkalinity (mg/L) and discharge (mm) for watershed 1. Pre-verses post harvest significance levels (P-values) are given, with error bars for monthly means.

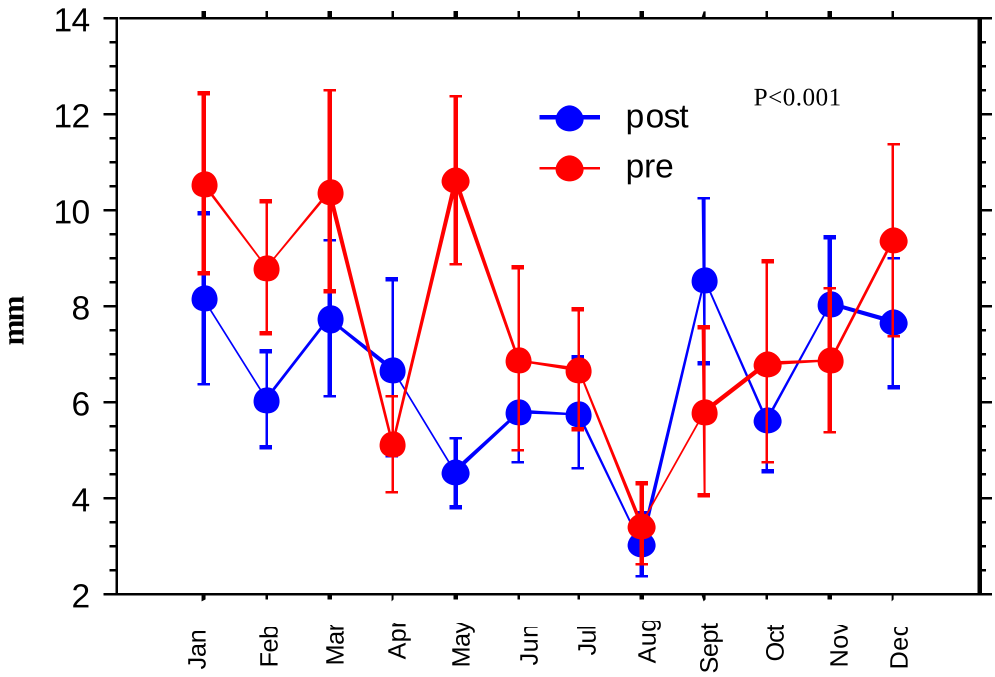

Figure 7.1.4 Mean monthly precipitation (mm). Pre-verses post harvest significance level (P-value) is given with error bars for monthly means.

observed changes in sodium and magnesium. Careful observation of the graph for calcium reveals similar post harvest trends, though not statistically significant.

The increase in post harvest potassium observed at watershed 1 was also noticed at watersheds 9 and 10. Jewett *et al.* attributed summer potassium peaks to be due to the warm soil temperatures enhancing microbial metabolism. Summer air temperature was significantly greater in the post-harvest period, and may have caused the higher potassium levels, though it was inconsistent with other watersheds. Jewett *et al.* noticed a 0.20mg/L increase in potassium levels following a 100% clearcut, and it is doubtful an intervention involving no clearcutting, and only 8% canopy removal would produce measurable increases. The increase in pH after harvest was also most likely due to the decreased discharge, and higher base content, as discussed in Chapter 5.

Watershed 2

Watershed 2 was selectively harvested on 17.7% of it's area, with a 60 meter buffer, resulting in roughly 6% canopy removal. Access to this site was limited, and no data was collected for the first four months of the calendar year after harvest, or October, making overall conclusions impossible. The minimal summer data, showed some differences in water quality parameters, all of which are within the natural variation, and not attributed to harvesting (see Figures 7.2.1 to 7.2.3). Though the differences in nitrate seem significant, all measurements of nitrate were recorded as being below the analytical detection limit, and absolute values not available.

Watershed 5

Watershed 5 is the largest watershed in the study (924 acres) with clearcut harvesting on less than 5% of the area. With no road crossings and a thirty-meter buffer,

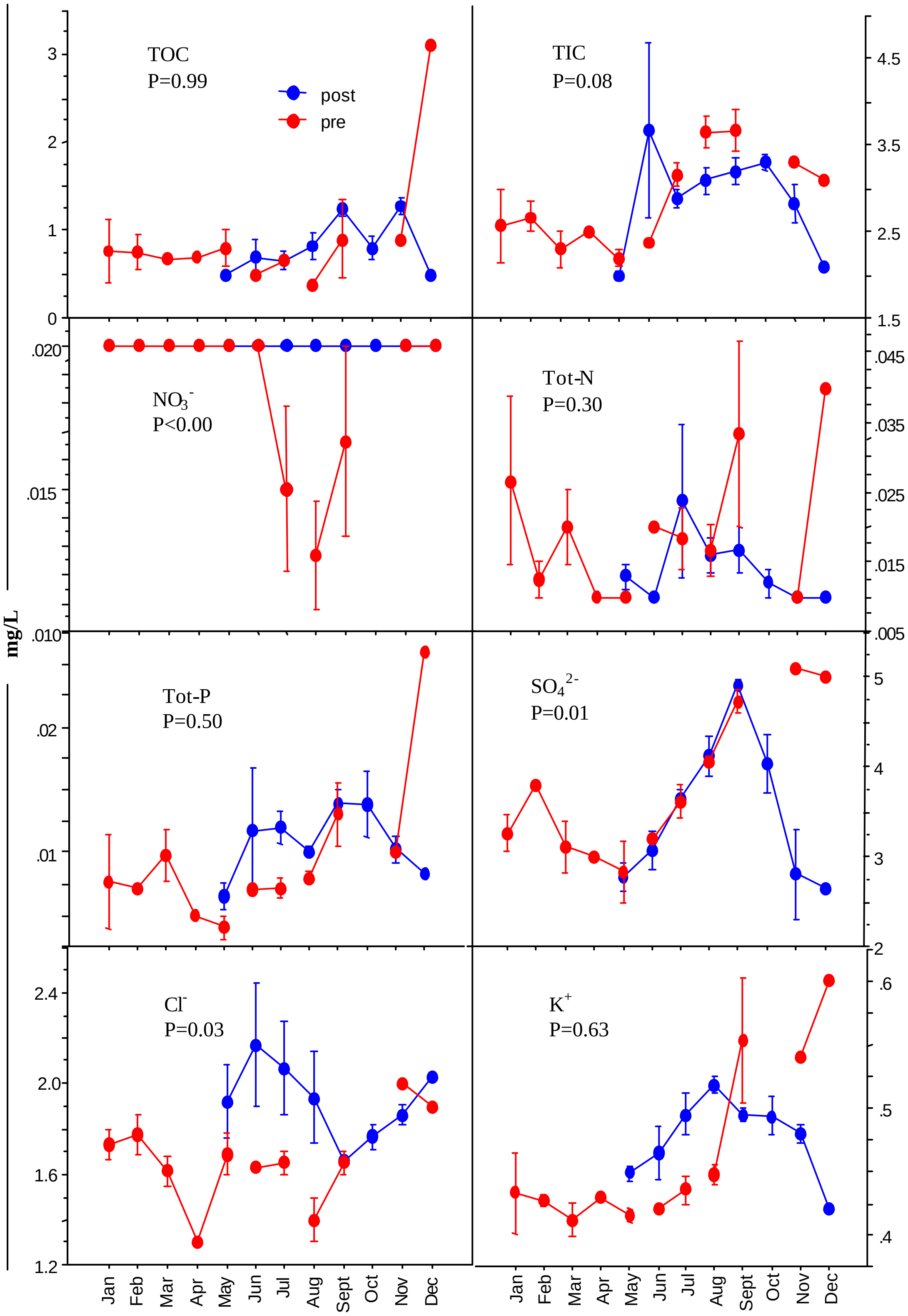

Figure 7.2.1 Mean monthly streamwater concentrations (mg/L) (Cl^-, K^+, P, SO_4^{2+}, NO_3^-, TOT-N, TOC, TIC) for watershed 2. Pre -verses post harvest significance levels (P-values) are given, with error bars for monthly means.

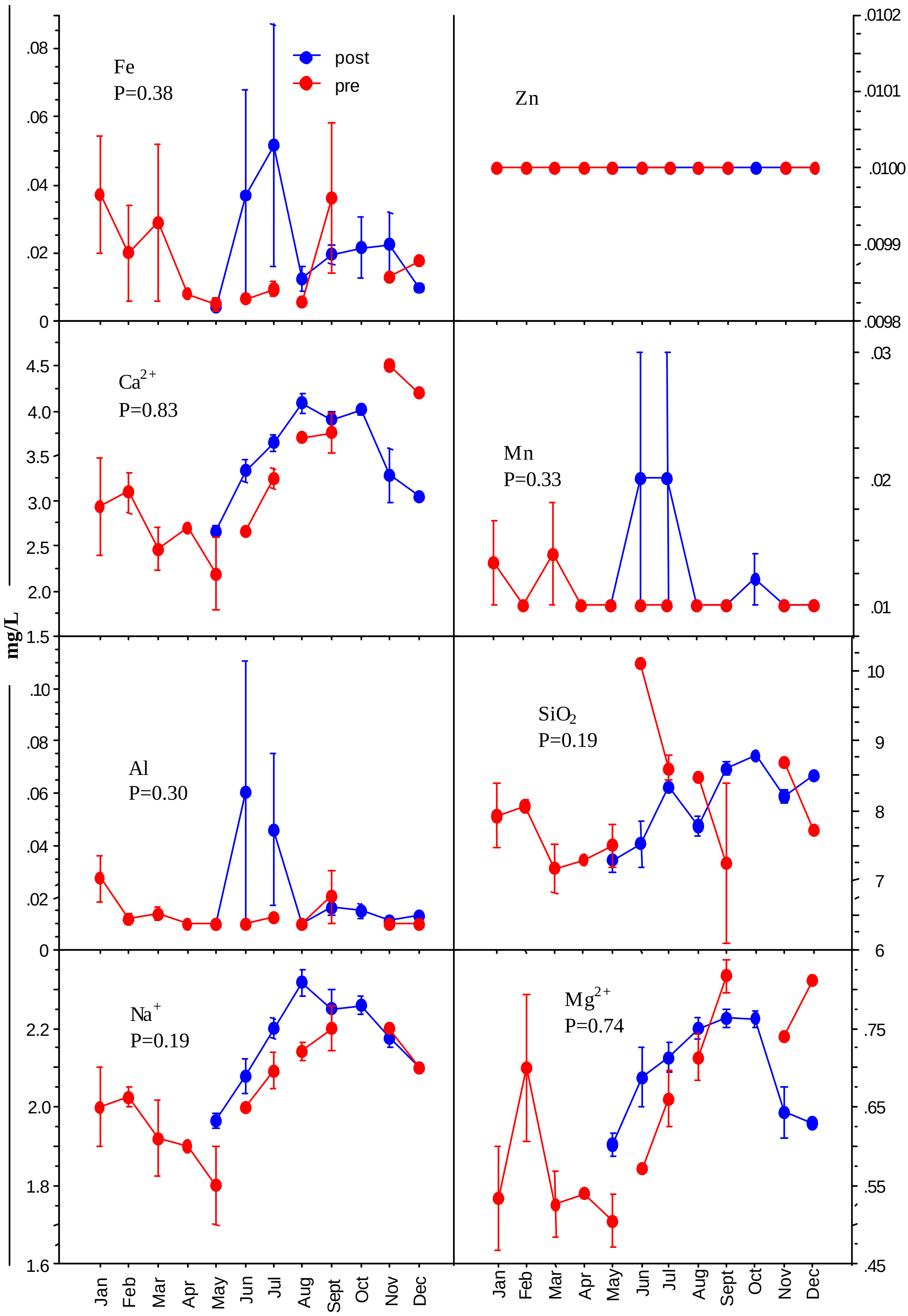

Figure 7.2.2 Mean monthly streamwater concentrations(mg/L) (Na^{+}, Mg^{2+}, Al, SiO_2,, Ca^{2+}, Mn, Fe, Zn) for watershed 2. Pre -verses post harvest significance levels (P-values) are given, with error bars for monthly means.

