The Effects on Water Quality of Restricting

Cattle Access to a Georgia Piedmont Stream

by

ZACHARY P. THOMAS

(under the direction of Miguel L. Cabrera)

ABSTRACT

Past research on the benefits of excluding cattle from streams has mostly been conducted in the western U.S. in arid regions with low grazing intensity (usually <1 AUM ha^{-1}). This study was conducted on a Georgia Piedmont stream bisecting an intensively grazed dairy pasture (4 to 7.5 AUM ha^{-1}). Nutrients, fecal coliforms, and the aquatic insect community were sampled before and after fencing was installed to restrict stream access. Restricting access resulted in decreases in nutrients (17 to 72%, $p<0.05$) and fecal coliforms (95%, $p<0.1$) and an increase in aquatic insect diversity. Coincident with fencing installation was cessation of a severe drought that caused an increase in nutrients (67 to 214%) and fecal coliforms (18%) and a decrease in aquatic insect diversity at an upstream reference site. Opposite trends at the study site suggest the effects of cattle access to streams outweigh those of severe drought.

INDEX WORDS: Water Quality, Fencing Effects, Cattle Grazing, Fecal Coliforms, Agricultural Pollution, Nutrients, Phosphorus, Drought

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And now, as my son is likes to say, “The END.”

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Introduction

Study Objectives

Because local environmental degradation of aquatic and riparian systems can have systemic consequences, this study focused on two main objectives. The first was to determine whether fencing a Georgia Piedmont stream flowing through a cattle pasture would result in significant reductions in nutrient and fecal coliform loading. The second was to quantify the change in pollutant loading attributable to fencing in order to estimate the environmental impact this best management practice (BMP) could have on downstream systems.

Background

Non-point source (NPS) pollution is defined as all sources of aquatic pollution that do not derive from permitted 'point' sources (USEPA, 1993) and includes all pollutants that enter a waterway from spatially dispersed and indirect sources. As loadings of pollutants from point sources have been reduced, the proportion of surface water impairment due to NPS pollution has steadily increased (USEPA, 1994). As an example, average nitrogen (N) and phosphorus (P) inputs from NPS sources nationwide now far exceed the contributions from point sources (Carpenter et al., 1998). Most anthropogenic NPS inputs can be assigned to two general sources: urbanization with its attendant effects and agriculture with its associated practices. In terms of NPS nutrient loading, residential septic systems can be a major source of non-point N pollution (Correll and Dixon, 1980). Urban areas can also contribute >50% of total nutrients through a combination of point and non-point sources (Carpenter et al., 1998). In the majority of watersheds, however, agricultural NPS sources outweigh urban sources in terms of overall nutrient loading (Carpenter et al., 1998) and also bacterial loading in some cases (USEPA, 1994). As an agricultural activity, the impacts of row cropping on nutrient inputs can exceed impacts due to cattle grazing (Correll and Dixon, 1980; Jordan et al., 1997). However, many areas, especially in parts of the southeastern U.S., have seen decreases in the amount of cropped acreage such that pasture is now the dominant agricultural land use and potential source of NPS pollution.

Among the various pollutants that enter aquatic systems from non-point sources, current concerns generally focus on two specific types: pathogenic bacteria and nutrients. High levels of bacterial loading are a concern because they indicate a potential human health risk thereby diminishing the "swimmable" and "fishable" use of surface water (e.g. USEPA, 1994). Bacterial loadings can also trigger fish and shellfish consumption advisories, which can decrease the commercial potential of water bodies (Wilson and Carpenter, 1999). Increased nutrient levels are a concern because they encourage surface water eutrophication (Dodds and Welch, 2000; Wetzel, 2001). Beyond the well-established link between nutrients and lake eutrophication (e.g. Schindler, 1975), excessive inputs of N and P are a common and growing problem in rivers, estuaries, and coastal waters (Smith, 1998). Eutrophication of lentic systems has been associated with shifts in chemistry (e.g. pH, dissolved oxygen, nutrient cycling), energy transfer (e.g. primary and secondary productivity, energy export), and biology (e.g. community structure, species dominance) (as summarized in Horne and Goldman, 1994; Smith, 1998; and Wetzel, 2001). Many similar effects have also been shown to occur in lotic systems (Dodds and Welch, 2000), with a shift from a heterotrophic to an autotrophic system upon addition of nutrients (Hershey et al., 1997). In both types of aquatic systems, eutrophication has been linked to diminished economic activity in the vicinity due to lowered property values, fewer visitors, and an overall lessening of perceived aesthetic value (Wilson and Carpenter, 1999).

As an agricultural practice, cattle grazing in riparian areas can be a major contributor to NPS water quality problems. Impacts that grazing cattle have on these systems have been divided into four general categories (from Kauffman and Krueger, 1984): removal/destruction of streamside vegetation, physical alteration of the channel and banks through destabilization, direct deposition of wastes that decrease water quality, and compaction of stream bank soil. Resulting effects of these impacts include increased sediment transport, increased nutrient loading and mobility, degraded aquatic and riparian habitat, increased erosion and runoff, and changes in local hydrological characteristics (Kauffman and Krueger, 1984; Belsky et al., 1999). When streams are present in pastures, cattle have been observed to concentrate their grazing activities, and thus the resulting impacts, in riparian areas (Armour et al., 1991; Fleichner, 1994). The magnitude of these impacts is suggested by the spatial distribution of cattle-produced excreta. In one case one third of total excreta was found on 5% of the pasture surface (Hilder, 1966 *in* Haynes and Williams, 1993) and in another case, 60% of dung and 55% of urine were deposited on 15-31% of total pasture surface (Saggar et al., 1988 *in* Haynes and Williams, 1993). In these studies, the areas of concentrated impact were termed ‘campsites’, i.e. locations where cattle either rest at night, find winter forage, seek drinking water, or shade themselves in the heat of the day (Haynes and Williams, 1993).

In the western U.S., the documented results of riparian cattle grazing include decreases in water quality, destruction of riparian and in-stream habitat, and changes in aquatic biota production and community composition (published reviews of this material include: Belsky et al., 1999; Strand and Merrit, 1999; Kauffman and Krueger, 1984). In humid eastern areas of the United States, the impacts of riparian cattle grazing have been associated with: increases in water borne sediments and bank erosion in Tennessee (Trimble, 1994); increased nutrient loading in the Maryland Piedmont (Shirmohammadi et al., 1996); lower aquatic insect abundance and diversity in Michigan (Strand and Merrit, 1999); lower fish population densities in Pennsylvania (Wohl and Carline, 1996); and, elevated bacterial levels in Virginia that are higher than levels protective of human health (Hagedorn et al., 1999).

One means of mitigating the impact of cattle grazing on streams has been to use fencing to either exclude or limit access to riparian areas. Exclusion has the effect of restoring a vegetated buffer between the upland impacts of the cattle and local waterways, which both mitigates runoff pollution (Daniels and Gilliam, 1996) and reduces sedimentation by preventing destabilization of the channel and banks (Owens et al., 1996). The benefits of vegetated riparian buffers to water quality and aquatic habitat are well established (see Wenger, 1999 for a thorough review of the scientific and secondary literature). In the eastern U.S., relatively little material has been published on the benefits of livestock exclusion by means of riparian fencing. In the studies that are available, researchers have found: substantial reductions (60-96%) in fecal coliforms in

Virginia streams (Hagedorn et al., 1999); reductions of total suspended solids, nutrient loads (excepting soluble reactive P), and an increase in benthic invertebrate richness in a Pennsylvania watershed (Galeone, 2000); reductions of nutrients loads (excepting NO_3^- /NO_2^-) in a North Carolina Piedmont stream (Line et al., 2000); and, reduction of sediment loads in Ohio (Owens et al., 1996).

Literature Review[1]

Effects of cattle on phosphorus and nitrogen loading in streams: In aquatic ecosystems, P is the least abundant of the major nutrients and therefore the most likely to be the limiting factor for biological growth (Wetzel, 2001). In most ecosystems not influenced by humans, the most significant input of P is released from the gradual weathering of primary and secondary soil and rock minerals. As P is released, most of it is taken up by living organisms. This P is then returned to the environment upon death, and a portion of the P moves directly into aquatic systems via groundwater flow and erosion of soil and surface materials. The majority of P that is biologically available is the soluble mineral fraction in the form of phosphate (PO_4^-). Because phosphate readily reacts with organic matter and particulate metal oxides to form insoluble precipitates, it is usually not a major constituent of the total P found in surface waters, constituting <10% on average (Wetzel, 2001). The major fraction of total P in aquatic systems is adsorbed or chemically bound to particulate matter. Particulate-P, while not immediately available for uptake by plants, is nonetheless a source of P over the long term. In systems not

[1] *For purposes of comparison amongst studies in the following sections, grazing intensity was converted into animal units ha^{-1} $month^{-1}$ (AUM ha^{-1}) in which each dairy heifer or beef cow is valued at 1AU and each mature dairy cow is valued at 1.4AU (USEPA 2001).*

heavily influenced by humans, particulate-P is released slowly as it desorbs from particulate matter and is released from organic matter (OM) as the OM is broken down.

Cattle grazing alters the dynamics of the natural aquatic P cycle mostly by increasing P inputs. Cows increase P loads delivered to streams in two main ways. First, in many cases, grazing has been shown to decrease the amount and density of stream bank vegetation thereby increasing the erodibiltiy of the banks (e.g. Trimble, 1994) and reducing the effectiveness of the riparian area as a filter for runoff P (e.g. Kauffman and Krueger, 1984). Trampling of the banks and channel sediments can also have the effect of increasing the rate of downstream transport of P contained in these materials. Second, cows can serve to increase P loads by concentrating P in consumed pasturage and feed and depositing the excreta in spatially concentrated locations, especially streams and riparian areas. As consumed material passes through the digestive tract, less soluble compounds are converted into more soluble inorganic forms (Haynes and Williams, 1993). Up to 80% of fecal P has been found to be in an inorganic and highly soluble form (Haynes and Williams, 1993). Phosphorus loads from cow defecation can be quite substantial. Cows defecate 11 to 16 times day^{-1} with each event averaging 1.5 to 2.7 kg (Haynes and Williams, 1993). Thus, there is a possible fecal load of 16.5 to 43.2 kg of feces cow^{-1} day^{-1} containing 1.2% total P by fresh weight (Safley et al., 1984 *in* Haynes and Williams, 1993). This indicates a possible fecal P load of 198 to 518.4 g P cow^{-1} day^{-1}. Because there is a strong correlation between total P intake and total fecal P (Bromfield and Jones, 1970 *in* Haynes and Williams, 1993), the use of feeds or nutritional supplements high in P could further increase the fecal P load. Although P in urine is highly soluble (Haynes and Williams, 1993), relative to fecal loads this is a minor contributor, only accounting for 2.56 to 5.28 g P cow^{-1} day^{-1} (using data from Safley et al., 1984 *in* Haynes and Williams, 1993).

In addition to phosphorus, cattle grazing can serve to increase nitrogen (N) loading to streams. The vast majority of biologically available N originates from biological or anthropogenic fixation of atmospheric N_2. Because of the availability of N from the atmosphere through fixation or as reactive gasses (e.g. N_2O, NH_3), sources of N loading to aquatic systems can be difficult to isolate and manage directly. Additionally, because of the high solubility of many of its molecular forms, N transport and availability to aquatic systems can also be difficult to quantify and manage. Anthropogenic increases in loading of N above historically natural levels has been associated with the eutrophication of streams (Dodds and Welch, 2000) and estuaries (Smith, 1998). Among commonly found N-species, nitrate (NO_3^-) enters usually enters aquatic systems from fertilizers or the mineralization of proteins and other reduced forms of N from wastes, fertilizers and decaying material. Because of its high solubility, nitrate easily enters groundwater and may pose a drinking water hazard in high concentrations. Ammonium (NH_4^+) and other reduced forms of N generally enter aquatic systems directly from waste materials and other sources of decomposing organic material. These forms of reduced N act as both an energy source for certain bacteria and as an N source for the construction of proteins and other organic molecules. Ammonia can also be a problem because it is toxic to aquatic organisms and levels above 10mg L^{-1} have been shown to produce acutely toxic results on fishes (Belsky et al., 1999).

Cattle grazing can increase N loading to streams by decreasing the effectiveness of the riparian filter and through direct deposition of wastes. The ability of intact riparian systems to remove N compounds from runoff and groundwater vary greatly (e.g. Wenger, 1999), but generally result in a decrease in concentration (e.g. Lowrance, 1992). The degradation of bank stability and vegetation has been shown to decrease the effectiveness of riparian systems to remove both nitrate and ammonium (Wenger, 1999). In terms of direct deposition, cows may introduce N into streams either in urine or feces. Cows urinate 8 to 12 times day^{-1} each event averaging 1.6 to 2.2 L (Haynes and Williams, 1993), for an average daily production of 12.8 to 26.4 L of urine cow^{-1}. The N content of cow urine is related to N in the diet and water consumption, but normally ranges from 8 to 15 g-N L^{-1} (Whitehead, 1970 *in* Haynes and Williams, 1993). This indicates a possible nutrient contribution from urine of 102.4 to 396.0 g-N cow^{-1} day^{-1}. Typically >70% of N present in urine is urea (Doak, 1952; Bathurst, 1952 *in* Haynes and Williams, 1993) which is readily decomposed into NH_4^+. By fresh weight, feces average 2.9% TN (Safley et al., 1984 *in* Haynes and Williams, 1993). This indicates a possible nutrient contribution from feces of 478 to 1252.8 g-N cow^{-1} day^{-1}. Only 20 to 25% of N in feces is immediately soluble (Mason, 1979 *in* Haynes and Williams, 1993), with much of the remainder requiring mineralization by microbial activity. Excess N in the diet serves to increase the amount excreted with the urine (Haynes and Williams, 1993), and thus dietary supplements and high N feeds may increase estimated urine loads above the numbers listed here.

Much of the recent research that has associated the impacts of grazing on P and N loading have used statistical comparisons of multiple watersheds wherein percent land use is compared to water quality metrics at the watershed outlet (e.g. Shirohammadi et al., 1996; Correll, 1998). While useful for getting at the general contribution of cattle grazing and other land uses to water quality, it is not possible to use this information to directly link cattle grazing intensity to specific water quality effects. Within the body of published research whose objective was to quantify the specific effects of cattle exclusion on nutrient loading there are mixed results. In the North Carolina Piedmont, Line et al. (2000) found reductions of 33, 78, 76, and 82% in nitrate/nitrite, total Kehldahl nitrogen (TKN), total P (TP), and total suspended solids (TSS) loading, respectively, after fencing the stream in a continuously and heavily grazed pasture (3.38 to 6.71 AUM ha^{-1}, yearlong). Fencing created a 10 to 16 m riparian buffer. In terms of mean nutrient concentrations, nitrate/nitrite did not change (2.7 mg L^{-1} pre-treatment, 2.9 mg L^{-1} post-treatment); however, mean TKN dropped from 18.3 to 9.6 mg L^{-1}, and mean TP from 7.7 to 4.0 mg L^{-1}. Owens et al. (1989) measured no change in nutrient or sediment loads after fencing an Ohio stream in a grazed pasture (0.61 AUM ha^{-1}, Apr through Aug). Nutrient concentrations were found to be 0.6 to 0.8 mg L^{-1} for nitrate/nitrite, 1.3 to 4.3 mg L^{-1} for total N (TN), and 0.1 mg L^{-1} for TP. Preliminary results of a study in Pennsylvania, also did not show any significant changes in nutrient loading before and after cattle access to 3 streams were restricted (1.6 to 4.1 m buffers established with fencing) (Galeone, 2000). However, there were few post-treatment data at the time of publication and a series of severe droughts occurred in the first post-treatment years (Galeone, 2000).

