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Title: The Influence of Contemporary Forest Management on Stream Nutrient Concentrations in an Industrialized Forest in the Oregon Cascades

Abstract approved:

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The increased demand for wood and fiber from a continually shrinking land base has resulted in the use of intensively managed forest plantations. The concentration of timber production on the most suitable sites allows the world's demand for forest products to be met on less land and enable native forests to be conserved. Because much of the water flowing in rivers in the U.S. originates as precipitation in forests, there is a justified concern about the impacts of forest management on water quality.

Nutrient concentrations were measured in eight streams from October 2002 to September 2011 to assess nutrient response to contemporary forest practices at the Hinkle Creek Paired Watershed Study in the Oregon Cascades. This period of time included a two-year pre-treatment calibration between control and treatment watersheds, a fertilization treatment of both basins in October 2004, and a post-treatment period from 2005 to 2011. A treatment schedule comprised of two temporally explicit harvest entries was used to assess the effects of clearcutting at the non-fish-bearing headwater scale and the fish-bearing watershed scale. Stream water samples were analyzed for nitrogen, phosphorus, calcium, sodium, potassium, magnesium, sulfate, chloride, and silicon as well as specific conductance, pH, and alkalinity. Programmable water samplers were used to take water samples during fall

freshets in November 2009 to assess the stream water discharge versus NO_3 + NO_2 concentration relationship.

All treatment watersheds showed a statistically significant increase in NO_3 + NO_2 concentrations after clearcutting ($p < 0.001$). The slope of the streambed through the disturbance was a stronger predictor of the magnitude of the response than was the magnitude of disturbance. Ammonia and organic nitrogen displayed notable increases after harvest treatment, but these increases were attributed to increases in the control watersheds. Phosphorus showed a response to timber harvest in one headwater stream. The remaining nutrients showed a small decrease in the control and treatment watersheds for the period after harvest. There was some evidence to suggest that the addition of urea nitrogen to both basins may have caused an increase in in-stream biota uptake of these nutrients. The storm response results showed that NO_3 + NO_2 concentrations in stream water increase with discharge during small storms that occur after periods of negligible precipitation.

Concentrations of NO_3 + NO_2 observed during the calibration period were similar to concentrations observed in an old-growth forest in the H.J. Andrews, suggesting that nutrient processing within the Hinkle Creek watershed had returned to levels that existed prior to its initial harvest sixty years ago. This finding helps to assess long-term impacts of shorter rotation timber harvest of regenerated Douglas-fir stands characteristic of industrialized timber harvest in Oregon.

The Influence of Contemporary Forest Management on Stream Nutrient Concentrations in an Industrialized Forest in the Oregon Cascades

by
William Scott Meininger

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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

William Scott Meininger, Author

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Chapter 1: Introduction

Background

Timber harvest is being restricted in Oregon and in many parts of the country and world despite a greater demand for forest resources from the rapidly increasing world population. The increased demand for wood and fiber from this reduced land base has resulted in the use of intensively managed timber plantations that optimize net present worth (Fox, 2000). The concentration of timber production on the most suitable sites will allow the world's demand for forest products to be met on less land and enable native forests to be conserved. Forest managers must understand and use sustainable practices to maintain long-term soil fertility for continued site productivity.

Paired watershed studies (PWS) have been used for more than a century to provide researchers with knowledge about impacts related to the disturbance of forest stands. A study done in the Alsea River watershed between 1959 and 1973 was one of the first of these studies in Oregon, and the results of that study played a major role in guiding foresters, researchers, and policymakers as they drafted the initial Oregon Forest Practices Act in 1971. Since then, a shift in society's values with regard to what amount of environmental impact is acceptable has prompted major changes to forest management practices and a shift in the paradigm of Best Management Practices (BPM's). Most of the current knowledge of contemporary forest practices comes primarily from dated studies such as the Alsea that evaluated historic practices used in the initial harvest of naturally grown forest stands. There is now a need to re-evaluate the environmental impacts of contemporary forest management practices to guide foresters, researchers, and policymakers as they look to the future of timber harvest in Oregon. The Hinkle Creek Paired Watershed Study was initiated to fill a gap in our current knowledge of contemporary, state-of-the-art practices carried out on harvest-regenerated, young stands located within privately owned industrialized forests in the Pacific Northwest (Skaugset et al., 2000).

The Paired Watershed Study

Paired watershed studies are simple in concept. Like many studies in other fields of research and science, PWS use a control and treatment to identify and quantify the effects of a treatment. Two watersheds are selected and monitored during a calibration period. Ideally, these watersheds are contiguous and similar for all relevant attributes, which include size, gradient, soil and parent material, precipitation, elevation, vegetation, land use, and history such as stand age and prior disturbance. After a calibration period, one of the watersheds undergoes treatment in the form of management activity. In the case of PWS undertaken to assess the impacts of forest management practices, management activites can include road construction, timber harvest, tree planting, fertilization, and site preparation such as the use of herbicides or burning to suppress vegetation.

Both watersheds are monitored during and after treatment to discern environmental effects. The control serves as a comparison to validate that changes observed in any of the monitored parameters after treatment were a response to the treatment and not a response to an external factor such as a shift in climatic patterns. This paired design helps to minimize the risk of detecting a treatment effect when there was no treatment effect, referred to in statistics as a Type I error.

PWS can and have provided knowledge in the development of Best Management Practices (Ice, 2004), but they have also received criticism for various deficiencies. One of the primary deficiencies of PWS that critics point out is that every watershed is unique, which makes any attempt at replication impossible. This variability across watersheds and the lack of replication means that extrapolation to other watersheds is limited. Because the results of the study are contingent on a number of environmental factors that may be specific to the time period when the study is conducted, any conclusions drawn from the study may quickly become invalid even within the same watershed after a shift in the behavior of one or more of these factors.

A second deficiency of PWS is that they are designed to detect a response to a treatment by means of a “black box” model, which provides very little, if any, knowledge of the processes that generate or influence the response. Without any understanding of the processes and mechanisms that link the treatment to the response, some argue we cannot fully understand or explain any of the observed results. Thus, any conclusions drawn, even within the same watershed, must be viewed as contingent on assumptions that may or may not be valid. Critics argue that by going after the processes rather than just the response, the conclusions drawn are more transferrable to other settings because the results give us a physically-based means to predict what the response to a particular treatment would be using hydrological models (Schnorbus and Alila, 2004; Seibert and McDonnell, 2010).

Additionally, PWS are expensive and finding undisturbed watersheds to conduct this type of research is difficult. The commitment from the land owner and those conducting the research can limit the calibration and post-treatment periods to lengths of time insufficient to measure important parameters, such as the duration of the response.

Environmental impacts of timber harvesting

Timber harvesting, particularly clearcutting, can produce negative effects (Hornbeck et al., 1987; Likens et al., 1970; Martin and Harr, 1989; Meehan, 1991). These include changes to stream chemistry, increases in stream temperature, changes to the sediment composition of the streambed, and alteration to the quantity and quality of large woody debris. Many of these impacts can be transported downstream. Some of these harvest-induced alterations of aquatic ecosystems are linked to declines in the diversity and abundance of native fish species in the Pacific Northwest, including some listed as threatened or endangered. Further, because 70 to 80 percent of the water flowing in rivers in the U.S. originates as precipitation in forests (Sedell et al., 2000), there is a justified concern about the impacts forest management may have on water quality.

Two major components of the ecosystem approach to forest management are water quality and long-term nutrient sustainability (Swanson and Franklin, 1992). Because streams are a primary way to redistribute nutrients within an ecosystem, the analysis of stream chemistry within an ecosystem following disturbance is viewed as a highly credible approach to detect treatment effects (Dahlgren, 1998). Over the past several decades in the Pacific Northwest, research has been done on understanding the disturbance to the nutrient cycle following timber harvesting (Brown et al., 1973; Cromack et al., 1999; Dahlgren, 1998; Gravelle et al., 2009; Martin and Harr, 1989). A review of this research indicates that the following five factors are the most important to determine stream chemistry: (1) geologic weathering; (2) atmospheric precipitation and climate, including precipitation chemistry, stream discharge, and temperature; (3) terrestrial biological processes; (4) physical and chemical reactions in soil; (5) and physical, chemical, and biological processes within aquatic ecosystems (Feller, 2005).

Geologic weathering is considered to be a dominant process in landscapes with younger soils derived from noncrystalline rocks (Gibbs, 1970) and has been shown to contribute K^+, Mg^+, Ca^+, and Si to streams (Cornwell, 1992; Wissmar et al., 1997). The rate of geologic weathering is strongly influenced by temperature (White and Blum, 1995), rock composition, mineral crystal size, rock texture and porosity, and degree of fissuring (Hem, 1985). Watersheds with finer-textured igneous rock such as basalt or tuff are more easily weathered and tend to have streams with relatively high chemical concentrations (Feller, 2005).

Precipitation in the western United States has much less influence on concentrations of most chemical species in streams (Dethier, 1979; Wissmar et al., 1997). However, it is the primary contributor of Cl and SO_4 ions, due to their low abundance in parent materials (Gorham, 1961; Hem, 1985).

Clearcutting removes the forest canopy. This increases the amount of solar radiation to the forest floor and reduces the interception of precipitation and transpiration of soil moisture (Bosch and Hewlett, 1982; MacDonald et al., 1991). This combination of increased soil moisture and surface temperature can enhance microbial decomposition of soil organic matter on the forest floor, which leads to increased rates of nitrogen mineralization and nitrification, producing NO_3 + NO_2. Nitrogen is normally conserved in an undisturbed ecosystem, but NO_3 + NO_2 can be exported in streams in a harvested ecosystem (Likens et al., 1970). Nitrification releases H^+ ions to cation exchange sites that facilitates the leaching of other ions into the streams (Martin and Harr, 1989), so an increase in NO_3 + NO_2 leaching in the soil matrix equates to an increase in the leaching of other nutrients.

The degree of nutrient losses after clearcutting is variable and is influenced by a number a factors (Feller, 2005). Streams in the deciduous forests of the Northeast increases in nutrient concentrations after timber harvest (Likens et al., 1970; Vitousek and Melillo, 1979) while only modest increases accompany the harvest of coniferous forests in the Pacific Northwest (Binkley et al., 2004). Douglas-fir forests of the western Cascades resist losses of nutrients after clearcutting because of the combination of high C:N ratios of the forest floor, the summer droughts which delay leaching, and the deep soils which provide ample storage and increased cation-exchange sites (Martin and Harr, 1989).

Objectives

The overall broad objectives of the Hinkle Creek Paired Watershed Study are to:

- determine the effects of forest management on the physical, chemical, and biological characteristics of habitat quality in small streams without fish;
- assess the influence of changes in the physical, chemical, and biological characteristics of habitat quality on amphibian and invertebrate abundance, distribution, and movement, in headwater streams with and without fish, and;
- evaluate the role of organism movement in maintaining abundance and diversity of fish and amphibians as habitat quality changes throughout the stream network.

To meet these research objectives at Hinkle Creek, a number of studies were carried out that investigated parameters of stream habitat, such as temperature, sediment, woody debris, canopy closure, concentrations of nutrients, and discharge regimes. Invertebrate and vertebrate populations were monitored for abundance and movement. The collaborative efforts of these individual studies will ultimately be synthesized in a report on the management impacts to stream ecosystems.

The objectives of the stream chemistry research are to:

- determine the local and downstream impacts of forest management carried out adjacent to non-fish-bearing streams where fixed-width buffers of overstory conifers are not left.

- determine the local and downstream impacts of forest management adjacent to fish-bearing streams where fixed-width buffer strips of overstory conifers were left.
- compare the results with those from other forested watersheds in the Pacific Northwest.

To meet these objectives, stream water samples were collected from fish-bearing and non-fish-bearing streams during a calibration and post-treatment period and analyzed for several nutrients to detect changes. The temporally and spatially explicit nature of the treatments allowed the stream water chemistry results to be correlated with treatment effects.

Chapter 2: Study site

Site characteristics

The Hinkle Creek Paired Watershed Study is located in southern Oregon in the foothills of the western Cascade Mountains about 40 kilometers northeast of Roseburg, Oregon. The 1,950 ha study site is owned by Roseburg Forest Products and is managed for timber production. The experimental watershed is situated in the transitional snow-zone with a climate dominated by higher pressure systems during the warm, dry summer months and by frontal Pacific storms during the fall, winter and spring. Mean annual precipitation is approximately 1,800 mm. Mean annual air temperature is 8.5˚C. The experimental area ranges from 430 to 1100 meters above sea level.

The study area is harvest regenerated forest comprised primarily of 60 year-old Douglas-fir (*Pseudotsuga menziesii*) with red alder (*Alnus rubra*) in the riparian areas. Under-story vegetation is mostly huckleberry (*Vaccinium parvifolium*) and sword fern (*Polystichum munitum*). The soils are characterized by deep, well-drained, loamy soils, consisting of Orford gravelly loam, Honeygrove gravelly clay loam, and Klikitat gravelly loam (Johnson et al., 1994). The parent material is basalt and rhyolite with deposits of volcanic sandstones and pyroclastic material (Sherrod, 2004). The lower portion of the watershed is classified as landslide debris. The area is mountainous with V-shaped valleys. The lower area of the basin has moderately sloped hillsides of 15 to 30 percent, and the upper area of the basin has sloped hillsides of 50 to 80 percent. Average stream gradients range from 12 to 21 percent (Table 2.1).

Table 2.1 Watershed characteristics. Refer to Figure 2.1

Watershed	Scale	Slope %	Size	Area (ha)	Harvest %	Distance to main stem (m)	Distance to main stem gauge (m)
North Fork	watershed	14	L	870	0	—	—
South Fork	watershed	12	L	1061	37	—	—
Myers	headwater	18	S	86.2	0	700	1,400
Fenton	headwater	12	S	22.7	75	1660	2,200
Clay	headwater	15	S	65.2	36	1240	2,500
DeMersseman	headwater	19	M	156	0	1560	2,800
BB	headwater	21	S	111	32	380	2,600
Russell	headwater	17	S	96	15	1500	2,800

Slope % for the watersheds is the average slope of the streambed from the top of the furthest headwater of the main stem to the gauging station. *Slope %* for the headwaters is the average slope of the streambed between the top of clearcut (or equivalent for control watersheds) and the gauging station. *Size* refers to the stream size classification put forth by the Oregon Department of Forestry. *Distance to main stem* indicates the distance from the station to the confluence with the North or South Fork. *Distance to main stem gauge* indicates the distance from the headwater station to the North or South Fork station.

Study design and site history

Hinkle Creek is a nested, paired watershed study comprised of two 3rd order watersheds and six 2nd order headwater basins nested within these two larger watersheds (Figure 2.1 and Table 2.1). The North Fork was designated the control watershed and the South Fork was designated the treatment watershed. Two headwater basins in the North Fork acted as controls for four headwater basins that were treated in the South Fork. The headwater basins were paired based on size, slope, and geology. Myers was paired with Fenton and Clay; DeMersseman was paired with Russell and BB.

The research at Hinkle Creek was designed to assess the impacts of timber harvest on two spatially and temporally explicit scales. The four discrete headwater basins were harvested in 2005-06 to measure the local impacts of intensive forest management to small, non-fish-bearing headwater basins and to measure the downstream impacts to the fish-bearing South Fork. Each headwater basin experienced a varying percentage of its area clearcut without a fixed width riparian buffer strip comprised of overstory conifers. to assess the relationship between magnitude of disturbance and magnitude of response. In 2008-09, four more clearcuts were added along the fish-bearing stream to assess local impacts of clearcutting with fixed width riparian zones comprised of overstory conifers.

In 2001, prior to the initiation of this research, three harvest units were clearcut in the South Fork that totaled 11 percent of the watershed. Two units were located in the headwaters of BB Creek and Russell Creek. The third unit was located 250 meters from the South Fork along Clay Creek below the Clay gauging station (Figure 2.1).

The calibration period for the stream chemistry research began in October 2002 and continued through October 2004. In the fall of 2004, the North Fork and South Fork were fertilized with urea-N (46% N) $CO(NH_2)_2$, an organic form of N (Norris and Moore, 1970). It was applied at a rate of 500 kg urea hectare^{-1} via helicopter (Figure 2.2). Because the harvest units from 2001 and those units scheduled for harvest in 2005-06 were not fertilized, the North Fork received 8 – 9 percent more urea than the South Fork, or about 25 percent more urea per hectare. This amounted to 436,500 kg of urea in the North Fork and 400,000 kg of urea in the South Fork.

The first harvest treatment took place between August 2005 and May 2006. It was comprised of five clearcuts in the headwater basins that totaled 14.3 percent of the South Fork basin (Figure 2.1). Fenton had 75 percent of its basin above the gauge clearcut; Clay had 36 percent, Russell had 15 percent, and BB had 32 percent. Timber was removed from both sides of these non-fish-bearing streams. Site preparation occurred in the fall of 2006 that included a broad-spectrum herbicide mixture which was applied to reduce competing vegetation for the Douglas-fir seedlings that were planted in the clearcuts in the winter of 2007.

The second harvest treatment took place between August 2008 and January 2009. It was comprised of four clearcuts below the headwaters that totaled 12.2 percent of the basin. Fixed width riparian buffers were left along all fish-bearing streams. All timber harvest and road construction activities by Roseburg Forest Products were carried out in accordance with the Oregon Forest Practices Rules (OFPR).

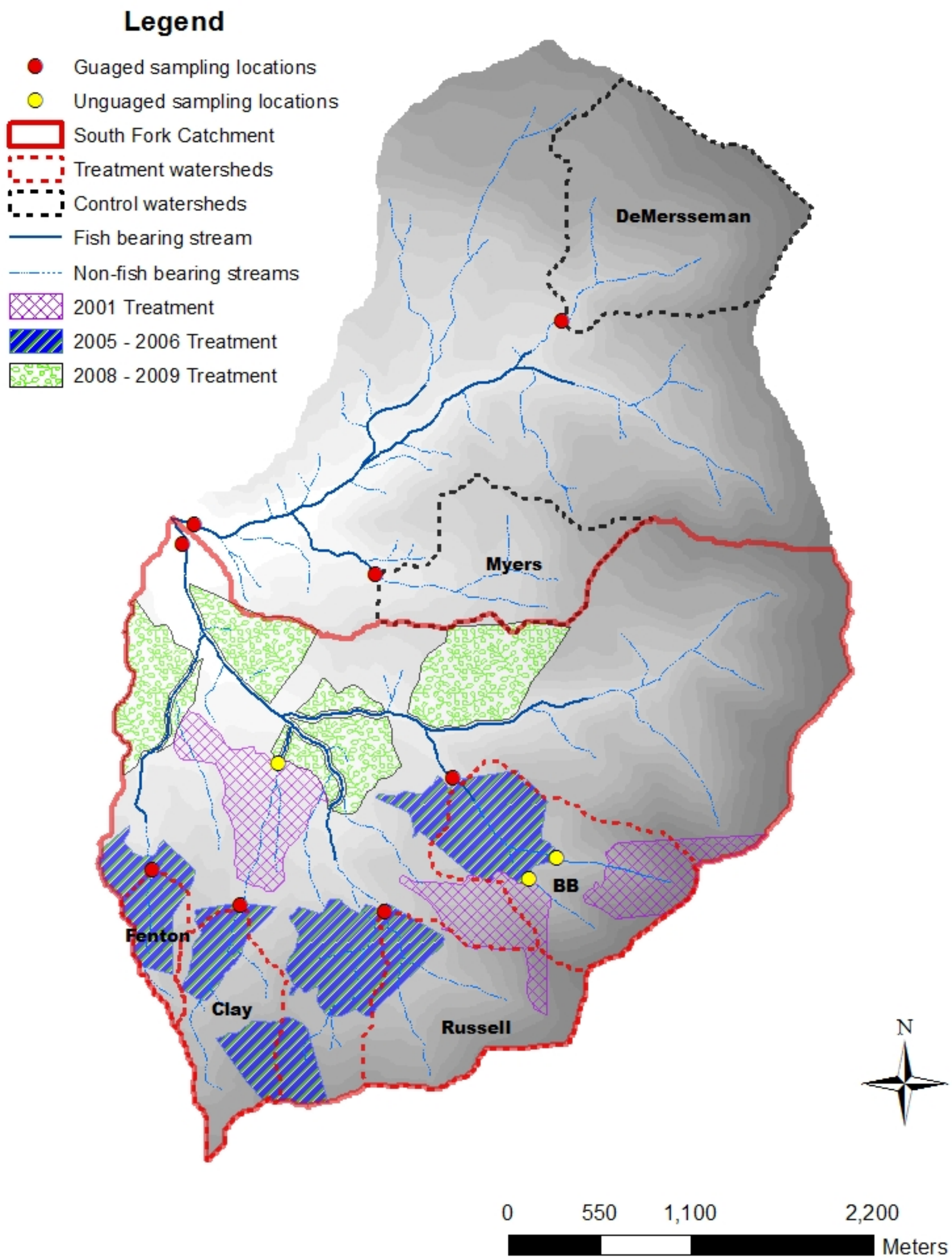


Figure 2.1 Map of Hinkle Creek showing clearcuts, basin boundaries, and water sampling locations.

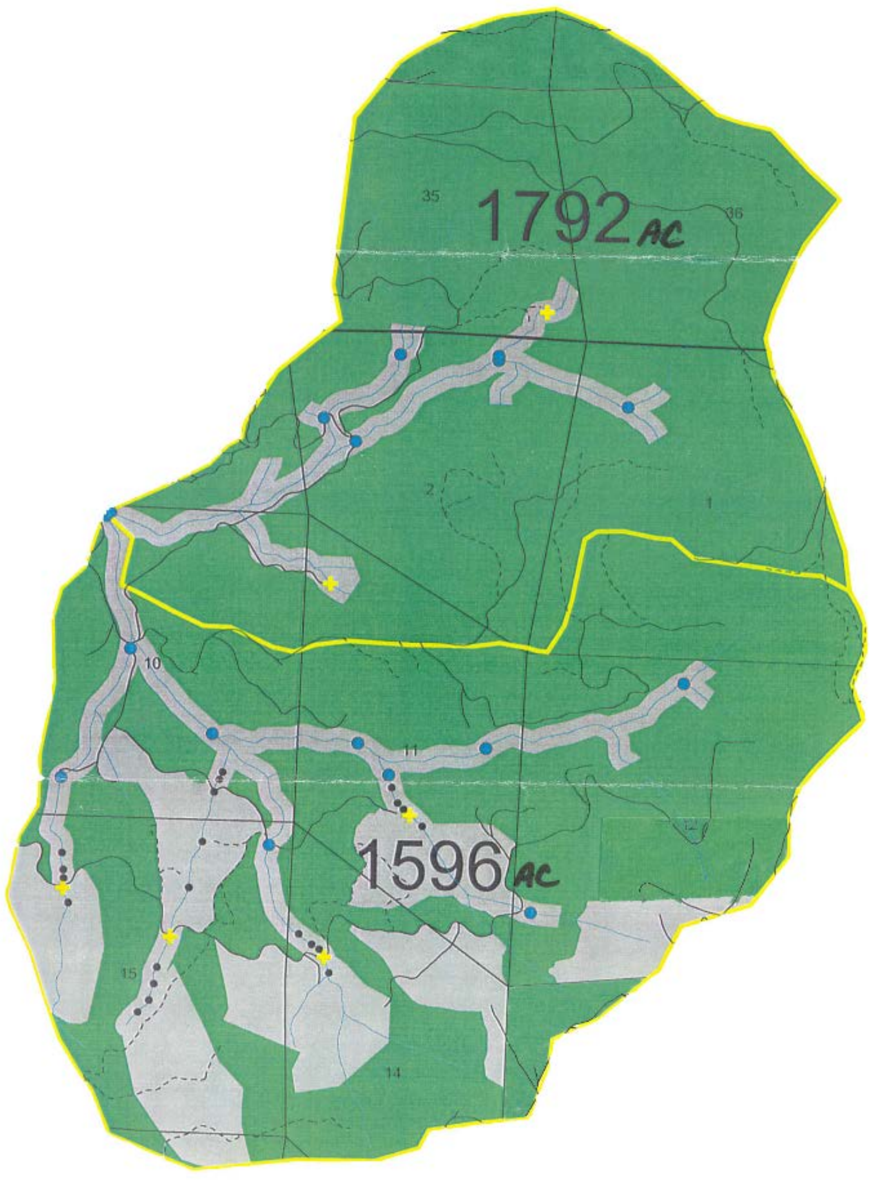


Figure 2.2 Fertilizer treatment. Fertilized areas are shown in green. Unfertilized areas are shown in gray. 60 – 100 foot buffers were left alongside fish-bearing streams.