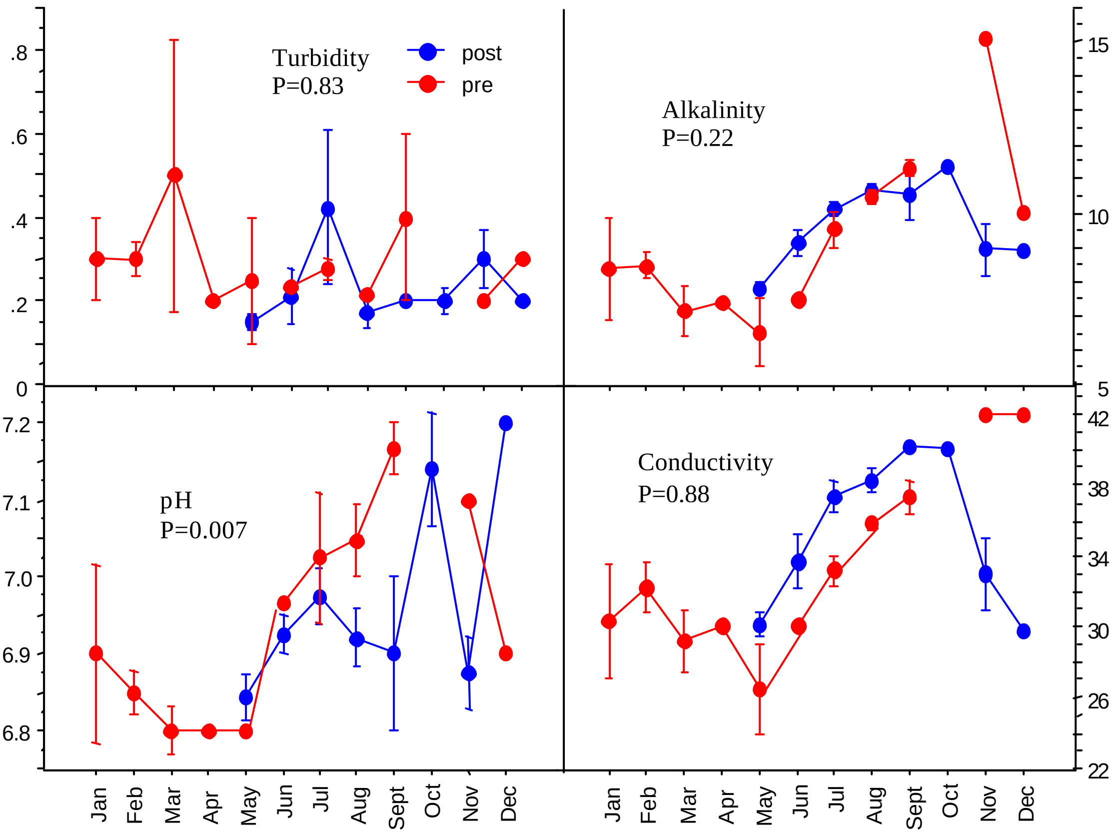

Figure 7.2.3 Mean monthly values for pH, conductivity (sie/cm), turbidity (JTU), alkalinity (mg/L) and discharge (mm) for watershed 2. Pre-verses post harvest significance levels (P values) are given, with error bars for monthly means.

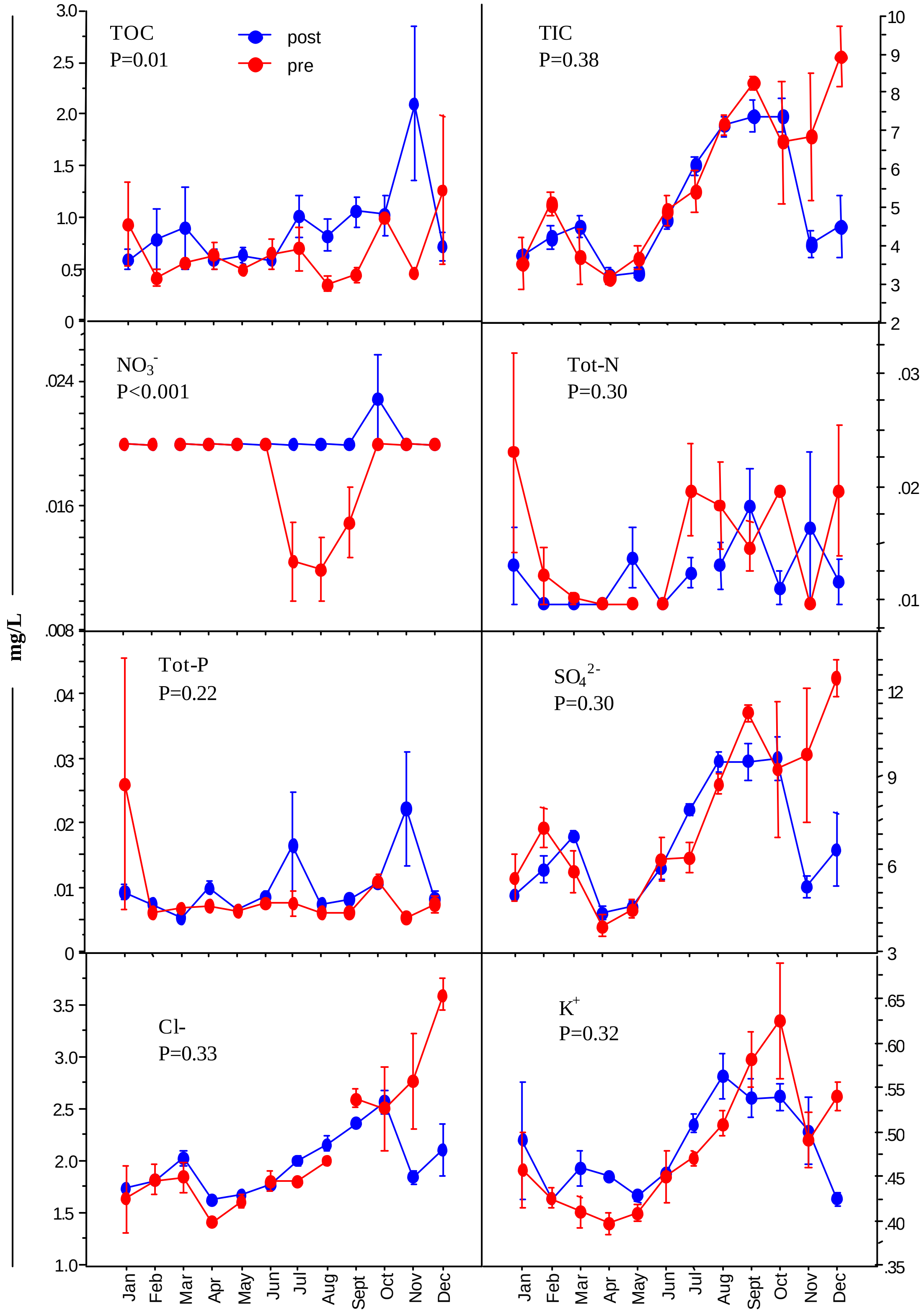

Figure 7.3.1 Mean monthly streamwater concentrations (Cl^-, K^+, P, SO_4^{2-}, NO_3^-, TOT-N, TOC, TIC) for watershed 5. Pre-verses post harvest significance levels (P-values) are given, with error bars for monthly means.

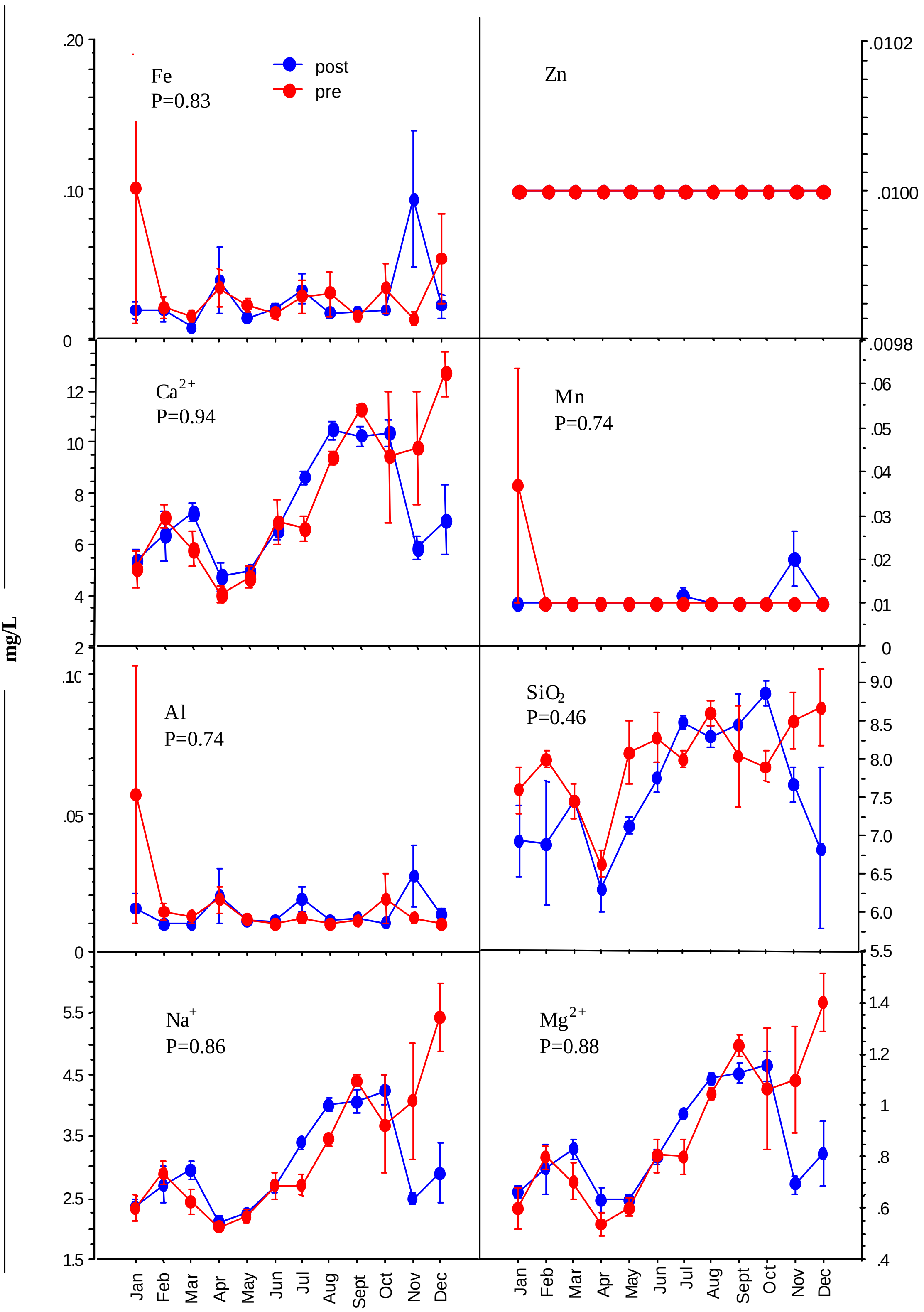

Figure 7.3.2 Mean monthly streamwater concentrations (Na^+, Mg^{2+}, Al, SiO_2,, Ca^{2+}, Mn, Fe, Zn) for watershed 5. Pre-verses post harvest significance levels (P-values) are given, with error bars for monthly means.