Effects of cattle on fecal coliform concentrations in streams: Although not usually considered pathogenic, fecal coliforms (FC) are regularly monitored in surface waters because they are indicators of fecal contamination that may include organisms that are pathogenic to humans (Geldreich, 1966). The presence of FC in surface waters originating from bovines have been associated with human pathogens such as *Cryptosporidium*, *Giardia*, *Salmonella*, *Shigella*, and enteric viruses (e.g. Bohn and Buckhouse, 1985). Bovine fecal contamination can also indicate the presence of bovine pathogens causing diseases such as bovine leptospirosis and mastitis (Crane et al., 1983). Because these organisms are carried by cattle (Belsky et al., 1999) and tend to increase with increasing grazing pressure (e.g. Gary et al., 1983), the probability of disease-causing organisms contaminating swimming areas, sport fishes, shellfish, and human water supplies increases with intensity of cattle use (Belsky et al., 1999).

FC concentrations in fresh cattle feces have been shown to range from 2.3 to 6 $x10^5$ FC-CFU g^{-1} wet weight (Crane et al., 1983). Since the mean mass of a defecation is 1.5 to 2.7 kg (Haynes and Williams, 1993), each fecal event contains a possible FC load of 3.5 to 16.2 $x10^8$ CFU. Deposition of this material in the channel or adjacent riparian areas can lead to significant increases in FC counts in the receiving waters. Degradation of the riparian area associated with cattle grazing can result in a greater possibility of this material being washed into the stream in storms. Many studies have shown a connection between stream discharge during storm events and FC levels (e.g. Robbins et al., 1972). This relationship, however, is complicated by the influence of temperature, livestock management practices, fecal deposit age, and the amount of source material in the channel and on the bank (Baxter-Potter and Gilliand, 1988). Antibiotics in the diet have also been associated with reductions in the fecal bacteria concentrations in feces (Crane et al., 1983). Regardless of the effects of these various factors, FC levels in agricultural storm runoff rarely meet the 200 fecal coliform colony forming units per 100 ml (FC-

CFU $(100\ mL)^{-1}$) standard for primary contact (Baxter-Potter and Gilliand, 1988; Crane et al., 1983).

Studies that have measured the effects of cattle grazing on FC levels uniformly report lower levels in the absence of cattle or after their access to streams has been restricted. Saxton and Elliot (1983 *in* Crane et al., 1983) found 1100 vs. 60 mean FC-CFU $(100\ mL)^{-1}$ at baseflow in streams near a grazed vs. an ungrazed Washington pasture, respectively. Pasture size and grazing intensity were not listed. In Oregon, Tiedemann et al. (1988) measured FC concentrations in streams draining pastures subjected to 4 different grazing strategies. These strategies included: no grazing, unmanaged grazing (0.13 AUM ha^{-1}), low intensity grazing managed with fencing and off-stream water sources (0.12 AUM ha^{-1}), and intensive grazing management for maximum production (0.36 AUM ha^{-1}). Resulting mean FC counts were 40, 90, 150, and 920 FC-CFU $(100\ mL)^{-1}$ for 0, 0.13, 0.12, and 0.36 AUM ha^{-1}, respectively. In terms of field derived runoff, in a central Nebraska study on small plots, Doran et al. (1981) found 113,700 vs. 13,280 FC-CFU $(100\ mL)^{-1}$ for grazed (0.97 AUM ha^{-1}, April through November) vs. ungrazed, respectively. Barker and Sewell (1973) measured 8,900 and 20,000 mean FC-CFU $(100\ mL)^{-1}$ for two 'heavily' grazed Tennessee pastures of unlisted sizes.

Impacts of cattle on aquatic and riparian biota: Riparian and aquatic ecosystems can be extremely diverse in terms of the native flora and fauna they support. For example, aquatic ecosystems in the southeastern U.S. contain about 800 fish species, a higher diversity than any other temperate geographic area on earth (Burkhead et al., 1997). Further, it has been suggested that riparian habitat provides the living conditions for a greater variety of wildlife than any other type of habitat in the U.S. (Johnson et al., 1977 *in* Kauffmann and Kreugar, 1984). Where records or studies are available, agriculture has generally had the effect of degrading aquatic and riparian ecosystems (Allan, 1996). While the effects of cattle grazing in particular can be beneficial from a range management perspective, especially in terms of increased forage production (Strand and Merritt, 1999), published reviews of scientific literature have failed to find any research demonstrating a net benefit of cattle grazing in terms of richness, production, or standing biomass for native organisms (Strand and Merritt, 1999; Belsky et al., 1999) except in the case of pollution tolerant taxa (e.g. Wohl and Carline, 1996). For example, cattle grazing has been associated with decreased densities of sensitive native fishes and lower overall fish diversity (Wohl and Carline, 1996), reduced usage or total avoidance of riparian areas by many local and migratory bird species (Kauffman and Kruegar, 1984; Belsky et al., 1999), and shifts in the functional feeding groups of the aquatic invertebrate community (Strand and Merritt, 1999). These types of reductions in the number, size, and productivity of native riparian or aquatic species are nearly always viewed as representing declining ecosystem health (Ohmart, 1996).

Cattle impact riparian and aquatic biota in many and diverse ways including, but not limited to, acute and chronic pollutant toxicity, direct destruction of habitat, and alterations to nutrient, energy, and material cycles. As mentioned previously, increased nutrient loading can cause toxicity problems (e.g. ammonia) and alter many of the natural energy cycles and processes (e.g. eutrophication). Examples of direct destruction include destabilization of the stream channel (Trimble, 1994) and trampling and overgrazing of stream bank vegetation (Kauffman and Kreugar, 1984). Degradation of stream bank vegetation can also alter stream energy cycles by increasing solar energy (Cummins, 1974) and destroying an important source of detrital material that can provide up to 90% of the organic matter necessary to support headwater stream communities (Cummins and Spengler, 1978 *in* Kauffman and Kreugar, 1984). Further, management of pastures can often have adverse impacts on stream biota as in the case of removing large woody debris from streams. This material, while considered a hazard to grazing cattle, nonetheless plays an important role in dissipating stream energy, routing sediment, and serves as a substrate for many organisms (Kauffmann and Kreugar, 1984).

Due to the large number of ways that cattle can impact an ecosystem's health, in terms of time, space, and magnitude, it can be difficult to determine the cumulative impact of cattle on an ecosystem. One way that many researchers have assessed the ecological impacts of activities of this nature has been to compare the abundance, diversity, and richness of some representative portion of the biota in the degraded system to that of a 'healthy' or reference system (Rosenberg and Resh, 1996). Within the field of aquatic ecology, the aquatic insect community has been the most frequently used means for gauging overall ecosystem health because: insects are ubiquitous; they encompass a large number of species each of which exhibits a range of responses to different environmental stresses; they are relatively sedentary in nature; and, they have life cycles that are long enough to cross over short term impacts (Rosenberg and Resh, 1996). Along these lines, Strand and Merritt (1999) sampled sites in two Michigan watersheds in order to compare the effects of cattle grazing on the aquatic invertebrate community. The researchers found two streams 1km apart with similar depth, current velocity, and water temperature (basin sizes were not listed). Based on these similarities they assumed that differences in riparian characteristics were the most important source of variation for invertebrate abundance, diversity, and community composition. One stream was considered 'overgrazed' and riparian cover was grass. The other study reach had extensive forest as its dominant riparian land cover and no cattle present. Sites were sampled monthly (two sites per reach) with drift nets, introduced rock, wood, and leaf packs. The researchers determined that the 'overgrazed' site had lower abundance (pasture vs. forest = ~33 individuals per sample vs. ~53) and lower taxonomic richness (33 vs. 44 insect taxa) and that the invertebrate community incorporated a larger proportion of insects tolerant of siltation and organic pollution (Strand and Merritt, 1999).

Relevance of this study: When cattle have access to streams there is a potentially major increase in water quality problems and a decrease in the biological integrity of the aquatic ecosystem. While the benefits of excluding cattle from streams are well understood in the western U.S., there is less published documentation for eastern areas. Because eastern pasture systems tend to have higher overall forage production, grazing intensities can be substantially higher than in the West. Also, the increase in wetness may play an influential role in increasing the likelihood that polluting material will be mobilized in runoff and transported into streams. In the state of Georgia there are 1.2 million ha of pasture land which cover 7.3% of the total surface area of the state (NASS, 2001). In many Georgia Piedmont counties, cattle grazing is now the dominant agricultural land use (e.g. Putnam, Morgan, and Greene counties, NASS, 2001). Bacterial loading from these areas is a concern because violations of the fecal coliform standard is the top reason that Georgia stream and river reaches are not meeting their designated usage (USEPA, 1998). Further, nutrient loading from these areas is a concern because most streams in this region contribute flow to reservoirs, many of which are listed as eutrophic already and all of which have seen substantial increases in indices of eutrophication in the last 20 years (GA-EPD, 1992). For these reasons there is an increasing interest in using fencing to exclude cattle from streams as a means to improve water quality. This study was conducted in order to increase the understanding of how excluding cattle will affect the nutrient and bacterial dynamics of streams in the southeastern Piedmont region.

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Chapter 2

The Effects on Water Quality of Restricting Cattle Access to a Georgia Piedmont Stream

Overview of Study

Two primary effects were hypothesized to result from using fencing to create a riparian buffer and restrict the access of cattle to a stream:

1. Water quality (in terms of measured nutrient and bacterial concentrations) would be improved leading to a decrease in the amount of nutrients and bacterial pollution exported to downstream systems; and,

2. The aquatic insect community (in terms of habitat and diversity indices) would be enhanced.

In order to test these hypotheses, a "before-after" experimental design incorporating an upstream site as a reference was employed. The reference site was located directly upstream of the area from which cattle were excluded, and the treatment site (henceforth referred to as the 'pasture' site) was located at the downstream end of the cattle exclusion area (Fig. 1). During a pre-treatment period, data on nutrients (N and P) and bacteria (fecal coliforms) were collected for 18 months. Additionally, one survey of the aquatic insect community was performed in each of three reaches (Fig. 2). At the end of this period, cattle access was restricted at the study site by closing the last open section of the riparian fencing. Post-treatment data were then collected for a further 10 (N and P) to 12 (fecal coliforms) months, and a second round of surveys of the aquatic insect community in the same three reaches was performed.

Site Description

The study site is located on the Southern Piedmont Plateau near Eatonton, Georgia (33º23’20”N latitude, 83º27’00”W longitude), in a 140-ha watershed that is drained by an unnamed second order tributary of Glady Creek, a tributary of the Little River in the Altamaha River basin. Within the watershed, mean annual rainfall is 1232 mm, and mean annual temperature is 17ºC. Slopes are 0 to 2%, and soils are moderately fertile with moderate to high erodibility. The area upstream of the study site is owned by the University of Georgia (Central Research and Education Center) and is used as pasturage for beef cattle. Within this area the stream is fenced including a riparian buffer of 3 to 15 m. During the period in which this study was conducted, management and use of the upstream pasture was not altered (cow-calf pasturage, 1.3 to 1.8 AUM ha^{-1}). The study pasture is approximately 4 ha and is used by a local dairy as a grazing area for restocking heifers (of Holstein stock, generally less than 1 year in age) from approximately March to December. In winter months the site typically becomes too wet, and the heifers are temporarily relocated to other pastures until the site dries out in the early spring. During the study, when animals were present, grazing intensity varied from 16 to 30 cows (4 to 7.5 AUM ha^{-1}) depending on availability of animals and pasture conditions. Vegetative cover within the pasture varied from approximately 95 to 99% depending on time of year and grazing intensity. Prior to excluding the cattle from the stream the majority of bare soil was along the banks of the stream. The stream of interest roughly bisects the site, flowing from north to south. Samples collected immediately upstream of the pasture were used as a reference. Samples collected immediately below the pasture were used to test the effects of the treatment (sampling sites shown in Fig. 1).

The treatment consisted of installing fencing along the stream to create a 15.2-m (50-ft) wide riparian buffer. Because the stream bisects the study site, a concrete stream crossing was installed near the north end of the site (Fig. 1). This crossing also allowed the cattle to access the stream for drinking. Prior to the commencement of the study, the crossing, all of the fence posts, and the majority of the fencing were installed. A section of the fencing was left open such that the cattle still had access to the stream (red line shown in Fig. 1). After 18 months of pre-treatment data collection, the final section of fencing was closed in October 2000. Data were then collected for 10 to 12 months to test the effects of the fencing.

Methods

Nutrient Sampling: Water samples were collected automatically using ISCO model 3700 samplers (ISCO Inc., Lincoln, NE) connected to CR-10x dataloggers (Campbell Scientific, Logan, UT). Sub-samples were drawn simultaneously once every 11 hours from both the reference and pasture sites. These sub-samples were composited as 5 sub-samples per sample bottle. In the field, samples were preserved using 1 mL of H_2SO_4 per 500 mL of sample. After collection, water samples were transported to the lab where 125 mL was filtered through a 0.45-µm Cellulose Nitrate filter paper that had been pre-wetted in a 0.5M solution of HCl. Filtered samples were frozen and later analyzed for soluble reactive P (SRP, Murphy and Riley, 1962), NH_4^+ (Crooke and Simpson, 1971) and NO_3/NO_2 (Keeney and Nelson, 1982) using an Alpkem autoanalyzer (OI Analytical, College Station, TX). A further 60 mL of each unfiltered sample was frozen and archived for potential additional analyses in the future. Finally, 250 mL of each unfiltered sample were stored in a dark refrigerator for later total P and N analyses. Total N and P analysis was accomplished using the persulfate oxidation method as per Korolef (1983) with the exception that the reagent to sample ratio was changed to 1:1 in order to insure complete oxidation of the potentially high dissolved organic C (DOC) content in the samples (Cabrera and Beare, 1993). After oxidation, the samples were frozen for later analysis on the Alpkem autoanalyzer. An attempt was made to collect the samples every 2 weeks (the outside limit of the preservation method), but this was not always possible. Samples that sat for more than 2 weeks before being filtered were identified and excluded from the statistical analysis.

Fecal Coliform Sampling: Grab samples of 100 mL of stream water were collected within a 20-minute interval at both the reference and pasture sampling sites. Samples were collected in autoclaved glass bottles. An attempt was made to collect samples approximately twice per month during the summer with approximately monthly winter sampling. However, because the stream dried up during both summers of the pre-treatment period, this was not always possible. Samples were stored on ice and analyzed within 3 hours of collection by membrane filtration with isolation on mFC agar and incubation at 44.5º C (in a water bath) for 24 hours (APHA, 1992). Because of the variability inherent in bacterial sampling, at least 3 dilutions (usually 10^0, 10^{-1}, and 10^{-2}) were performed on each sample. Three replications were performed for each dilution and the number of colony forming units (CFUs) recorded for each sampling date represent the mean of these replicates.

Hydrology: At both sampling sites, stage was monitored hourly using pressure transducers connected to CR-10x dataloggers (Campbell Scientific, Logan, UT). From March 1999 to February 2000, Model 236 PC (0 to 34.5 kPa range) MicroSwitch pressure transducers (Honeywell, Freeport, IL) were employed until it was determined conclusively that inaccurate stage data collected at the pasture site was a result of the malfunction of the pressure transducer. The same model pressure transducer installed at the reference site functioned properly during this period. In February 2000 both pressure transducers were replaced with Druck model 1833 (0 to 17.25 kPa range) pressure transducers (Druck Inc., New Fairfield, CT), which provided reliable and consistent stage data during the remainder of the study. For the time periods when stage data were available for each site, stage-discharge curves were developed using USGS standard techniques to estimate discharge. Velocity measured at 0.6 of water column depth for ~10 equally spaced sections across a stream transect at both the reference and the pasture sampling sites. Measurements were made on multiple occasions at different stages. Fortunately, due to the close proximity of the sampling sites (~300 m), discharge was found to vary little between locations. Hence discharge data for the pasture site during the period with a malfunctioning transducer (3/99 to 2/00) were derived from a regression based on data collected from both sampling sites using the Druck pressure transducers.