Chapter 3: Methods

Data collection

Stream discharge measurements for the North Fork and South Fork were carried out by the United States Geological Survey via continuous stage recorders and current meter readings used to establish a stage discharge relationship curve. Stage recordings were reported in 15 or 30 minute data. For the six headwater basins, Montana flumes and Druck PDCR 1830 pressure transducers were installed to measure stage at a 10 minute interval. All eight of these gauged locations were equipped with Campbell Scientific CR10x data loggers to record stage, temperature, and turbidity.

Water samples were collected for stream chemistry analysis each month for the first year of the study beginning in October 2002. After October 2003, samples were collected seasonally. In the original study design, water samples were taken at the eight gauged locations only - in the North and South Fork just above the confluence and in the headwaters just above the flumes. Three additional water sampling sites were added in January 2003 to address elevated NO_2 + NO_3 concentrations observed after the results from the first samples were received from the lab. These new sampling sites were located at ungauged sites below the 2001 clearcuts (Figure 2.1).

Water samples were taken with a grab sample technique used by the Environmental Protection Agency. Two 1-liter bottles were filled at each sampling location. Each bottle and lid was rinsed three times in the stream prior to filling the bottle on the fourth grab. Samples were transported in coolers to the laboratory on the day of collection and placed in a freezer. Nitrogen and phosphorus were analyzed within a day or two of collection; the other nutrients were analyzed over the following weeks.

Water samples were analyzed for dissolved organic nitrogen (DON), inorganic nitrogen (nitrate + nitrite and ammonia), phosphorus, calcium, sodium, potassium, magnesium, silicon, chloride, and sulfate, as well as pH, specific conductance and alkalinity. Nitrite (NO_2) is immediately oxidized to nitrate (NO_3) under aerobic conditions (Stednick, 1991), but the lab analysis methods reduce NO_3 back to NO_2 for analysis purposes. Even though this requires results to be reported as NO_2 + NO_3, it is comparable to NO_3. This becomes important when comparisons are made with older data sets that used different analysis methods and reported results as NO_3 only. All samples were analyzed at the Cooperative Chemical Analytical Laboratory (CCAL) at Oregon State University in Corvallis, OR. Nutrient concentrations were reported in milligrams per liter (mg/L). Values that were at-or-below the level of measurement precision/detection were used as reported. Detection levels, methodology, and instrumentation used by CCAL are shown in Appendix F.

Data analysis

A Before-After/Control-Impact Paired-Series (BACIPS) study design (Stewart-Oaten et al., 1986) was used to assess changes to stream chemistry as a result of timber harvest. The calibration period was October 2002 through October 2004, just prior to fertilization. The terms *harvest* and *treatment* are used interchangeably. The term *treatment watersheds* refers to any of the harvested watersheds **before or after harvest**. The terms *pre-harvest* and *pre-treatment* refer to the calibration period for **control or treatment watersheds**.

The treatment period was December 2006 to September 2011 for all watersheds except the South Fork. Treatment data for the South Fork were split into two time periods to account for the two temporally explicit treatments. The first treatment period is December 2006 to October 2009, and the second treatment period is October 2009 to September 2011. Sample size was n = 17 for the calibration period and n = 18 for the treatment period for all watersheds except the South Fork. The first and second treatment period sample size for the South Fork was n = 10 and n = 8, respectively.

Data were analyzed for statistical significance using Student's t-tests between observed and predicted values for post-treatment time periods comparing differences between the North and South Fork, Myers and Clay, Myers and Fenton, DeMersseman and Russell, and DeMersseman and BB. All statistical computations were done using TIBCO Spotfire S+ statistical software (TIBCO, 2008). Changes in nutrient concentrations were tested for statistical significance at the $\alpha \leq 0.05$ level.

Total annual export of NO_2 + NO_3 was estimated for each gauged location. Average concentration (mg/L) for each water year was multiplied by the stream discharge (L/s), which was multiplied by the number of seconds in a year. Final results represent a yearly average for each period which was calculated by adding the total load for each year within a given period and dividing by the number of years in that period. The calibration period is comprised of a single year because discharge data was not available the first year of the two-year calibration period. Water year 2005 was considered a separate period in these calculations due to the fertilization treatment. The post-treatment period is comprised of water years 2006 - 2010. Total load was determined as kilograms per year and kilograms per hectare per year.

In the fall of 2009, water samples were collected during storms to observe the relationship between stream discharge and concentration of NO_3 + NO_2 during fall freshets. To do this, we used the water samples collected for on-going sediment research. These water samples were collected by ISCO 3700-c portable water sampling systems located at all six flumes and the North and South Fork. Each ISCO was programmed to take one liter water samples when discharge reached levels indicative of a storm. The samples were retrieved within a couple of days and brought to Corvallis for sediment analysis. Upon their arrival in Corvallis, 250 mL were removed, filtered, frozen and then analyzed for NO_3 + NO_2 at CCAL. The samples in the ISCO received no sun light during their wait between sampling and pick-up, and this analysis was done on storms during November 2009, so the samples remained dark and cold prior to analysis. Not all locations experienced adequate discharge to commence sampling for each storm and some locations never produced samples.

Chapter 4: Results

The results of this study focus on changes to nutrient concentrations as a result of contemporary forest management practices that include harvest activities, fertilization, and vegetation suppression. Although some road work did occur during this study, it was limited in scope and occurred during the harvest operations; thus, road effects are lumped with harvest effects.

Graphs that illustrate the response for each nutrient for the treatment vs. control pairs are shown in Appendix A. Graphs that illustrate the mean concentrations for each nutrient before and after treatment are shown in Appendix B. Summary tables that show the mean concentrations for select nutrients before and after treatment are shown in Appendix E.

Nitrate + nitrite (NO_3 + NO_2)

Graphs that illustrate the response for NO_3 + NO_2 in the treatment vs. control pairs are shown in Figures 4.2 - 4.4. Table 4.1 shows the mean concentration of NO_3 + NO_2 before and after treatment with p-values for all treated basins. Mean concentration of NO_3 + NO_2 in the South Fork was 6.6 times greater than in the North Fork during the calibration period (Figure 4.1). In the first treatment period, mean concentration of NO_3 + NO_2 decreased by 0.021 mg-N L^{-1} in the South Fork. In the second treatment period, mean concentration of NO_3 + NO_2 increased by 0.032 mg-N L^{-1} from the first treatment period, an increase of 0.011 mg-N L^{-1} from the calibration period. No reliable statistical comparison could be made for the South Fork between calibration and treatment periods because a credible calibration could not be established (see Discussion). Annual estimates of total NO_3 + NO_2 export are shown in Table 4.2.

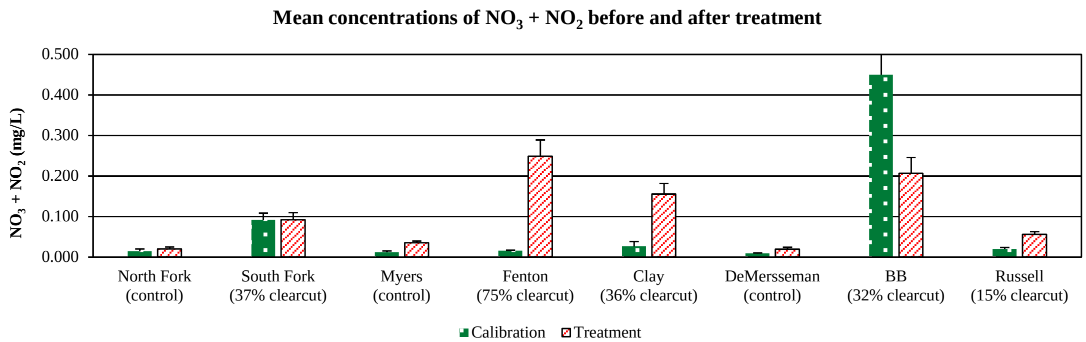


Figure 4.1 Mean concentrations of NO_3 + NO_2 for each stream during the calibration and treatment periods. The treatment results for the South Fork reflect a combination of both treatment periods. Error bars indicate S.E.

The responses for NO_3 + NO_2 in Myers, Fenton, and Clay are shown in Figure 4.3. Both treatment sites showed a statistically significant increase ($p < 0.002$) in concentrations after timber harvest (Table 4.1). Increases in Fenton (75% clearcut) were greatest, where mean concentrations increased by almost 17 times from the calibration period. Concentrations in Clay (36% clearcut) increased by almost six times. Myers (control) displayed a three-fold increase between the calibration and treatment periods (Figure 4.1). Estimated total annual loads are shown in Table 4.2.

The responses for NO_3 + NO_2 in DeMersseman, BB, and Russell are shown in Figure 4.4. Mean concentrations in BB (32% clearcut) decreased by 0.244 mg-N L^{-1} after timber harvest. Mean concentrations in Russell (15% clearcut) were more than twice the concentrations observed during the calibration period ($p < 0.0002$). DeMersseman (control) displayed a two-fold increase between the calibration and treatment periods. Estimated total annual loads are shown in Table 4.2.

Table 4.1 Mean concentrations of nitrogen and phosphorus before and after treatment with Student's t-test p-value results. All values in mg/L.

		$NO_3 + NO_2$	NH_3	DON	DP	PO_4
North Fork	(Pre)	0.014	0.004	0.039	0.018	0.009
(control)	(Post)	0.017	0.006	0.047	0.024	0.010
	p-Value	—	—	—	—	—
South Fork	(Pre)	0.092	0.006	0.037	0.018	0.009
(25% clearcut)	(Post)	0.071	0.007	0.036	0.024	0.010
	p-Value	0.200	0.590	0.305	0.930	0.140
South Fork	(Pre)	0.092	0.006	0.037	0.018	0.009
(37% clearcut)	(Post)	0.103	0.010	0.037	0.017	0.010
	p-Value	0.800	0.990	0.925	0.300	0.140
Myers	(Pre)	0.012	0.009	0.031	0.024	0.015
(control)	(Post)	0.035	0.009	0.045	0.028	0.015
	p-Value	—	—	—	—	—
Fenton	(Pre)	0.015	0.006	0.026	0.040	0.032
(75% clearcut)	(Post)	0.248	0.008	0.047	0.040	0.027
	p-Value	**0.000**	0.160	0.323	0.170	0.090
Clay	(Pre)	0.026	0.007	0.033	0.022	0.014
(36% clearcut)	(Post)	0.155	0.009	0.045	0.020	0.010
	p-Value	**0.002**	0.240	0.605	**0.003**	**0.010**
DeMersseman	(Pre)	0.009	0.006	0.027	0.017	0.010
(control)	(Post)	0.019	0.006	0.042	0.020	0.010
	p-Value	—	—	—	—	—
BB	(Pre)	0.450	0.008	0.031	0.019	0.012
(32% clearcut)	(Post)	0.206	0.007	0.043	0.021	0.012
	p-Value	**0.003**	0.530	0.775	0.270	0.600
Russell	(Pre)	0.020	0.004	0.020	0.020	0.013
(15% clearcut)	(Post)	0.056	0.007	0.029	0.022	0.012
	p-Value	**0.000**	0.080	0.195	0.150	0.600

Table 4.2 Average total annual load for $NO_3 + NO_2$ in kg yr^{-1}. Values in parenthesis indicate average total annual load per hectare in kg yr^{-1} ha^{-1}.

	Calibration	Fertilization	Treatment
North Fork	50.5 (0.06)	837.4 (0.96)	130.5 (0.15)
South Fork (37% clearcut)	721.4 (0.68)	1011.7 (0.95)	1017.7 (0.96)
Myers	5.2 (0.06)	188.3 (2.18)	17.7 (0.21)
Fenton (75% clearcut)	1.4 (0.06)	4.0 (0.18)	56.3 (2.49)
Clay (36% clearcut)	11.5 (0.18)	109.0 (1.67)	96.9 (1.49)
DeMersseman	11.8 (0.08)	694.4 (4.44)	30.7 (0.20)
BB (32% clearcut)	375.0 (3.38)	489.9 (4.42)	303.2 (2.74)
Russell (15% clearcut)	11.8 (0.12)	97.6 (1.01)	62.4 (0.65)

All streams showed an increase in total annual load except BB. The greatest increase in estimated total annual load per hectare was observed in Fenton, which had an increase of almost 4200 percent over its pre-treatment total load. The increases observed in the control watersheds ranged from 150 percent to 250 percent compared to 80 percent to 4200 percent in the treatment watersheds (excluding BB), but the low end of this scale was influenced by the 2001 treatment. BB was not included in this comparison because it showed a decrease in total load. Post-treatment increases in total load for the treatment headwaters had a high correlation to the slope of the clearcut basin, discussed below.

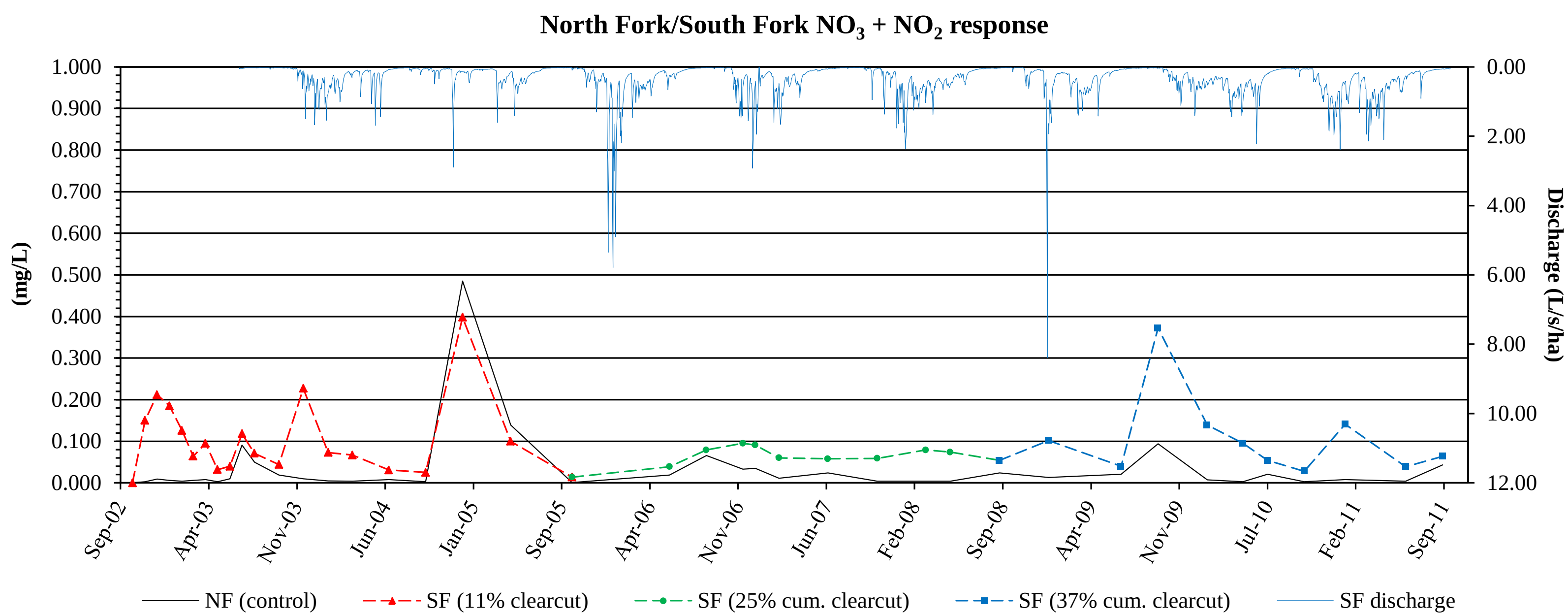


Figure 4.2 North Fork and South Fork $NO_3 + NO_2$ response. The spike in January 2005 reflects the first sample taken after fertilizer treatment. The spike in October 2009 reflects the influence of storm discharge.

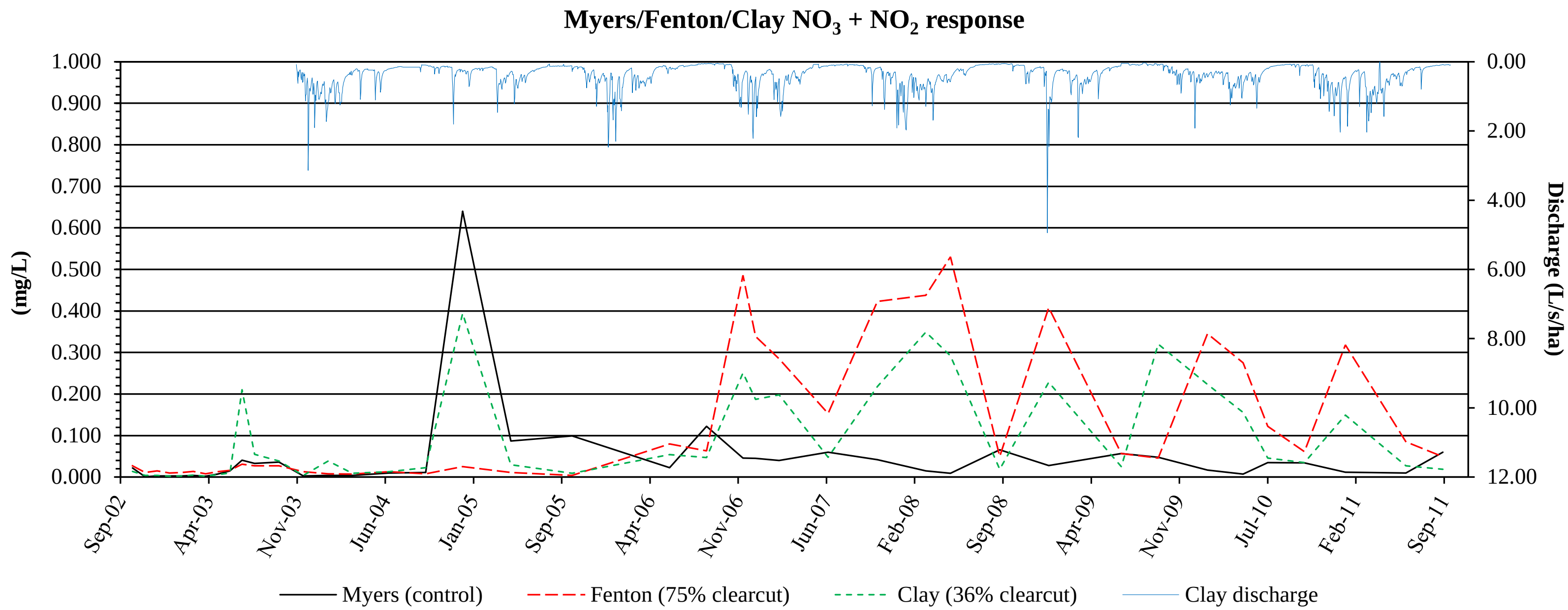


Figure 4.3 Myers, Fenton, and Clay $NO_3 + NO_2$ response. The spike in January 2005 reflects the first sample taken after fertilizer treatment.

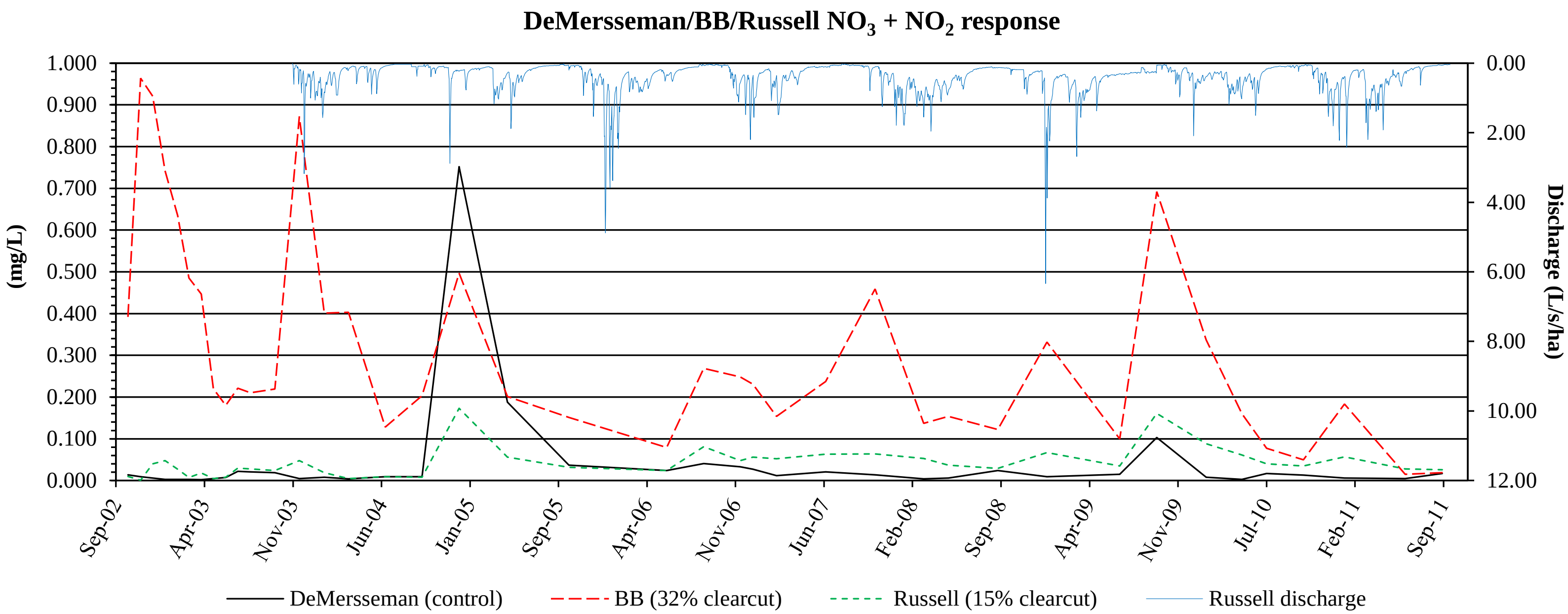


Figure 4.4 DeMersseman, BB, Russell $NO_3 + NO_2$ response.

Ammonia and dissolved organic nitrogen (DON)

Mean concentrations of ammonia were low (~0.007 mg-NH_3 L^{-1}) for all streams for the duration of the study. Many values were below the limit of detection. Some treatment watersheds displayed a notable increase in concentrations of ammonia after treatment, but these increases were attributed to increases observed in the control watersheds. A statistically significant change was not found between the calibration and treatment periods for any of the streams (Table 4.1). In October 2009, during the second treatment period of the South Fork, the concentration of ammonia was 0.038 mg-NH_3 L^{-1}, the maximum observed for any stream during the study.

Concentrations of dissolved organic nitrogen (DON) displayed a notable increase in all six headwaters (Table 4.1). Observed increases in the treatment watersheds were attributed to increases in the control watersheds. A statistically significant change was not detected.

Dissolved phosphorus (DP) and orthophosphates (PO_4)

Mean concentrations of DP and PO_4 before and after treatment are shown in Table 4.1. Concentrations of DP and PO_4 did not respond to timber harvest for any streams except Clay. Seasonal response for both DP and PO_4 was evident. Peak concentrations occurred more often in the summer. Fenton showed elevated phosphorus and orthophosphate levels before and after treatment, but did not respond to timber harvest.

Cations (Ca, Na, Mg, K) and sulfate (SO_4), chlorine (Cl), and silicon (Si)

Mean concentrations of all the cations for the control and treatment watersheds are shown in Table 4.3. All four cations showed a decrease in concentration across all watersheds. Concentrations were higher in the control watersheds during calibration, but the greatest decreases were observed in the treatment watersheds. Strong seasonal fluctuations were apparent for all species. Peak concentrations occurred in the fall.

Table 4.3 Mean concentrations of Ca, Na, Mg, and K in the control and treatment watersheds before and after treatment. All values are in mg/L.