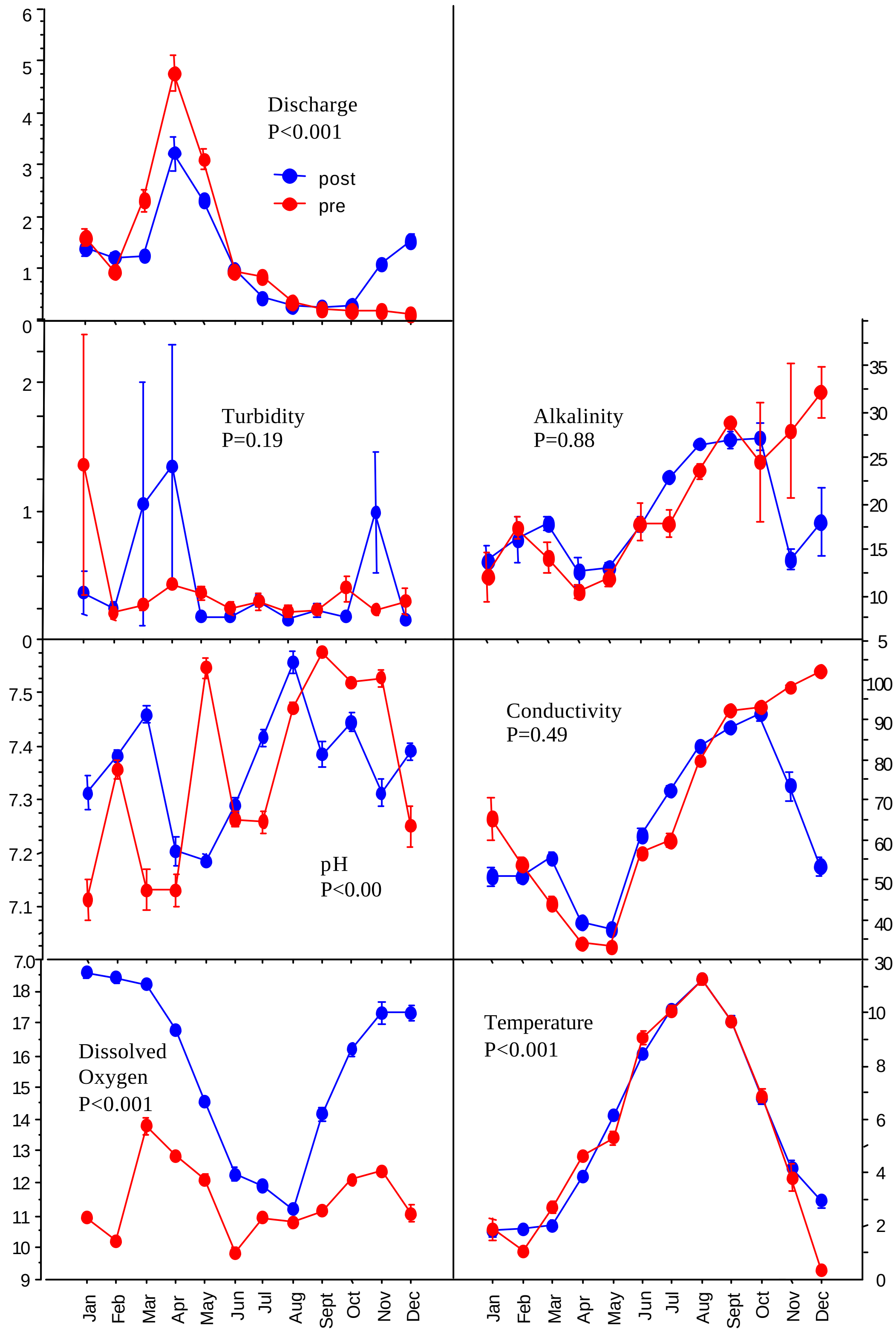

Figure 7.3.3 Mean monthly values for dissolved oxygen (mg/L), temperature (^{O}C), pH, conductivity (sie/cm), turbidity (JTU), alkalinity (mg/L) and discharge (mm) for watershed 5. Pre-verses post harvest significance levels (P values) are given, with error bars for monthly means.

any differences in pre- and post-harvesting water quality, are likely due to natural variability (see Figures 7.3.1 to 7.3.3). This applies to temperature, discharge and pH, and have been discussed above. Nitrate levels were also at the analytical detection limit for all samples in this study. Total organic carbon levels were noted to have increased in a number of watersheds, including reference watersheds. This is most likely due to the higher summer/fall temperatures inducing faster decomposition of summer litter (see Chapter 5).

Watershed 6

Watershed 6 was clearcut on 11% of its area, with a 30 meter buffer. The significantly higher levels of TOC, and pre- and post-harvest differences in discharge and temperature have already been discussed, and will not be repeated (see Figures 7.4.1 to 7.4.3). Watershed 6 provided the only confident example of a pre and post harvest difference that can be attributed to forest management. During road construction, a galvanized steel culvert was installed, just above the station. This induced significantly higher turbidity, and zinc levels for a period of five months after installation. The bound zinc was occasionally re suspended during high flow periods.

Watershed 9

Watershed 9 was the second largest watershed in the study, and the most intensively managed. It was over 10% clearcut, and 22% selectively cut, resulting in roughly 17% canopy removal. There are numerous roads in the watershed, though none crossing major tributaries. Potassium was the only element with significantly higher levels during the post harvest period, probably for the same reasons as discussed above (see Figures 7.5.1 to 7.5.3).

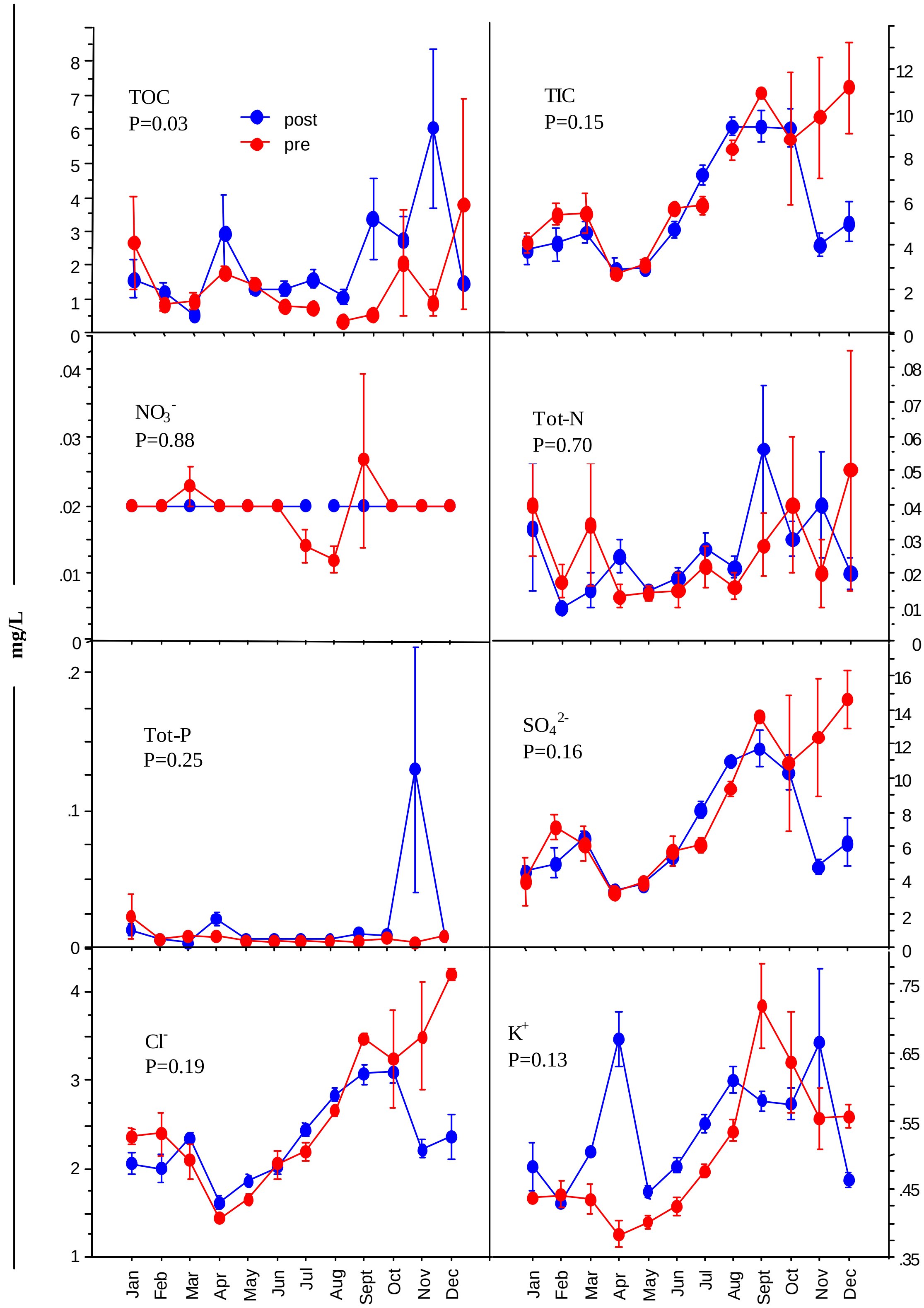

Figure 7.4.1 Mean monthly streamwater concentrations (Cl^-, K^+, P, SO_4^{2-}, NO_3^-, TOT-N, TOC, TIC) for watershed 6. Pre-verses post harvest significance levels (P-values) are given, with error bars for monthly means.

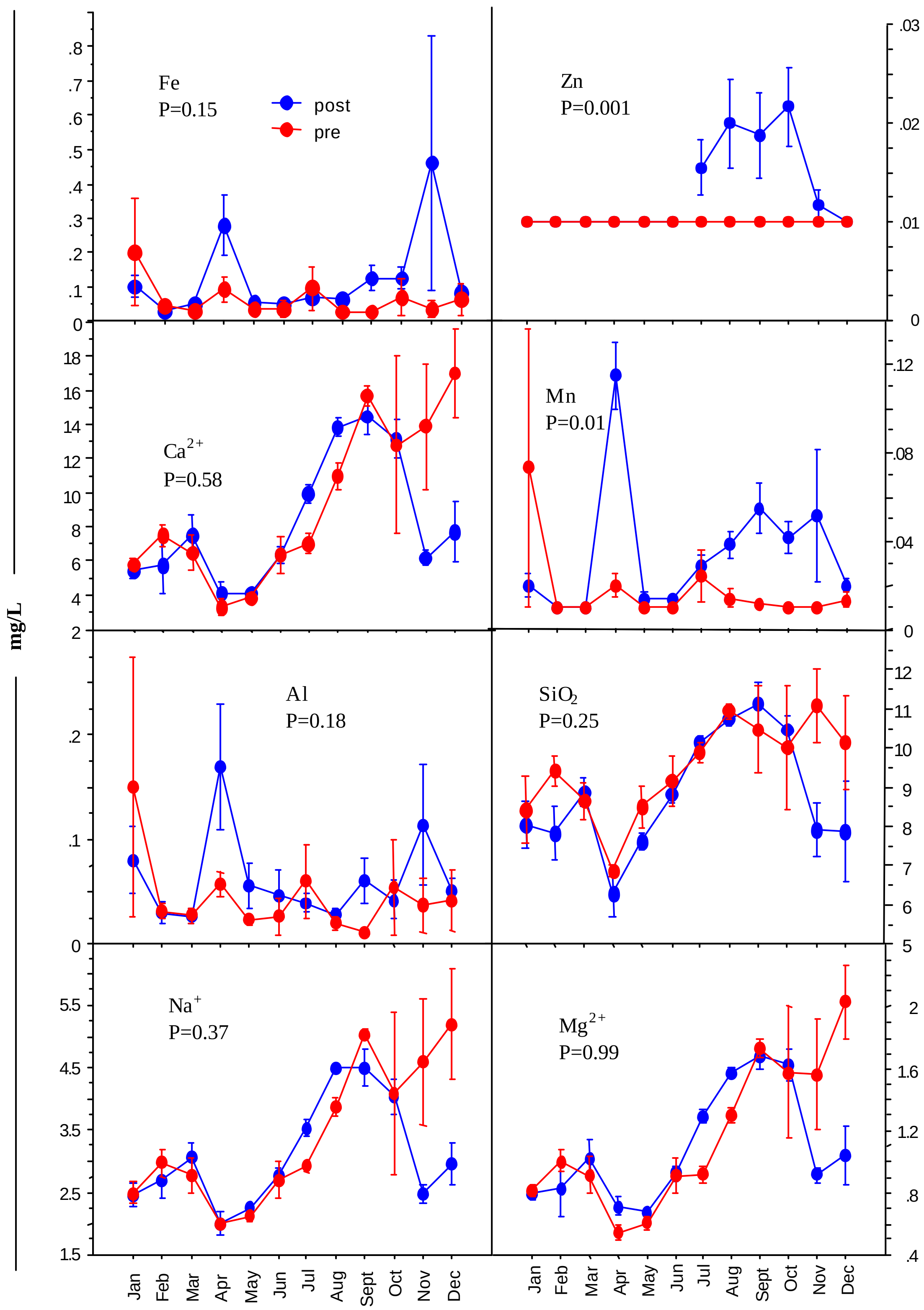

Figure 7.4.2 Mean monthly streamwater concentrations (Na^+, Mg^{2+}, Al, SiO_2, Ca^{2+}, Mn, Fe, Zn) for watershed 6. Pre-verses post harvest significance levels (P-values) are given, with error bars for monthly means.