Aquatic Invertebrate Sampling: Aquatic invertebrates were collected twice during the study period. Both samples were taken in late spring, once before (5/9/2000) and once after (6/9/2001) closing the riparian fencing. Samples were collected from 100-m reaches at 3 locations: a ‘reference reach’ (located immediately upstream of the reference sampling site), a ‘pasture reach’ (located within the study site), and a ‘downstream reach’ (starting at 50 m downstream of the pasture sampling site) (Fig. 2). Sampling was qualitative and included the same timed effort (2 person-hours per site) for all sites on both sampling occasions. Within the time constraints, a representative amount of all habitats were sampled and as many invertebrates as possible picked and preserved in 70% ethanol. Invertebrates were later identified to family in the laboratory.

Data Analysis: Randomized Intervention Analysis (RIA, Carpenter et al., 1989) was employed to analyze the results of the water quality sampling. This statistical process has been shown to be effective for the analysis of non-parametric, ecological data when the experimental treatments are not replicated (Carpenter et al., 1989). To satisfy the requirements of the analysis, at least 40 data points were collected for each tested parameter before and after installation of the fencing for both the reference and pasture sampling sites. Additional analyses of the nutrient and fecal coliform data were performed using SAS (SAS Institute Inc., Cary, NC). The effects of organic pollution and habitat quality on the aquatic insect community were determined using the Hilsenhoff family level biotic index (Hilsenhoff, 1988). Differences in aquatic insect diversity between sites and sampling dates were calculated using Shannon’s H’ and Fisher’s index of alpha diversity (both indices calculated as outlined in Magurran, 1988).

Results

Climate and Hydrology: Precipitation and discharge data were considered by water year (from 10/1 to 9/30). In temperate climates, because most plant growth has ceased by 9/30, rainfall after this date tends to increase the amount of groundwater and soil moisture available to plants in the following spring and summer. For this reason, precipitation and discharge measured by water years are more directly relevant to impacts on vegetation, other organisms, and natural systems. During the 1999 (starting on 10/1/98) and 2000 (starting on 10/1/99) water years (the pre-fencing data collection period), the region in which the study area is located experienced two of the driest water years in recorded history. Rainfall in 1999 and 2000 were 746 mm and 831 mm, respectively, which are 39% and 33% lower than the 50-year mean of 1232 mm (Table 1). These droughts led to the gradual stagnation of the stream in early spring and complete de-watering of the channel by mid-summer for substantial periods of time (10 weeks in 1999, 7 weeks in 2000). Precipitation for water year 2001 (starting on 10/1/00), the post-treatment period, was also below average (1065 mm); however, the stream did not stop flowing at any point in 2001. The effects of the 2000 drought can be seen in the 2000 mean annual flow which is less than half that measured for water year 2001 (Table 1). Unfortunately, before the beginning of this study, the study stream (or one of similar watershed size in the region) was not gauged and thus the long-term mean annual flow for this system is unknown. A graphical representation of the hydrograph and precipitation during the study period is presented in Fig. 3.

Phosphorus dynamics: At the pasture sampling site, both the mean concentrations of SRP and TP decreased after the riparian fencing was closed. Mean SRP concentration decreased 34% and mean TP concentration decreased 18% (Table 2). In contrast to this trend, in the post-fencing period at the reference site mean SRP concentration increased 214% and mean TP concentration increased 92% (Table 2). In the post-fencing period, mean SRP concentrations for both sites were within ± 1 standard error of each other (reference = 0.66 ± 0.07, pasture = 0.67 ± 0.09). The changes in SRP and TP concentration at both sites are apparent in the individual samples (Fig. 4). Box plots of the differences between the pasture site and reference site (as Pasture - Reference) for individual samples show that in the post-fencing period, differences in SRP concentration between the sites decreased and were centered around 0 (Fig. 5), indicating no consistent difference in concentration between the sites. The same trend in differences in concentration between sites is visible in the post-fencing period in the box plots for TP (Fig. 6). The mean difference between paired samples dropped from ~1.0 mg L^{-1} to ~0.5 mg L^{-1} in the post-fencing period. In the case of both box plots, differences that register below the zero line indicate paired samples for which SRP or TP concentration at the reference site was higher than at the pasture site. RIA demonstrated a significant treatment effect; the pasture site and the reference site were more similar in terms of both SRP ($p < 0.001$) and TP ($p = 0.005$) in the post-fencing period (Table 3).

Plotting pooled data from pre- and post-fencing periods for SRP vs. TP indicated that a large proportion of TP for both sampling sites was SRP (Fig. 7). A regression of the relationship between SRP and TP indicated that, for both sampling sites, SRP constituted approximately 76 to 79% of TP. Thus the majority of the reduction in TP can be attributed to the drop in SRP and may not be attributable to other changes in the system after the fencing was closed.

Nitrogen dynamics: At the pasture sampling site, the mean concentrations of NH_4^+, NO_3^-/NO_2^-, and TPN decreased in the post-fencing period. Mean NH_4^+ decreased 72%, mean NO_3^-/NO_2^- decreased 17%, and mean TPN decreased 47% (Table 2). Similar to the results seen with mean P-concentrations, mean concentrations of the N species measured increased in the post-fencing period at the reference site. Mean NH_4^+ increased 137%, mean NO_3^-/NO_2^- increased 67%, and mean TPN increased 119% (Table 2). The changes in NH_4^+ and TPN concentrations for both sites are clearly visible in individual samples (Fig. 8). The same changes are evident in NO_3^-/NO_2^- concentrations (Fig. 9) although not quite as pronounced as with NH_4^+ and TPN. Box plots of the differences between the sites (as Pasture - Reference) for individual samples show that, in general, many of the differences in NH_4^+ concentration between the sites did not change in the post-fencing period (especially within the 25th to 75th percentiles). However, the outliers representing huge differences in NH_4^+ in the pre-fencing period are entirely absent in the post-fencing period (Fig. 10). Box plots of the NO_3^-/NO_2^- data also show little decrease in the magnitude of the differences and fewer outliers at the upper end in the post-fencing period (Fig. 11). Box plots of the TPN data show a clear decrease in differences between the sites, with a drop in mean difference from ~6.0 mg L^{-1} (pre-fencing) to ~2.0 mg L^{-1} (post-fencing) (Fig. 12). RIA demonstrated a significant treatment effect of fencing on nitrogen dynamics; the pasture site and the reference site were more similar in terms of NH_4^+ ($p < 0.001$), TPN ($p = 0.001$), and NO_3^-/NO_2^- ($p = 0.012$) in the post-fencing period (Table 3).

In plots of NH_4^+ vs. TPN, no trend is apparent at the reference site for the pooled pre- and post-fencing data (Fig. 13). A plot of the pooled data for the pasture site shows that the majority constituent in several TPN samples was NH_4^+ (Fig. 13). By splitting the pasture data into pre- and post-fencing periods it becomes evident that prior to fencing, higher levels of TPN mostly consisted of NH_4^+ (Fig. 14), with the exception of 5 samples (circled in Fig. 14). These samples were collected in mid-June during a 4-day period at the very beginning of the study and a review of field and lab notes did not turn up any explanation for their unexpectedly lower concentrations of NH_4^+. With the exception of these samples, there is a contrast between a strong, linear trend between TPN and NH_4^+ in the pre-fencing period and no relationship after the fencing was closed (Fig. 14). As with the relationship witnessed between SRP and TP, the majority of the data represent samples collected during baseflow conditions and so do not represent the system's performance during stormflows.

During the first two summers of the study period (the pre-fencing period), the study stream dried up completely. Data collected for NH_4^+ and NO_3^-/NO_2^- during the final two months prior to desiccation of the stream show an interesting trend: at the pasture site, as the concentration of NH_4^+ increases there is an increase in NO_3^-/NO_2^- that closely tracks the NH_4^+ trend (Fig. 15). This trend, however, is not in evidence in the data from the reference site during the same period (Fig. 15). During both of these periods flow was slowing to a halt and then the channel was gradually drying out. It is possible that as the stream ceased to flow it took on the qualities of a shallow, eutrophic pond in the springtime: high primary production during the day accompanied by high oxygen output. Personal observations of the stream during this period revealed massive algal blooms on the water surface in the pasture reach that had entrapped large bubbles of oxygen underneath (Fig. 16). Under these circumstances, it is possible a portion of the high concentrations of NH_4^+ (introduced by the cattle as urine) were nitrified during day light hours when high levels of dissolved oxygen were available. A similar algal bloom was not observed in the reference reach probably due to the lower level of nutrients and the presence of large trees that completely shaded the channel.

Nutrient export from pasture site at base flow: In the post-fencing period at the pasture site, decreases in nutrient concentrations (Table 2) were accompanied by a doubling of mean annual flow (Table 1). Except for NO_3^-/NO_2^-, the reductions in concentrations more than offset the increase in flows resulting in a net decrease in the total amount of N and P exported from the site (Table 4). Reductions in export were most pronounced in terms of SRP (71%) and NH_4^+ (61%).

Aquatic Invertebrates: In the reference reach, the Hilsenhoff family level biotic index (FBI) increased indicating a decrease in habitat quality (Table 5). This decrease in quality at the reference reach after fencing closure was mirrored in a decrease in diversity as indicated by Shannon's H' and Fisher's alpha, a decrease in total taxa collected and total sample abundance, and an increase in the % of animals that were chironomids (Table 5). At the pasture site, the Hilsenhoff FBI score decreased after fencing indicating an improvement in habitat quality (Table 5). An increase in diversity at this site after fencing was indicated by an increase in Shannon's H' and Fisher's alpha (Table 5). Additionally, total taxa collected and total sample abundance increased and the % of animals that were chironomids decreased (Table 5). At the downstream reach, the Hilsenhoff FBI score did not change after fencing closure indicating no change in quality rating (Table 5). An increase in insect diversity at the downstream site after fencing closure was indicated by both Shannon's H' (from 0.64 to 1.62) and Fisher's alpha (from 3.09 to 3.51), however, this increase was not as dramatic as the changes at either the reference or pasture reaches (Table 5). The total number of taxa collected decreased slightly, total sample abundance was nearly halved, and the % of animals that were chrironomids decreased (Table 5).

Fecal coliforms: While fecal coliform (FC) levels did not vary much at the reference site between the pre- and post-fencing periods, concentrations measured at the pasture site dropped an order of magnitude in the post-fencing period (Fig. 17). In 95% of paired samples taken in the pre-fencing period FC concentrations were lower at the reference site than at the pasture site. In comparison, only 40% of samples taken in the post-fencing period registered lower FC concentrations at the reference site. Mean FC concentration at the pasture site decreased 95% after fencing (Table 2). In comparison, mean FC concentration increased 18% at the reference site after the fencing was closed (Table 2). Hence the mean difference in FC concentration between the sites decreased by 97% (Table 3), a trend that is also apparent in the differences between sites in the individual samples as shown in box plots in Fig. 18. Although in the post-fencing period the pasture and reference sites were substantially more similar in terms of mean FC concentration, RIA indicated that the effect of the fencing on FC dynamics was significant at the $\alpha = 0.1$ level ($p = 0.088$). This outcome (high p-value despite the substantial increase in similarity between the sites) may be a result of the small number of samples drawn ($n = 40$), which is the minimum recommended number of samples needed to obtain good results from the RIA.

Discussion

Phosphorus: At both the pasture and reference site for both the pre- and post-fencing periods, SRP was found to constitute 76 to 79% of TP at baseflow, suggesting that the majority of the reductions in TP for the pasture site were driven by reductions in SRP. The low proportion of particulate P measured in TP at baseflow is not unexpected, especially for the study stream, which has a wide, shallow channel, low channel slope (~0.2%), and low thalweg velocity at baseflow (~0.01 m sec^{-1}). Also, because of the drought, lower than normal flows prevailed during the duration of the study, which may have further reduced the load of particulate P at baseflow. A further effect the drought may have had on SRP levels at both sampling sites may have been to increase the concentration over what would be expected in wetter years. In a grazed watershed experiencing a drought in Pennsylvania, Galeone (2000) linked elevated SRP levels at baseflow to higher than normal reducing conditions in organically enriched sediments due to increased stream stagnation. A similar circumstance possibly occurred at both sampling sites during this study. Because of the early problems associated with trying to accurately gauge stream flow at the pasture site, all samples were drawn on a timed basis and thus storm flows are not well represented in the data. Because much of the P exported by streams from a system can be in particulate form and can be removed during spates (Meyer and Likens, 1979), the strong relationship found here between SRP and TP is probably only a characterization of baseflow conditions and may not hold for more intense hydrologic circumstances.

Nitrogen: The relationship observed between TPN and NH_4^+ at the pasture site in the pre-fencing period disappeared in the post-fencing period. Because cattle urine contains a high concentration of N that is readily converted to NH_4^+ (Haynes and Williams, 1993), moving the location of urine deposition away from the stream had a substantial effect on the NH_4^+ dynamics of the stream at baseflow. Because NH_4^+ was a major constituent of TPN at the pasture site in the pre-fencing period, fencing also had the effect of decreasing TPN in the post-fencing period at baseflow. This is similar to the trend found by Line et al. (2000) where total Kehldahl nitrogen (TKN) was found to be consistently high (mean = 18.1 mg L^{-1}) prior to using fencing to exclude cattle from a North Carolina stream. In the post-fencing period, mean TKN dropped (mean = 9.6 mg L^{-1}), especially during baseflow. After fencing, TKN was tied to discharge with the majority moving only during storms when surface runoff was a significant factor (Line et al., 2000).

Other nutrient trends: In the later part of the post-fencing period, NO_3^-/NO_2^- levels generally decreased as water flowed from the reference site to the pasture site (Fig. 6) between 1/15/01 and 4/15/01, a period of higher precipitation and baseflow discharge (Fig. 3). Although not as pronounced, this same trend is visible in SRP and TP (Fig. 4) and NH_4^+ and TPN (Fig. 5) and is indicated in the box plots by the negative difference in paired post-fencing observations, especially for SRP (Fig. 5), NO_3^-/NO_2^- (Fig. 11), and TP (Fig. 6). Because these higher nutrient concentrations in the reference occurred over more than 20 samples and were found in all nutrient species measured, error in analysis was ruled out. Further, since this period was wetter with higher overall base and storm flows, increased removal either from biological processes or settling of particulate matter does not seem a logical explanation. Although there is no way to verify what really happened, a preferential flow path of enriched water was entering the stream near the sampling intake point at the reference site. Near the intake for the reference sampling site, there is a shaded area where the beef cattle in the upstream pasture area tend to seek shelter, especially on hot or rainy days. This highly impacted area is less than 5 m from the reference sampling intake point, highly trampled, and heavily loaded with excrement. It is likely that during the wetter post-treatment period, more material was mobilized from this location and flowed preferentially to the reference sampling intake point where it artificially increased the measured reference nutrient concentrations. By the time the stream water had reached the pasture sampling intake downstream, the stream water was more thoroughly mixed so that the spike in concentrations was not apparent. Since removing the sampling dates from the randomized intervention analysis had no effect on the significance of the changes in the pasture systems between pre- and post-fencing, all of the data were retained.