	Ca			Na			Mg			K		
	(Pre)	(Post)	Change	(Pre)	(Post)	Change	(Pre)	(Post)	Change	(Pre)	(Post)	Change
Control	6.34	6.12	-0.22	4.10	3.94	-0.15	1.86	1.54	-0.32	0.42	0.38	-0.04
Treatment	5.48	5.01	-0.47	3.51	3.30	-0.22	1.39	1.09	-0.30	0.37	0.33	-0.04

Mean concentrations of sulfate (SO_4), chloride (Cl), and silicon (Si) for the control and treatment watersheds are shown in Table 4.4. Mean concentrations of sulfate (SO_4) were greater in the treatment watersheds during the calibration period, and showed a greater decrease in the treatment watersheds after treatment. Fenton (75% clearcut) displayed the greatest decrease of -0.06 mg L^{-1}. As with the cations, sulfate, chlorine and silicon displayed strong seasonal fluctuations. Peak concentrations occurred in the fall.

Mean concentrations of Cl were higher in the control watersheds during the calibration period, but the treatment watersheds displayed a greater decrease after treatment. Myers displayed the only increase after treatment of 0.01 mg L^{-1}. BB (32% clearcut) displayed the greatest decrease of 0.37 mg L^{-1}.

Mean concentrations of Si were greater in the control watersheds than in the treatment watersheds. As with Cl, concentrations of Si in the treated watersheds showed a greater decrease than the control watersheds. The North Fork and

DeMersseman control watersheds were the only two to show an increase in mean concentrations of Si during the treatment period, with increases of 0.03 mg L^{-1} and 0.02 mg L^{-1}, respectively. Fenton (75% clearcut) showed the greatest decrease, with 0.59 mg L^{-1}.

Table 4.4 Mean concentrations of SO_4, Cl, and Si in the control and treatment watersheds before and after treatment. All values are in mg/L.

	SO_4			Cl			Si		
	(Pre)	(Post)	Change	(Pre)	(Post)	Change	(Pre)	(Post)	Change
Control	0.15	0.14	-0.01	1.56	1.54	-0.02	8.59	8.55	-0.04
Treatment	0.16	0.13	-0.03	1.44	1.23	-0.21	8.40	8.05	-0.35

Specific conductance, pH, and alkalinity

Mean values for the control and treatment watersheds for specific conductance, pH, and alkalinity for the calibration and treatment periods are shown in Table 4.5. Specific conductance decreased for all streams during the study, with the greatest decreases observed in the treatment streams. The minimum specific conductance for any stream during the study was 32.5 µS cm^{-1}, observed once in Fenton during calibration, and the maximum was 111.7 µS cm^{-1}, observed once in DeMersseman during calibration. There was no change to mean pH levels for any of the streams after treatment. The minimum pH for any stream during the study was 7.2, observed once in Fenton during calibration. The maximum pH for any stream was 7.9, observed once in post-treatment DeMersseman, which also had the highest pre- and post-harvest mean pH of 7.7. Alkalinity decreased for all streams between pre- and post-treatment, with the greatest decreases observed in the treatment streams. The minimum detected alkalinity, 3.57 HCO_3, occurred in pre-treatment Fenton, and the maximum detected alkalinity, 13.90 HCO_3, occurred in pre-treatment DeMersseman.

Table 4.5 Mean values for specific conductance (µS cm^{-1}), pH, and alkalinity (mg-HCO_3 L^{-1}) in the control and treatment watersheds.

	Conductance			pH			Alkalinity		
	(Pre)	(Post)	Change	(Pre)	(Post)	Change	(Pre)	(Post)	Change
Control	61.4	59.9	-1.4	7.6	7.6	0.0	7.2	7.0	-0.2
Treatment	53.0	49.0	-4.0	7.5	7.5	0.0	6.0	5.6	-0.5

Chapter 5: Discussion

Non-fish-bearing streams are not afforded the same protection during harvest activities as fish-bearing streams. Over-story canopy and fertilizer-free buffer strips are not required to be left along non-fish-bearing streams as they are along fish-bearing streams. The Hinkle Creek Paired Watershed Study used two temporally and spatially explicit scales to determine what the downstream impacts of these practices adjacent to non-fish-bearing streams might have on fish-bearing streams, and to compare these downstream impacts to the impacts from harvest practices carried out directly along fish-bearing streams.

Ideally, all the watersheds in this paired watershed study would have been devoid of any disturbance prior to the study, but this was not the case. The 2001 clearcuts resulted in elevated NO_3 + NO_2 concentrations in BB and the South Fork during the calibration period. The levels were pronounced enough to prevent a reliable calibration between the North and South Fork and BB and DeMersseman. Consequently, a credible statistical comparison of means could not be made for these streams between the calibration and treatment periods.

The uneven application of fertilizer between the North and South Fork in the fall of 2004 further complicated the analysis. The North Fork received more fertilizer than the South Fork because the 2001 harvest units and the planned 2005 harvest units were not fertilized. Despite the addition of these confounding variables, an analysis of temporal and spatial trends in the data was informative, and credible statistical comparisons were possible for Fenton, Clay, and Russell.

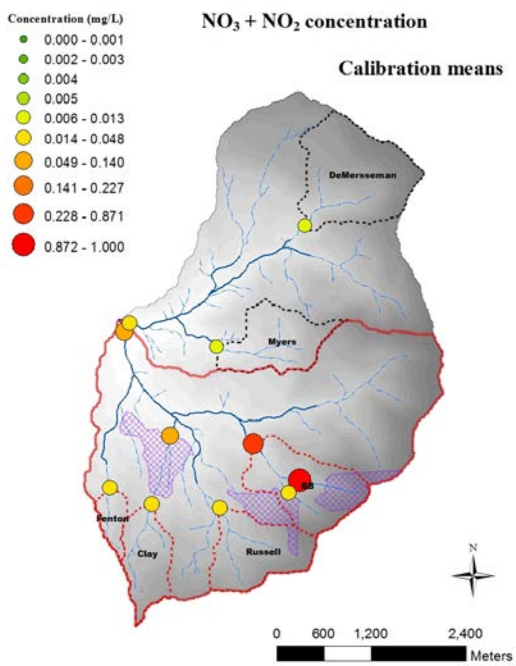
Old-growth forests and regenerated forests: A comparison

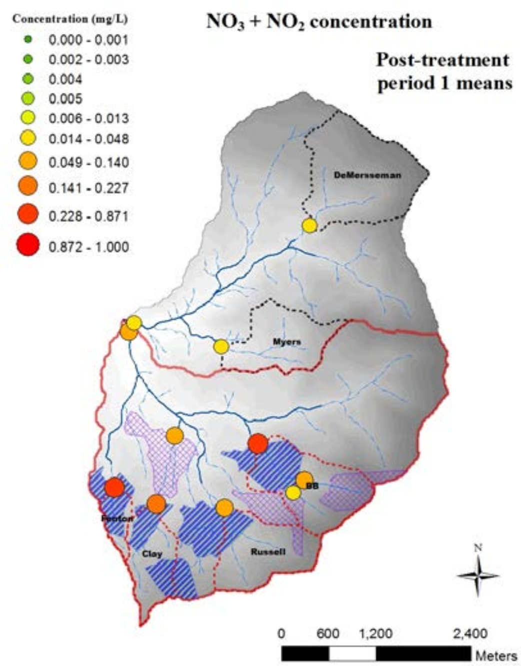
Critics of studies that involve assessment of impacts from timber harvest on second-growth, harvest regenerated stands argue that credible estimates of a treatment response cannot be made because the calibration period itself represents an altered state of the system, so any conclusions about long-term impacts from these studies are speculative. While this criticism may be valid, this theory is unlikely to be verified because most of the remaining old-growth forests are protected for conservation. So, a comparison of calibration data with data collected from streams draining old-growth forests is the best available method to measure how well the ecosystem has returned to conditions that existed prior to the first [harvest] disturbance. This metric can then be used as a surrogate baseline with which to evaluate the effects of current harvest practices.

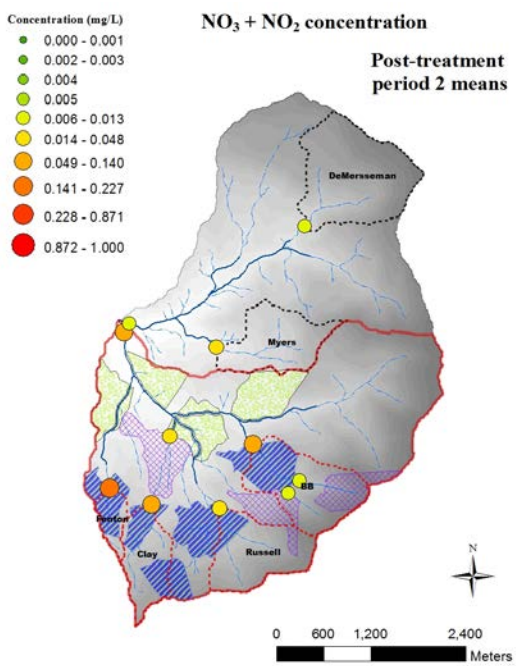
Concentrations of NO_3 + NO_2 measured in the North Fork during the calibration period are shown below with stream water concentrations of NO_3 measured in four watersheds dominated by old-growth Douglas-fir stands located 104 kilometers north-northeast at the H.J. Andrews Experimental Forest (Table 5.1). Concentrations of NO_3 + NO_2 observed in Hinkle Creek during the calibration period were below concentrations observed at the H.J. Andrews WS10 (Sollins et al., 1980), but greater than concentrations observed in WS2, WS9, and WS8 (Vanderbilt et al., 2003). This comparison indicates that the processing of NO_3 + NO_2 at Hinkle Creek had returned to conditions that probably existed prior to harvest 60 years ago.

Table 5.1 Concentrations of NO_3 observed in four old-growth forests in WS2, WS8, WS9, and WS10 at H.J. Andrews compared to concentrations NO_3 + NO_2 observed in the North Fork during calibration. All values are in mg/L.

	North Fork	WS2	WS8	WS9	WS10
NO_3	0.014	0.001	0.004	0.003	0.019







	Mean $NO_3 + NO_2$ (mg/L)		
	Calibration	Period 1	Period 2
North Fork	0.015	0.017	0.023
South Fork	0.098	0.075	0.117
Myers	0.011	0.039	0.028
Fenton	0.015	0.346	0.163
Clay	0.027	0.199	0.122
DeMersseman	0.008	0.017	0.022
BB	0.453	0.230	0.192
Russell	0.020	0.052	0.062
Lower Clay	0.054	0.149	0.024
North BB	0.888	0.077	0.002
South BB	0.034	0.025	0.001
Treatment Headwaters	0.129	0.207	0.135

Figure 5.1 Mean concentration of $NO_3 + NO_2$ during the calibration, post-treatment period 1, and post-treatment period 2 for all sampling locations.

$NO_3 + NO_2$

Although this research analyzes several nutrients and other water chemistry parameters, a strong emphasis was placed on the analysis of the nitrogen species because nitrogen is the nutrient required in the largest abundance by vegetation and the one considered to be the most limiting nutrient in this western Oregon ecosystem. $NO_3 + NO_2$ in particular is known to show a notable response to timber harvest due to its mobility (Chapin et al., 2002).

A number of questions arose when a comparison of the response for $NO_3 + NO_2$ in the South Fork was made between the calibration period and both treatment periods. Mean concentration in the South Fork during the first treatment period was 20 percent less than levels observed during calibration, even though the mean concentration in the treatment headwaters was 63 percent greater during this treatment period than the calibration period. It was thought that the 2001 clearcut along lower Clay imparted a stronger signal to the South Fork than the headwater clearcuts because of its proximity to the South Fork (Figure 2.1). However, the mean $NO_3 + NO_2$ concentration observed in Lower Clay during the calibration period was 0.054 mg-N L^{-1}, too low to account for the 0.092 mg-N L^{-1} mean concentration observed in the South Fork.

Although the mean concentration across all the treatment headwaters was 0.128 mg-N L^{-1} during the calibration period, the mean concentration in BB during this period was 0.450 mg-N L^{-1}, well above the 0.202 mg-N L^{-1} mean concentration observed in the headwaters after treatment. The location of the gauging station where the water samples are collected for BB is about the same distance to the South Fork station where water samples were collected as the other headwater gauging stations, but it is three-to-four times closer to the main stem of the South Fork than the other headwater gauging stations (Table 2.1, Figure 2.1). The shorter distance to the main stem from the BB station relative to the other headwater stations meant less exposure to the greater attenuation of the lower-volume headwater streams compared to the attenuation of the higher-volume main stem. The lower ratio of volume-to-surface area for the main stem reduces attenuation compared to the higher ratio of volume-to-surface area for the smaller tributaries (Binkley et al., 2004). It appeared that the elevated NO_3 + NO_2 concentrations observed in the South Fork during calibration were a result of the 2001 clearcuts in the upper BB watershed.

The source of the NO_3 + NO_2 observed in BB and the South Fork during the calibration period was determined to be from the clearcut located on the north BB tributary, which was 70 percent clearcut. The mean concentration observed below this clearcut during the calibration period was 0.888 mg-N L^{-1}, more than four times greater than the concentrations observed below the 2006 BB clearcut during the first treatment period of the South Fork, and almost three times greater than the concentrations observed in Fenton during this same period. It was concluded that this clearcut was responsible for the elevated concentrations of NO_3 + NO_2 observed during the calibration period.

In addition to these unexpected NO_3 + NO_2 responses, the temporal trends in the South Fork after the 2005-06 and 2008-09 treatments were surprising. The observed response during both treatment periods was visibly different from the response to the 2001 harvest (Figure 4.2). During the calibration period, there was a pronounced seasonal response, with winter peaks above 0.200 mg-N L^{-1}. A seasonal response was barely apparent during the first treatment period, when peak values never reached above 0.100 mg-N L^{-1}. Even after the 2008-09 treatments, which were located directly along the South Fork and should have produced a strong signal because of minimal attenuation, seasonal response was still less than the response from the 2001 harvest (the elevated concentration observed in October 2009 can be partially attributed to the elevated concentration in the North Fork at that time, discussed below). It appeared that the NO_3 + NO_2 concentrations produced by the clearcut in the north BB tributary were responsible for eliciting a greater seasonal response than all the 2008-09 clearcuts combined.

Concentrations of NO_3 + NO_2 in the treatment headwaters (except Russell) displayed a strong seasonal response, with peak concentrations during the fall and high-precipitation winter months. Fenton (75% clearcut) and Clay (36% clearcut) showed the highest concentrations the second year after harvest. This display of peak concentrations in the second year is consistent with the observations of other studies carried out in the Pacific Northwest (Brown et al., 1973; Gravelle et al., 2009). BB was on track to do the same, but the concentrations observed in October 2009 were higher. Russell (15% clearcut) displayed minimal seasonal response, with almost no visible difference between peaks and valleys of the response curve.

Dissolved phosphorus and orthophosphate

Concentrations of dissolved phosphorus and orthophosphate observed at Hinkle Creek were similar to concentrations and responses observed in watersheds 6 (WS-6) and 7 (WS-7) at the H.J. Andrews (Martin and Harr, 1989) where the biogeochemistry is more heavily influenced by the younger volcanic parent material than in other regions of the United States. The two-fold DP and OP concentrations observed in Fenton demonstrate the high variability of geologic influence within a single watershed (Feller, 2005).

Cations (Ca, Na, Mg, K) and sulfate (SO_4), chloride (Cl), and silicon (Si)

Concentrations of Ca, Na, and Mg after treatment were nearly twice the concentrations of these nutrients observed by Martin and Harr (1989) in the H.J. Andrews after harvesting and burning but a small fraction of those observed at Hubbard Brook after harvesting followed by herbicide (Likens et al., 1970). The decline in observed concentrations of these nutrients before and after treatment across all watersheds was a bit perplexing, but even more perplexing was that the greatest decreases occurred in the treatment watersheds. Patterns in the data suggest that fertilization may have enhanced the in-stream up-take of these nutrients, and research has shown that in-stream nutrient uptake can be further enhanced by the removal of the riparian zone, which may explain why the greatest decreases were observed in the harvested watersheds (Bernhardt et al., 2003).

Specific conductance, pH, and alkalinity

Specific conductance is a measure of the electrical conductance of water, thus it is indicative of the concentration of total dissolved ions. The decrease in specific conductance observed in all watersheds after treatment reflects the pattern of decreased ion concentrations observed.

The pH of stream water showed no response to timber harvest. This response was similar to that observed by Tiedemann in the Blue Mountains of eastern Oregon (Tiedemann et al., 1988). Mean pH showed no change between the calibration and treatment periods, and the maximum pH detected, 7.9, was well below the maximum set by the Oregon Department of Environmental Quality (OAR 340-041-0033).

The alkalinity levels in Hinkle Creek were about a third of those observed by Martin and Harr (1989) in H.J. Andrews. The decrease in alkalinity across all watersheds coincides with the decrease in concentration of several nutrients.

Relationship between nutrient concentration and stream discharge

Investigation of the notably elevated NO_3 + NO_2 concentrations observed in the North and South Fork in January 2005 and October 2009 revealed some important circumstances. After an inspection of the antecedent discharge behavior for each sample, it was found that the majority of the samples throughout the study were taken during baseflow, when discharge was not associated with precipitation. But the samples taken in January 2005 and October 2009 were taken on the falling limb of a hydrograph, at or near the peak, where the results from the storm response analysis indicated peak concentrations often occur (Appendix C). Just as important, both of these small storms occurred after significant dry spells; the January 19 storm occurred after a month of negligible precipitation during an uncharacteristically dry winter, and the October 27 storm occurred after several dry summer months.

The storm response results show that concentrations of NO_3 + NO_2 in stream water increase with discharge during small storms that occur after periods of negligible precipitation. This seems counter-intuitive, because increased amounts of water added to a system typically results in dilution. Precipitation in the Oregon Cascades possesses appreciable concentrations of NO_3 + NO_2 (Martin and Harr, 1989; Vanderbilt et al., 2003), but this contribution cannot account for the increases observed during storm runoff. Dahlgren (1998) and Feller (2005) attribute the large increases observed in concentrations of NO_3 + NO_2 during storm events to changes in hydrologic flowpaths, noting that the larger macropores that remain dry during unsaturated conditions serve as a reservoir for NO_3 + NO_2 until they are flushed out during periods of saturation when preferential flowpaths develop.

Figure 5.2 shows South Fork hydrographs for the samples taken January 2005 and October 2009. The other two hydrographs show the concentrations of NO_3 + NO_2 at different intervals during small storms in the South Fork and Clay Creek in November 2009. The January 2005 and November 2009 South Fork storms are nearly equal in magnitude but have different durations. The 2005 storm occurred over several days and the 2009 storm occurred over several hours. The Clay Creek storm response demonstrates that this pattern was observed in other streams during different storms of great magnitude. A complete display of storm response data is provided in Appendix

C. Hydrographs for each water year that indicate the hydrologic circumstances for each sample are shown in Appendix D.

For the storm sampled on November 6, concentrations were 0.040 mg-N L^{-1} halfway up the rising limb of the hydrograph, 0.147 mg-N L^{-1} near the peak, and 0.200 mg-N L^{-1} halfway down the falling limb. This trajectory is consistent with patterns seen in other studies that analyzed the nutrient concentration vs. stream discharge relationship during storms and found that concentration of NO_3 + NO_2 often reaches its maximum during the recession phase of the hydrograph (Hill, 1993; McDiffett et al., 1989; Newbold et al., 1995).

The five-fold increase observed between the rising limb and falling limb concentrations suggests that the sample taken on January 19, 2005 was strongly influenced by timing. Although this sample was expected to show elevated concentrations because of the application of fertilizer two months prior (Bisson et al., 1992), it appears the perceived effects of fertilization may have been amplified because the sample was taken on the falling limb of the hydrograph after nearly a month of negligible precipitation. Even without considering the influence of discharge, the elevated concentrations in the South Fork after fertilization were well below the concentrations found in other fertilized forests (Binkley et al., 1999; Binkley et al., 2004).

The sample taken October 2009 was taken near the peak of the hydrograph after a four month summer dry period, which allowed NO_3 + NO_2 to build up in the soil solution (McDiffett et al., 1989). Again, this position on the hydrograph was shown by our storm response data to yield concentrations of NO_3 + NO_2 up to five times greater than those observed halfway up the rising limb and was similar to the responses observed in a headwater streams in southern Ontario (Hill, 1993) and Pennsylvania (McDiffett et al., 1989).

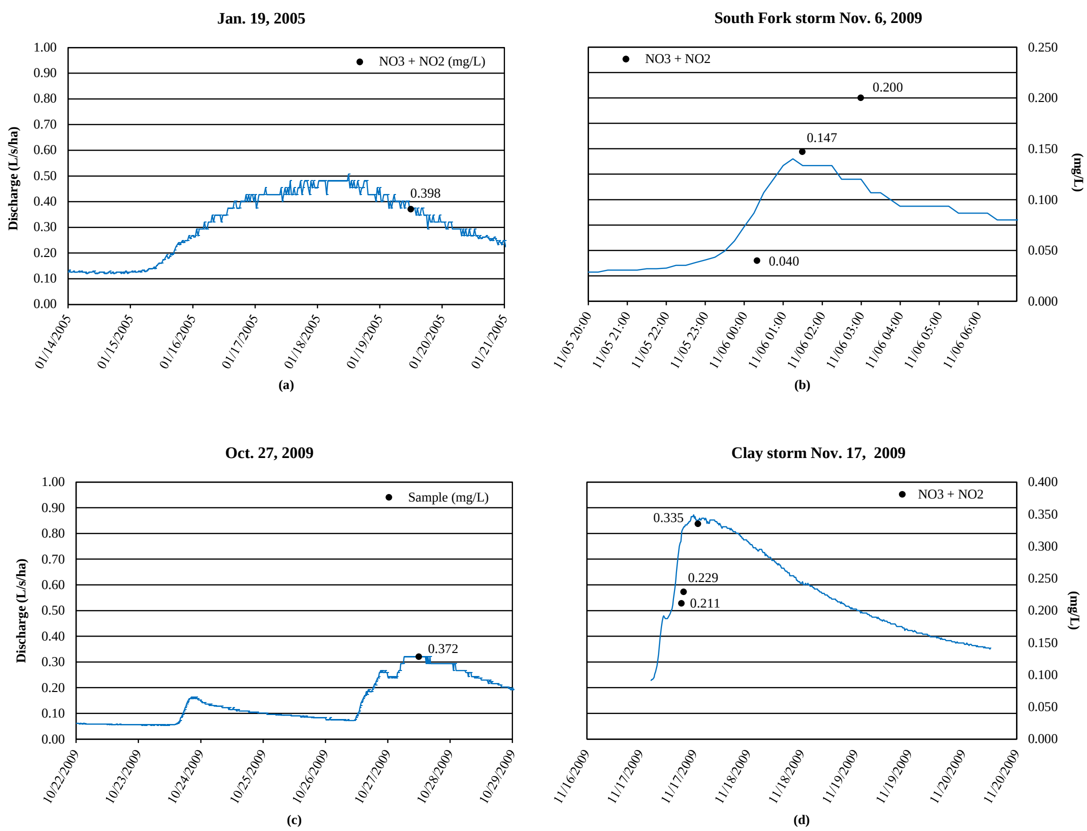


Figure 5.2 Concentration of $NO_3 + NO_2$ vs. discharge relationship during storms. Graph (a) shows the hydrologic circumstances for the samples collected in January 2005. Graph (b) shows the concentration trajectory observed in the South Fork during a November storm in 2009. Graph (c) shows the hydrologic circumstances for the samples collected in October 2009. Graph (d) shows the concentration trajectory observed in Clay Creek during a November storm in 2009.

The influence of timber harvest on stream flow

Removal of the canopy eliminates interception and transpiration, which increases storm runoff and stream discharge (Beschta et al., 2000; Thomas and Megahan, 1998; Ziemer, 1998). This effect was evident at Hinkle Creek after timber harvest (Zegre, 2008). Even though this effect was not included in this stream chemistry analysis, the possible effects are worth mentioning. Post-treatment summer low flows experience the greatest percentage increase compared to winter high flows (Ziemer, 1998). Because an increase in the concentration of NO_3 + NO_2 with discharge has been shown here and in other studies, the possibility exists that concentrations of NO_3 + NO_2 observed during the summer were greater not only because of increased nutrient leaching but also because of an increase in discharge.