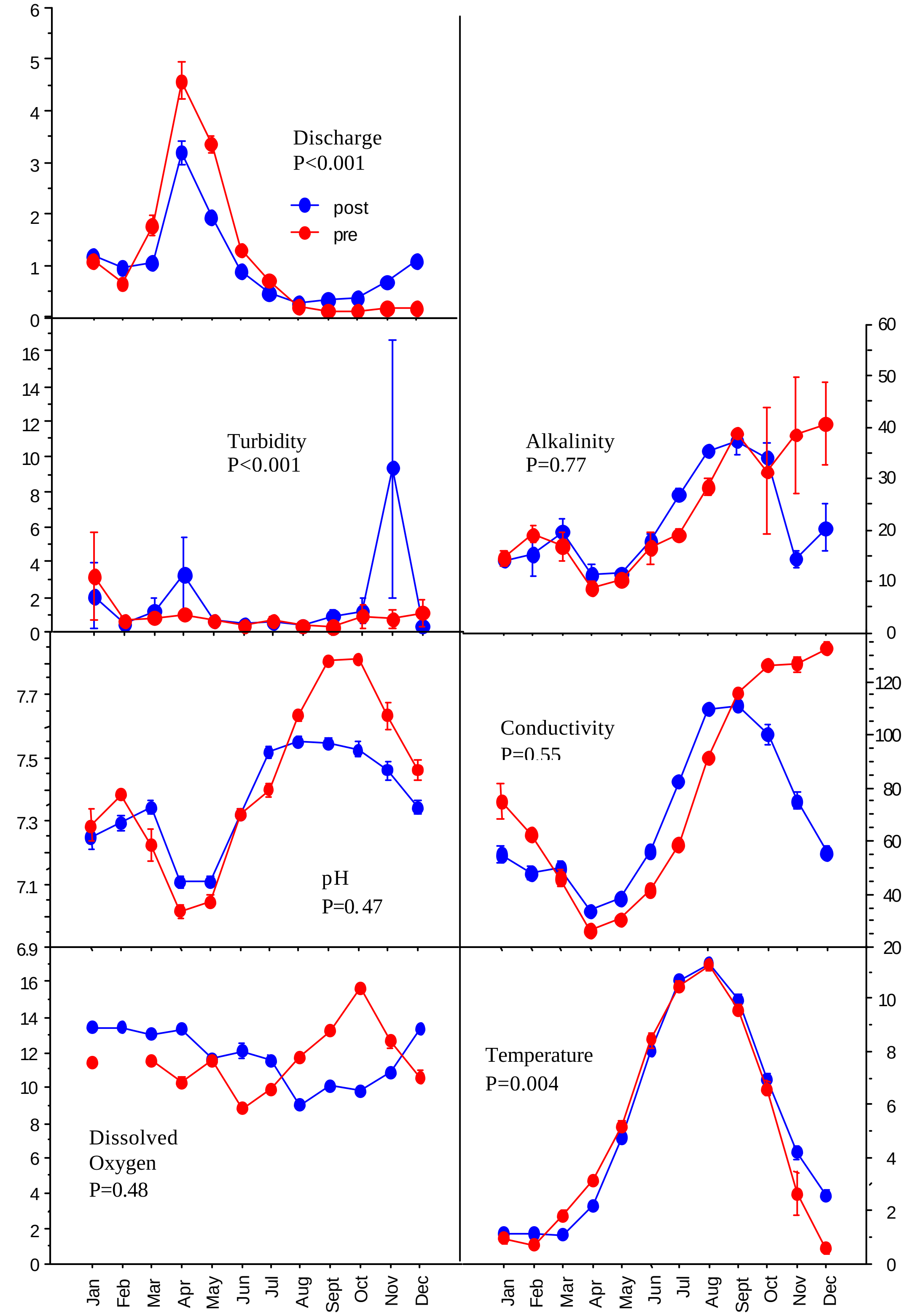

Figure 7.4.3 Mean monthly values for dissolved oxygen (mg/L), temperature (^{O}C), pH, conductivity (sie/cm), turbidity (JTU), alkalinity (mg/L) and discharge (mm) watershed 6. Pre-verses post harvest significance levels (P-values) are given, with error bars for monthly means.

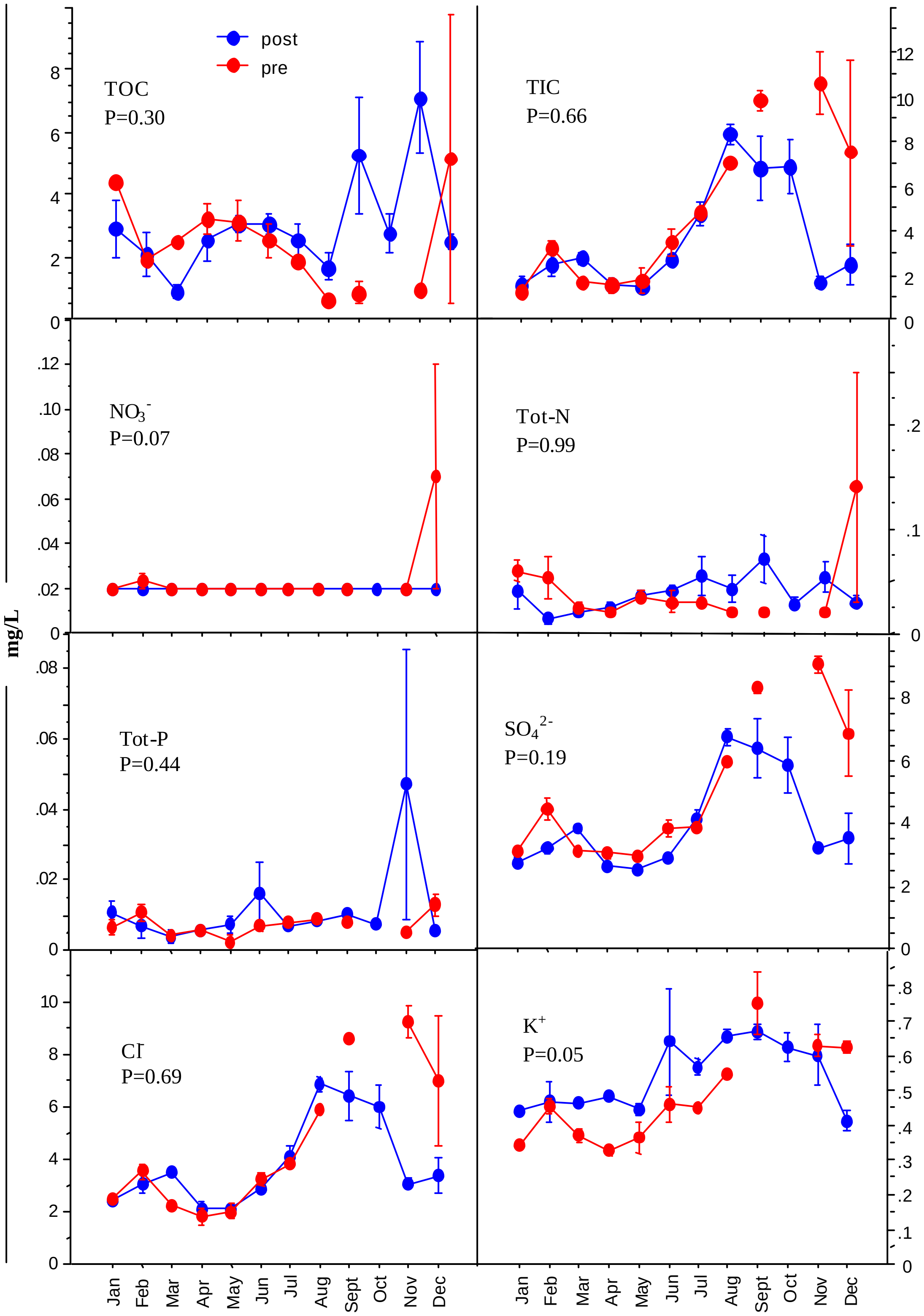

Figure 7.5.1 Mean monthly streamwater concentrations (Cl^-, K^+, P, SO_4^{2-}, NO_3^-, TOT-N, TOC, TIC) for watershed 9. Pre-verses post harvest significance levels (P-values) are given, with error bars for monthly means.

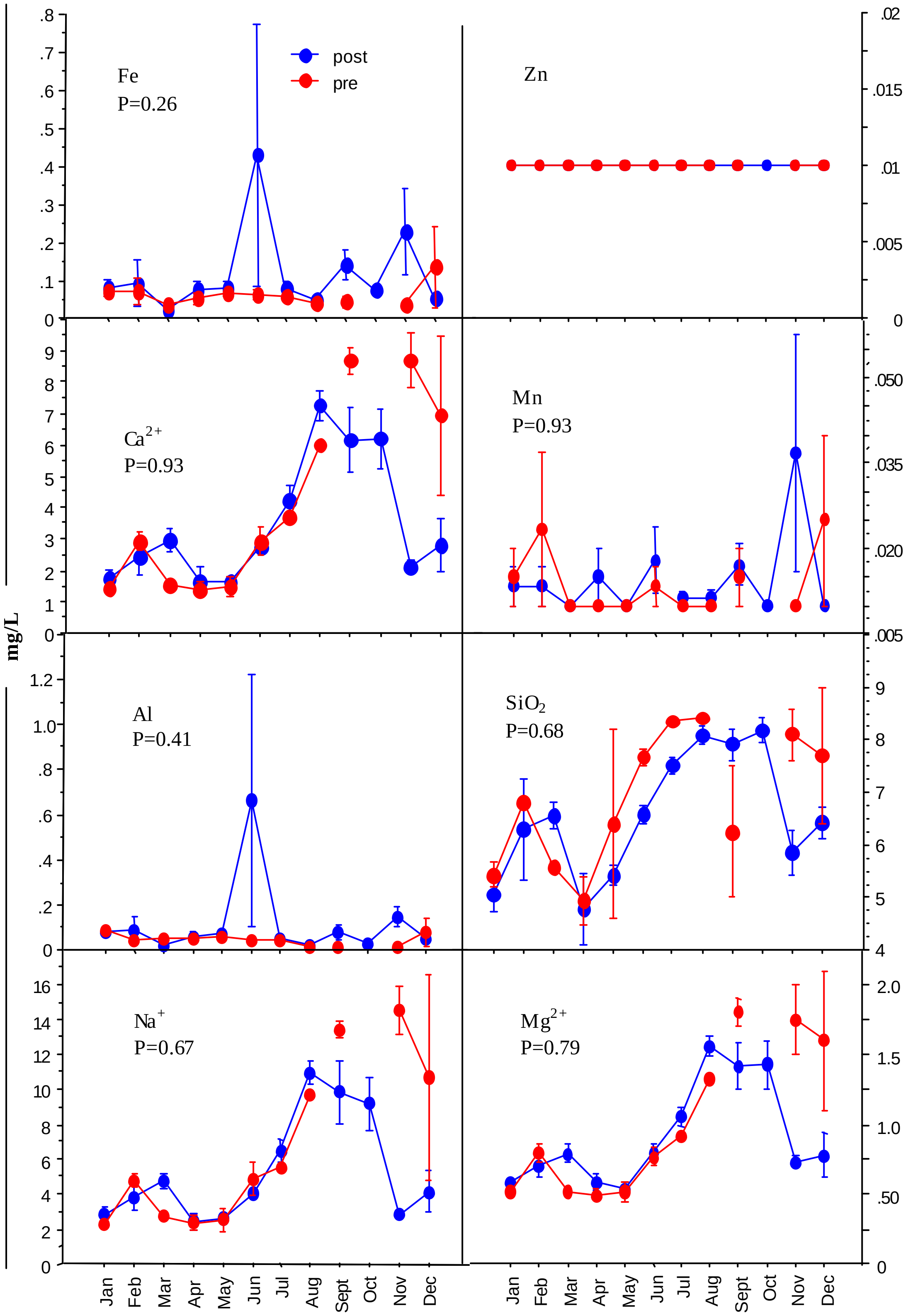

Figure 7.5.2 Mean monthly streamwater concentrations (Na^+, Mg^{2+}, Al, SiO_2, Ca^{2+}, Mn, Fe, Zn) for watershed 9. Pre-verses post harvest significance levels (P-values) are given, with error bars for monthly means.