Secondary treatment effects of the drought: As can happen in many field experiments, an uncontrolled factor (e.g. the weather) can introduce a secondary treatment effect. In this case, the severe 2-year drought that occurred in the pre-fencing period is just such an unanticipated factor. The confounding effects of this 'secondary treatment' can be difficult if not impossible to remove as a source of increased variability in the data, especially (as in this case) where it was not possible to replicate the experiment. For just this sort of situation Carpenter et al. (1989) developed randomized intervention analysis (RIA) which uses data from paired samples from reference and treatment sites from pre- and post-treatment periods to account for 'secondary treatment' effects.

Effects of drought on water quality and aquatic biota: The reference system used in this study is also a grazed system with a fenced riparian corridor (3- to 15-m width), albeit of larger size (140 ha compared to 4 ha for the study site) and lower grazing intensity (1.3 to 1.8 AUM ha^{-1} compared to 4 to 7.5 AUM ha^{-1}). The severe 2-year drought that occurred during this study began during the pre-treatment period and effectively ended coincidently about the time that the riparian fencing was closed. The post-fencing period was also a drought year but less severe than the pre-fencing drought (Table 1). A visual comparison of the overall effects of the drought on the pasture is presented in Fig. 19. Because no alteration in land use or management occurred in the reference system during this period, the changes measured at the reference site in the post-fencing period can be associated with the effects of drought on a moderately grazed watershed with a fenced riparian corridor.

The effects of droughts on aquatic systems have not been as extensively studied as those of other naturally occurring disturbances, especially floods (Lake, 2000), and this is particularly true for streams draining grazed areas. In grazed areas that are not artificially fertilized, the majority of nutrient and bacterial inputs to streams originate from animal wastes. For this reason, the major impacts that cattle have on streams, in terms of nutrient and bacterial inputs, relate to the ability of their wastes to enter and then move down the stream. Because riparian area along the stream in the reference system was already fenced, direct deposition of wastes into the channel was not possible. For this reason, factors that affected the generation and movement of field runoff drove nutrient and fecal coliform dynamics in this system. The drought may have altered the transport and mobility of runoff pollution in two ways: by decreasing the movement of material into the channel (fewer and less intense rainstorms) and by altering the hydraulic dynamics of the stream itself (lower overall baseflow and fewer, less intense stormflows). Lower total rainfall also leads to decreased overall wetness in the pasture. This increase in dryness has been shown to decrease the rate at which urine moves through soil and the rate of physical and biological disintegration of fecal excreta (Haynes and Williams, 1993), further decreasing the movement of pollutants to nearby streams during storms. Additionally, since most fecal coliforms tend to be mobilized either from the stream sediments or from the field surface during storms (Crane et al., 1983), the lower overall number of these events during the pre-treatment period droughts may have decreased the amount of FC source material during baseflows. Thus, an increase in rainfall, mean annual flow, and the number and intensity of storms in the post-fencing period resulted in a substantial increase (67 to 214%) in mean nutrient concentrations and an 18% increase in mean FC concentration. In a watershed wherein grazing is the dominant land use, the most important factors driving the effects of drought on mean annual nutrient and bacterial concentrations in streams may be the difference in frequency, intensity and timing of rainfall from one year to the next.

Under the climatic conditions that occurred during this study, it is not surprising to see an increase in mean nutrient and bacterial concentrations in the reference system with an increase in precipitation in the post-fencing period. Because many of the mechanisms behind this effect of the drought are particular to grazed systems the same effect may not be applicable to other agricultural, urban, or natural land uses. Further, in already arid regions the level of effect that droughts have on nutrient and bacterial dynamics in grazed systems may be less significant (e.g. Caruso, 2001) because fecal desiccation and other mechanisms associated with dry conditions are already significant factors even under normal rainfall regimes.

The particular effects that a disturbance, such as drought, has on the aquatic community are more related to the severity, duration, and return frequency of the disturbance itself (Lake, 2000). Relative to other types of disturbances, such as floods, recovery of the biotic community from droughts can take considerably more time and post-drought community dynamics can be unpredictable (Lake, 2000). Further, in the case of a grazed system, pollutant-bearing residues that would normally be flushed from the system on a continuous basis may build up during extreme droughts; then, with a return to a wetter hydrologic regime, this residual material may become the source of increased water pollution and impact on the aquatic community. Particular effects that droughts have been shown to have on the aquatic biota include reductions in richness of sensitive taxa and overall diversity of the community (e.g. Morrison, 1990; Lugthart and Wallace, 1992) and an increase in the amount of richness associated with chironomids (Lugthart and Wallace, 1992). For the aquatic insect samples taken in the reference system, both the diversity and richness of the community was found to be reduced in the post-fencing period and the score for the Hilsenhoff FBI indicated a decrease in habitat quality (Table 5). Also, the proportion of Chironomids increased after the drought (Table 5). Thus, while the severity of the drought lessened in the post-fencing period, the aquatic community responded with a decrease in diversity and an increase in the proportion of tolerant taxa. The response of the aquatic insect community in the post-fencing period may be indicative of the lag time before recovery (slow re-colonization, flushing of pollutants) after a small stream is subjected to a severe drought of long duration.

Effects of fencing on nutrient concentrations and export at baseflow: In contrast to the trends observed at the reference site, mean concentration of all measured nutrients decreased in the post-fencing period at the pasture site. The primary effect that closing the riparian fencing had on the cattle was to prevent them from accessing the shade of the trees in the riparian corridor. This meant that the nearest source of shade was an old barn located ~150 m from the stream. Although the concrete crossing still afforded some access to the stream, after the fencing was closed the cattle spent very little time in this un-shaded area. Thus, the fencing had the effect of shifting the primary resting (and waste deposition) area of the cattle from the stream channel and its vicinity to an upland area (pre-fencing and post-fencing locations indicated in Fig. 1). Fencing the riparian area also had the effect of preventing trampling of the streambank vegetation. In the post-fencing this led to rapid re-establishment of vegetation on the banks, in the riparian area, and in shallow channel areas (Figs. 20a to 20c). By causing the majority of wastes to be deposited uphill, nutrients mobilized from excreta then had to pass through a vegetated buffer before reaching the stream – a situation that has been associated with nutrient and bacteria reductions (e.g. Trimble, 1994; Daniels and Gilliam, 1996). Nutrient species representing labile fractions in fresh manure (SRP) and urine (NH_4^+) were most substantially reduced (34% and 72%, respectively). Much of the overall reductions in N and P concentrations can be attributed to the decreases in these fractions. Thus, despite the increase in both rainfall and mean flow in the post-fencing period, which increased all mean nutrient concentrations at the reference site, mean nutrient concentrations decreased under baseflow conditions at the pasture site.

In a similar study, Owens et al. (1989) measured no change in nutrient concentrations in an Ohio stream after cattle were excluded with fencing. Further, Owens et al. (1989) measured mean concentrations of TP (0.1 mg L^{-1}) and total N (2.9 mg L^{-1}) that are lower than the post-fencing mean concentrations of TP (1.07 mg L^{-1}) and

TPN (4.17 mg L^{-1}) measured in this study. Grazing intensity, however, was much lower (0.61 AUM ha^{-1}) than was the case in this study (4 to 7.5 AUM ha^{-1}). In the one published study of the effects of cattle exclusion where grazing intensity was comparably high (3.38 to 6.71 AUM ha^{-1}), mean concentrations of TP and total N were higher overall and found to decrease significantly after the installation of fencing on a North Carolina stream (Line et al., 2001). This suggests that there may be a threshold of grazing intensity above which allowing cattle access to streams is significantly more destructive to water quality.

In terms of the effects that fencing had on downstream systems, annual nutrient export at baseflow was decreased substantially for SRP (71%), NH_4^+ (61%), and TPN (34%). While mean nutrient concentrations decreased significantly in terms of all nutrients, the increase in mean annual flow in the post-fencing period increased TP export enough such that there was no substantial reduction in total annual export (-2%, Table 5). Also, this increase in flow coupled with only a small drop in NO_3^-/NO_2^- concentration lead to an increase in the total export of NO_3^-/NO_2^- (53%). Since NO_3^-/NO_2^- dynamics can be more influenced by groundwater rather than surface inputs, the increase in precipitation may be more of a driver for this trend than any effect of the fencing. In total, annual export calculated at baseflow suggests that, in terms of total nutrient loading, the quantity of P and N exported to downstream systems was decreased after the installation of fencing. Since much of total annual nutrient export can occur during storm events, especially in the case of P where >80% was found to be exported in a single storm in one study (Meyer and Likens, 1979), baseflow export numbers are not quantitatively representative of the whole system's behavior.

Effects of fencing on fecal coliform concentrations: The difference in mean FC concentration between the pasture and reference sites decreased substantially in the post-fencing period (Table 3). This decrease was found to be significant at the $\alpha = 0.1$ level and occurred despite the wetter and rainier conditions prevalent in the post-fencing period. As mentioned previously, closure of the fencing changed the location of primary excrement deposition away from the stream and introduced an intervening vegetated buffer (as indicated in Fig. 1). As in the situation with runoff nutrient pollution, the majority of FC and other bacteria from manure that were mobilized in storms had to pass through this buffer to reach the stream. Such buffers have been shown to be very effective in decreasing the concentration of bacteria in runoff (Wenger, 1999). For this reason, it is not surprising that mean FC concentrations at the pasture site decreased 95% after the fencing was closed, despite the fact that the post-fencing period was wetter and experienced larger and more intense rainfall events, which led to an increase in mean FC concentration at the reference site.

Baseflow FC concentrations measured at the pasture site (range: 233 to 262500 FC-CFU $(100\ mL)^{-1}$) in the pre-fencing period were 1 to 2 orders of magnitude higher than values found in published literature for other grazed systems at baseflow (range: 90 to 1100 FC-CFU $(100\ mL)^{-1}$, from Tiedemann et al., 1988 and Crane et al., 1983). In fact, pre-fencing FC concentrations at the pasture site were found to be more in the range of values measured in direct field runoff from grazed plots during storms (range: 8900 to 113700 FC-CFU $(100\ mL)^{-1}$, from Barker and Sewell, 1973 and Doran et al., 1981). The similarity in FC concentration measured in the pre-fencing pasture site of this study and direct runoff from fields in other studies is probably due to the similarity between the systems: both are intensively grazed and incorporate wet conditions that maximize the possibility of bacterial mobility. The decrease in FC concentrations in the post-fencing period at the pasture site, however, did not bring mean FC concentration to within the range of values measured at baseflow in other published studies (e.g. Crane et al., 1983). This phenomenon is probably a result of the high level of grazing intensity in this study (4 to 7.5 AUM ha^{-1}), which is 11 to 20 times higher than the highest listed density for other grazing studies that measured FC at baseflow (max = 0.36 AUM ha^{-1}, Tiedemann et al., 1988).

Effects of fencing on aquatic biota and habitat: Because of the long term effects that drought can have on the aquatic insect community (Lake, 2000), it is difficult to disentangle the effects of the drought from that of the fencing, especially in the pasture and downstream reaches that were sampled. Using the reference reach as a gauge for the effects of the drought, all indicators point to a resulting decrease in the biological integrity of the system. No such trend is evident in the data collected in the pasture reach; rather, an increase in both diversity and habitat quality was observed in the pasture reach in the post-fencing period. It appears that, in the pasture reach, the load placed on the system by the presence of cattle in the stream was of greater magnitude than the impact of the drought. When this impact was mitigated (by preventing direct access to the channel and moving the primary resting location to an upland area), the aquatic community responded quickly with an increase in diversity and the presence of sensitive taxa. In the downstream reach, there were mixed results – whereas the habitat conditions did not change (as indicated by the Hilsenhoff FBI), overall diversity increased, % Chironomids decreased, and abundance dropped by nearly 50%. Drawing a conclusion from these changes in the aquatic community is particularly difficult for this reach for two reasons: the drought had less of an impact here as the channel retained some small pools throughout all years; and, cattle impacts have been shown to have highly localized effects on aquatic biota (see Strand and Merritt, 1999) so the changes upstream may have had a weaker effect on the downstream reach than on the pasture reach.

Conclusion

Because waste deposition is often the primary cause of elevated nutrient and bacterial concentrations in streams draining grazed areas, controlling the amount and location of deposition is of primary importance in improving water quality. In this study, despite leaving a portion of the stream accessible to the cattle for drinking, the cattle could not access streamside resting locations after fencing installation and so moved to an upland resting area. This effectively shifted the primary location of manure and urine deposition to an upland area and introduced a vegetated buffer between the cows and the stream. The effects of this change in cattle behavior were witnessed in decreased mean concentrations of SRP, TP, NH_4^+, NO_3^-/NO_2^-, TPN, and FC. Constituents of fresh cattle excreta that are particularly available (showing up as SRP, NH_4^+, and FC) were the most substantially reduced by the shift in cattle location and waste deposition. The decreased impact of the cattle on the stream due to fencing the riparian area was also witnessed in the aquatic insect community, which increased in diversity and the presence and abundance of sensitive taxa. Within the context of other studies, the water quality and biotic impacts that cattle have on streams may be a function of grazing intensity. Fencing a pasture stocked at low density had no effect (0.61 AUM ha^{-1}, Owen et al., 1989) whereas there was a significant reduction of nutrient concentrations here (4 to 7.5 AUM ha^{-1}) and under similar stocking rates (3.38 to 6.71 0.61 AUM ha^{-1}, Line et al., 2000).

Decreases in pollutant concentrations at the pasture site were measured despite a drought that occurred during the study that caused nutrient and FC concentrations to increase at the reference site. The decrease in pollutant concentrations at the pasture site suggests that, in this system, the effects of high intensity cattle grazing on stream nutrient and FC dynamics outweigh the effects of drought. The aftermath of the drought was also indicated in the aquatic insect community in the reference reach wherein decreases in diversity and presence of sensitive taxa were witnessed. The converse increases measured in the pasture reach suggest that the effects of high intensity cattle grazing outweighed the effects of the severe drought on the aquatic insect community in this system.

Unfortunately, it was not possible to quantify the change in total annual nutrient and bacterial export from the pasture site that fencing may have had. Calculated decreases in total annual export of P and N under baseflow conditions suggest that there was a decrease in overall export, especially species representing labile fractions in fresh cattle excreta (SRP and NH_4^+); however, this decrease was not comprehensively verified for all hydrologic conditions. This type of information is critically needed especially now as state and federal agencies are attempting to use a watershed level perspective to improving water quality and need to weigh the costs and benefits of available best management practices. While not conclusive, extrapolating from the data collected in this study does, however, indicate that restricting cattle access to streams with fencing will likely reduce loading to downstream systems.

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Table 1. Mean annual precipitation and stream flow at the study site. Data are summarized by water year (e.g. 10/1/1998 to 10/1/1999 for water year 1999).

	1999†	2000†	2001	50-yr mean
Precipitation (mm)	746	831	1056	1232
Mean annual flow (m^3 s^{-1}) ‡	0.007	0.028	0.068	
Approx. time stream desiccated (weeks)	10	7	0	
Number of storms§	2	7	11	

† Pre-fencing data collected during these water years.

‡ Using data collected at reference sampling site. Data for 1999 only include 5/1/99 to 10/1/99.

§ Storms counted as precipitation events with an intensity >25mm hr^{-1}.

Table 2. Mean nutrient and bacteria concentrations for both sampling sites before and after fencing closed. Values are mean ± SE.