Conversely, the less mobile cations can show a negative correlation with discharge because of dilution (Bond, 1979; McDiffett et al., 1989). This dilution effect was eliminated as a possible explanation for the decrease in mean post-treatment concentration of several nutrients because a similar decrease was observed in the control watersheds.

Longitudinal effects and post-harvest recovery

The decision to add three sampling locations early in the study provided some valuable information that would have otherwise gone undiscovered. The addition of two sampling locations on the headwater tributaries of BB provided more than just an explanation for the source of the elevated NO_3 + NO_2 levels observed in the beginning of the study. The north tributary drains a basin that was 70% clearcut in 2001, providing the study's longest record of treatment response and the only record of a stream without subsequent disturbance to have returned to [assumed] pre-treatment concentrations. In November 2007, six years after treatment, when all the other streams displayed a spike in concentrations of NO_3 + NO_2, concentrations observed in the north tributary of BB were near the limit of detection, and remained there for the rest of the study. This length of post-treatment recovery time is consistent with the findings of other studies that investigated the effects of timber harvest on stream water nutrient dynamics (Dahlgren, 1998; Gravelle et al., 2009; Martin and Harr, 1989). The Lower Clay location showed a return to [assumed] pre-treatment levels by July 2010.

The addition of the sampling locations on upper BB and Lower Clay also made it possible to assess longitudinal effects of timber harvest on nutrient concentrations. The Lower Clay sampling location was at the bottom of a large clearcut along Clay Creek, 1,000 meters downstream of the first Clay sampling location. Once increased concentrations of NO_3 + NO_2 were observed in Clay after the 2005-06 treatments the effects of attenuation could be measured. Mean concentrations of NO_3 + NO_2 after treatment in Lower Clay were almost half of those observed in Clay (36% clearcut). In one instance in October 2009 when the concentration of NO_3 + NO_2 in Clay was 0.320 mg-N L^{-1}, no NO_3 + NO_2 was detected at Lower Clay. No riparian buffer was left in this clearcut because it is a non-fish-bearing stream, and the trees are 5 – 7 feet tall at the time of this writing, so this result sheds light on stream attenuation rates during the very early phases of reforestation. Unfortunately, no discharge data was available for the Lower Clay location, so no comparison of total load estimates was possible.

While the addition of the Lower Clay sampling location made it possible to assess NO_3 + NO_2 attenuation in a stream passing through a very early succession clearcut, the addition of the upper BB sampling location made it possible to assess NO_3 + NO_2 attenuation in a stream with high gradient underneath the canopy of ~55 year-old Douglas firs. Between January 2003 and October 2004, mean decrease in NO_3 + NO_2 concentrations between the upper BB and lower BB sampling location was 0.314 mg-N L^{-1}. During one instance in December 2003, concentrations in upper BB were 1.746 mg-N L^{-1} (the highest observed for any stream for any period of the study) while concentrations in lower BB were 0.871 mg-N L^{-1}, half of what they were about 800 meters upstream.

Fertilization

The use of urea for the fertilization of forests has been shown to temporarily increase stream nitrogen concentrations by up to two orders of magnitude (Binkley and Brown, 1993), but such concentration increases decrease when unfertilized buffer strips were left along stream channels (McClain et al., 1998). Three species of dissolved nitrogen increase after the application of urea: dissolved organic nitrogen (DON), which is generally present for a few days; ammonia, which can often show elevated concentrations for several weeks to several months; and nitrate, which can show elevated concentrations for a year or more (Anderson, 2002; Hetherington, 1985; Moore, 1975). At Hinkle Creek, all three of these forms of nitrogen demonstrated a lasting effect in all streams that were affected, but this cannot be attributed to fertilization because this effect was observed in Fenton, which was not fertilized. DON showed a notable increase in all the headwater streams, but a muted effect in the North and South Fork (Figure 5.3). Ammonia showed a marked increase in the North and South Fork and three of the headwater streams, but showed only a small increase in DeMersseman and a decrease in Myers and BB (Figure 5.4)

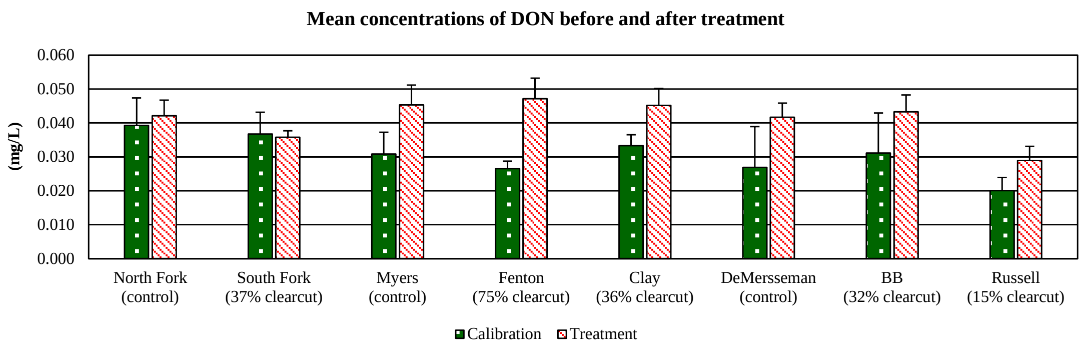


Figure 5.3 Mean concentrations of DON for all streams. Error bars represent S.E.

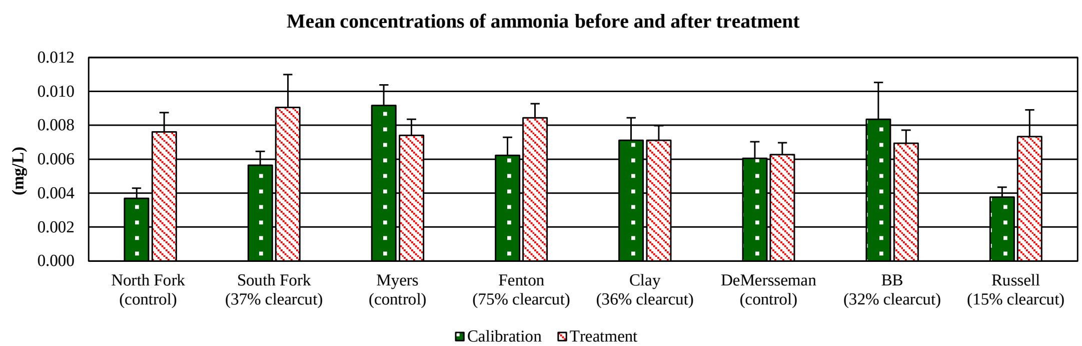


Figure 5.4 Mean concentrations of ammonia for all streams. Error bars represent S.E.

Vegetation suppression

Vegetation suppression at Hinkle Creek was done with herbicide treatment. Herbicides were applied two or three times within the first two years after harvest, depending on results. The method for herbicide treatment was one of prevention rather than reduction, meaning the herbicide was applied after logging and then again in the spring(s) to prevent vegetation from becoming established, as opposed to killing already-established vegetation.

Magnitude of disturbance relative to magnitude of response

$NO_3 + NO_2$ response in this study had a low correlation to the magnitude of disturbance. A linear regression comparing mean concentrations observed during the second and third year after treatment to the size of the clearcut (% of basin clearcut) for Fenton (75% clearcut), BB (32% clearcut), Clay (36% clearcut), Russell (15% clearcut) and North BB (70% clearcut) had an R^2 value of 0.42 (Figure 5.5a).

Conversely, topography was a strong predictor of the magnitude of response. A linear regression comparing the mean concentration after treatment to the average slope of the streambed through the clearcut for Fenton (12% slope), BB (21% slope), Clay (15% slope), Russell (17% slope), and North BB (53% slope) had an R^2 value of 0.91 (Figure 5.5b). Combining the parameters of slope and clearcut percentage was the strongest predictor of the magnitude of response. A linear regression comparing mean concentrations after treatment to clearcut percentage multiplied by slope had an R^2 value of 0.99 (Figure 5.5c). A comparison of average total annual load to clearcut percentage multiplied by slope had an R^2 value of 0.84 (Figure 5.5d).

The concentrations of $NO_3 + NO_2$ observed in the South Fork after each of the three temporally explicit disturbances showed that the downstream impact from a single steep clearcut located in the furthest reaches of the watershed was greater than the downstream cumulative impacts of the four moderately sloped clearcuts. The North BB tributary (70% clearcut), with a slope of 53 percent, was responsible for producing elevated concentrations of $NO_3 + NO_2$ 3,600 meters downstream in the South Fork. These elevated concentrations were 30 percent greater than the cumulative downstream concentrations from the four clearcuts located no further than 2,800 meters from the South Fork, and only 13 percent less than the cumulative concentrations produced by the four clearcuts located directly along the South Fork. The North BB clearcut was just 3 percent of the area of the watershed, and the four South Fork clearcuts were 12.2 percent of the area of the watershed.

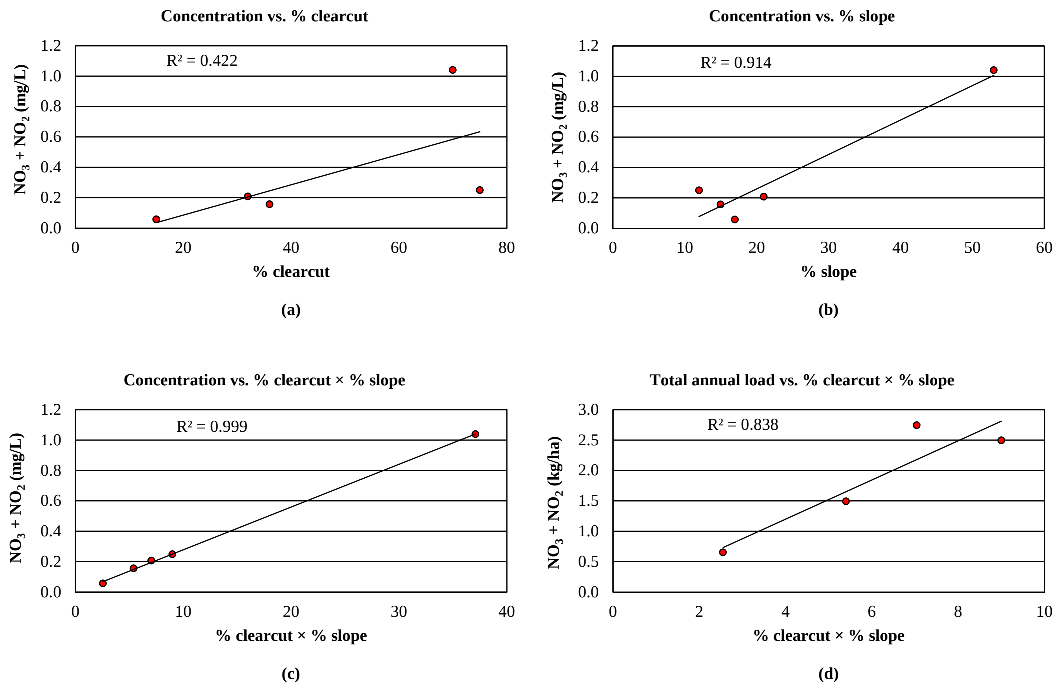


Figure 5.5 Magnitude of response relative to magnitude of disturbance. Graph (a) shows the relationship between % clearcut and mean concentration of NO_3 + NO_2 after treatment. Graph (b) shows a comparison between % slope and mean concentration of NO_3 + NO_2 after treatment. Graph (c) shows a comparison between % clearcut × % slope and mean concentrations of NO_3 + NO_2 after treatment. Graph (d) shows a comparison between % clearcut × % slope and total annual load after treatment. Only four data points are shown in graph (d) because discharge data was not available for the Upper BB clearcut.

It can be immediately observed in graph (b) that the far right data point had a strong influence on the position of the regression line and thus the R^2 value. Likewise, the highest point in graph (a) also had a strong influence on the outcome of the regression. The standard error and p-value for the regressions shown in (a), (b), and (c) are shown in Table 5.2. The regression shown in (d) is based on values that were a combination of two values whose temporal resolution was too crude to arrive at a reliable number; hence, no statistics are presented for this regression.

Table 5.2 Regression statistics for Figure 5.5

Figure	SE	p-value
(a)	0.35	0.236
(b)	0.13	0.011
(c)	0.01	0.001

Chapter 6: Conclusion

One of the objectives of this study was to determine the downstream impacts of forest management practices carried out in headwater basins that are not afforded the protection of unharvested riparian zones or unfertilized buffer strips. To increase logging efficiencies, multiple headwater basins are harvested in clusters, so there is a concern that the cumulative impacts of this type of intensive management on several headwater basins might pose a threat to fish-bearing streams further downstream. A second objective of this study was to assess the efficacy of unharvested riparian zones and unfertilized buffer strips required along fish-bearing streams and compare these impacts to the cumulative impacts from headwater basins that are not afforded such protection.

Local effects were measured in four non-fish-bearing headwater basins that experienced varying degrees of timber removal. The cumulative impacts to these headwater streams were measured downstream in the larger fish-bearing South Fork. Two years later, local effects to the South Fork were measured after the completion of four treatments located directly along the South Fork.

Our data indicated that industrialized timber harvest temporarily but significantly increased local and downstream stream concentrations of $NO_3 + NO_2$ at Hinkle Creek, but the maximum levels detected were well below all levels set for water quality by the EPA. Concentrations of ammonia, DON, and phosphorus did not show an appreciable response to timber harvest. The remaining nutrients analyzed showed a decrease after treatment, but this was not attributed to timber harvest. With regards to $NO_3 + NO_2$, the cumulative impact from the four treated headwaters was equal-to-or-less than the cumulative impact of the four treatments located along the South Fork.

Because there was no legitimate calibration period free of disturbance for the South Fork due to the effects of the 2001 harvest units, a comparison between calibration and post-treatment periods to discern treatment effects was not justified and no conclusion was drawn by doing so. A comparison of the North Fork calibration data with the South Fork post-treatment data revealed a much more reliable assessment of the treatment effects. This is because data from the headwaters of the South Fork that were not affected by the 2001 harvest disturbance indicate that NO_3 + NO_2 concentrations in the South Fork would have been similar to concentrations in the North Fork prior to the 2001 disturbance. A comparison of calibration data from the North Fork with post-treatment data in the South Fork shows that there was probably a six-fold increase in NO_3 + NO_2 concentrations in the South Fork as a result of timber harvest and fertilization. This increase observed at Hinkle Creek was an order of magnitude less than the increases observed in the Needle Branch of the Alsea watershed after clearcut (Brown et al., 1973), but greater than the increases observed in watershed F7 at Mica Creek (Gravelle et al., 2009).

The responses of the other nutrients suggest that geologic and atmospheric processes have a more influential role over the response of these nutrients than anthropogenic disturbances. A comparison of all watersheds shows that stream water chemistry and nutrient concentrations can vary considerably within a 3rd-order stream basin, even among the smaller, contiguous watersheds. For example, the phosphorus concentrations observed in Fenton were nearly twice those observed in neighboring Clay, and the calcium concentrations observed in DeMersseman were nearly twice those observed in Myers. The apparent decline in concentrations for all the nutrients aside from nitrogen and phosphorus is an anomaly. Other studies in the Pacific Northwest that measured these nutrient responses to timber harvest have shown either no increase or a small increase in stream water concentrations after disturbance. One difference between those studies and the Hinkle Creek study, however, was the addition of the fertilization treatment at Hinkle Creek. Although no clear link was established, the pattern of fluctuation for several of the nutrients suggests that fertilization may have caused a temporary increase in the rate of in-stream nutrient uptake.

At the headwater and basin scales, topography appeared to be a stronger predictor of the magnitude of the response than was the size or proximity of the clearcut (magnitude of disturbance). The downstream impact from a single, steep headwater clearcut was 30 percent greater than the combined downstream impact from four moderately sloped headwater basins and only 13 percent less than the combined local impact of four clearcuts located directly along the South Fork. This finding indicates that downstream impacts can be mitigated the most by minimizing the number of harvest units on steep slopes that are completed within five years of each other.

Fertilization garnered the greatest response in stream NO_3 + NO_2 export compared to timber harvest, even if it was ephemeral. Some streams showed an increase of 400 – 500% in concentrations of NO_3 + NO_2 immediately after fertilization. There was no way to measure downstream cumulative impacts of fertilization using this study design because fertilization was basin-wide. To do this would have required a sampling location downstream of the confluence of the North and South Fork to allow for some amount of in-stream uptake and dilution. Sustained elevations of DON and ammonia were observed in several streams after fertilization.

The Lower Clay sampling location provided information on the ability of streams without riparian zones to mitigate the effects of upstream disturbances that cause increases in stream water NO_3 + NO_2 loss. In the sixth year after the harvest along lower Clay Creek, concentrations of NO_3 + NO_2 in the Lower Clay site appeared to return to [assumed] pre-harvest conditions. The effects of attenuation through this clearcut from concentrations of NO_3 + NO_2 produced by the clearcut in the headwater basin were significant, and might have been helped by the lack of canopy that results along non-fish-bearing streams. This finding may corroborate a study done at Hubbard Brook by Bernhardt (2003) that showed an increase in in-stream nitrogen-processing efficiency after a canopy-removing disturbance (ice storm) led to increased light availability and large inputs of woody debris to the stream, similar to the outcome of harvesting without riparian buffer strips. They determined that without the increased in-stream processing ability, stream export of NO_3 + NO_2 would have been 80 - 140% higher than observed, pointing to "an intriguing negative feedback mechanism whereby the same disturbance that causes watershed NO_3 + NO_2 loss may simultaneously lead to increased in-stream retention and transformation."

With so few data points allowed to portray the behavior of the response, it was important to scrutinize each data point to understand how the results might have been influenced by other variables. One conclusion taken from this study is that hydrologic conditions that exist antecedent to and at the time of the sampling can exert significant influence on the perceived results of the treatment effect. The discovery of the hydrologic influence that affected the January 2005 and October 2009 concentrations highlights the need for further research into the discharge vs. concentration relationship for larger storms and for periods of prolonged precipitation to more accurately assess NO_3 + NO_2 losses under a variety of discharge regimes.

Although nutrient concentration levels observed in several studies of forested ecosystems in the Pacific Northwest have never reported levels of any nutrients that approached drinking water standards or levels toxic to fish (Binkley and Brown, 1993), even after fertilization (Hetherington, 1985) none of these studies have been done on private industrial land. The findings from this study help show that contemporary forest practices pose minimal risk to local and downstream water quality. The desire to better understand nutrient dynamics, their reaction to disturbance and their resiliency in the aftermath continues to drive research on the subject, and Hinkle Creek provided an excellent opportunity to add to the existing base of knowledge.

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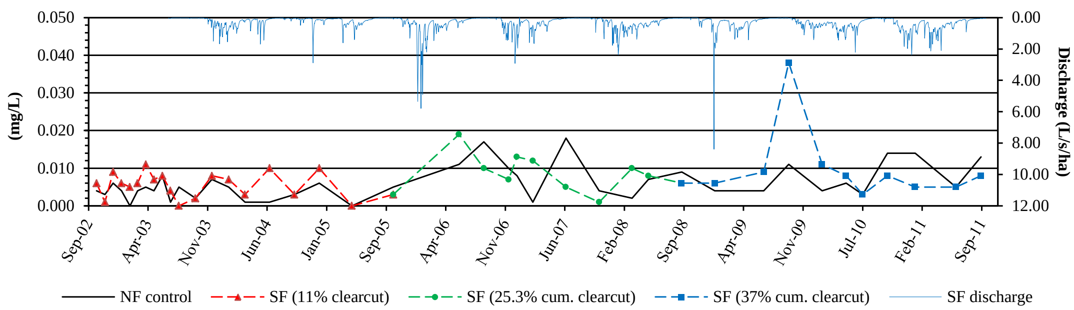
Appendices

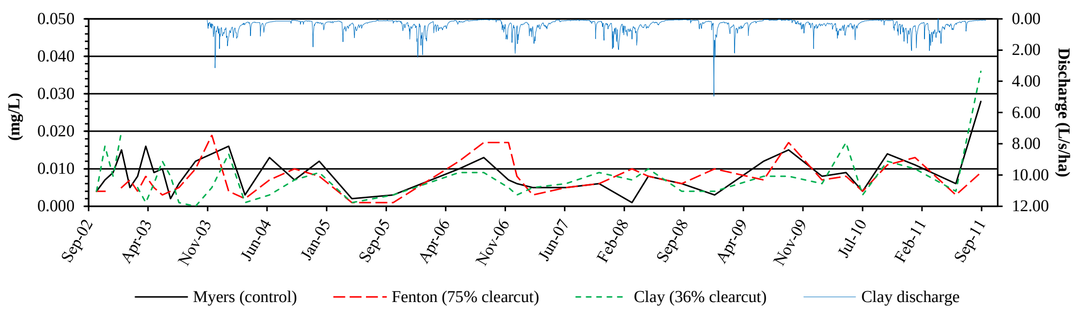
Appendix A Graphs of nutrient response

Ammonia

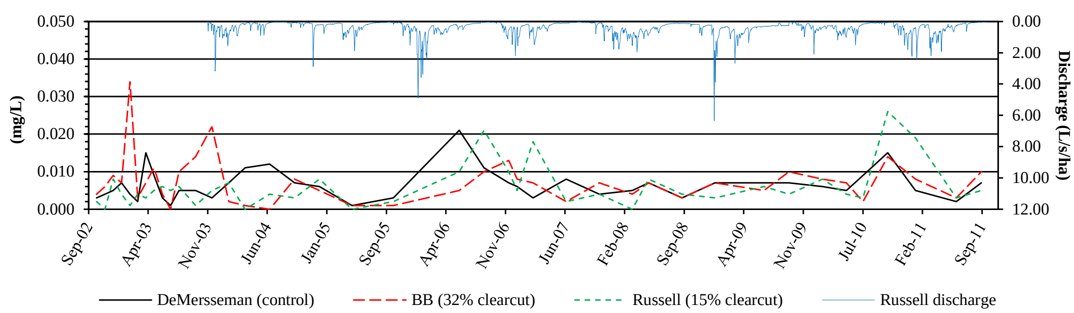
North Fork/South Fork ammonia response



Myer/Fenton/Clay ammonia response

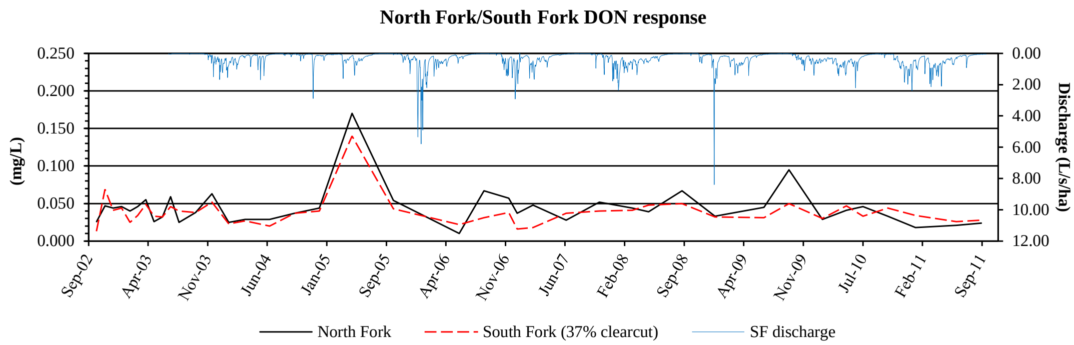


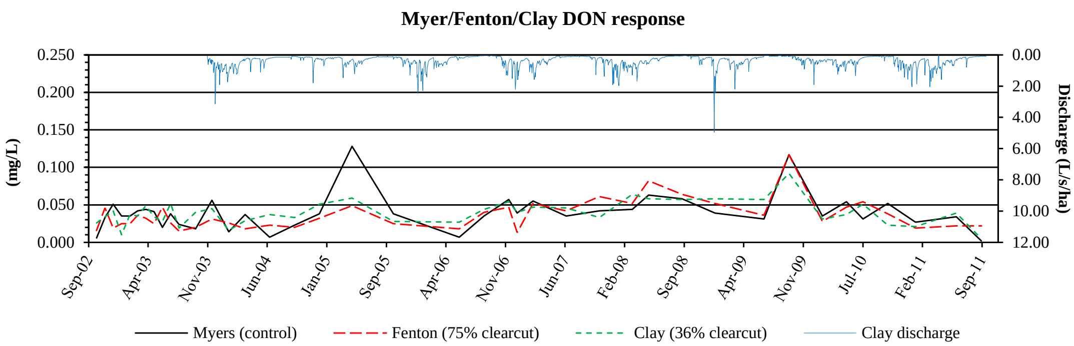
DeMersseman/BB/Russell ammonia response

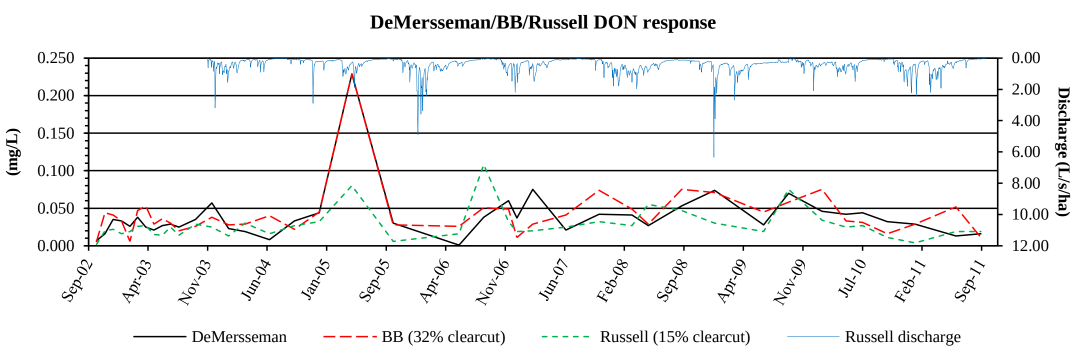


Appendix A Graphs of nutrient response (continued)