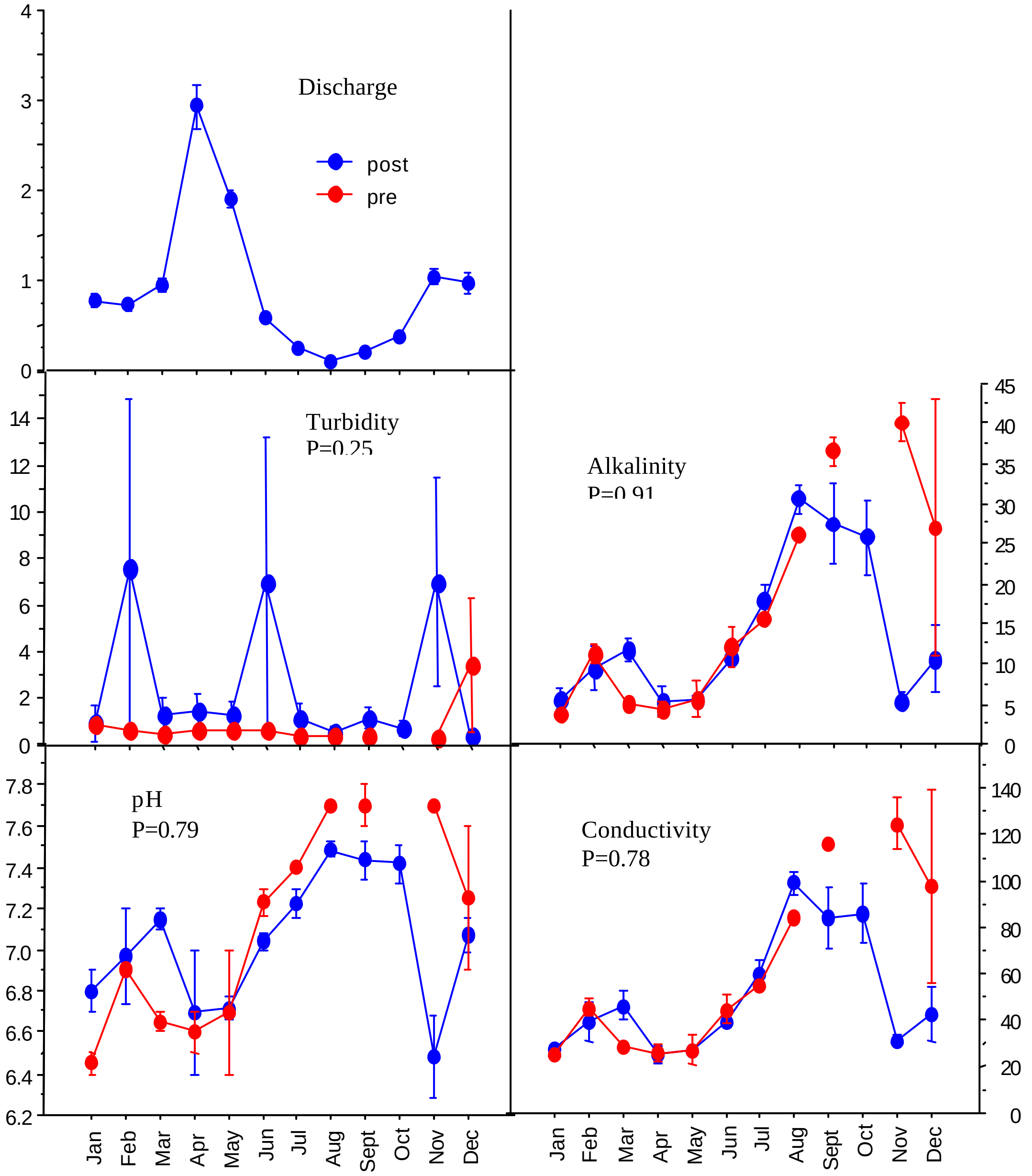

Figure 7.5.3 Mean monthly values for dissolved oxygen (mg/L), temperature (^{O}C), pH, conductivity (sie/cm), turbidity (JTU), alkalinity (mg/L) and discharge (mm) for watershed 9. Pre-verses post harvest significance levels (P-values) are given, with error bars for monthly means.

Watershed 10

Watershed 10 was selectively cut over 21% of its area, with no significant increases in any element during the post harvest period (see Figures 7.6.1 to 7.6.3).

CONCLUSIONS

The findings of this chapter supported the findings of other authors, as described in the literature reviews in Chapters 3 and 4. Harvesting of such small portions of a watershed (<20% canopy removal) does not produce a measurable effect on water quality and discharge, especially with such minimal pre-harvest data. The exceptions are in situations where watercourse crossings disturb streambeds sufficiently to increase turbidity, and galvanized pipes are used, which contribute zinc to levels above background. In order to isolate smaller effects, if present, the pre-harvest period must be long enough to gain a clear understanding of the natural geochemical and hydrological characteristics of the watershed. This enables the researcher to say with greater confidence that a small difference before and after harvesting is due to the actual modification of the natural environment, and not naturally changing conditions.

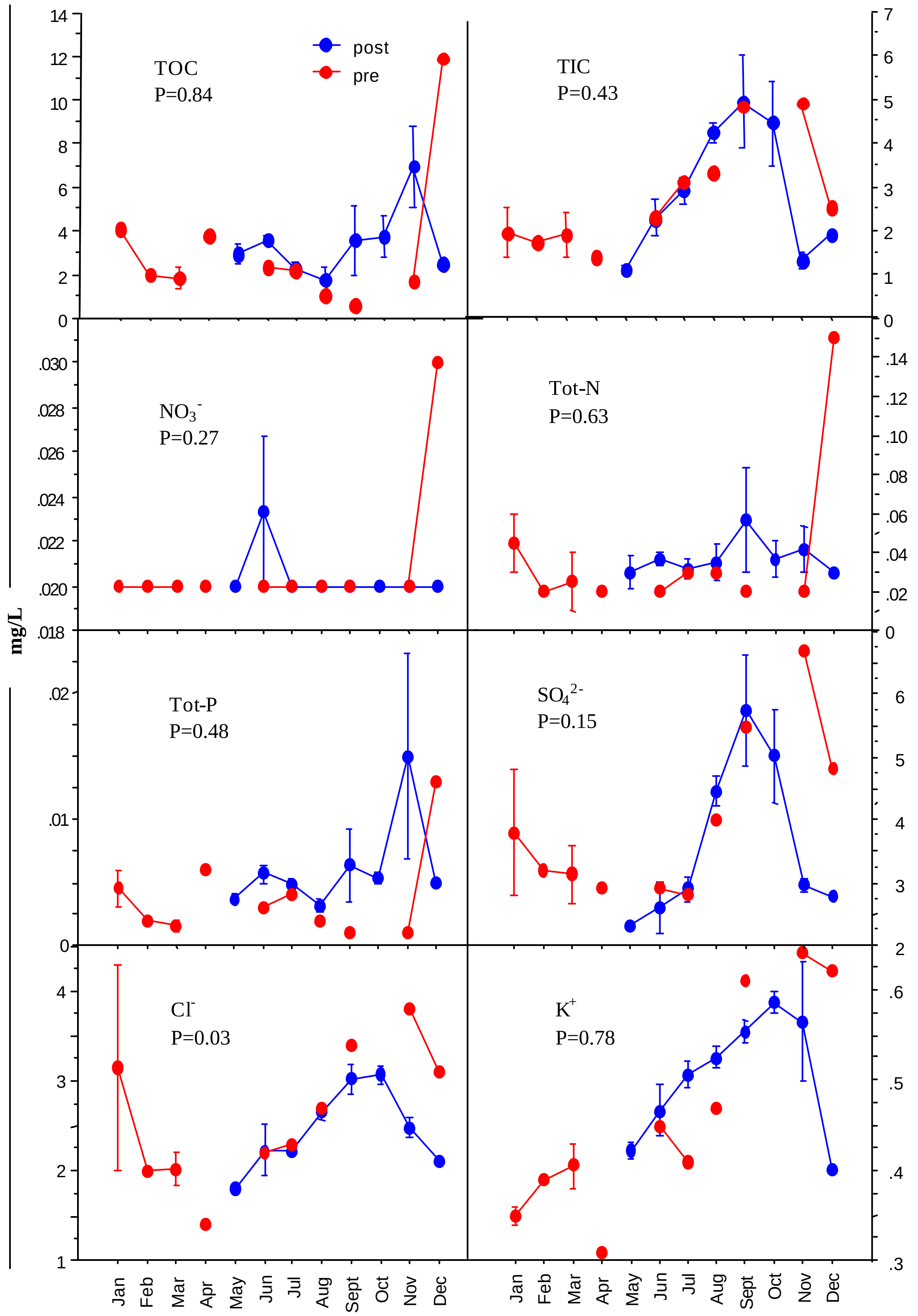

Figure 7.6.1 Mean monthly streamwater concentrations (Cl^-, K^+, P, SO_4^{2-}, NO_3^-, TOT-N, TOC, TIC) for watershed 10. Pre-verses post harvest significance levels (P-values) are given, with error bars for monthly means.

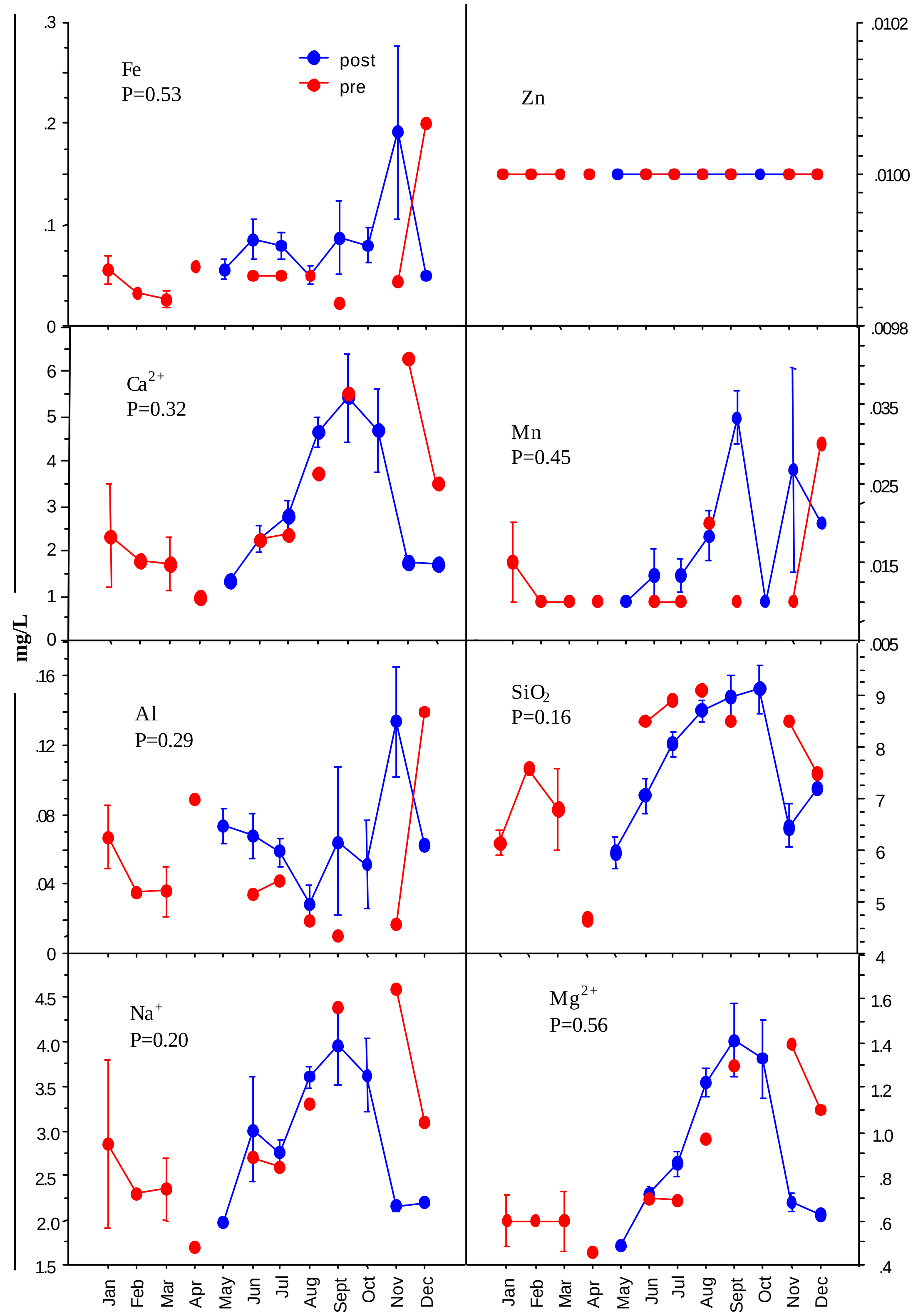

Figure 7.6.2 Mean monthly streamwater concentrations(mg/L) (Na^+, Mg^{2+}, Al, SiO_2, Ca^{2+}, Mn, Fe, Zn) for watershed 10. Pre-verses post harvest significance levels (P-values) are given, with error bars for monthly means.