	Pasture site		Reference site	
	Pre-fencing	Post-fencing	Pre-fencing	Post-fencing
	------------------------------ ($mg\ L^{-1}$) ------------------------------			
Soluble reactive P	1.01 ± 0.11	0.67 ± 0.09	0.21 ± 0.01	0.66 ± 0.07
NH_4^+-N	3.39 ± 0.61	0.96 ± 0.07	0.19 ± 0.02	0.45 ± 0.03
(NO_3^-/NO_2^-)-N	0.61 ± 0.08	0.51 ± 0.09	0.12 ± 0.01	0.20 ± 0.02
Total P	1.31 ± 0.10	1.07 ± 0.31	0.36 ± 0.02	0.69 ± 0.08
Total persulfate N	7.82 ± 1.14	4.17 ± 0.94	1.06 ± 0.08	2.32 ± 0.25
	------------------------- (CFU (100 mL)$^{-1}$) -------------------------			
Fecal coliforms	51831 ± 16360	2581 ± 1022	974 ± 232	1146 ± 327

Table 3. Differences between mean nutrient and bacteria concentrations for the sampling sites (Pasture – Reference) before and after fencing closed. P-values were derived from analysis with RIA software.

	Pre-fencing	Post-fencing	p-value
	-------------------- ($mg\ L^{-1}$) --------------------		
Soluble reactive P	0.804	0.014	< 0.001
NH_4^+-N	3.200	0.511	< 0.001
(NO_3^-/NO_2^-)-N	0.484	0.303	0.012
Total P	0.952	0.377	0.005
Total persulfate N	6.754	1.855	0.001
	--------------- ($CFU\ (100\ mL)^{-1}$) ---------------		
Fecal coliforms	50857	1435	0.088

Table 4. Calculated annual nutrient export from pasture site at baseflow conditions. Export from pasture site considered as change in respective nutrient concentration for stream water passing through site (i.e. Pasture – Reference).

	Pre-fencing	Post-fencing	Difference (Post – Pre)	% Change
	------------------ ($kg\ ha^{-1}\ yr^{-1}$) ------------------			
Soluble reactive P	18.9	5.4	- 13.5	- 71%
NH_4^+-N	706.4	273.4	- 433.0	- 61%
(NO_3^-/NO_2^-)-N	108.2	166.2	+ 58.0	+ 53%
Total P	209.7	203.7	- 6.0	- 2%
Total persulfate N	1492.3	991.8	- 500.5	- 34%

Table 5. Summary of aquatic invertebrate data and indices used.

	Reference reach		Pasture reach		Downstream reach	
	Pre-fencing	Post-fencing	Pre-fencing	Post-fencing	Pre-fencing	Post-fencing
Hilsenhoff FBI						
Score:	6.1	7.8	10.0	7.6	7.9	8.0
Rating:	Fair	Poor	Very poor	Poor	Poor	Poor
Fisher's alpha	3.17	2.57	0.97	3.37	3.09	3.51
Shannon's H'	1.87	1.26	0.36	1.65	0.64	1.62
EPT	0.14	0.00	0.00	0.02	0.01	0.00
Taxa per sample	11	8	5	15	12	11
Total sample abundance†	98 (33%)	55 (62%)	190 (92 %)	290 (49%)	146 (88%)	77 (46%)

† Numbers in parentheses represent the percentage of abundance represented by chironomids.

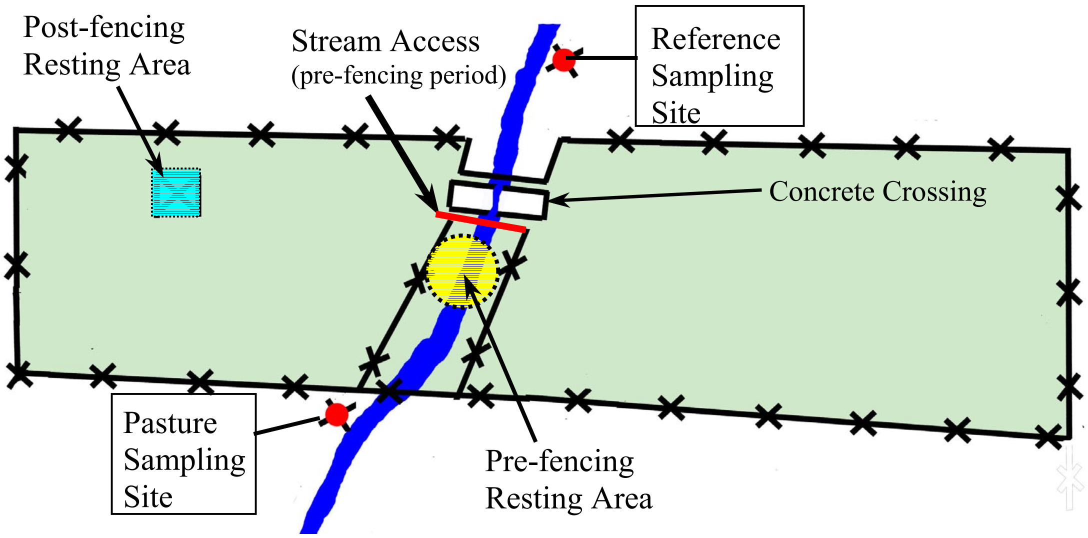

Figure 1. Map of study site including nutrient/bacteria sampling sites. Red line indicates pre-fencing stream access point that was closed in post-fencing period. Pre-fencing resting area indicated in yellow. Post-fencing resting area indicated in blue.

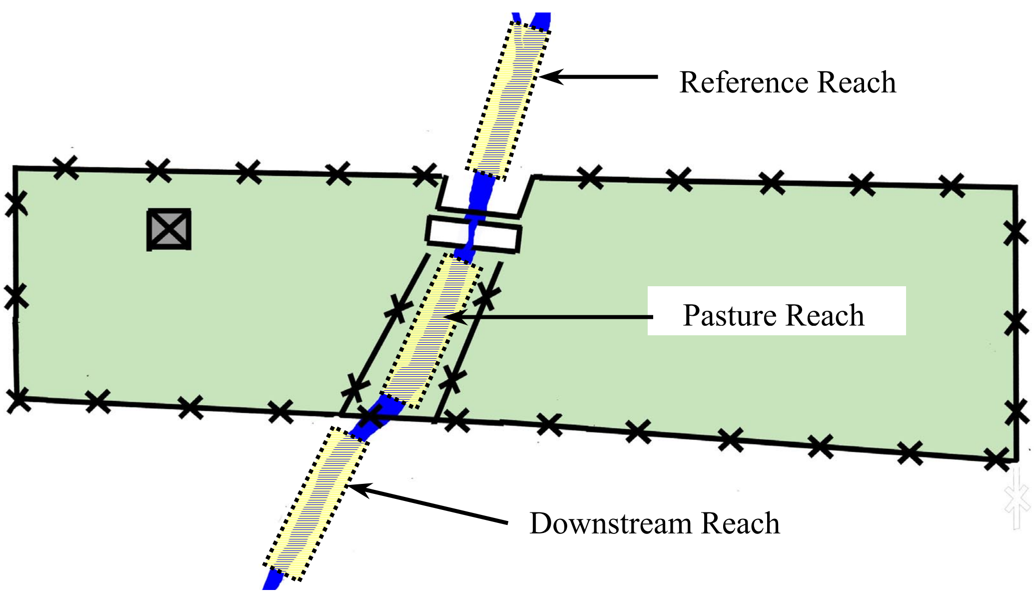

Figure 2. Map of reaches sampled for aquatic insects. Reach areas are shown in yellow.

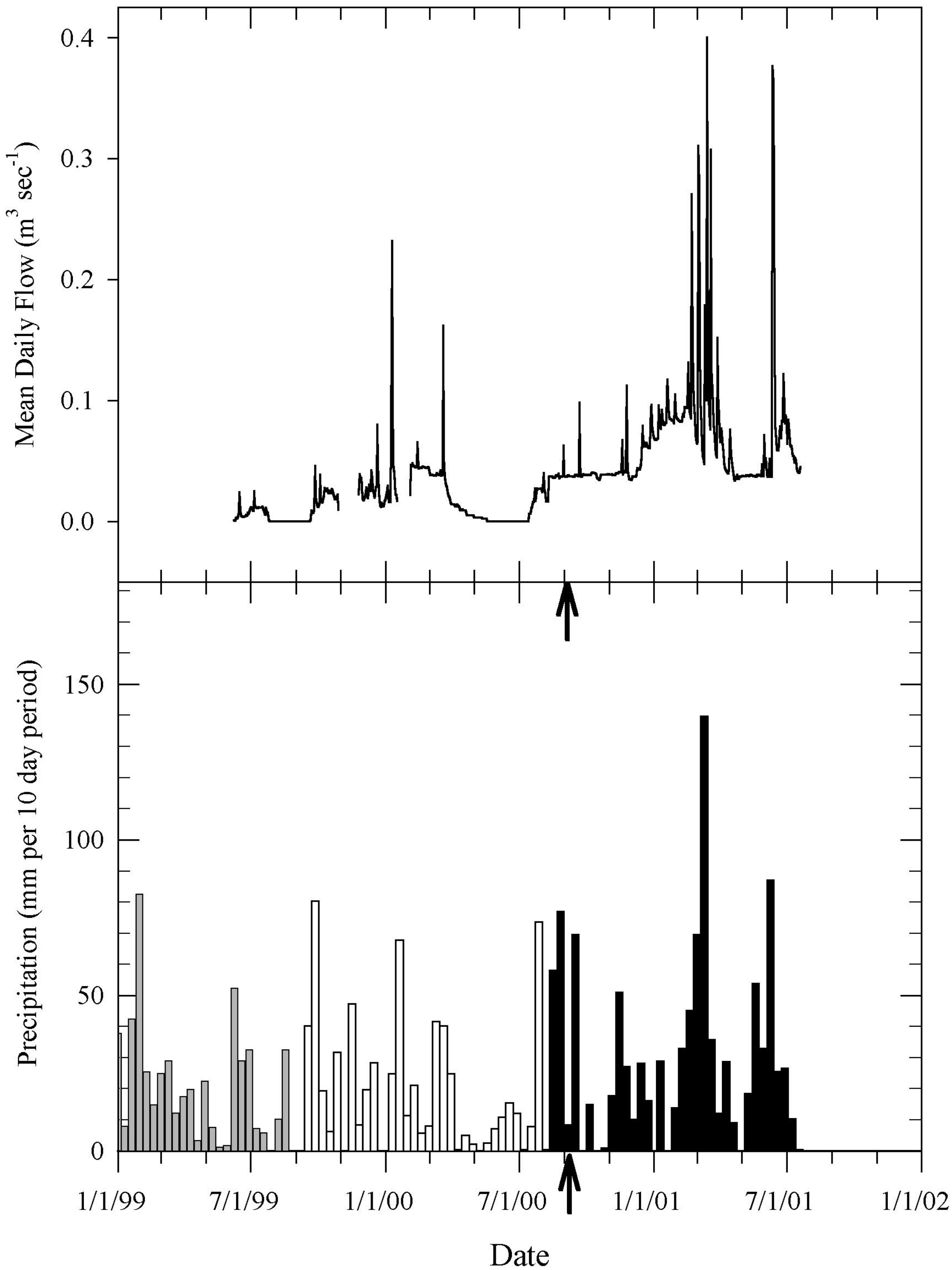

Figure 3. Precipitation and mean daily discharge at study site. Precipitations shown in 10 day intervals. Arrows indicate date fencing closed. White bars show precipitation for water year 9/99 to 9/00; black bars show precipitation for water year 9/00 to 9/01

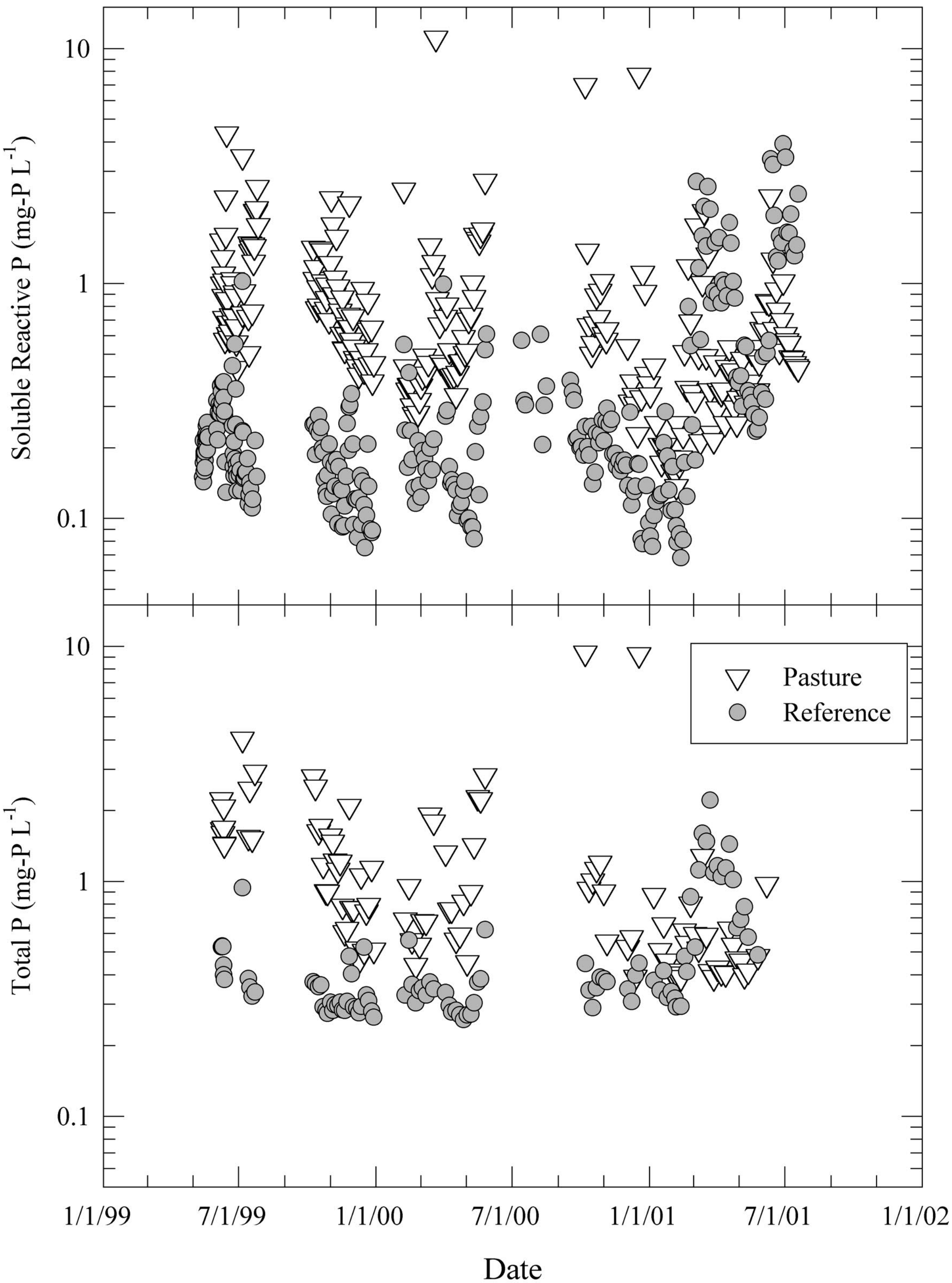

Figure 4. Soluble reactive P and total P vs. time. Arrows indicate date fencing closed.