Dissolved organic nitrogen (DON)

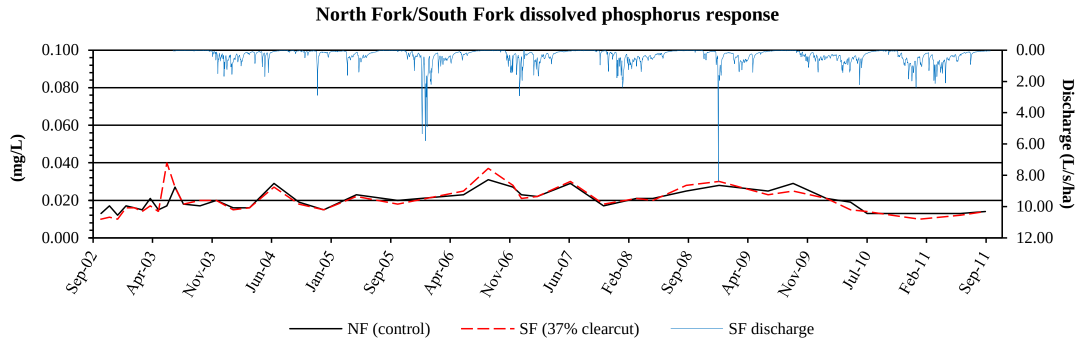


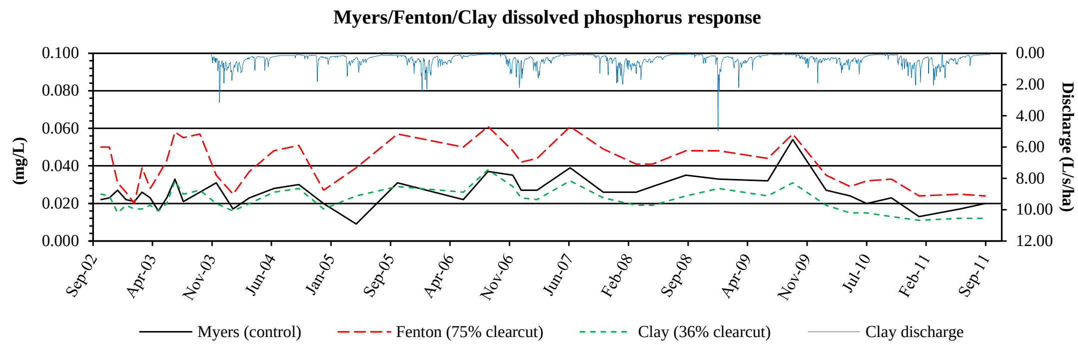


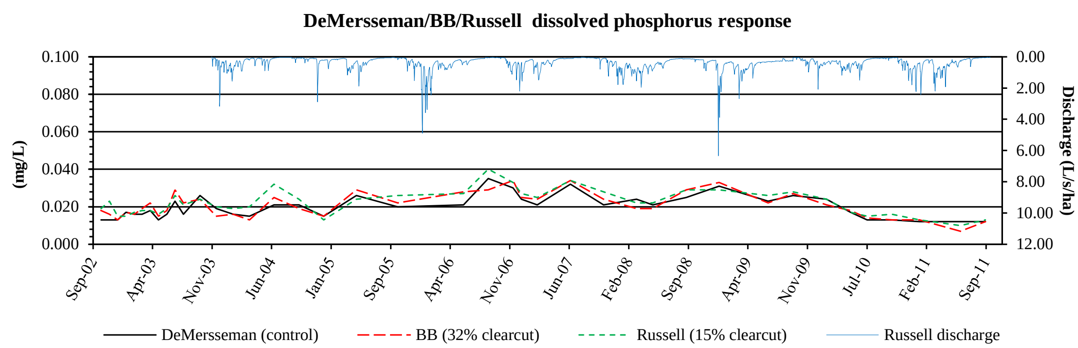


Appendix A Graphs of nutrient response (continued)

Dissolved phosphorus (DP)

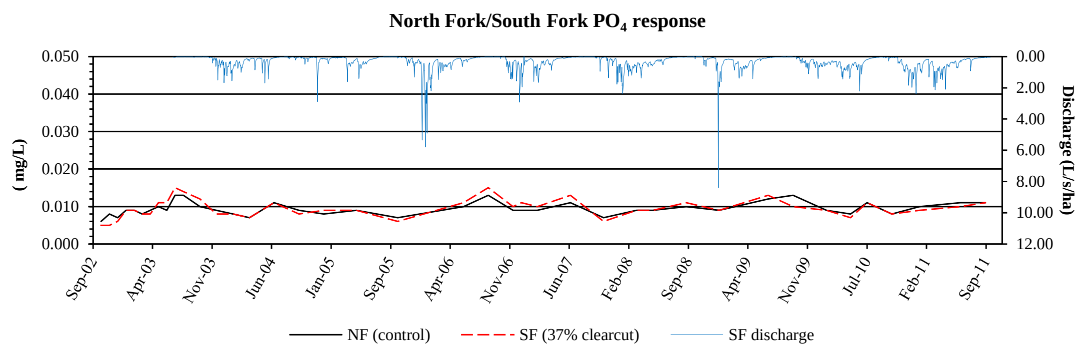


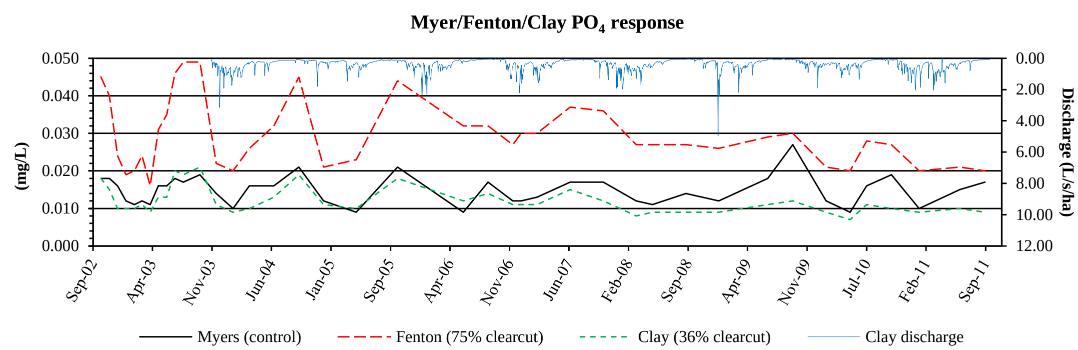


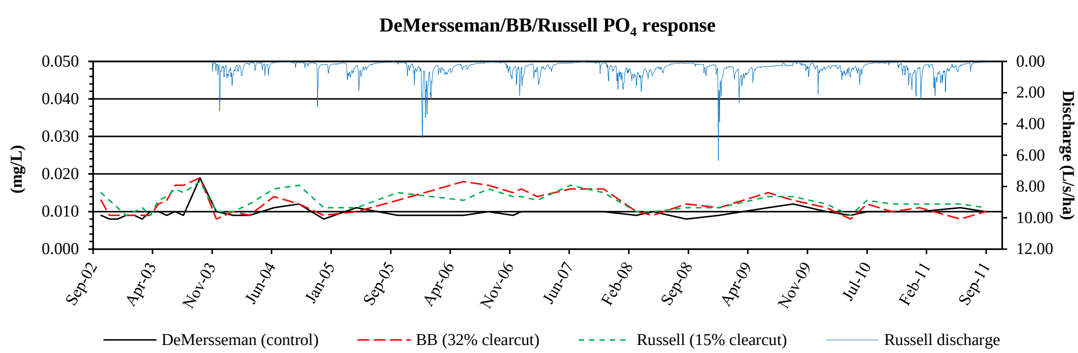


Appendix A Graphs of nutrient response (continued)

Orthophosphate (PO_4)

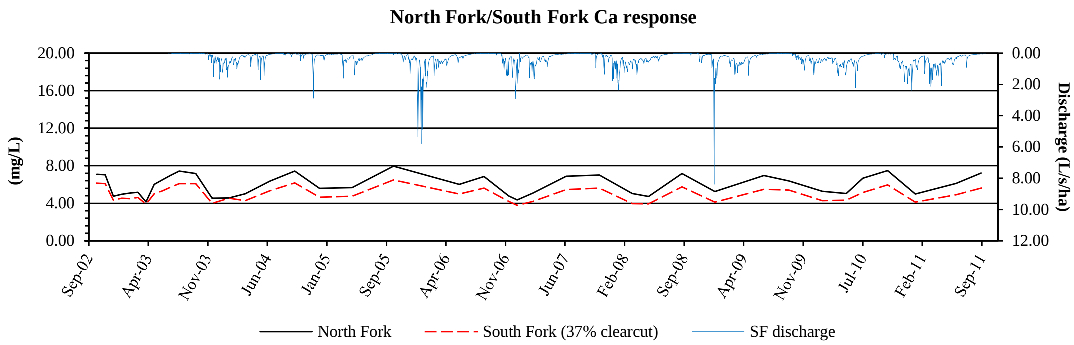


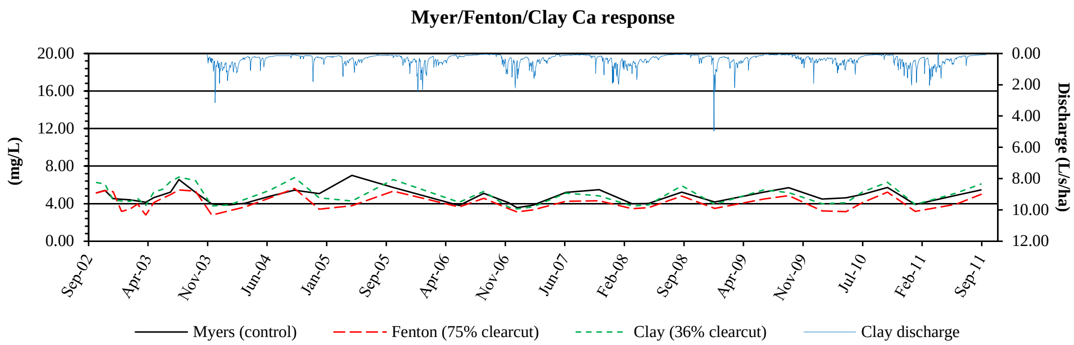


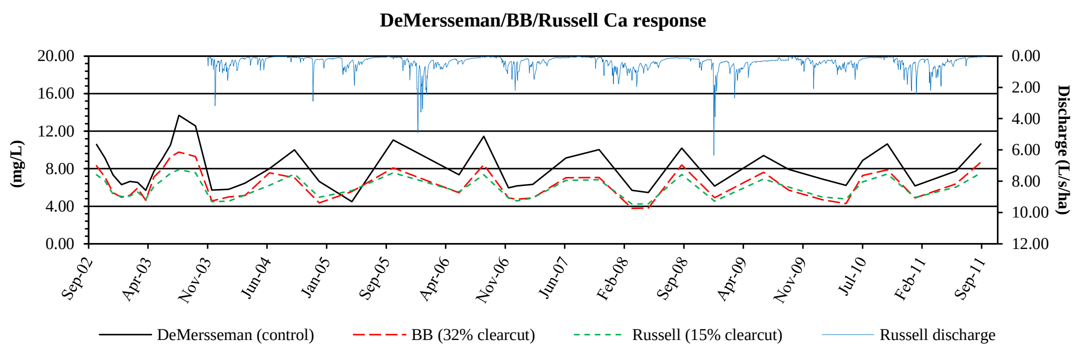


Appendix A Graphs of nutrient response (continued)

Calcium (Ca)

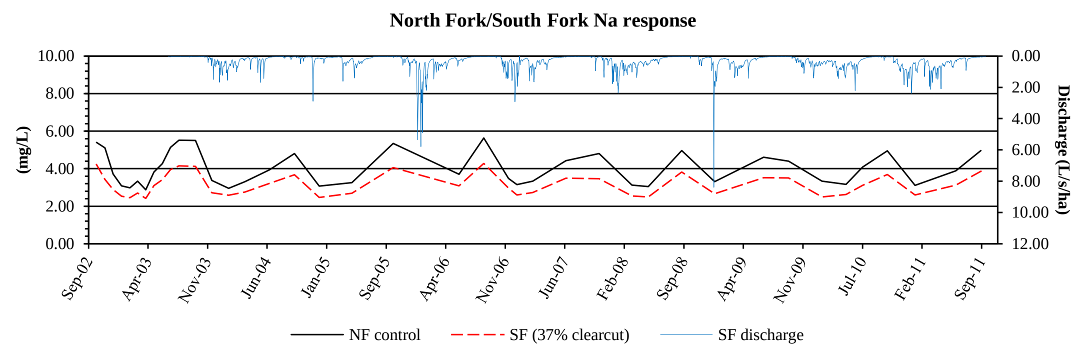


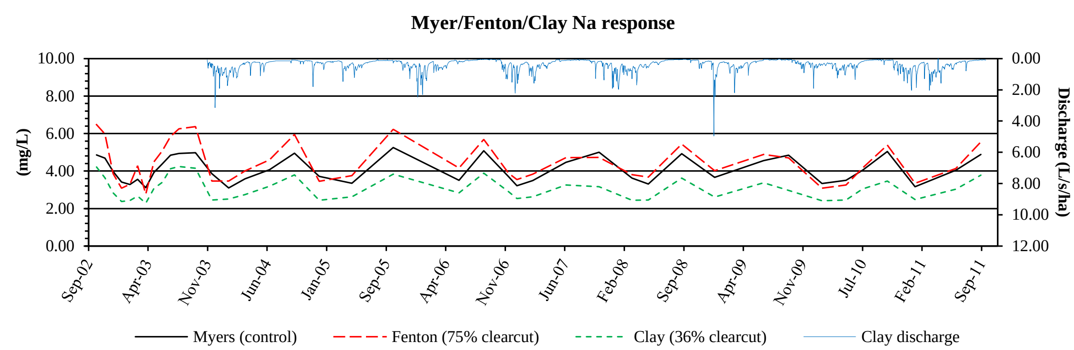


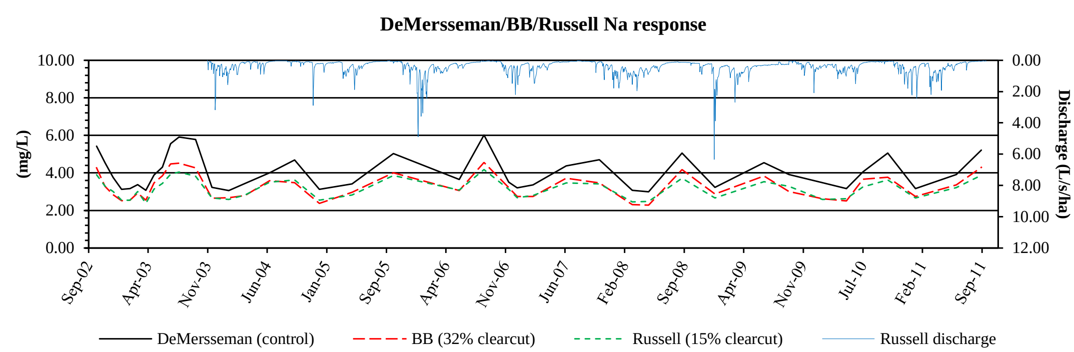


Appendix A Graphs of nutrient response (continued)

Sodium (Na)



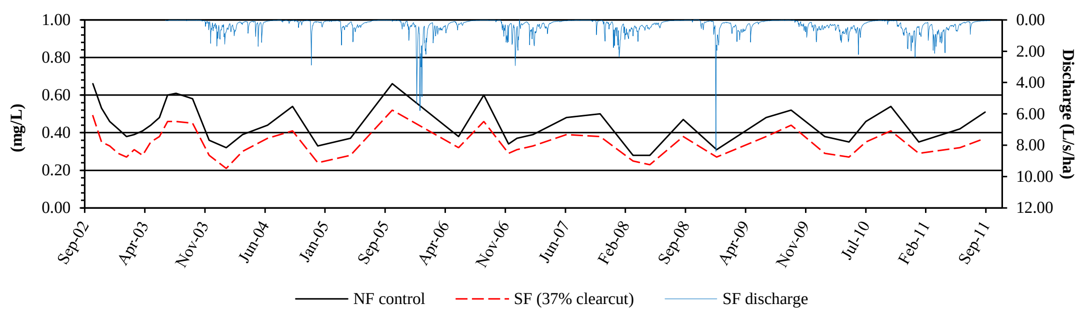




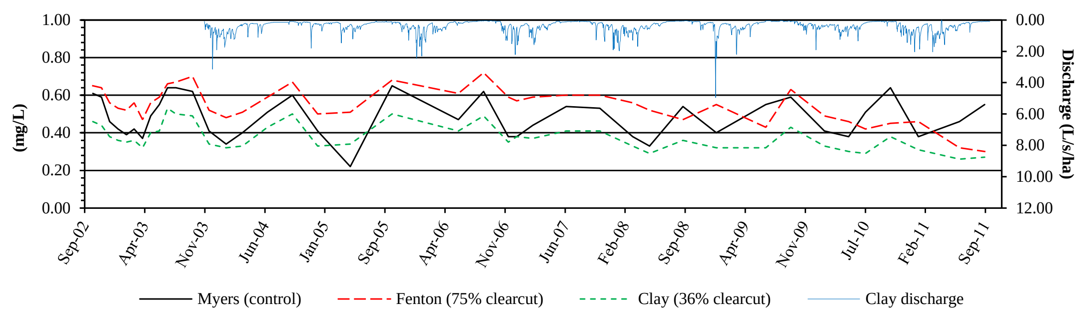
Appendix A Graphs of nutrient response (continued)

Potassium (K)

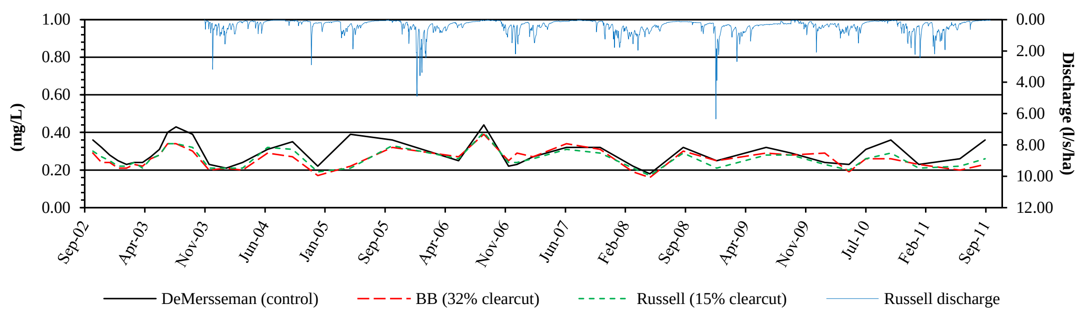
North Fork/South Fork K response



Myer/Fenton/Clay K response

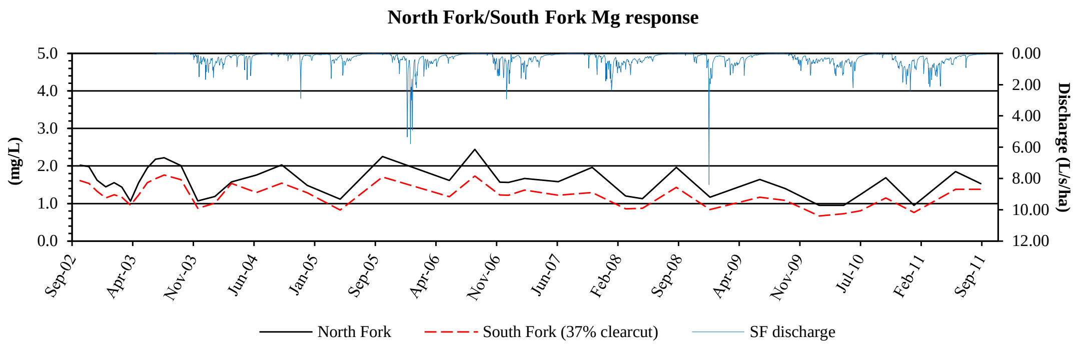


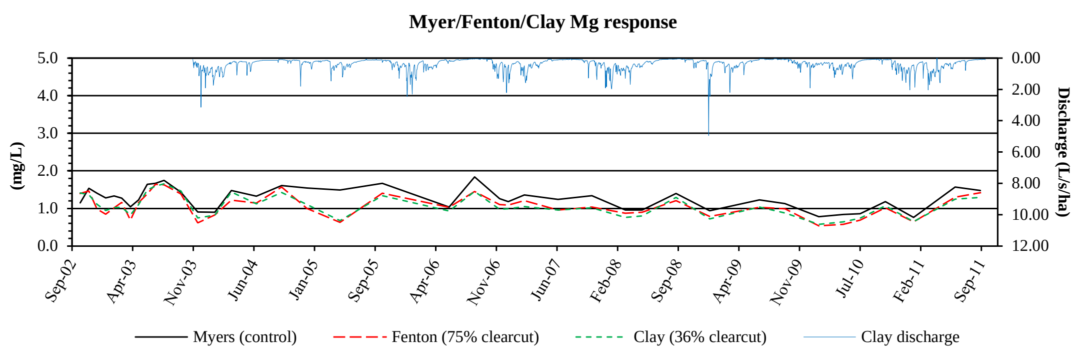
DeMersseman/BB/Russell K response

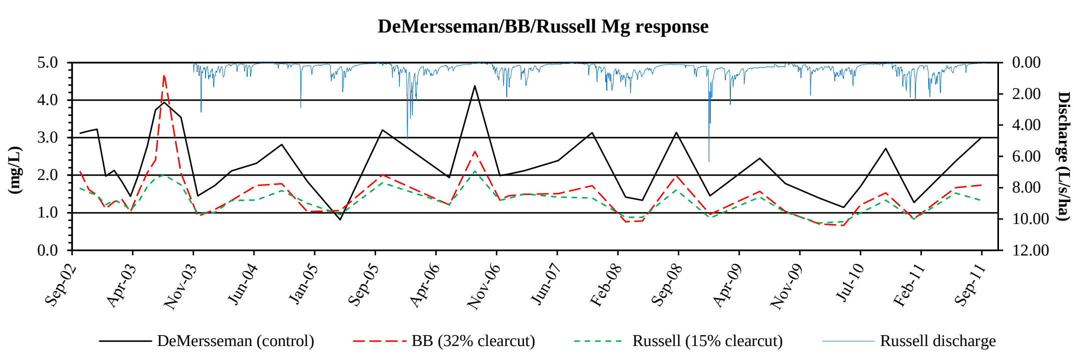


Appendix A Graphs of nutrient response (continued)

Magnesium (Mg)