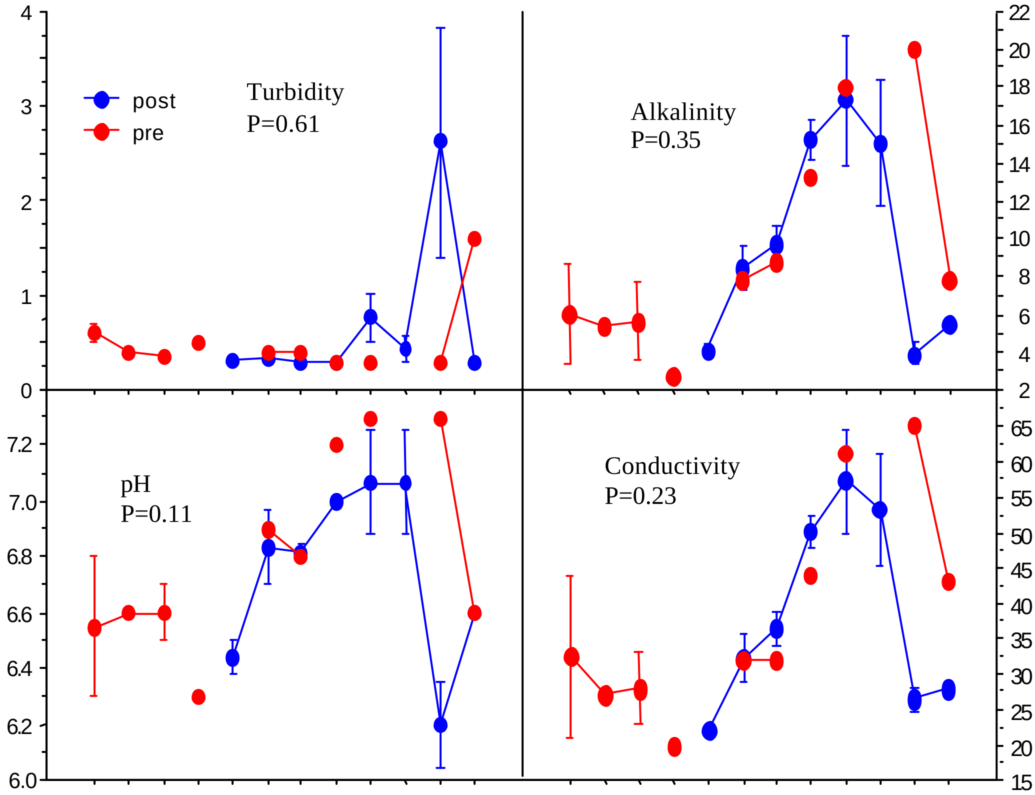

Figure 7.6.3 Mean monthly values for pH, conductivity (sie/cm), turbidity (JTU), and alkalinity (mg/L) for watershed 10. Pre-verses post harvest significance levels (P-values) are given, with error bars for monthly means.

CHAPTER 8

ESTIMATING DAILY STREAM ION CONCENTRATIONS BETWEEN SAMPLE DATES

INTRODUCTION

The calculation of total ion fluxes (ion budgets) from watersheds has been approached in numerous ways in the literature (Dann 1986). The ideal scenario is continuous and simultaneous determination of precipitation input, water chemistry and discharge through all seasons. Somewhat less intensive would be a simultaneous evaluation of mean daily ion concentrations and stream discharge. However, very few projects have budgets or human resources to support such an intensive sampling. As a result, weekly or bi-weekly sampling procedures are usually adopted for streamwater chemistry. This means primary nutrient exports from watersheds must be interpolated between sampling dates. Daily measurements of some water quality parameters, and discharge can now be measured with automated stream gauging methods and are available for most modern studies. The opportunity exists to find correlations between these parameters and less frequently sampled water chemistry variables with regression analysis. Where this method is effective, it should prove more realistic than typical linear interpolation techniques.

OBJECTIVE

There were two main objectives for this chapter.

(a) to evaluate the outcome of five commonly used methods for estimating daily ion concentrations in monitored forest streams, using calcium and discharge as a case study;

(b) to investigate the use of other probe parameters to estimate daily ion concentrations.

This chapter was designed to help decide the best means of developing formulas for the estimation of ion concentrations between sample dates.

METHODS

Dann (1986) described numerous methods for estimating annual ionic exports. Of these, three were derived from summations of daily estimation techniques. He advised using subsets of data specific to different times of the year, verses the whole data set, to eliminate biasing for times of the year when sampling intensity is greater than other times.

1. *Regression Equation*: one regression is generated for each solute per month,

by relating the concentration of each ion on a given day to the corresponding daily flow. Daily flow is then entered into the formula to give daily ion concentrations which are summed for the annual export.

2. *Weighted regression procedure based on Flow Quartile*: Flow rates [m^3/(time)] are divided into quartiles such that each quartile contributes 25% of the mean annual stream discharge over the study period (see Figure 8.2). The same number of x-y pairs (daily discharge and ion concentration respectively) are collected randomly from each quartile for use in the regression analysis. One correlation equation is generated for the entire year, with data pooled for each quartile.
3. *Weighted regression procedure based on Flow Duration*: Flow rates are divided into quartiles as in method 2. The number of samples coming from a quartile is reflective of the percentage of time during the average year that the streamflow rate falls into that quartile. For example, if 20% of the days of the year were in the high flow quartile, then 20% of the x-y pairs chosen for the regression equation should come from that flow quartile.

Two other techniques will be examined here.

4. *Curvilinear regression with complete data set*: The entire annual data set is used to create curvilinear regressions relating the concentration of the ion on a given day to the corresponding daily flow for each year.
5. *Linear interpolation*: This, the simplest method, interpolates linearly between

samp ling points as follows:

$$\frac{(1st\ number - 2nd\ number)}{(number\ of\ days\ missing + 1)}$$

where “2nd number” refers to the ion concentration on the later of the two points in the year, and the 1st number is the earlier. The value this formula gives is the daily interval between each unknown.

For purposes of comparison, the discharge and ion concentration data from HBWS watershed 1 were used from the period of 1994 to 1997 inclusively. Calcium was chosen as it varied greatly over the study period and seemed to be flow dependent. Figure 8.1 characterizes the relationship between calcium and discharge. As recommended by Dann (1986), methods 1-3 were evaluated with both linear regression techniques, and curvilinear regressions.

Regression techniques, one to four give daily estimates of the given ion as predicted from daily discharge, and are thus variable between sample days, though these methods differ in how samples are chosen to generate the regressions (Figure 8.2).

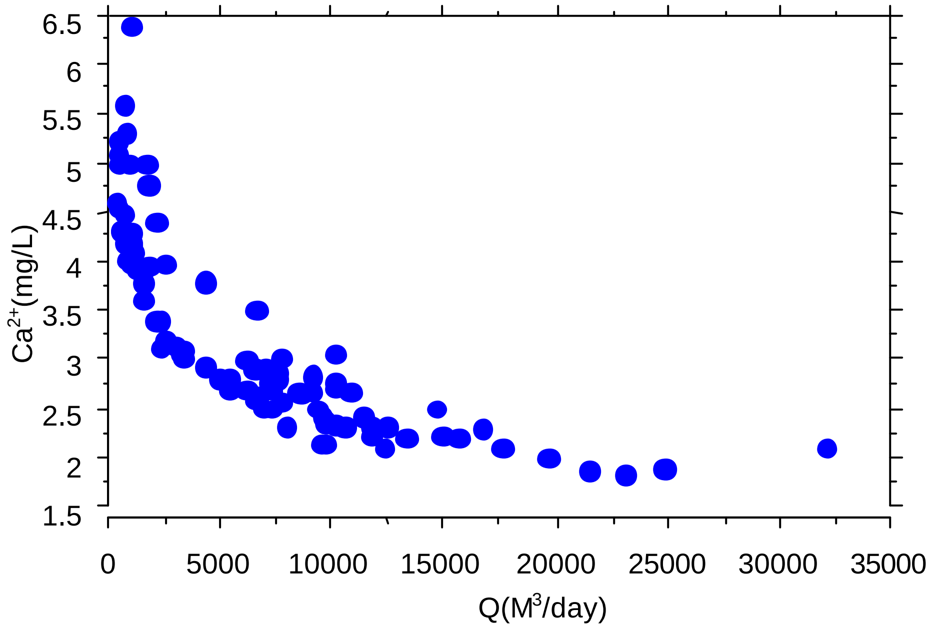

Figure 8.1 Graph illustrating the relationship between the calcium concentration in streamwater and daily total discharge.

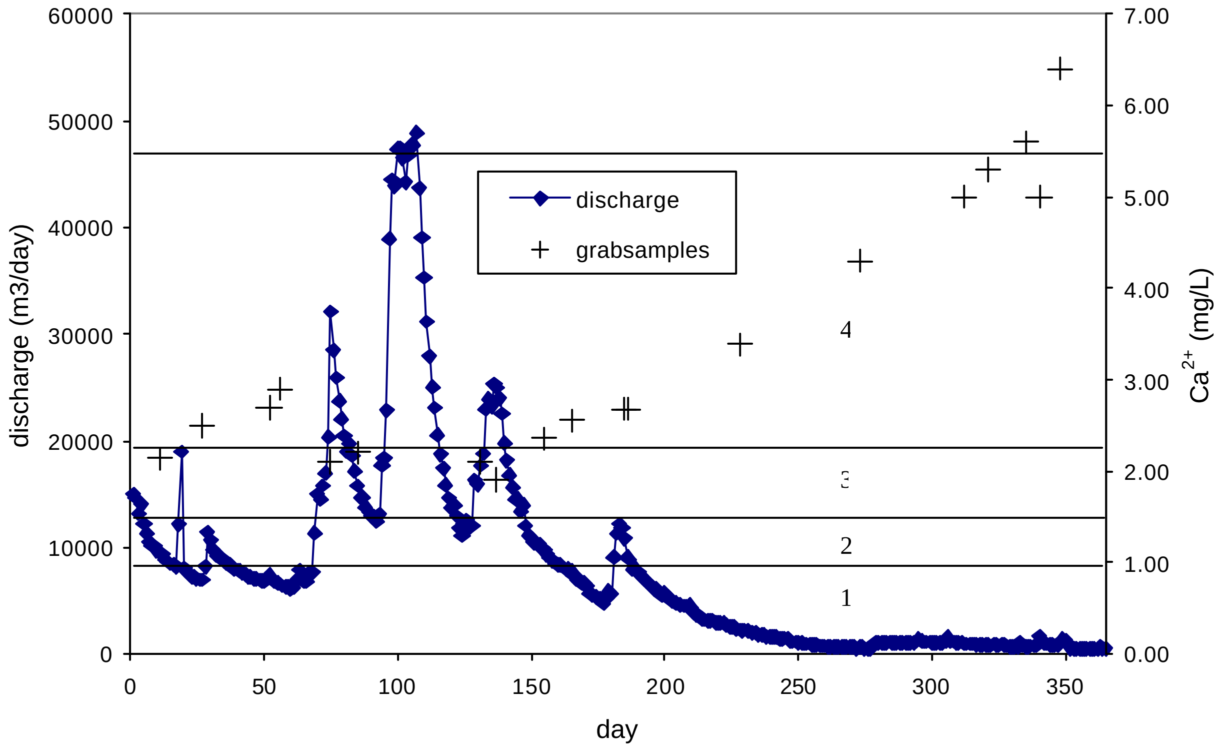

Figure 8.2 Graph illustrating flow quartiles (numbered 1 to 4) as used in methods 2 and 3. The sum of all daily flows falling within the range of each quartile equals 25% of the annual discharge. Samples are chosen randomly from each quartile as described above.