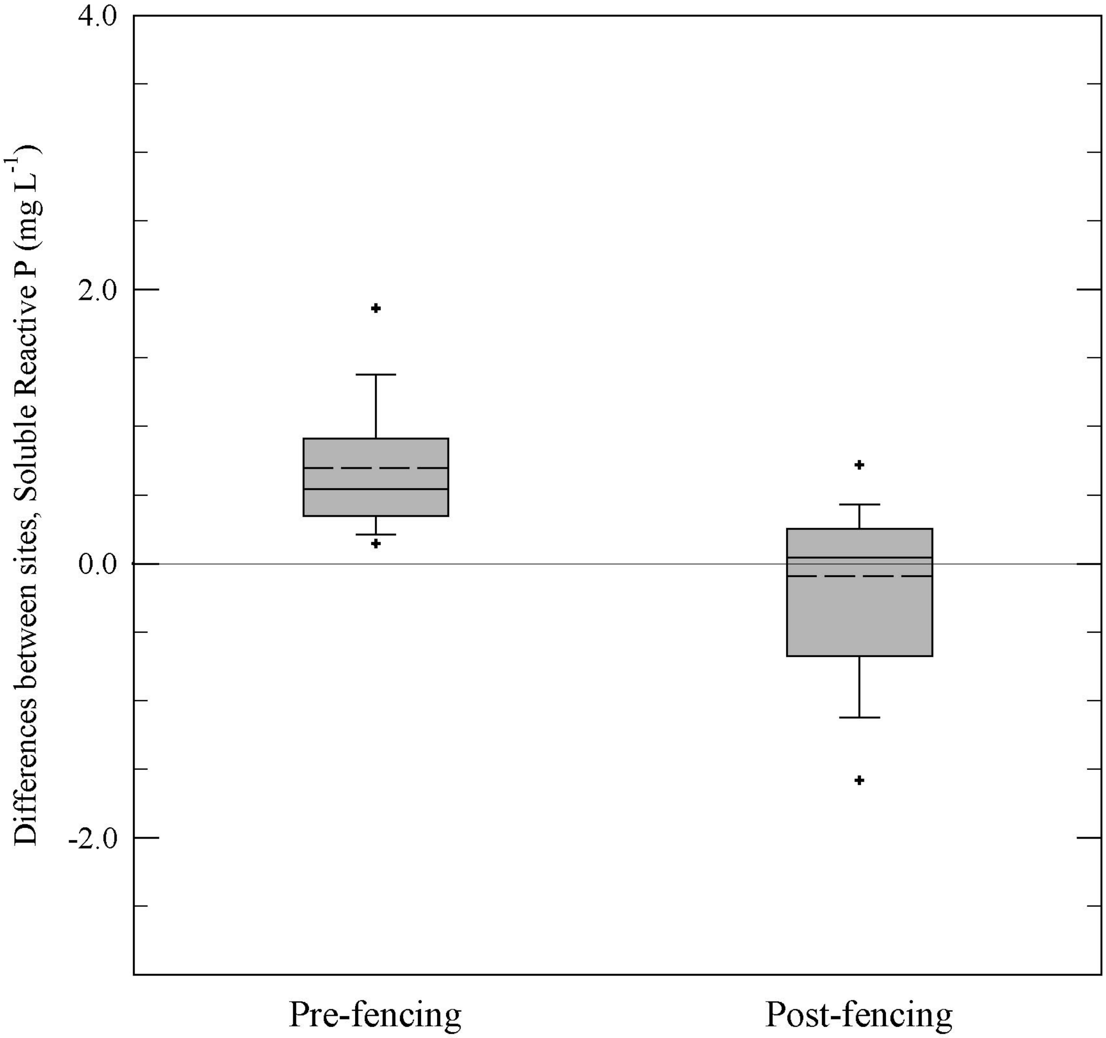

Figure 5. Box plots of differences in soluble reactive P between pasture and reference sites. Difference computed as (Pasture - Reference). Dashed line indicates mean of differences between sites.

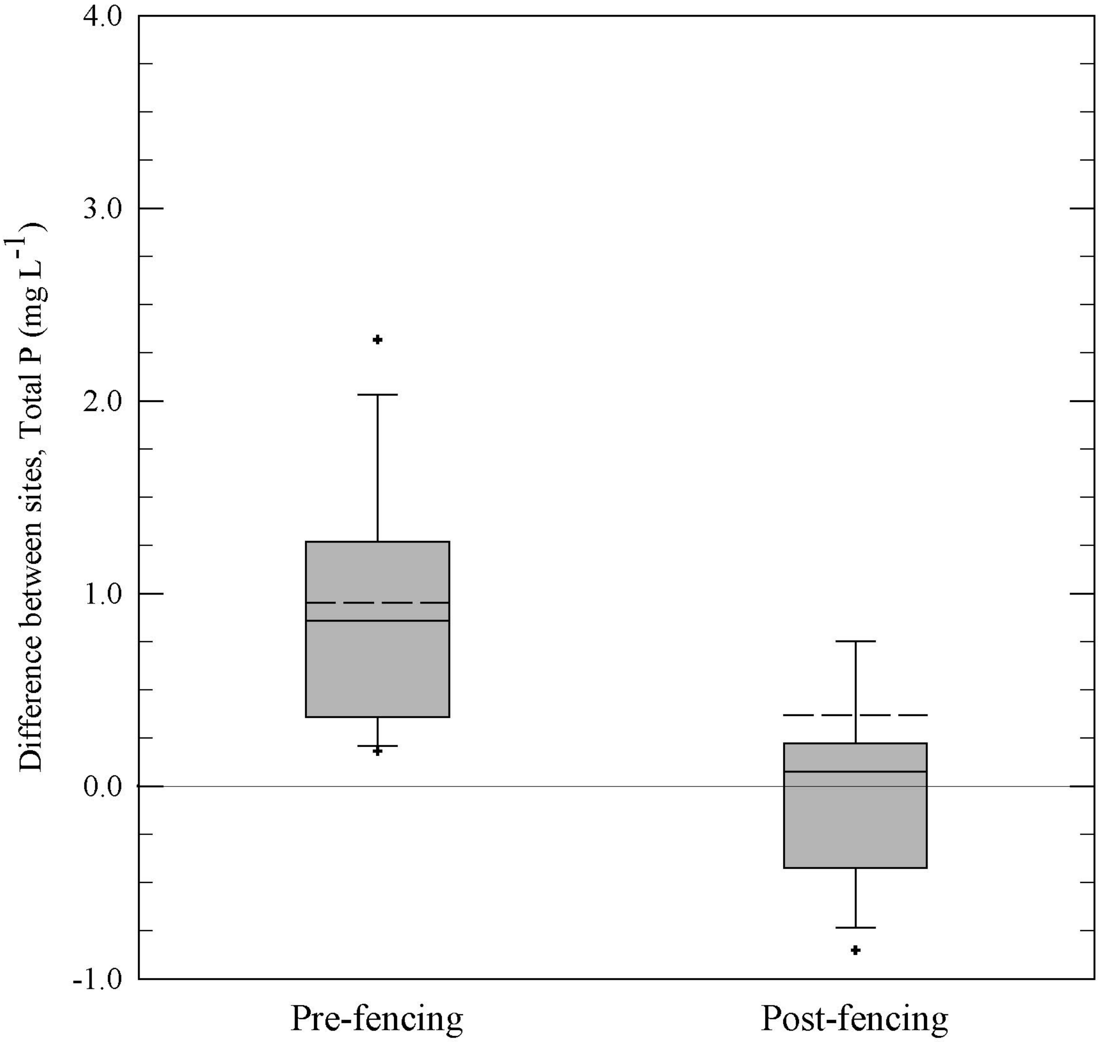

Figure 6. Box plots of differences in total P between pasture and reference sites. Difference computed as (Pasture - Reference). Dashed line indicates mean of differences between sites.

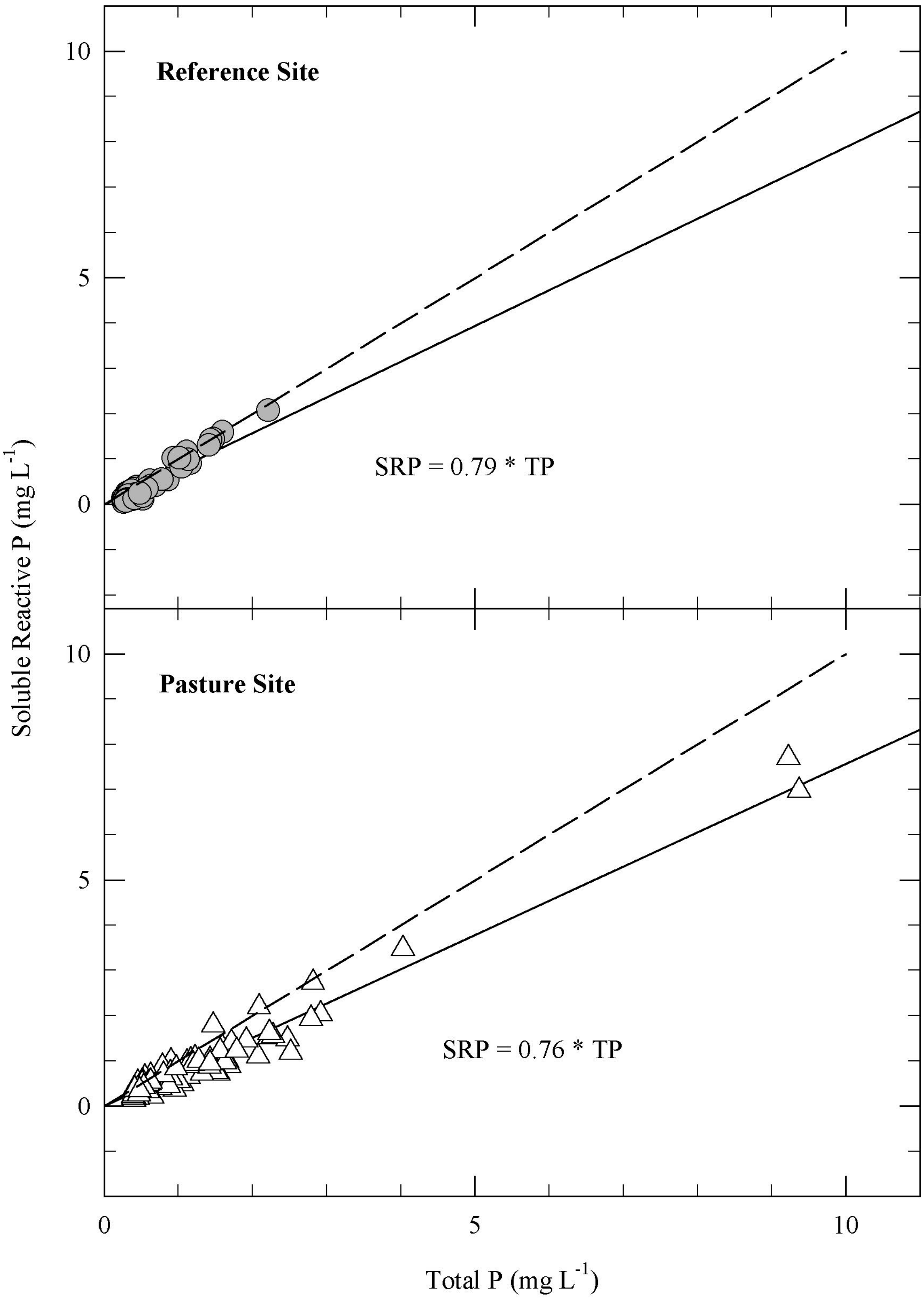

Figure 7. Soluble reactive P vs. total P for both sampling sites. Dashed lines show 1:1 line. Solid lines show regression lines.

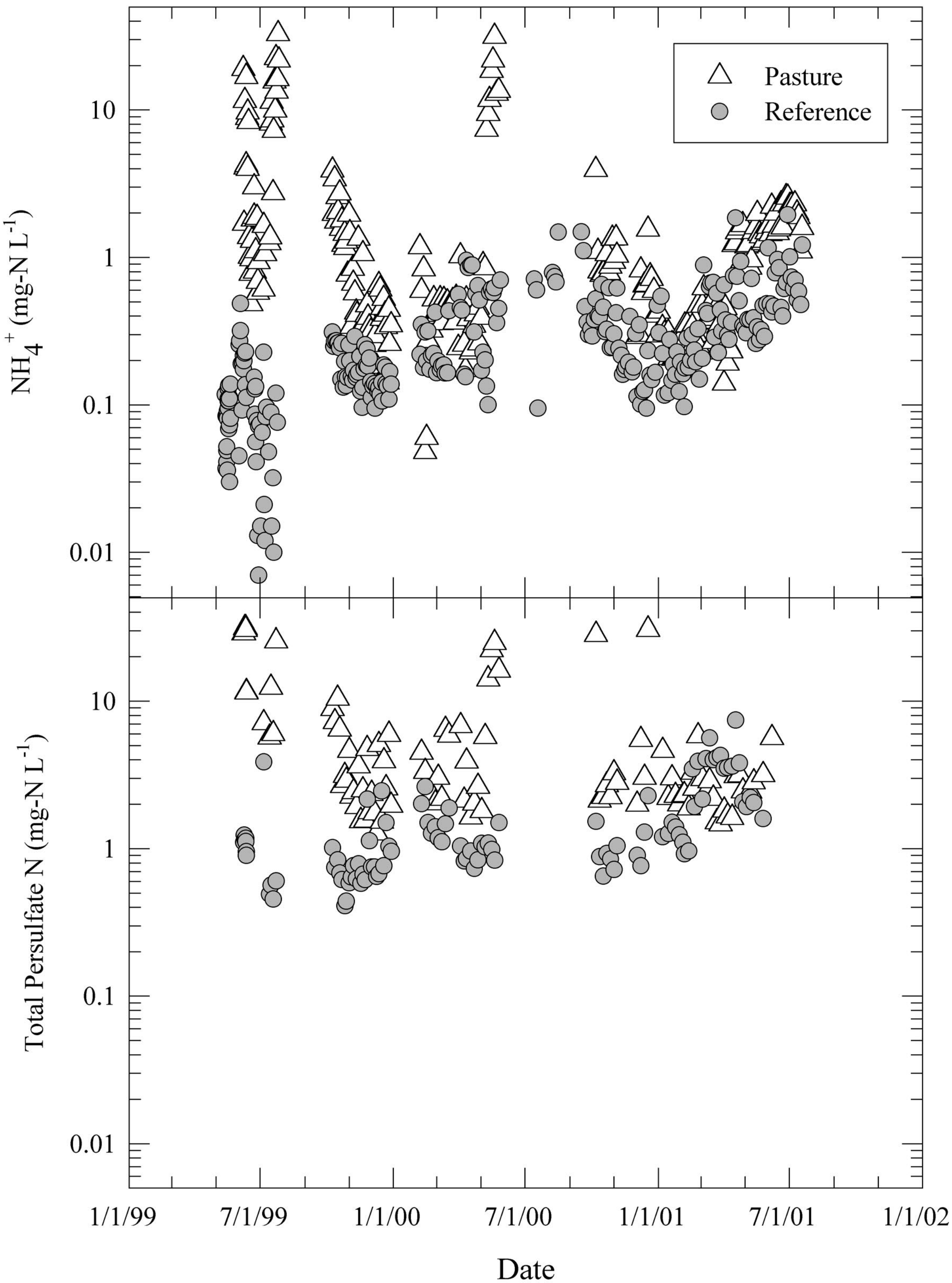

Figure 8. Ammonium and total persulfate N vs. time. Arrows indicate date fencing closed.