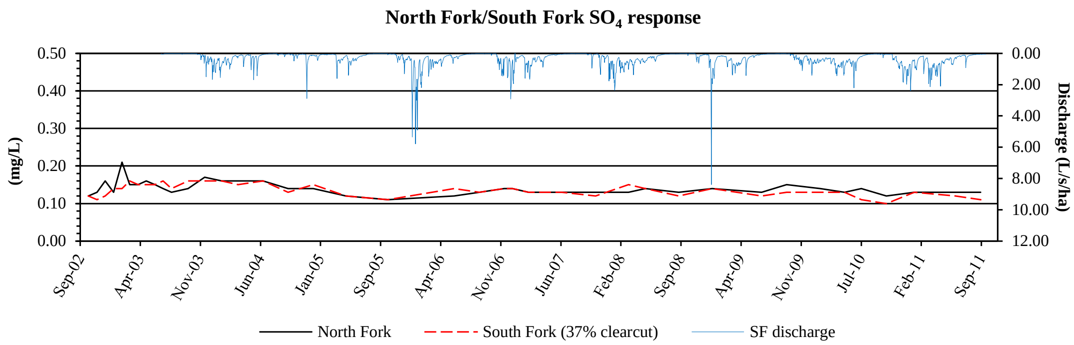


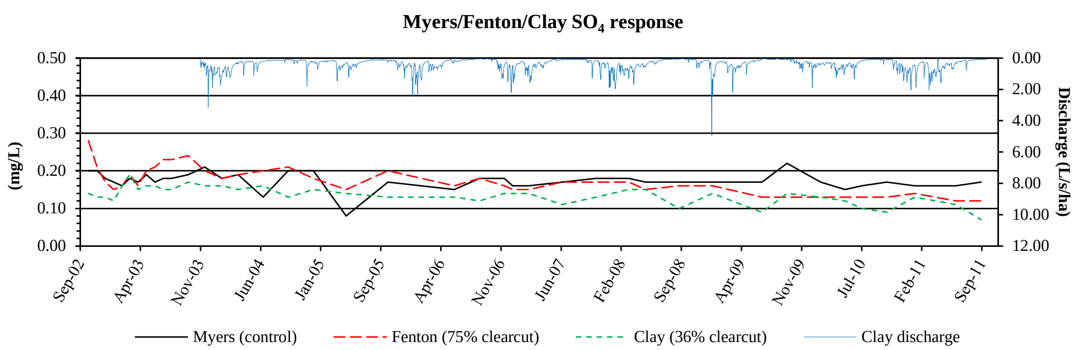


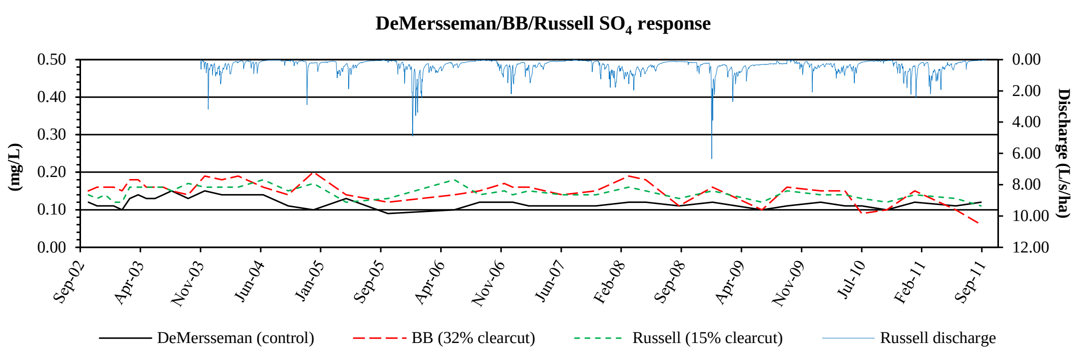


Appendix A Graphs of nutrient response (continued)

Sulfate (SO_4)

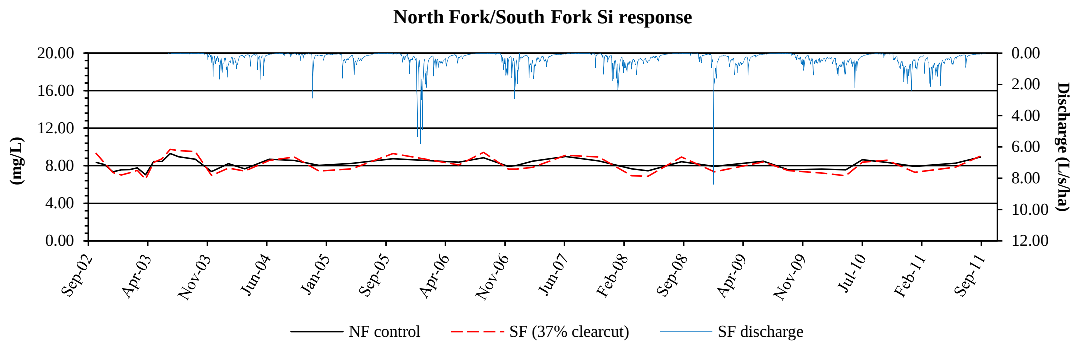


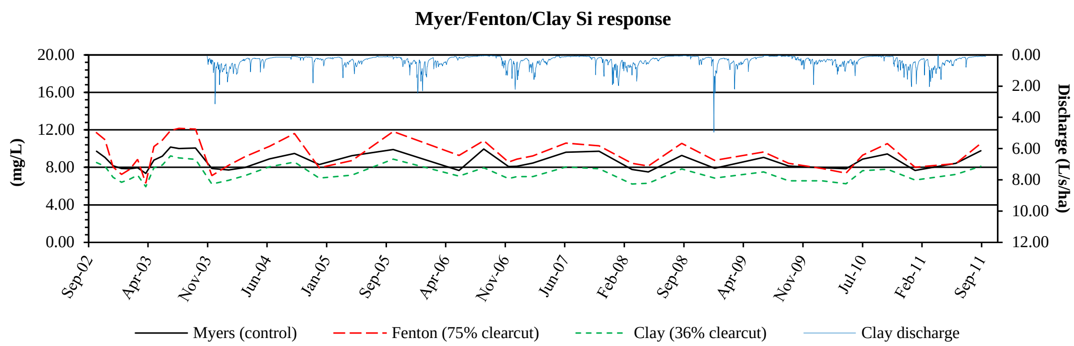


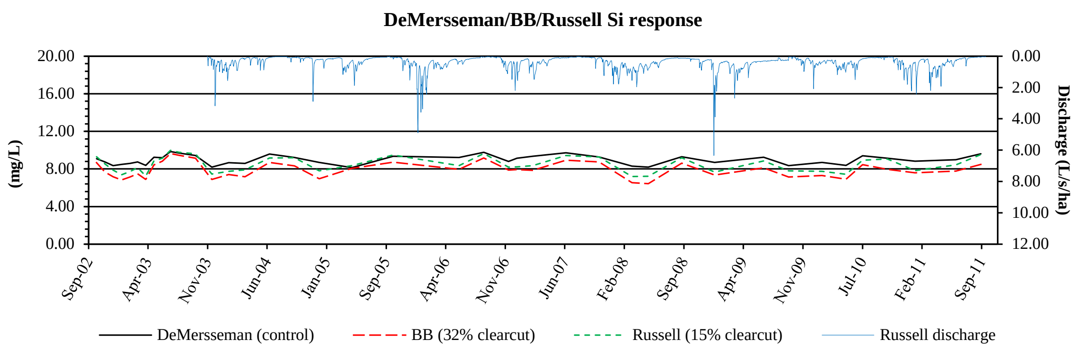


Appendix A Graphs of nutrient response (continued)

Silicon (Si)

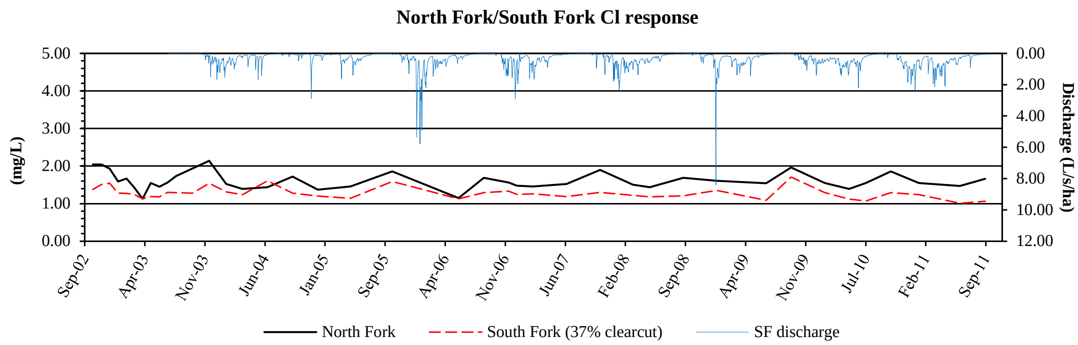


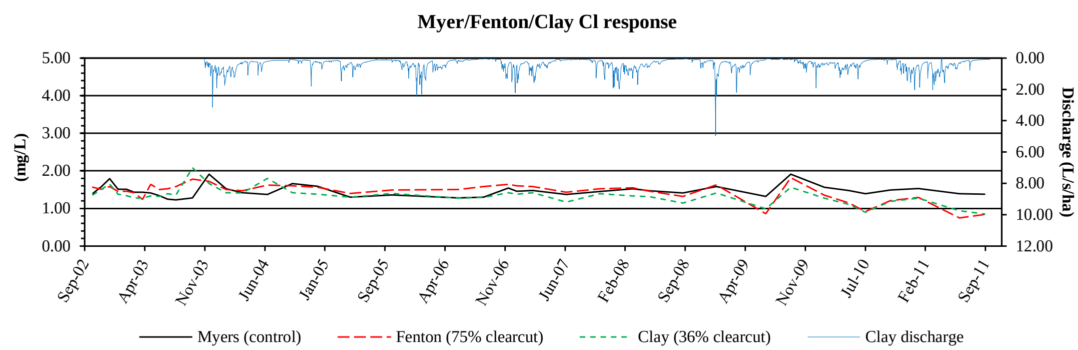


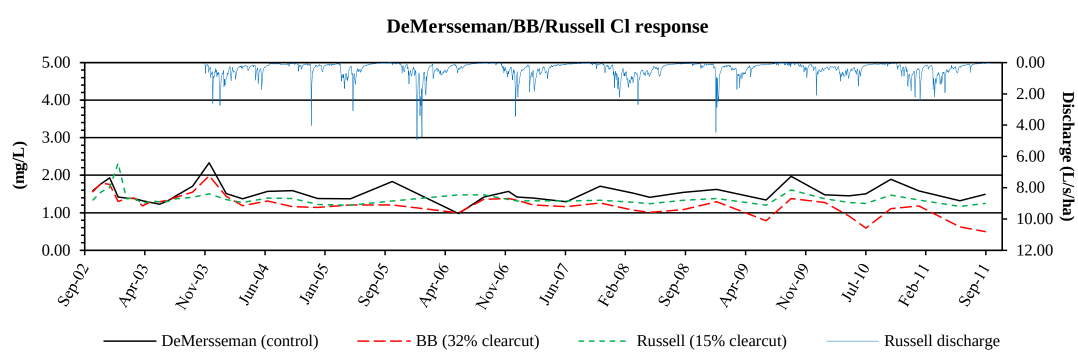


Appendix A Graphs of nutrient response (continued)

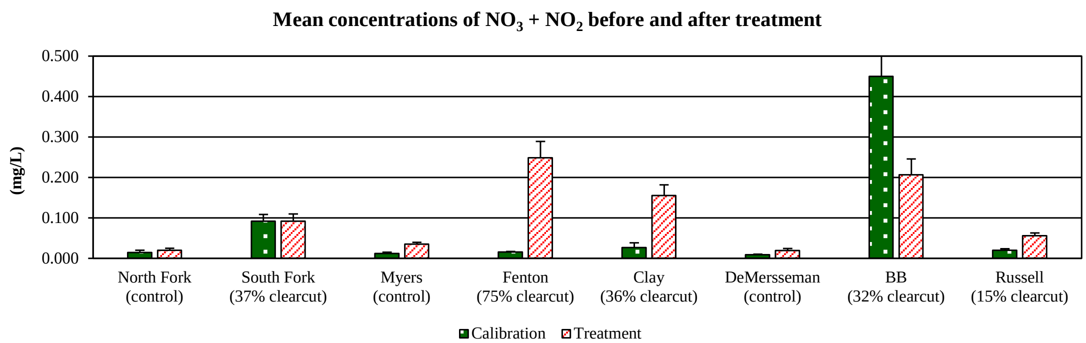
Chloride (Cl)

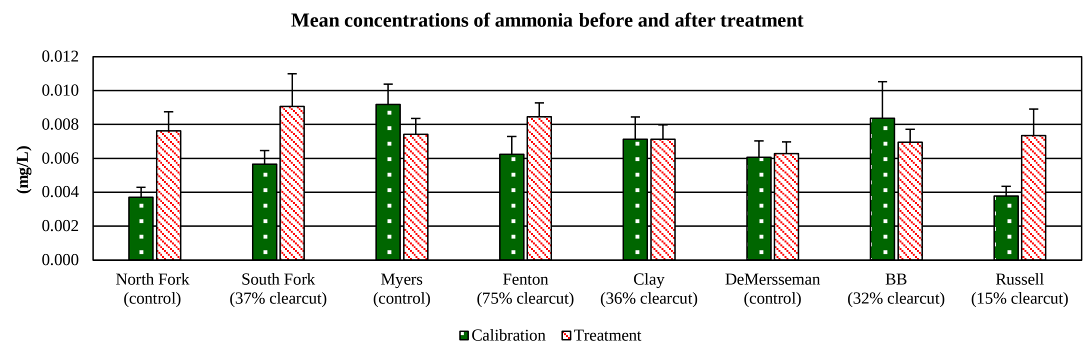


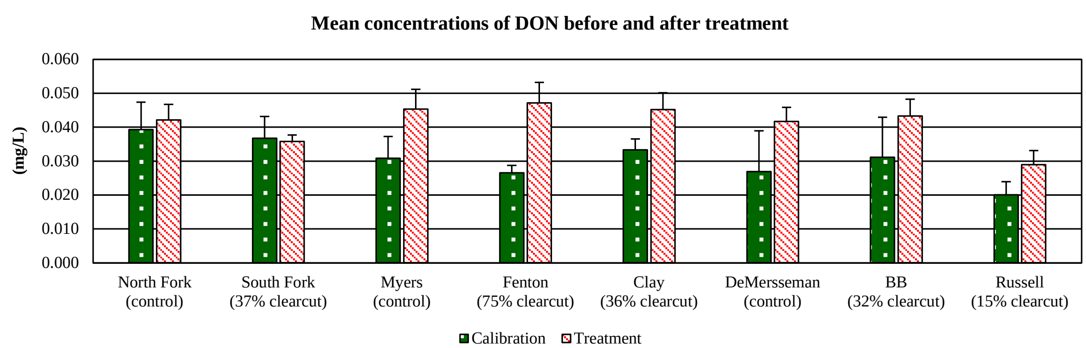




Appendix B Mean concentration of nutrients before and after treatment

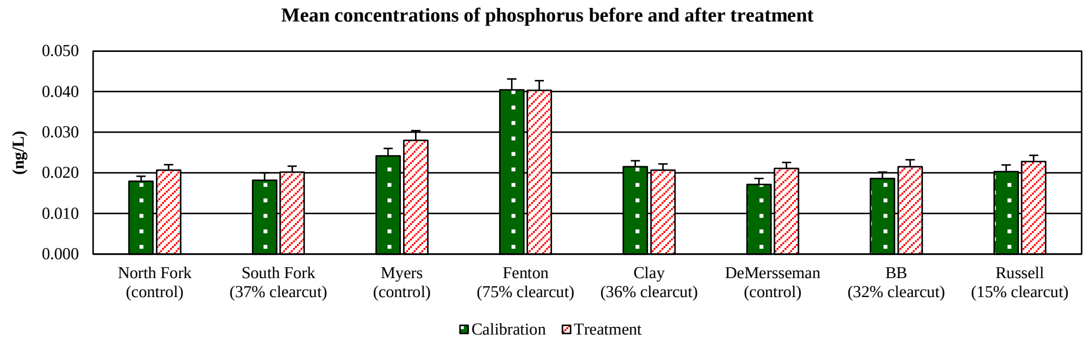


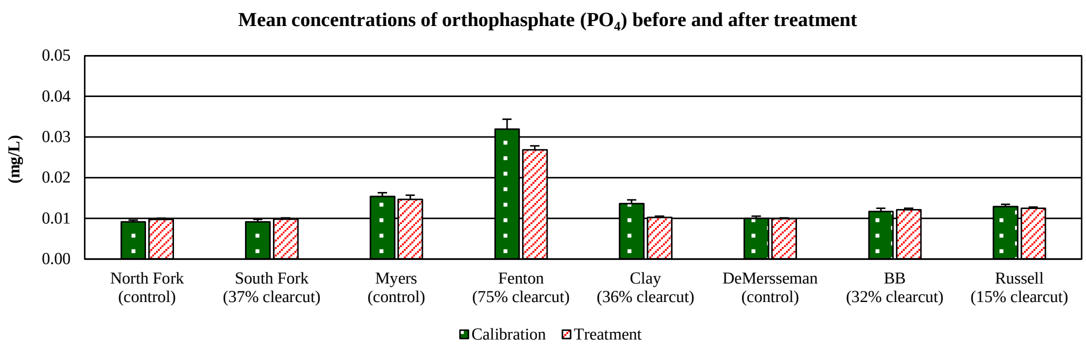


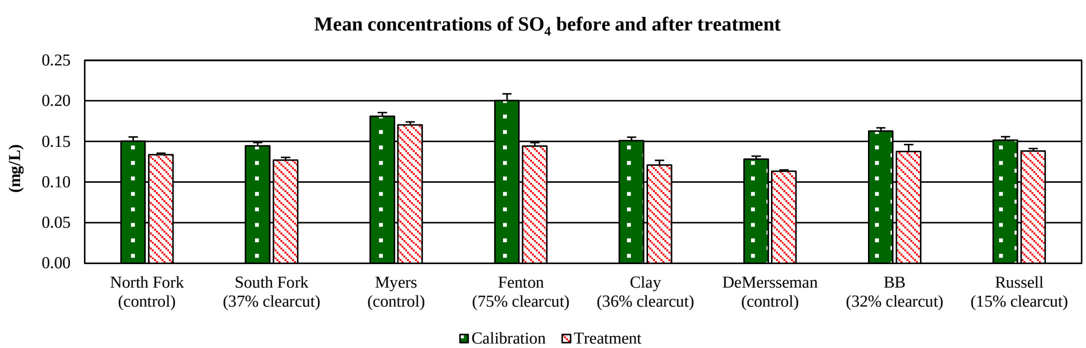


* Error bars denote S.E.

Appendix B Mean concentration of nutrients before and after treatment (cont'd)

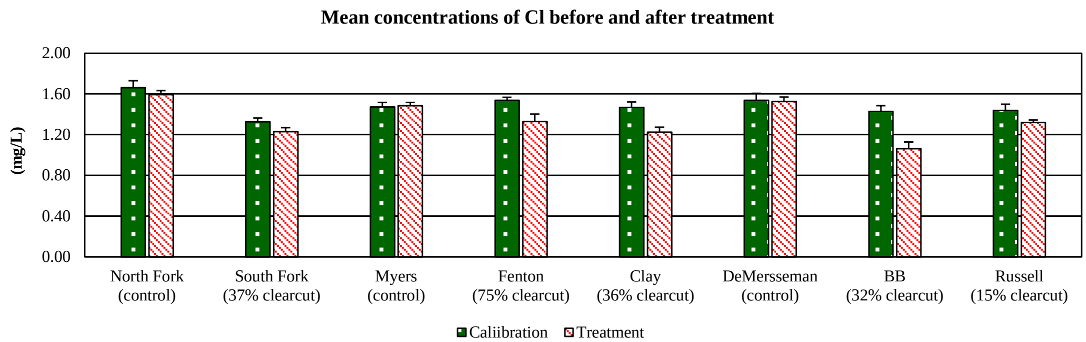


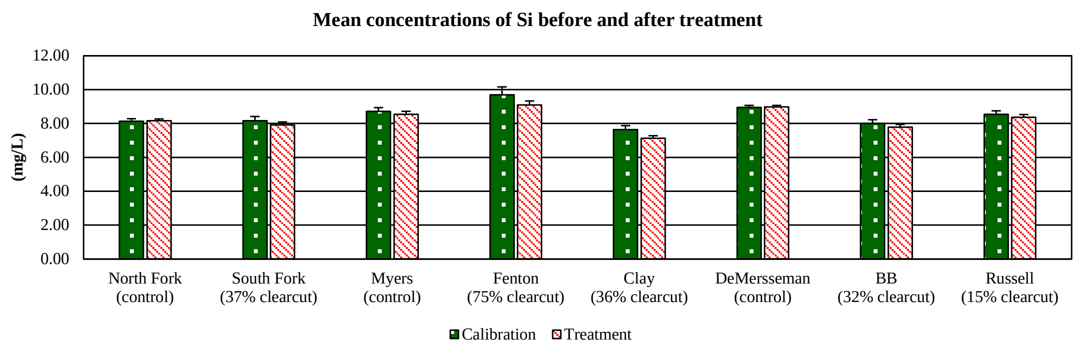


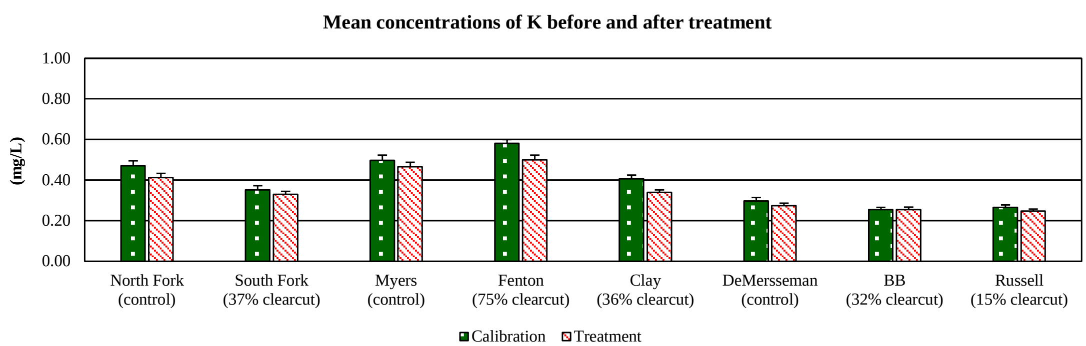


* Error bars denote S.E.