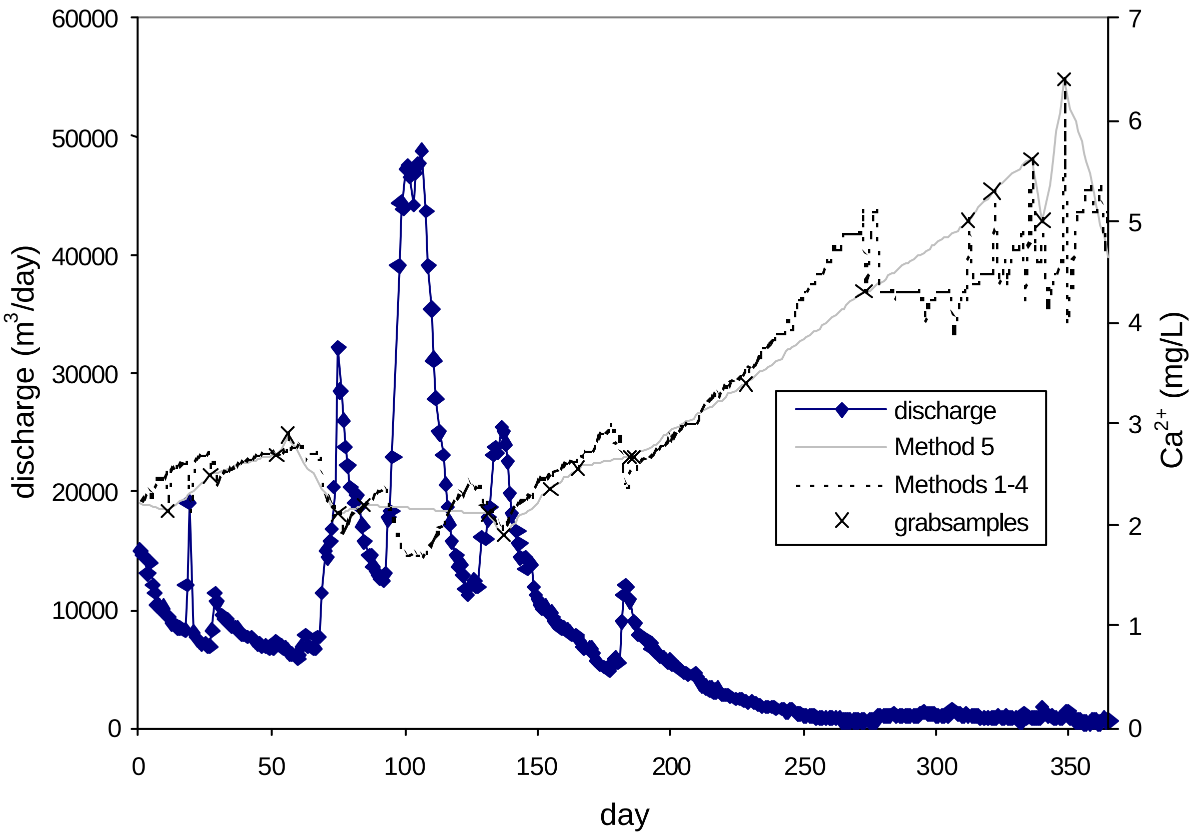

Figure 8.3 Graph illustrating the difference between ion estimation with daily data, verses monthly mean data.

RESULTS AND DISCUSSION

The results of calculating calcium flux from watershed 1 via methods 1 to 5 are displayed in Table 8.1. The different methods showed considerable differences in yearly ion flux. Johnson (1979) suggested that regression equations may be appropriate when discharge and concentration are highly correlated, as these equations help to remove bias created by the fact that a higher proportion of samples are generally taken during summer when access roads are open. This time of year is when flow dependant ions are generally

Table 8.1 Yearly calcium fluxes from watershed one for each method. Note: *=linear regression ($Y=b_0+b_1x$); §=curvilinear regression ($Y=b_0x^{b_1}$)

Method	1994	1995	1996	1997
	Year			
	Kg/year			
1a*	17879	17948	16105	19188
1b§	18194	17942	16443	20126
2a	17603	18958	17868	19209
2b	17310	17877	15731	19393
3a	16631	18209	16929	18500
3b	17645	18236	15977	19837
4	18041	17802	16250	19340
5	17678	18264	16032	19845
Mean	17623	18155	16417	19430
SD	429	326	606	448
Lowest estimate as percent of highest	91.4	93.9	88	91.9

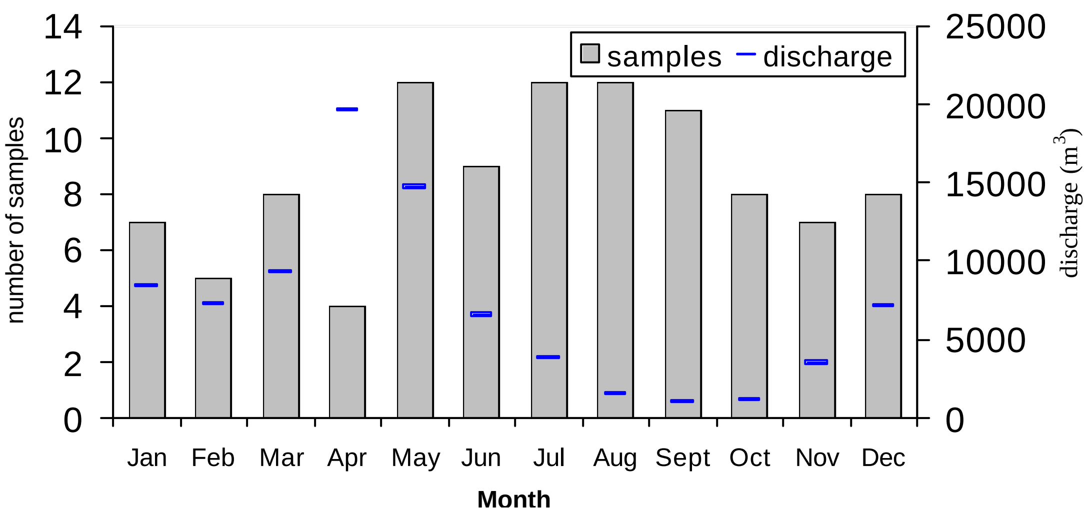

Figure 8.4 Graph illustrating sampling intensity by month and the mean total discharge by month for the study period for watershed 1.

Table 8.2 Linear regression coefficient summary for each month from 1994 to 1997 inclusive

Month	n	b_0	b_1	R^2
January	7	2.87	-4.05×10^{-5}	0.09
February	5	3.56	-1.15×10^{-4}	0.87
March	8	2.84	-3.03×10^{-5}	0.53
April	4	3.16	-5.69×10^{-5}	0.78
May	12	2.70	-3.49×10^{-5}	0.90
June	9	3.31	-8.41×10^{-5}	0.66
July	9	3.63	-1.01×10^{-4}	0.65
August	8	4.65	-0.001	0.85
September	7	4.53	-3.14×10^{-4}	0.65
October	7	4.96	-3.6×10^{-4}	0.433
November	6	5.13	-2.25×10^{-4}	0.900
December	8	5.62	-2.68×10^{-4}	0.83

at their highest concentrations with others less so (see Figure 8.1).Dann (1986) points out that when the dependent variable is related to the independent in a curvilinear fashion, then linear regressions are inappropriate. Due to the non-linear trend obvious in Figure 8.1, the same procedure was repeated but with a non-linear regression with the form $Y=b_{0x}^{b1}$ (see Table 8.3).

Table 8.3 Non-linear regression coefficient summary for each month from 1994 to 1997 inclusive

Month	n	b_0	b_1	R^2
January	7	9.06	-0.14	0.12
February	5	75.24	-0.38	0.81
March	8	13.07	-0.18	0.84
April	4	68.52	-0.36	0.75
May	12	22.57	-0.25	0.85
June	9	16.30	-0.20	0.60
July	9	13.00	-0.17	0.86
August	8	18.13	-0.22	0.81
September	7	7.11	-0.08	0.59
October	7	8.13	-0.09	0.41
November	6	21.1	-0.21	0.99
December	8	31.29	-0.26	0.89

For methods 2 and 3, the discharge data was divided up into four quartiles such that each totaled 25% of the flow for the study period. The intervals were <7249m^3/day, 7249 to 11059m^3/day, 11059 to 19008m^3/day, and <19008m^3/day. Method 2 requires an equal number of samples from each quartile, and since the upper quartile only had five samples, five samples were chosen at random from the other three quartiles for a total of twenty samples.

Method 3, based on flow duration, required the percentage of samples taken from each quartile to be reflective of the percentage of time the river was flowing within the discharge range of that quartile. Therefore, 62% of samples were taken randomly from the first (low flow) quartile, 19% from the second quartile, 13% from the third quartile, and 6% from the fourth (highest flow) quartile, for a total of 67 samples. The results of method 2 and 3 with linear and non-linear regressions are summarized in Table 8.4.

Table 8.4 Linear and non-linear regression coefficients for methods 2 and 3

Method	n	b_0	b_1	R^2
2a*	20	4.055	-9.978×10^{-5}	0.56
2b§	20	21.889	-0.238	0.77
3a	67	4.013	-1.162×10^{-4}	0.68
3b	67	23.583	-0.245	0.83

*= linear regression ($Y=b_0+b_1x$); §= non-linear regression ($Y=b_0x^{b1}$).

Non-linear regressions are not only more appropriate for reasons stated above,

but also gave more reliable estimates for both methods. When each of the two non-linear models developed from method 2 and 3 are applied to the remainder of the data set for model verification, and actual data compared to predicted, the reliability of method 3 is much greater than method 2, as illustrated by the linear regression results in Table 8.5. The intercept is very close to zero and slope almost 1, probably because this method utilizes a greater portion of the data for development of the model (i.e. 67 x-y pairs versus only 20).

Table 8.5 Model verification results for methods 2 and 3

Method	n	b_0	b_1	R^2
2b	70	0.651	0.769	0.78
3b	23	-0.016	0.994	0.94

Method 4 utilizes the entire data set in a curvilinear regression to predict calcium concentration based on discharge. This method does not necessarily allow for model verification, but in methods 2 and 3, increasing the sample size from 20 to 76 greatly increased the reliability of the model (see Table 8.4), and using the full set of data increased this further (see Figure 8.3).

Method 5 simply interpolated linearly between sample dates, and was the least sophisticated of all methods. Methods such as this may be more appropriate when samples were taken at two or three day intervals as there is less chance of significant hydrogeological events occurring between samples that will be missed by this method. In the case of the HBWS where sampling was generally at one to two week intervals, this method simply would not account for any possible variation in between sampling dates.