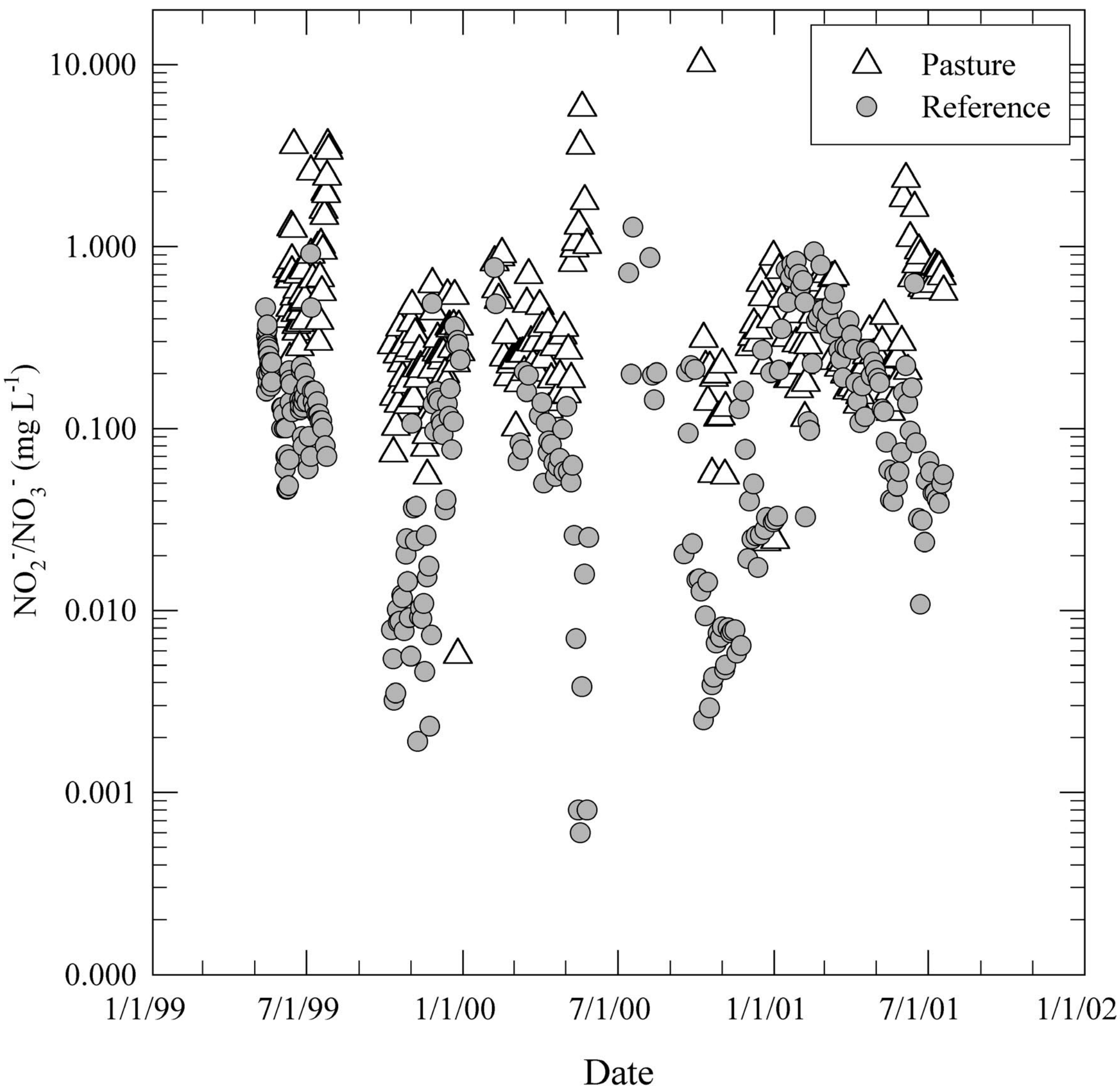

Figure 9. Nitrate/nitrite vs. time. Arrows indicate date fencing closed.

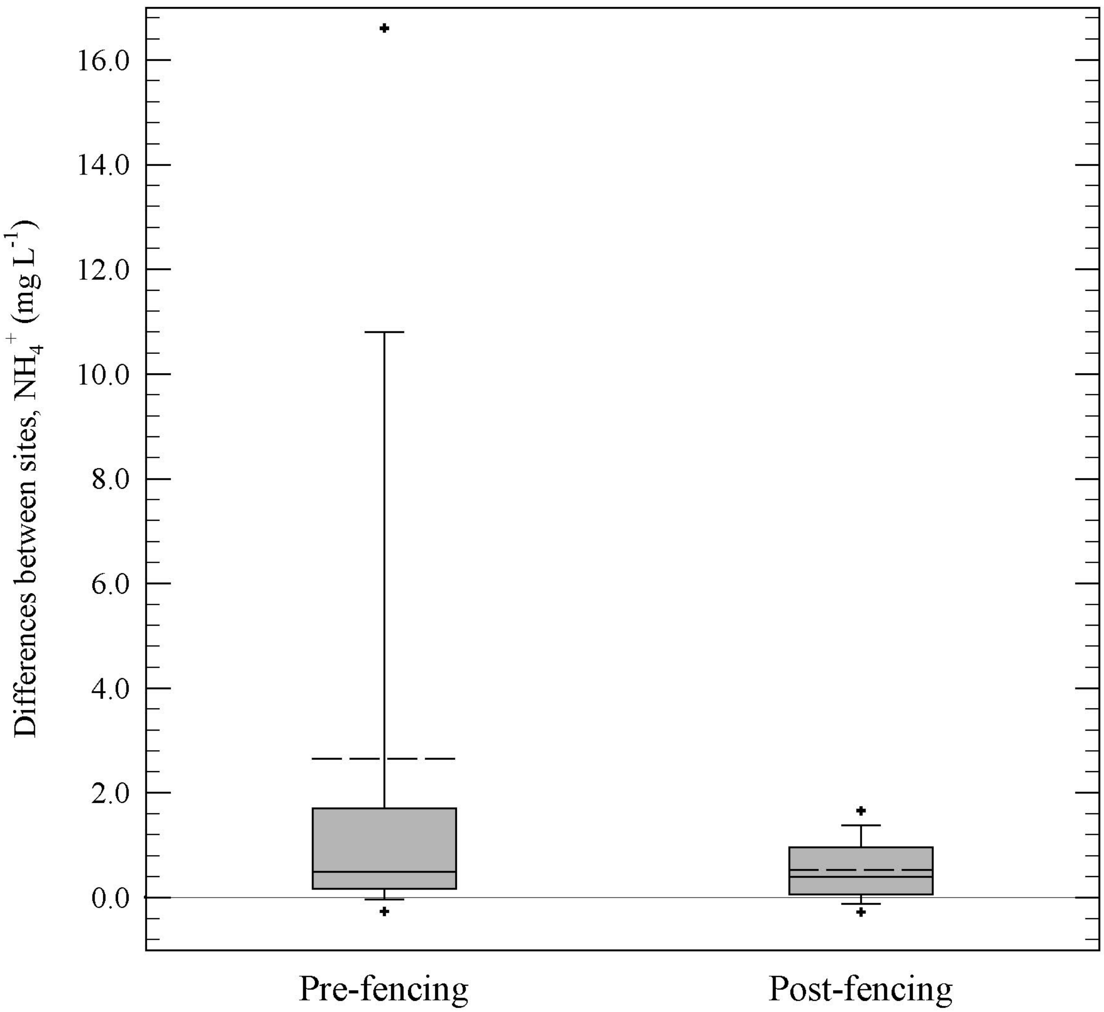

Figure 10. Box plots of differences in ammonium between pasture and reference sites. Difference computed as (Pasture - Reference). Dashed line indicates mean of differences between sites.

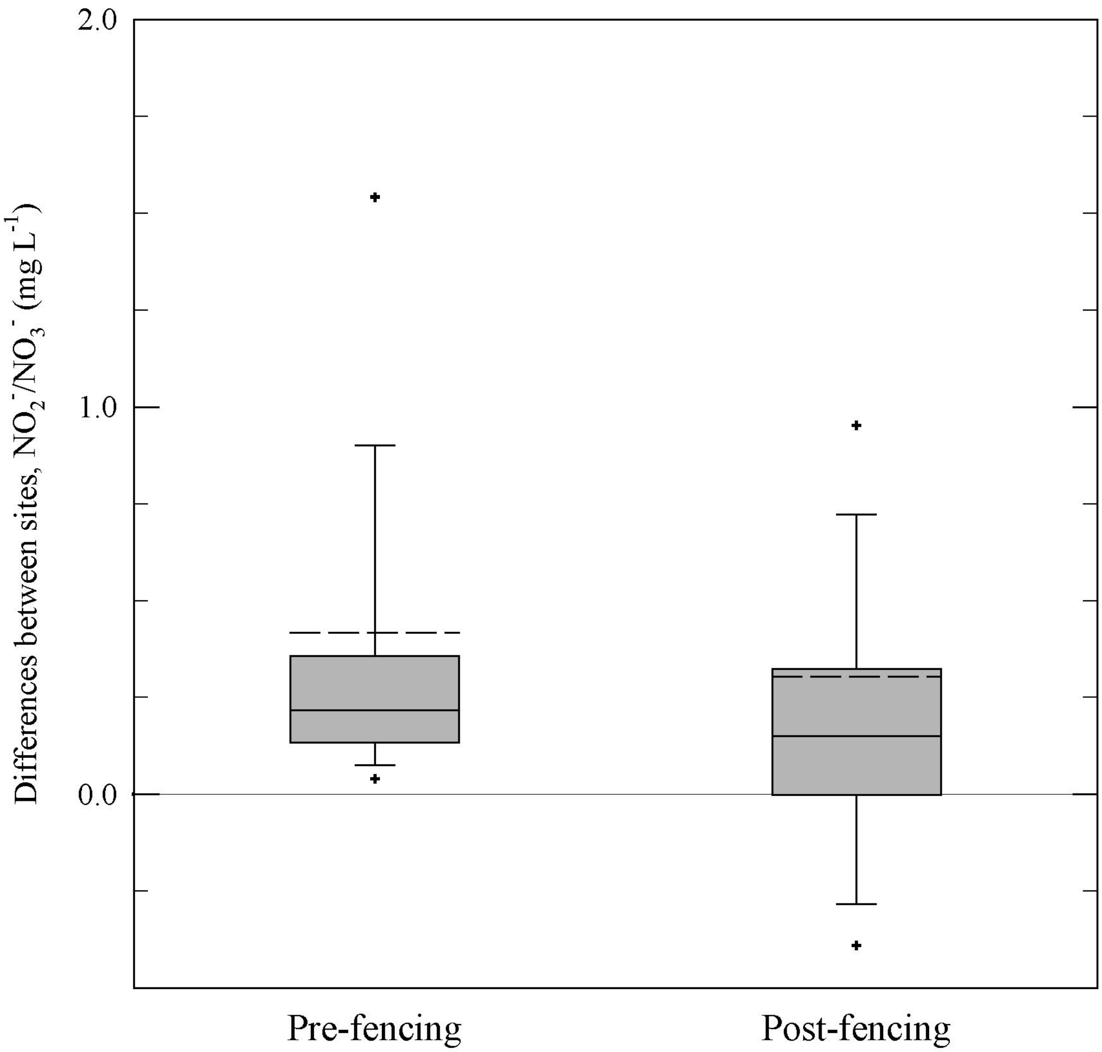

Figure 11. Box plots of differences in nitrate/nitrite between pasture and reference sites. Difference computed as (Pasture - Reference). Dashed line indicates mean of differences between sites.

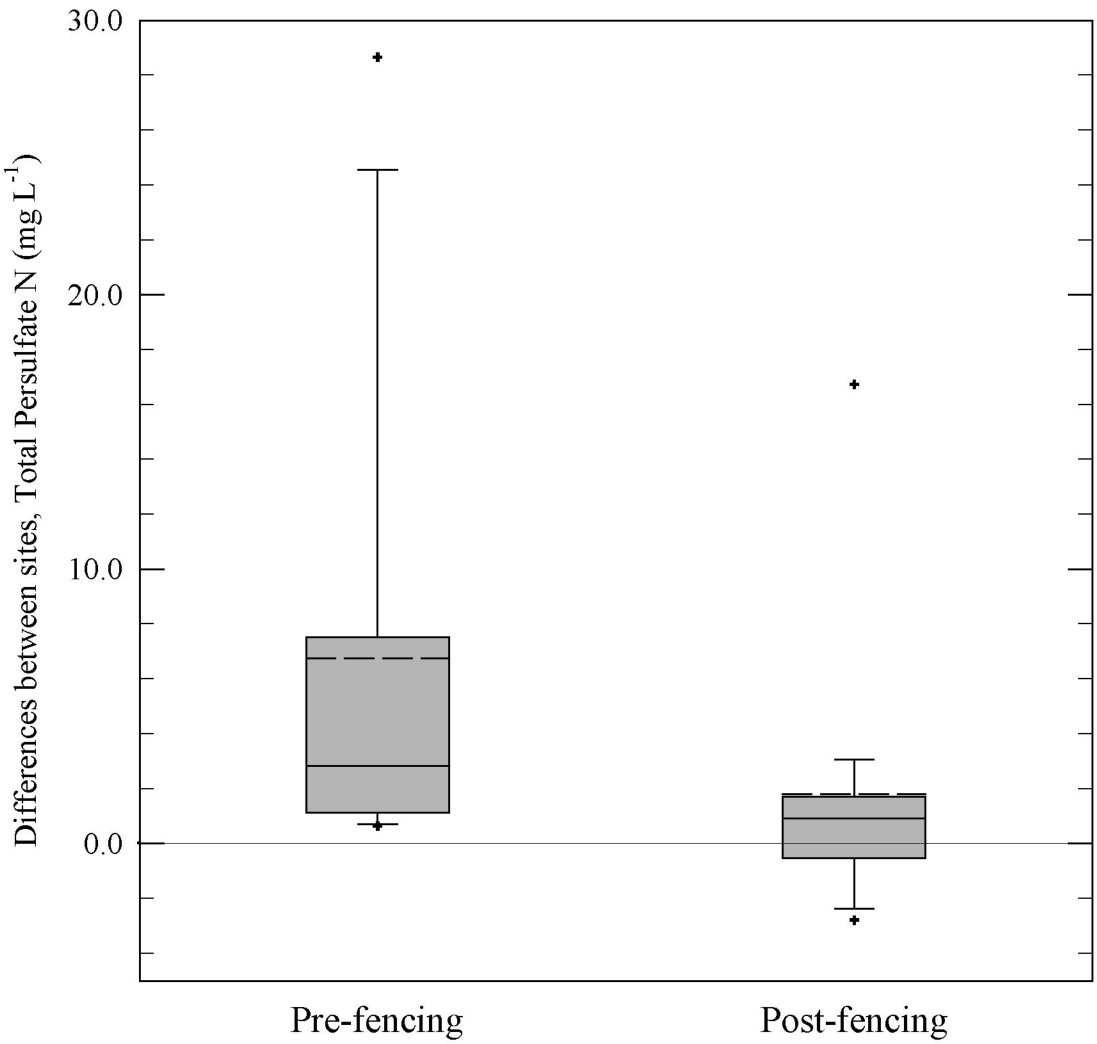

Figure 12. Box plots of differences in total persulfate N between pasture and reference sites. Difference computed as (Pasture - Reference). Dashed line indicates mean of differences between sites.