Appendix B Mean concentration of nutrients before and after treatment (cont'd)

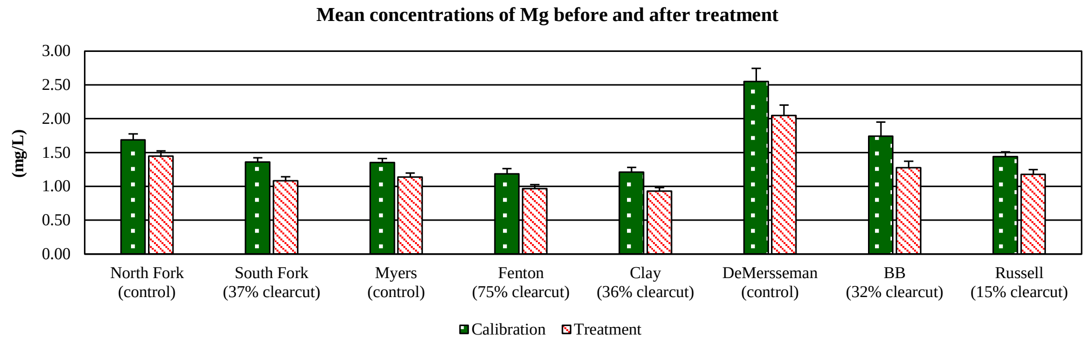


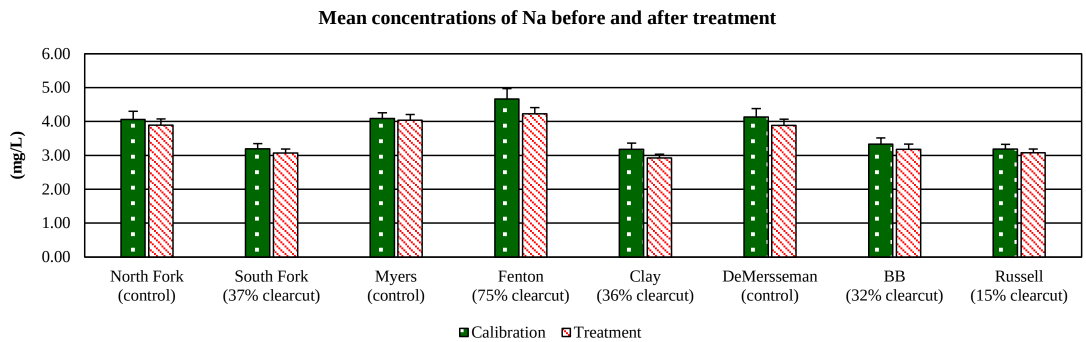


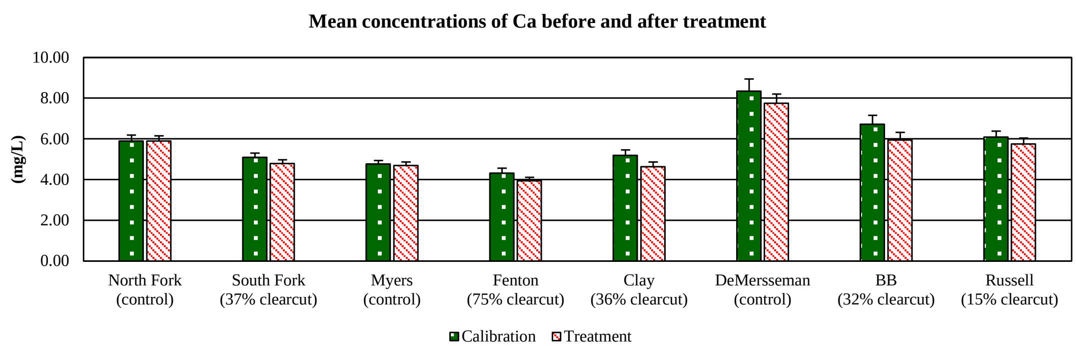


* Error bars denote S.E.

Appendix B   Mean concentration of nutrients before and after treatment (cont'd)







* Error bars denote S.E.

Appendix C   Storm $NO_3 + NO_2$ response

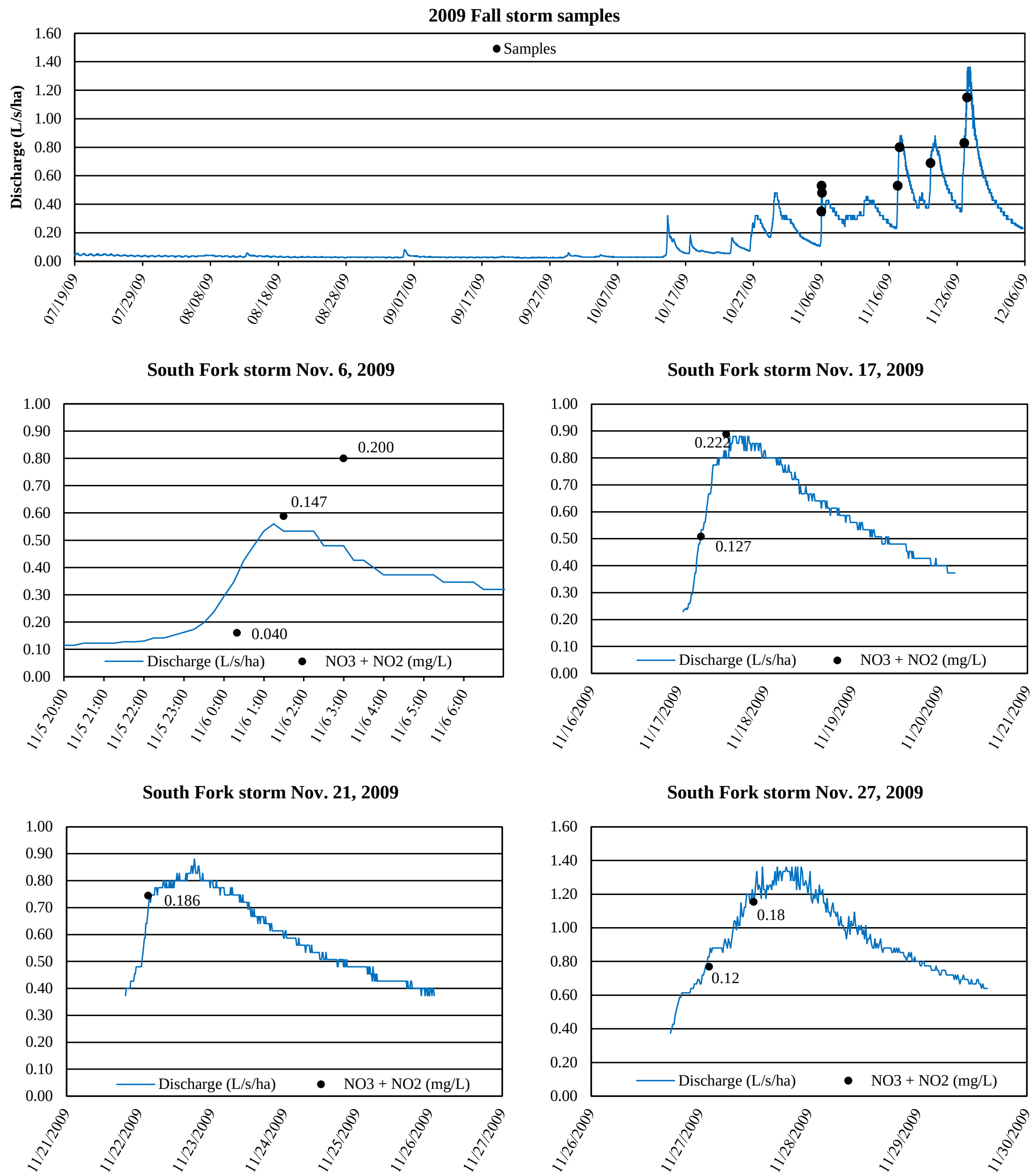
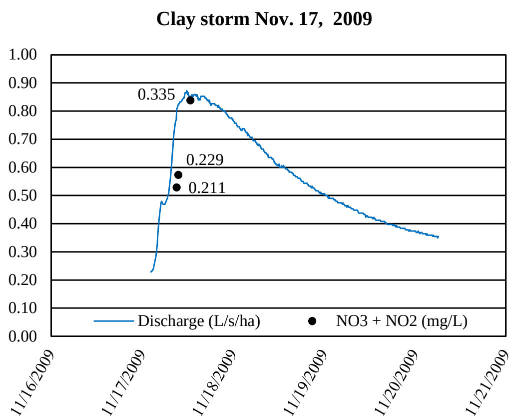
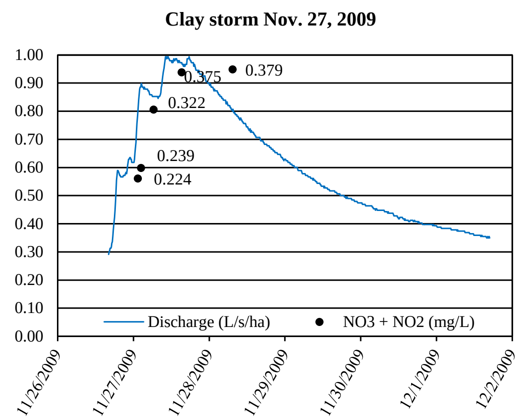
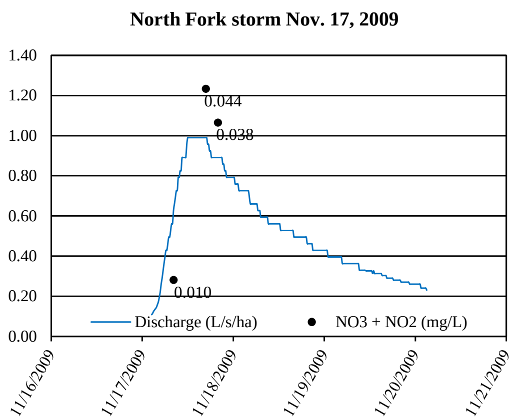


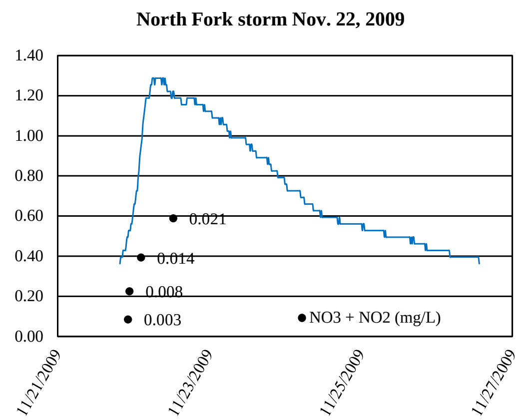
Figure C.1   Storm hydrographs with storm concentration response.

Appendix C   Storm NO_3 + NO_2 response (cont'd)









Appendix D   Hydrographs of the South Fork for each water year

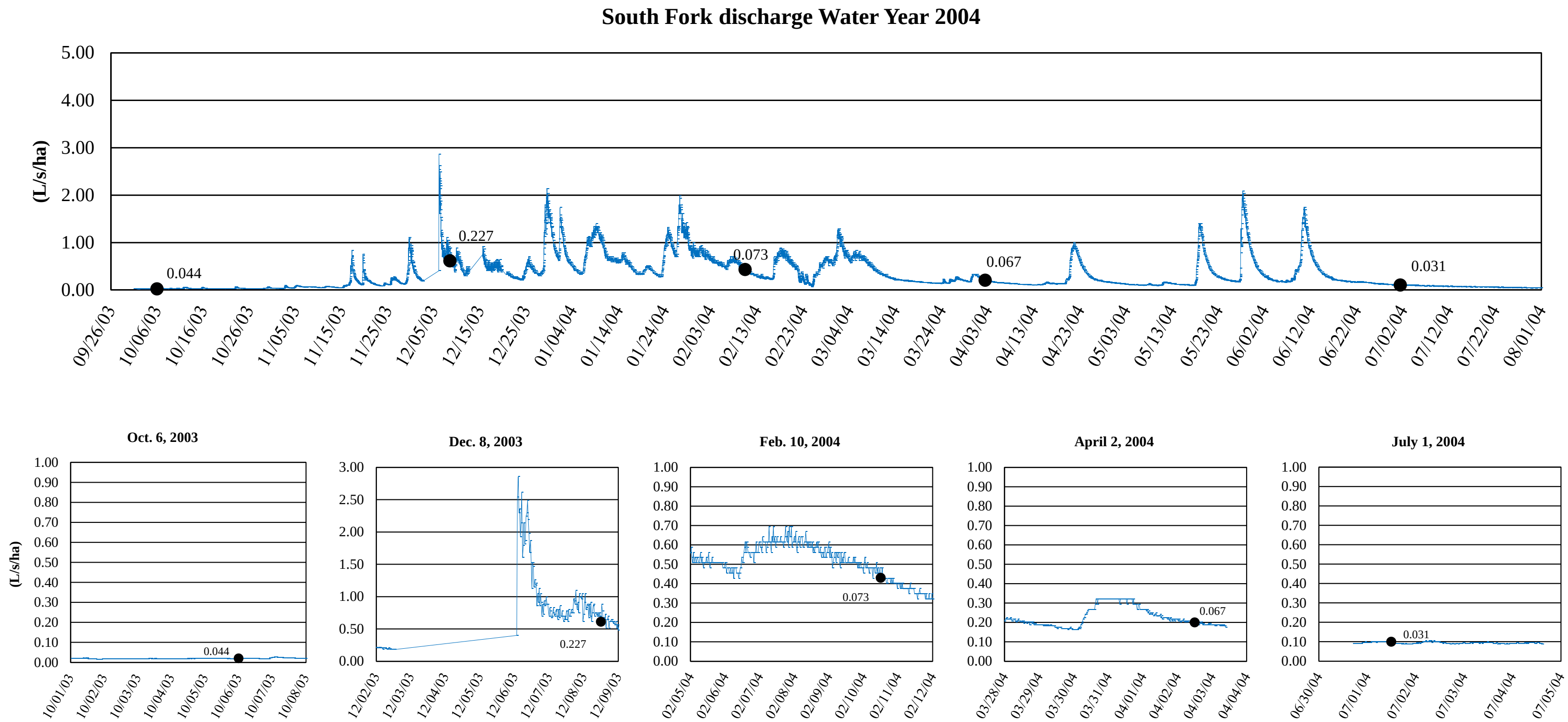


Figure D.1 South Fork hydrograph paired with stream water NO_3 + NO_2 concentration for Water Year 2004. Concentrations of NO_3 + NO_2 are in mg/L.

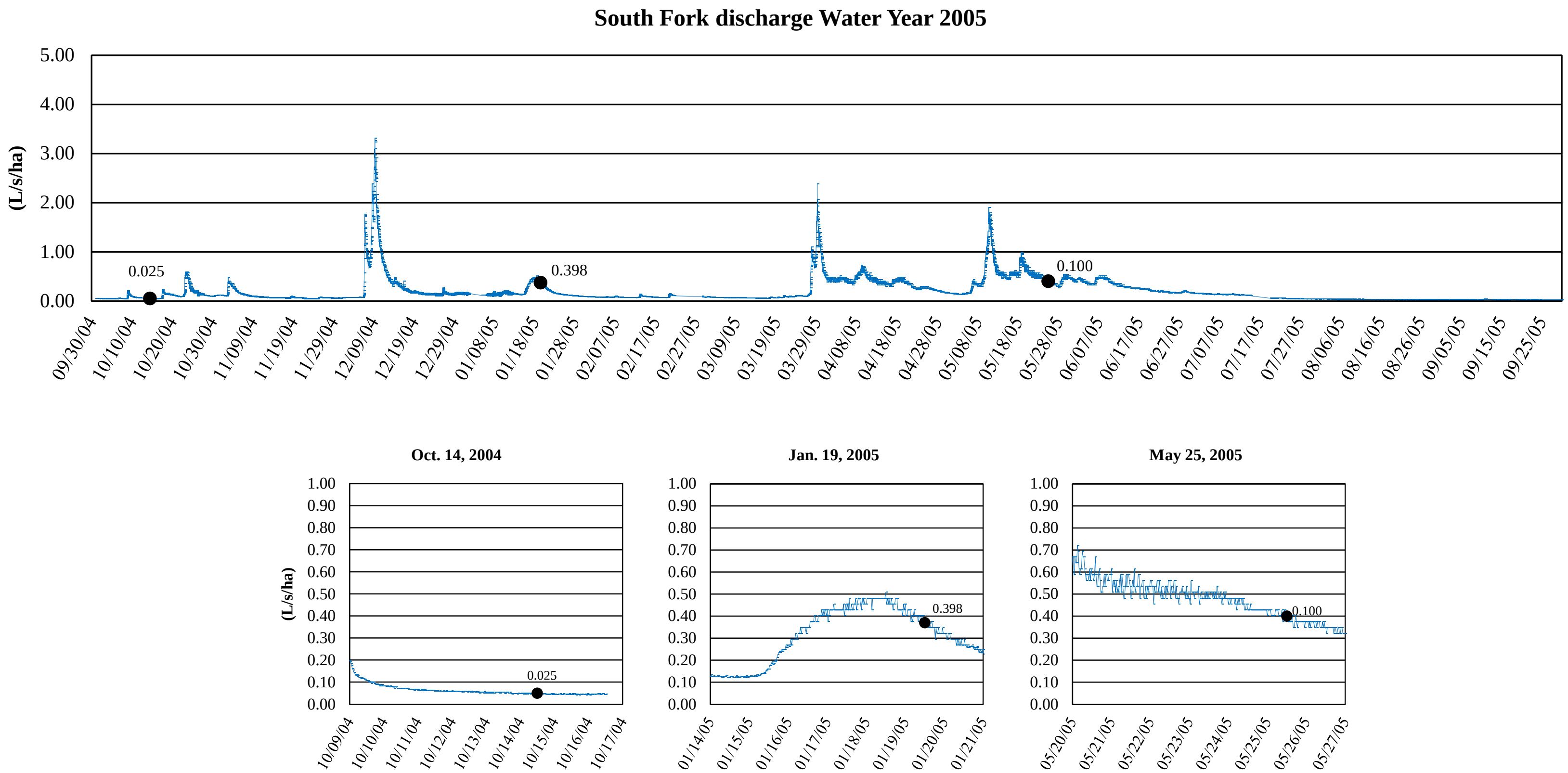


Figure D.2   South Fork hydrograph paired with stream water $NO_3 + NO_2$ concentration for Water Year 2005. Concentrations of $NO_3 + NO_2$ are in mg/L.

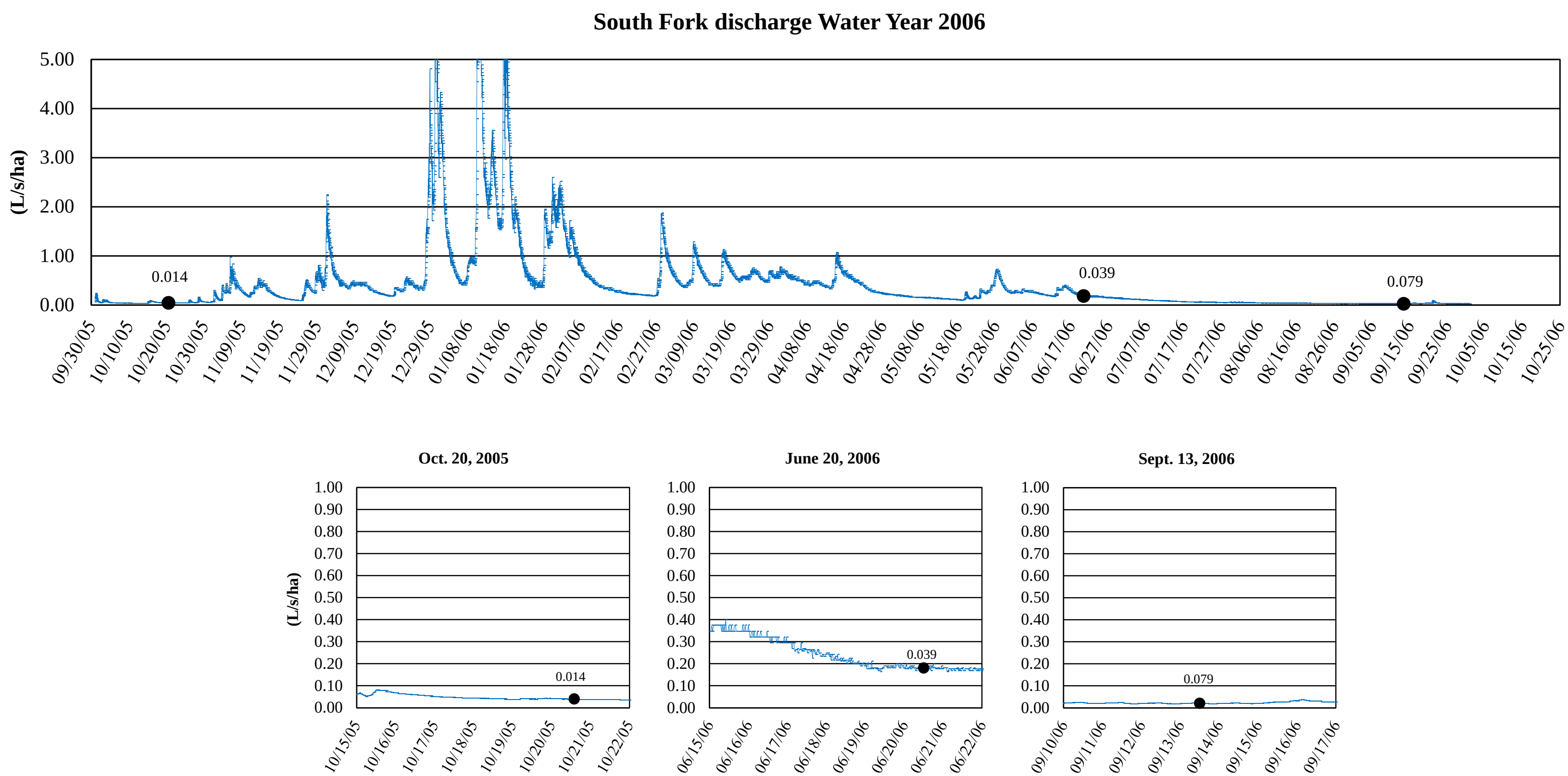


Figure D.3   South Fork hydrograph paired with stream water $NO_3 + NO_2$ concentration for Water Year 2006. Concentrations of $NO_3 + NO_2$ are in mg/L.

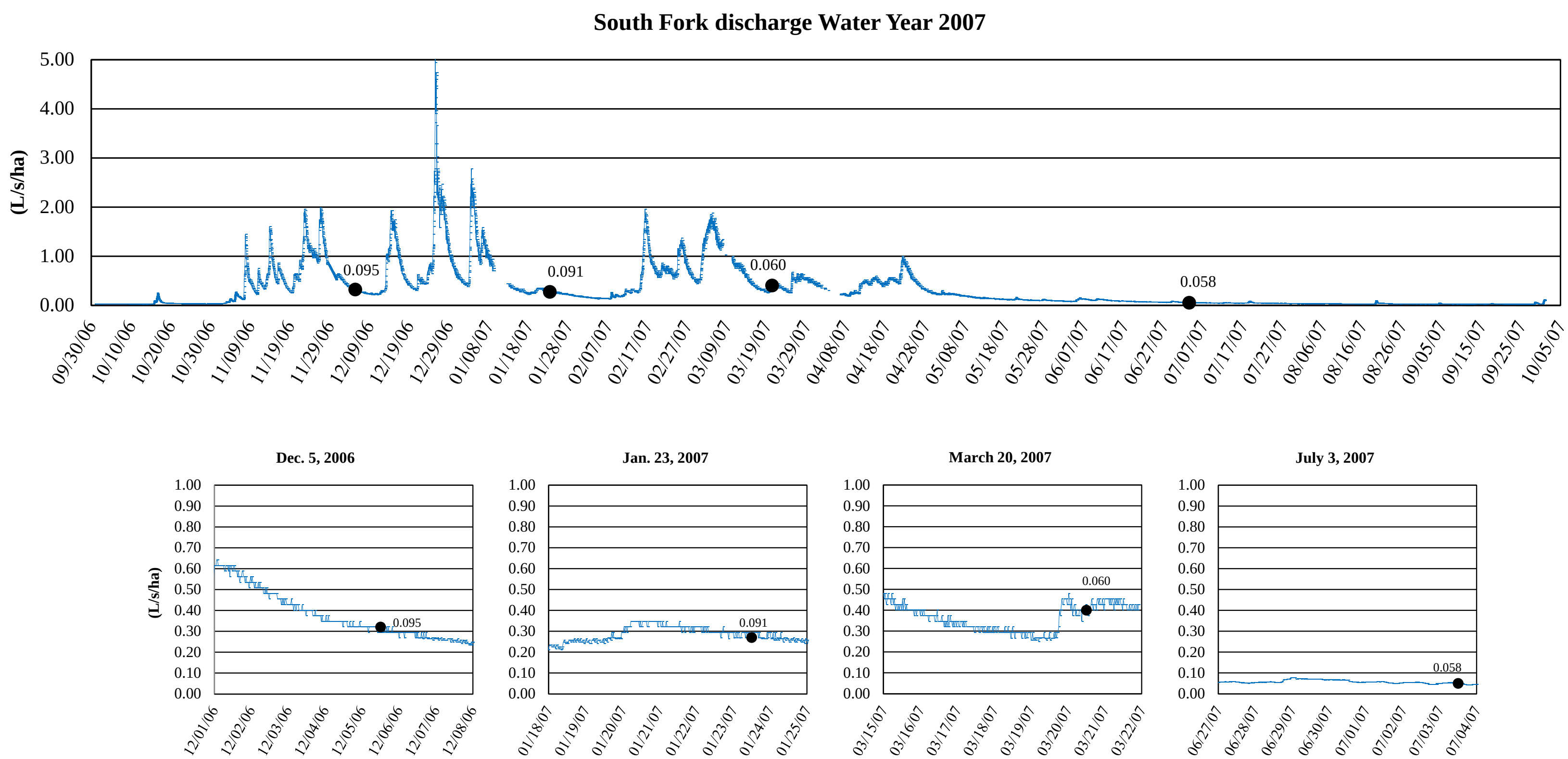


Figure D.1   South Fork hydrograph paired with stream water NO_3 + NO_2 concentration for Water Year 2007. Concentrations of NO_3 + NO_2 are in mg/L.