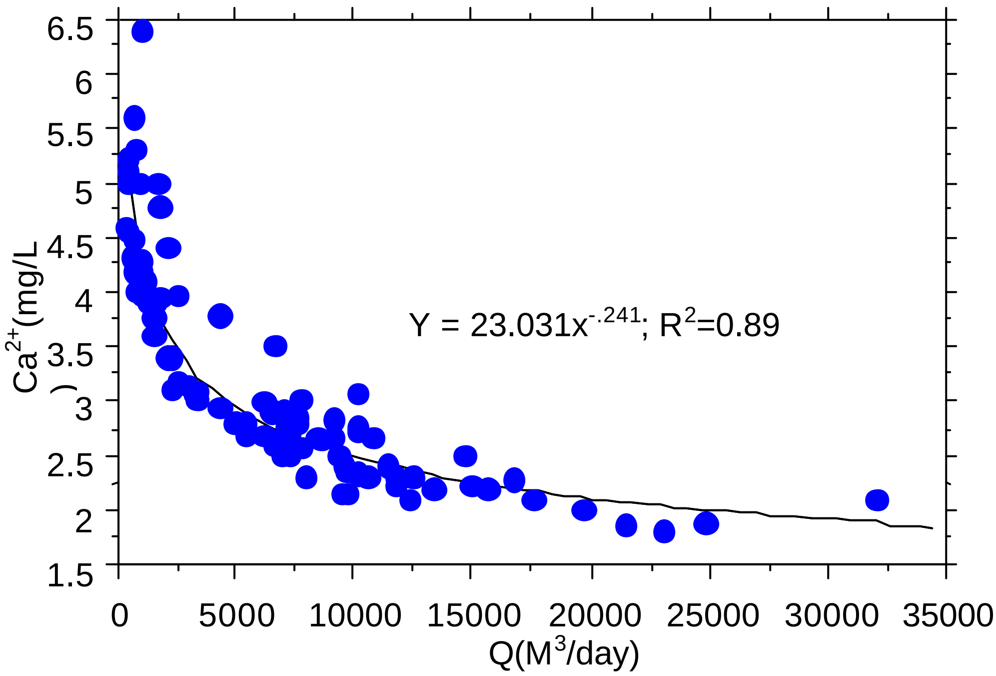

Figure 8.5 Non-linear regression for the relationship between calcium and daily discharge. R^2=0.89.

Though time consuming and complicated, method 1 shows promise. Unfortunately the months of the year that were poorly sampled, resulting in poor regression estimates, were also the months with some of the highest discharges. Months were used in this method as the sub groups However, these are in no way reflective of actual hydrogeologic time frames, and it is possible that any one or more month may overlap significant hydrogeologic events, such as spring snowmelt, or soil freezing or thawing. A more appropriate breakup of the hydrogeologic year would respect these events.

Methods 2 and three required random sampling from the entire data set to obtain the x-y pairs for the regression analysis. This assumes independence and a normal distribution of data however, which is not the case for discharge concentration data. As well, method 2 was limited by the minimal sampling in the highest flow quartile, resulting in only twenty samples being used from the entire data set. Method 3, though using more overall data, actually used fewer samples from the highest flow quartile that was already poorly represented.

Method 4 used the entire data set to create the model, and seemed to give the most accurate and reproducible results, and was favored over the above methods for estimating calcium between sampling days. Since regression models give more weight to higher flow periods, it is important to have good data for high flow times. Though the HBWS was poorly sampled during these periods, the variability of calcium concentrations at high water levels tends to be minimal.

Method 5, the only method that does not functionally relate one hydrogeological factor to another, may be reasonable for estimating yearly ion fluxes (Dann 1986), but cannot realistically evaluate calcium concentrations between sampling dates. In the case of the HBWS, where there is often significant time between samples during wet times of the year, significant hydrogeologic events can be missed.

In light of the above, the possibility of using this technique with other daily probe data was investigated. Raw data, as well as transformations of data were examined with linear and non-linear regressions against the data from the different probes. Some general conclusions were immediately obvious.

For the purposes of estimating ion concentrations, the most valuable probe information came from the discharge and conductivity probes. Dissolved oxygen, temperature, and turbidity neither depend on, nor determine water chemistry to any significant degree, and thus are not used in this discussion. Though pH indeed depends greatly on water chemistry, there seems to be too many compounding factors influencing the pH values to allow it to reliably predict individual ion levels. It was found to be fairly reliable for estimating alkalinity, though not helpful for individual ions.

As has been discussed in previous chapters, conductivity and discharge are closely related. In summary, this is due primarily to the fact that discharge is dependent on groundwater levels, and groundwater levels in turn relate to residence time. Residence time strongly relates to solute concentration. For these reasons, both probes seem to only be useful for those ions that are discharge dependent. Further analysis of the graphs from Chapter 5, and the review of literature in Chapters 3 and 4 reveal the following to be most heavily discharge dependent, of those measured at the HBWS:

$$SiO_2, SO_4^{2-}, Cl^-, K^+, TIC, Na^+, Mg^{2+}, Ca^{2+}.$$

The feasibility of predicting these ions via linear and non-linear regressions with conductivity and discharge was investigated. Of the numerous model types investigated for the different ions, the most consistent and reliable were linear ($Y=b_0+b_1x$), power ($Y=b_0x^{b1}$) and exponential regressions ($b_0(e^{(b1*X)})$). The Table 8.6 summarizes the equations that were deemed to be the most appropriate for each ion, by watershed, based on residual statistics, as well as visual assessment of how realistically they fulfilled their purpose.

Table 8.6 Summary of regression equations predicting major ions based on conductivity and discharge.

Watershed 1			Watershed 4			Watershed 5		
ION	EQUATION	R^2	ION	EQUATION	R^2	ION	EQUATION	R^2
SiO_2	5.5+.07cond	.53	SiO_2	$3.99cond^{.27}$	.36	SiO_2	$3.4Q^{.2}$	.52
SO_4^{2-}	0.86+.08cond	.72	SO_4^{2-}	$3.2(e^{(.003cond)})$	.30	SO_4^{2-}	$111.3Q^{-.32}$	.90
Cl^-	1.86-.08Q	.13	Cl^-	$.56cond^{.34}$	.48	Cl^-	$8.8Q^{-.17}$	.69
K^+	$1.2Q^{-.115}$	.69	K^+	$.08cond^{.47}$	.79	K^+	$1.1Q^{-.09}$	.47
TIC	$16.4Q^{-.22}$	.77	TIC	-1.4+0.12cond	.97	TIC	$88Q^{-.32}$	.92
Na^+	$2.06Q^{-.09}$	.78	Na^+	$.03cond^{1.33}$	.97	Na^+	$30.1Q^{-.26}$	.92
Mg^{2+}	$0.92Q^{-.17}$	.82	Mg^{2+}	$.09cond^{.6}$	.97	Mg^{2+}	$8.5Q^{-.26}$	.94
Ca^{2+}	$23.1Q^{-.24}$	.87	Ca^{2+}	$.21cond^{.70}$	.94	Ca^{2+}	$130.5Q^{-.33}$	.95

Table 8.6 con't

Watershed 6			Watershed 9		
ION	EQUATION	R^2	ION	EQUATION	R^2
SiO_2	$30.7Q^{-.16}$	.65	SiO_2	$29.6Q^{-.15}$	.67
SO_4^{2-}	1.2+.06cond	.88	SO_4^{2}	1.194+.06cond	.94
Cl^-	$15.9Q^{-.25}$	.74	Cl^-	$15.9Q^{-.25}$	.74
K^+	$.41x10^{(.003cond)}$	.47	K^+	$.41x10^{(.003cond)}$	.43
TIC	.08+.087cond	.90	TIC	-1+.1cond	.97
Na^+	$1.65x10^{(.009cond)}$	.90	Na^+	-.34+.114cond	.98
Mg^{2+}	.24+.013cond	.92	Mg^{2+}	.25+.013cond	.98
Ca^{2+}	-.42+.13cond	.94	Ca^{2+}	-.23+.08cond	.99

The above results clearly indicate that those ions that are highly related to weathering reactions, are the most reliably predicted with conductivity and discharge. As has been noted, as discharge decreases, the ion concentrations for these ions increases. Since the concentrations of these ions are very low in rainwater, a significant rain event that increases discharge, and promotes the dilution effect, does not contradict this trend. Other ions, such as potassium, chloride and sulfate, the concentration of which depends on numerous factors including the fact that they exist in significant amounts in precipitation, are less often predicted with a high degree of confidence.

CONCLUSIONS

The intelligent and consistent prediction of ion concentrations between sample dates is indeed possible for some ions, especially with the assistance probes measuring conductivity and discharge. With these probes, ions which are mainly related to discharge and groundwater levels are the most reliable. For situations like the HBWS, where some months were sampled at a very low level, regression relationships should be generated using the entire data set (method 4). In other situations where there are sufficient numbers of samples to generate reliable results, regressions can be developed either by month, or within time windows in the year that reflect significant hydrologic periods, such as the frost period, snowmelt period, summer low flow period, or during periods of heavy rain.

CHAPTER 9

CONCLUSIONS AND CONTRIBUTIONS

Hypothesis 1. *Streams very close to each other, within the identical ecophysical region can have significant differences in streamwater chemistry and water quality parameters.*

The water quality portion of the Hayward Brook Watershed Study was designed with little pre-harvest information, based on the assumption that these watersheds should be very similar in many ways. This thesis effectively illustrates that this assumption was inappropriate, and that for future studies, more extensive pre-harvest investigations are required to select those basins that best qualify for establishing good experimental watershed treatment protocols, and to facilitate direct basin-to-basin comparisons. For example, the eight catchments of the HBWS varied by an order of magnitude for pre-harvest pH, electrical conductivity, and base cation concentrations in the streamwater. These differences were due to differences in soil substrate, for the main part.

Hypothesis 2: *Harvesting less than 20% of Acadian forests has no significant impact on stream discharge or water quality.*

Firm conclusions regarding this hypothesis are difficult to obtain from the data presented in this thesis, in spite of the extensive and intensive character of these data. This is in part due to the fact that harvesting was limited to a few percent per basin, and also due to the large base-to-basin variations, as mentioned above. In combination with the results from the extensive literature review, however, it can be concluded at this point that the commercial felling and extraction of trees within the Hayward Brook Watersheds did not results in a measurable change in stream discharge or water quality, for the most part. In contrast, road construction and the accompanying culvert installation produced an immediate peak in zinc concentrations above background levels.

> Hypothesis 3. *The continuous recording of readily measured water quality parameters can be used to predict other, less easily measured water quality parameters.*

Of all the water quality parameters that were measured, electrical conductivity proved to be most useful for predicting many of the other streamwater quality variables, such as pH, Ca, Mg, TIC, and bicarbonate content. In turn, electric conductivity can in part be predicted from the stream discharge observations. Already, there have been several studies to relate stream chemistry to stream discharge, primarily for the purpose of hydrograph separation (Caissie et al. 1996; Laudon and Slaymaker 1997). This thesis has shown both discharge and conductivity can become a tool for estimating the concentrations of many water quality parameters for each basin, once the basin-specific relationships have been developed. In general, the electrical conductivity probe and the pressure transducer to gauge stream height are most reliable in terms of generating, e.g., hourly data records, thereby capturing even small hydrologic events, including diurnal variations.

Hypothesis 4. *Streamwater chemistry and water quality parameters, as they vary over time within each basin, are strongly affected by weather and season, in accordance with rate of stream discharge.*

Typically during summer when streamflow was low, streamwater ion concentrations and pH were high. Low ion concentrations occurred during snowmelt. Summer storms, in general, provide a brief flush of solutes the soil matrix, followed by dilution. Low intensity summer rains (<1mm) often produced no measurable change in stream discharge, or steam gauge height. The continuous electrical conductivity probe, however, was more sensitive than the stage height sensor and detected change even though there was no indication of change in streamflow rate.

RECOMMENDATIONS

This thesis provides a summary of the data of the HBWS only, and explores some of the relationships between the data, variable-to-variable, and basin-to-basin. More work could be done to model stream discharge, electric conductivity, pH and various ion concentrations for each basin, and day after day, depending on the actual weather record. The data presented here can serve both as input for the model (weather, basin characteristics), as well as verifying model output (stream discharge, conductivity, ion concentrations).

The data fall short in terms of knowing what to expect in term of whole-basin treatment response to harvesting. Future studies could concentrate on monitoring several of these eight basins with a whole basin treatment. In the mean time, a model could be developed that would suggest what such changes should be for each basin. In the end, combining the field results with the model results would lead to the establishment of a well calibrated and well verified model that could be used to generalize expected harvest treatment effects on many other similar forest basins.

Since the conductivity probe is capable of capturing even small hydrological events in a generally reliable manner, one should explore the potential of this probe to continuously monitor many other forest watersheds in terms of their short- and long-term reactions to changes in surficial conditions, be these changes related to changes in land-use, forest harvesting, plantation establishment, forest fire, and climate change, to name a few. Doing so would, for example, expand on the work done by Laudon and Slaymaker (1997) to assist in hydrograph separation. Based on basin-specific calibrations, the same probe can then generate estimates for stream discharge, ion concentrations and pH as well. While this probe does not substitute the need for measuring the other quantities directly, this probe would serve as a general purpose, cost-effective devise to inform about the streamwater environment as it changes in response to weather and within basin activities.

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