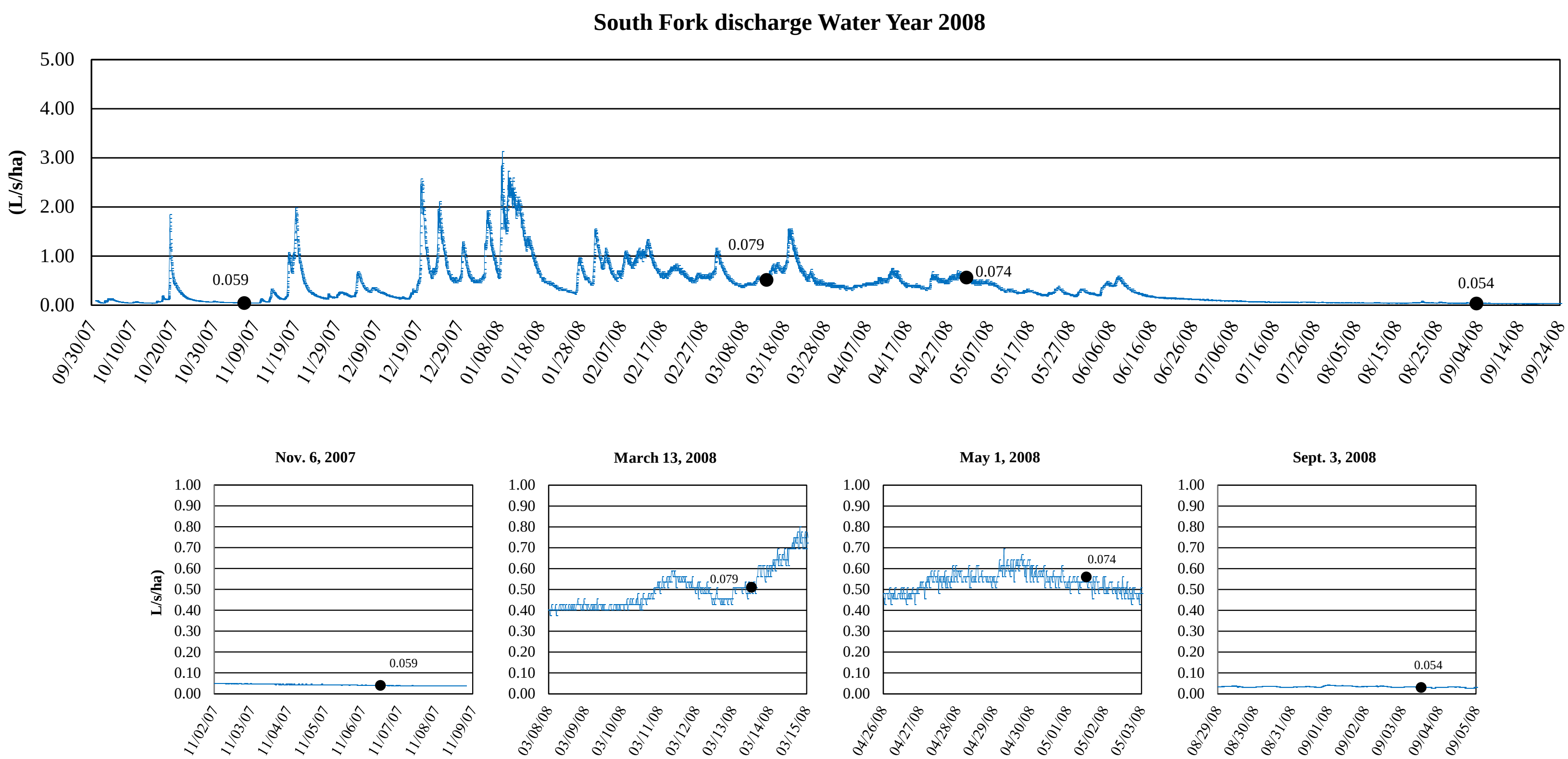


Figure D.1   South Fork hydrograph paired with stream water $NO_3 + NO_2$ concentration for Water Year 2008. Concentrations of $NO_3 + NO_2$ are in mg/L.

Appendix D   Hydrographs of the South Fork for each water year (continued)

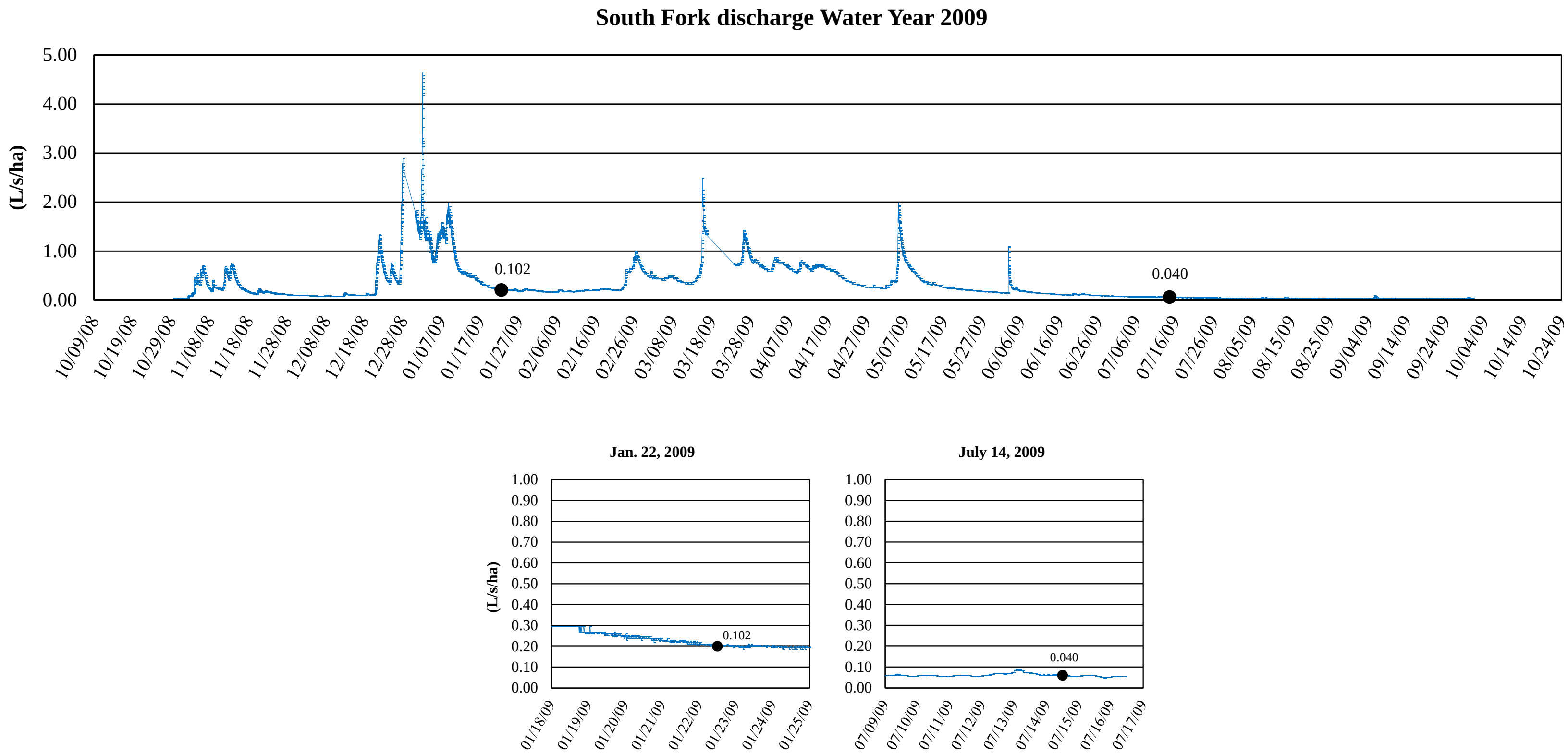


Figure D.1   South Fork hydrograph paired with stream water $NO_3 + NO_2$ concentration for Water Year 2009. Concentrations of $NO_3 + NO_2$ are in mg/L.

Appendix D   Hydrographs of the South Fork for each water year (continued)

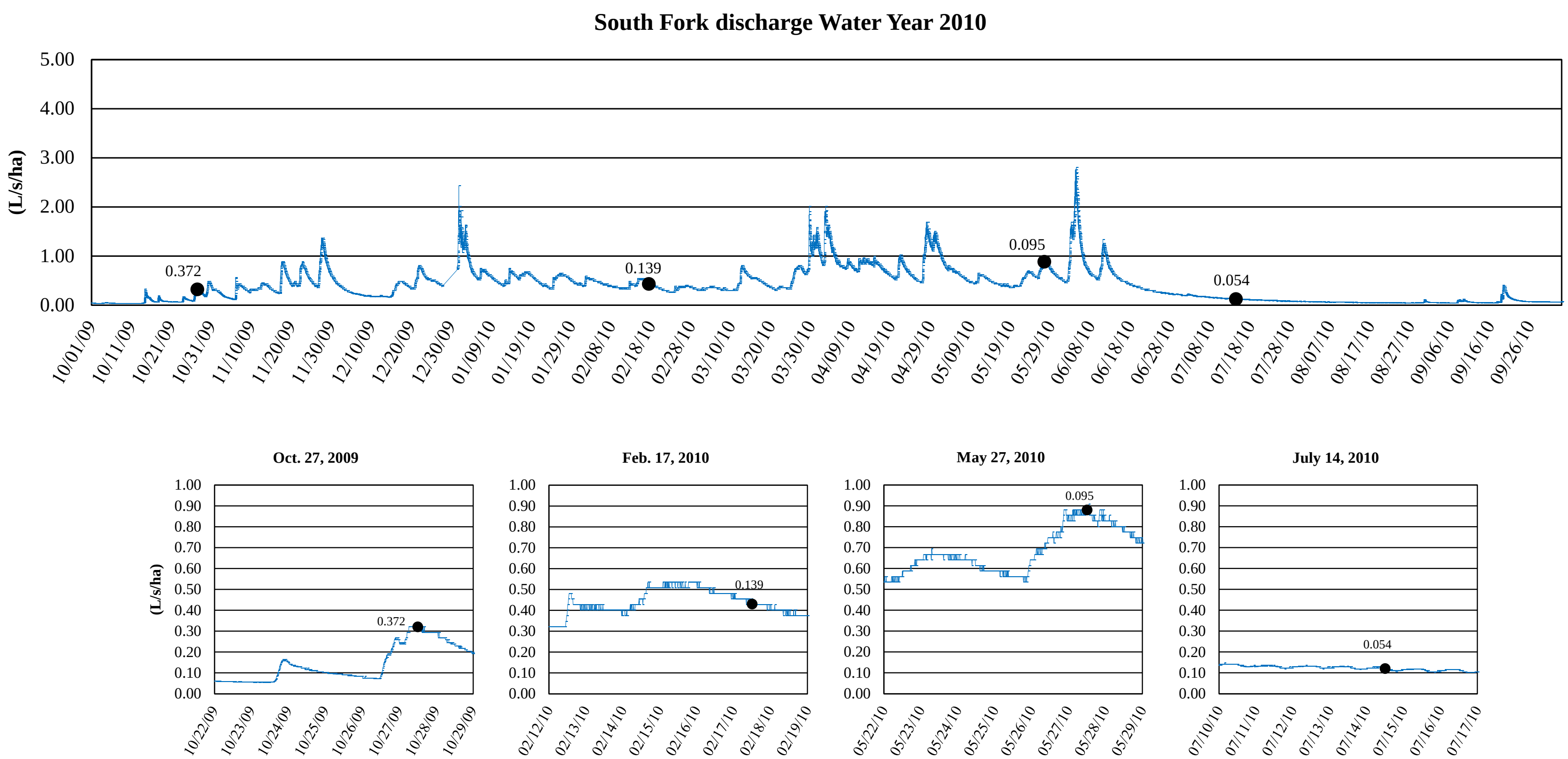


Figure D.1   South Fork hydrograph paired with stream water $NO_3 + NO_2$ concentration for Water Year 2010. Concentrations of $NO_3 + NO_2$ are in mg/L.

Appendix E   Summary tables of select stream chemistry parameters

	Conductance			pH			Alkalinity		
	Pre	Post	Change	Pre	Post	Change	Pre	Post	Change
North Fork	59.1	58.3	-0.8	7.58	7.64	0.1	6.86	6.74	-0.1
	3.0	*2.5*		*0.02*	*0.03*		*0.37*	*0.31*	
South Fork	49.2	46.8	-2.5	7.51	7.55	0.0	5.62	5.34	-0.3
	2.1	*1.7*		*0.02*	*0.03*		*0.28*	*0.23*	
Myers	52.0	50.7	-1.3	7.46	7.53	0.1	5.98	5.80	-0.2
	1.8	*1.8*		*0.02*	*0.02*		*0.23*	*0.25*	
Fenton	51.5	47.8	-3.7	7.49	7.44	0.0	5.86	5.25	-0.6
	3.1	*1.8*		*0.04*	*0.03*		*0.38*	*0.28*	
Clay	49.3	44.4	-4.8	7.51	7.51	0.0	5.60	4.97	-0.6
	2.7	*1.8*		*0.03*	*0.04*		*0.32*	*0.28*	
Demersseman	73.0	70.7	-2.3	7.68	7.71	0.0	8.74	8.41	-0.3
	4.8	*3.9*		*0.02*	*0.03*		*0.60*	*0.52*	
BB	60.0	53.7	-6.2	7.58	7.63	0.1	6.58	6.21	-0.4
	3.6	*3.2*		*0.03*	*0.04*		*0.52*	*0.45*	
Russell	54.9	52.1	-2.8	7.52	7.57	0.0	6.40	6.01	-0.4
	2.5	*2.3*		*0.02*	*0.02*		*0.32*	*0.30*	

Table E.1   Summary table for mean specific conductance, pH, and alkalinity before and after treatment for all streams. Values in italics denote S.E.

Appendix E   Summary tables of select stream chemistry parameters

			Na^+	K^+	Ca^+	Mg^+
North Fork	Pre	Mean	4.07	0.47	5.91	1.69
(control)		*SE*	*0.24*	*0.02*	*0.28*	*0.09*
	Post	Mean	3.90	0.41	5.91	1.44
		SE	*0.18*	*0.02*	*0.26*	*0.08*
South Fork	Pre	Mean	3.20	0.35	5.10	1.36
(37% clearcut)		*SE*	*0.15*	*0.02*	*0.20*	*0.07*
	Post	Mean	3.07	0.33	4.79	1.08
		SE	*0.12*	*0.01*	*0.18*	*0.06*
Myers	Pre	Mean	4.09	0.50	4.77	1.35
(control)		*SE*	*0.17*	*0.03*	*0.17*	*0.06*
	Post	Mean	4.04	0.47	4.70	1.14
		SE	*0.17*	*0.02*	*0.17*	*0.06*
Fenton	Pre	Mean	4.66	0.58	4.32	1.18
(75% clearcut)		*SE*	*0.31*	*0.02*	*0.24*	*0.08*
	Post	Mean	4.23	0.50	3.94	0.97
		SE	*0.19*	*0.02*	*0.17*	*0.06*
Clay	Pre	Mean	3.18	0.41	5.19	1.21
(36% clearcut)		*SE*	*0.17*	*0.02*	*0.27*	*0.07*
	Post	Mean	2.93	0.34	4.64	0.93
		SE	*0.11*	*0.01*	*0.23*	*0.05*
DeMersseman	Pre	Mean	4.14	0.30	8.34	2.55
(control)		*SE*	*0.25*	*0.02*	*0.59*	*0.19*
	Post	Mean	3.89	0.27	7.76	2.04
		SE	*0.18*	*0.01*	*0.45*	*0.16*
BB	Pre	Mean	3.34	0.25	6.72	1.74
(32% clearcut)		*SE*	*0.18*	*0.01*	*0.43*	*0.21*
	Post	Mean	3.18	0.26	5.94	1.28
		SE	*0.15*	*0.01*	*0.39*	*0.10*
Russell	Pre	Mean	3.19	0.27	6.09	1.44
(15% clearcut)		*SE*	*0.13*	*0.01*	*0.29*	*0.07*
	Post	Mean	3.08	0.25	5.75	1.18
		SE	*0.11*	*0.01*	*0.29*	*0.07*

Table E.2   Summary table for Na, K, Ca, and Mg before and after treatment for all streams. Values in italics denote S.E.

Appendix E   Summary tables of select stream chemistry parameters

			SO_4	**Cl**	**Si**
North Fork	Pre	Mean	0.15	1.66	8.12
(control)		*SE*	*0.01*	*0.07*	*0.15*
	Post	Mean	0.13	1.59	8.15
		SE	*0.00*	*0.04*	*0.11*
South Fork	Pre	Mean	0.14	1.33	8.16
(37% clearcut)		*SE*	*0.00*	*0.03*	*0.25*
	Post	Mean	0.13	1.23	7.91
		SE	*0.00*	*0.04*	*0.19*
Myers	Pre	Mean	0.18	1.47	8.71
(control)		*SE*	*0.00*	*0.04*	*0.23*
	Post	Mean	0.17	1.49	8.53
		SE	*0.00*	*0.03*	*0.19*
Fenton	Pre	Mean	0.20	1.54	9.68
(75% clearcut)		*SE*	*0.01*	*0.03*	*0.48*
	Post	Mean	0.14	1.33	9.09
		SE	*0.00*	*0.08*	*0.25*
Clay	Pre	Mean	0.15	1.47	7.63
(36% clearcut)		*SE*	*0.00*	*0.05*	*0.25*
	Post	Mean	0.12	1.23	7.12
		SE	*0.01*	*0.05*	*0.16*
DeMersseman	Pre	Mean	0.13	1.54	8.95
(control)		*SE*	*0.00*	*0.07*	*0.12*
	Post	Mean	0.11	1.53	8.97
		SE	*0.00*	*0.04*	*0.11*
BB	Pre	Mean	0.16	1.43	7.99
(32% clearcut)		*SE*	*0.00*	*0.06*	*0.23*
	Post	Mean	0.14	1.06	7.78
		SE	*0.01*	*0.07*	*0.18*
Russell	Pre	Mean	0.15	1.44	8.53
(15% clearcut)		*SE*	*0.00*	*0.06*	*0.22*
	Post	Mean	0.14	1.32	8.35
		SE	*0.00*	*0.02*	*0.19*

Table E.3   Summary table for SO_4, Cl, and Si before and after treatment for all streams. Values in italics denote S.E.

Appendix F   Lab instrumentation and methods

Analysis	**Instrumentation**
Alkalinity	Radiometer TIM840 Auto-Titrator
Ammonia	Technicon Auto-Analyzer II
Calcium	Varian SpectrAA220
Carbon, Inorganic	Shimadzu TOC-VCSH Combustion Analyzer
Carbon, Organic	Shimadzu TOC-VCSH Combustion Analyzer
Chloride	Dionex 1500 Ion Chromatograph
Specific Conductance	YSI model 3200
Iron	Varian SpectrAA220
Magnesium	Varian SpectrAA220
Nitrate + Nitrite	Technicon Auto-Analyzer II
Nitrogen-Total	Technicon Auto-Analyzer II
pH	Radiometer TIM840 Auto-Titrator
Phosphate- Ortho (SRP)	Technicon Auto-Analyzer II
Phosphorous- Total	Technicon Auto-Analyzer II
Potassium	Varian SpectrAA220
Silicon	Technicon Auto-Analyzer II
Sodium	Varian SpectrAA220
Sulfate	Dionex1500 Ion Chromatograph

Appendix E Lab instrumentation and methods (cont'd)

Analysis	Method Detection Limit (MDL)[1]	Minimum Level of Quantification (ML)[2]	Precision[3]
Alkalinity	0.2 mg/L	0.6mg/L	+/- 0.2
Ammonia-nitrogen*	0.010 mg/L	0.032 mg/L	+/- 0.003
Calcium	0.06 mg/L	0.19 mg/L	+/- 0.06
Chloride	0.01 mg/L	0.03 mg/L	+/- 0.01
Magnesium	0.02 mg/L	0.06 mg/L	+/- 0.02
Nitrate-nitrogen	0.001 mg/L	0.003 mg/L	+/- 0.001
Nitrogen, total (Persulfate)	0.010 mg/L	0.032 mg/L	+/- 0.010
Phosphate, ortho	0.001 mg/L	0.003 mg/L	+/- 0.001
Phophorous, total	0.002 mg/L	0.003 mg/L	+/- 0.002
pH	0.1 pH units[4]	0.3 pH units	+/- 0.1
Potassium	0.03 mg/L	0.10 mg/L	+/- 0.03
Silica	0.20 mg/L	0.6 mg/L	+/- 0.05
Sodium	0.01 mg/L	0.03 mg/L	+/- 0.01
Specific conductance	0.4 us/cm	1.3 us/cm	+/- 2%
Sulfate	0.01 mg/L	0.06 mg/L	+/- 0.01

1 The semiquantitative limit is the EPA MDL from 40 cfr part 136 as explained in detail in the epa publication http://www.epa.gov/waterscience/methods/det/rad.pdf.
2 The ML is defined as the lowest level at which the entire analytical system must give a recognizable signal and acceptable calibration point for the analyte.
3 Precision evaluated by repeated analysis of near detection level standard solutions.
4 Limitation of instrument scale on instrument currently in use.
* Please note that for ammonia-nitrogen that the laboratory has been able to produce data with the same precision as stated above at a LOD of 0.005 mg/1.

Appendix E   Lab instrumentation and methods (cont'd)

Analysis	* Method # (with Modifications)
Alkalinity	APHA 2320, titrate to pH 4.5. Modifications: Use 0.02N Na_2CO_3 and 0.02N H_2SO_4
Ammonia	APHA 4500-NH_3 G; EPA 350.1
Calcium	APHA 3111 D; flame atomic absorption spectroscopy. Modifications: nitrous oxide/ acetylene flame. Addition of 1 ml 50 g/1 lanthanum oxide to 10 ml sample
Chloride	APHA 4110 B; EPA 9056A
Specific Conductance	APHA 2510; Wheatstone bridge
Magnesium	APHA 3111 B; flame atomic absorption spectroscopy
Nitrate + Nitrite	APHA 4500-NO_3 F; EPA 353.2. Cadmium reduction method
Nitrogen-Total	APHA 4500-NO_3 F; APHA 4500-P J. Persulfate digest
pH	APHA 4500 H; stirred measurement with temperature compensation
Phosphate- Ortho (SRP)	APHA 4500-P F; EPA 365.1. Ascorbic acid method
Phosphorous- Total	APHA 4500-P J; APHA 4500-P F; EPA365.2. Persulfate digest
Potassium	APHA 3111 B; flame atomic absorption spectroscopy
Silicon	APHA 4500-SiO_2 E; Technicon industrial method 105-71W/B.
Sodium	APHA 3111 B; flame atomic absorption spectroscopy
Sulfate	APHA 4110 B; EPA 9056A

* Method References (note: CCAL procedures developed primarily from APHA methods; comparable EPA reference included for informational purposes only)
APHA 2005. Standard Methods for the Examination of Water and Wastewater; 21st Edition; American Public Health Association, Washington, D.C.
U.S. EPA Office of Solid Waste (OSW) Methods Team; Ariel Rios Bldg. (5307W); 1200 Pennsylvania Ave. NW; Washington, DC 20460; Phone: 703-308-8855; Fax: 703-308-0511; URL http://www.epa.gov/epaoswer/hazwaste/test/index.htm
U.S.EPA National Exposure Research Laboratory (NERL); Microbiological and Chemical Exposure Assessment Research Division (MCEARD); [formerly the Environmental Monitoring Systems Laboratory (EMSL), Cincinnati, OH]; 26 West Martin Luther King Drive; Cincinnati, Ohio 45268-0001; Fax: 513-569-7757