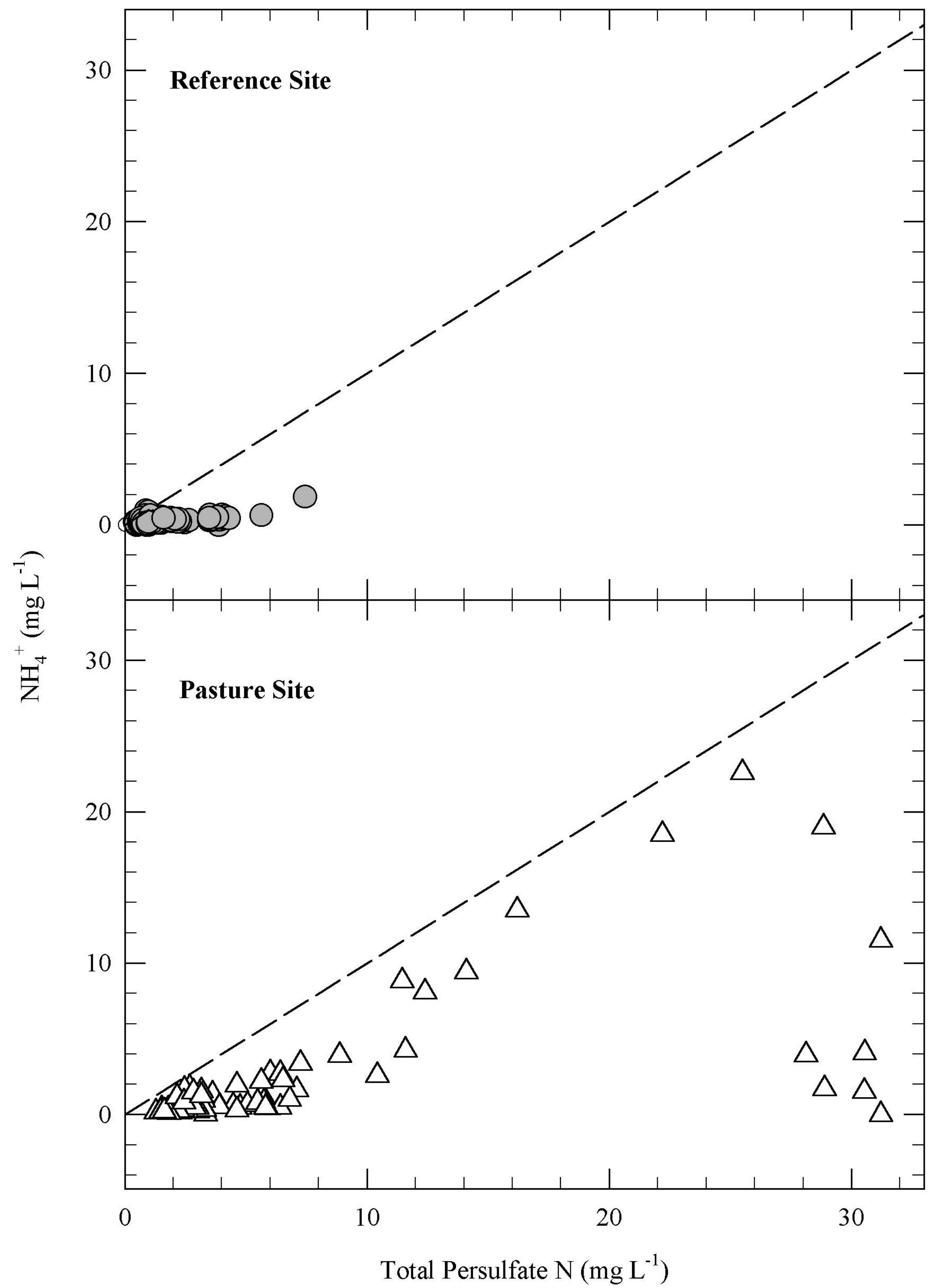

Figure 13. Ammonium vs. total persulfate N for both sampling sites. Dashed lines show 1:1 line.

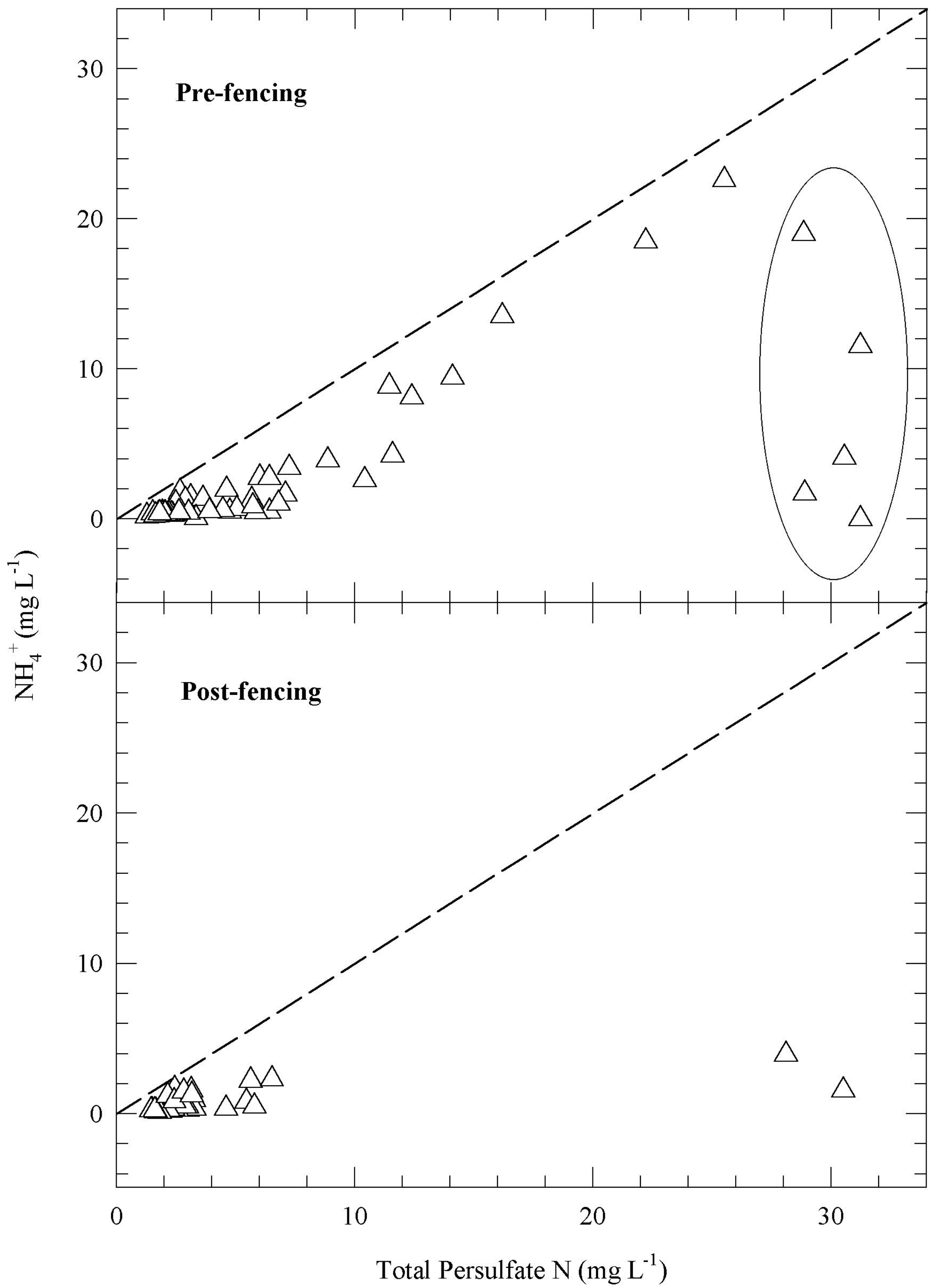

Figure 14. Ammonium vs. total persulfate N at pasture sampling site for pre- and post-fencing periods. Dashed lines show 1:1 line. Circled triangles represent samples taken at the beginning of the study within a four-day period (6/9/99 to 6/11/99).

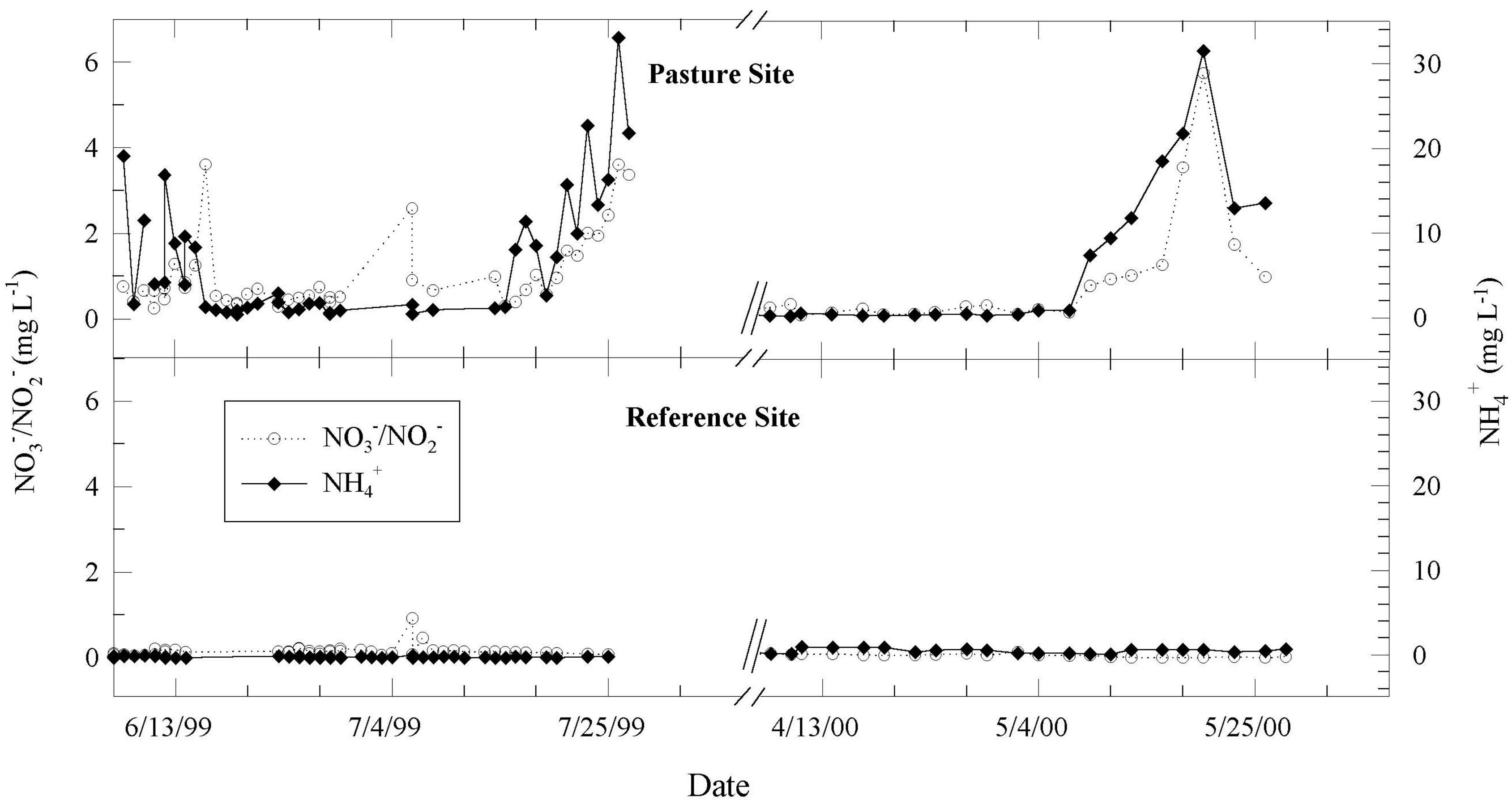

Figure 15. Ammonium and nitrate/nitrite vs. time for both sampling sites. Concentrations shown for 2 month period leading up to cessation of stream flow during pre-fencing drought years of 1999 and 2000.

Figure 16. Close-up view of oxygen bubbles generated by algal bloom in stream channel. Photo taken in March 2000 (pre-fencing period) during second drought year and prior to complete stagnation of stream flow. Indicated bubble is ~3 cm in diameter.

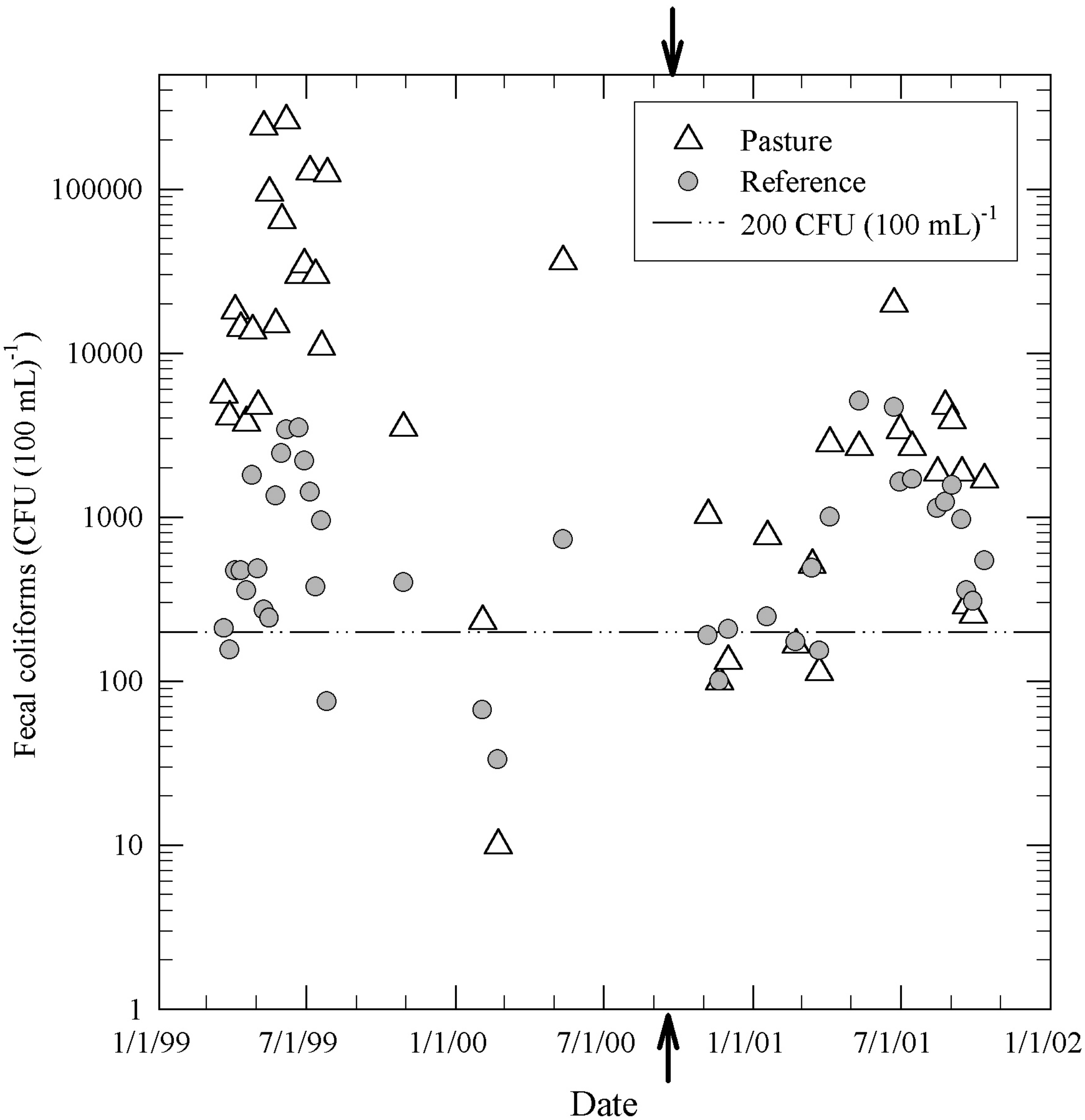

Figure 17. Fecal coliform concentrations vs. time. Arrows indicate date fencing closed. Dashed line indicates safety standard for primary human contact.

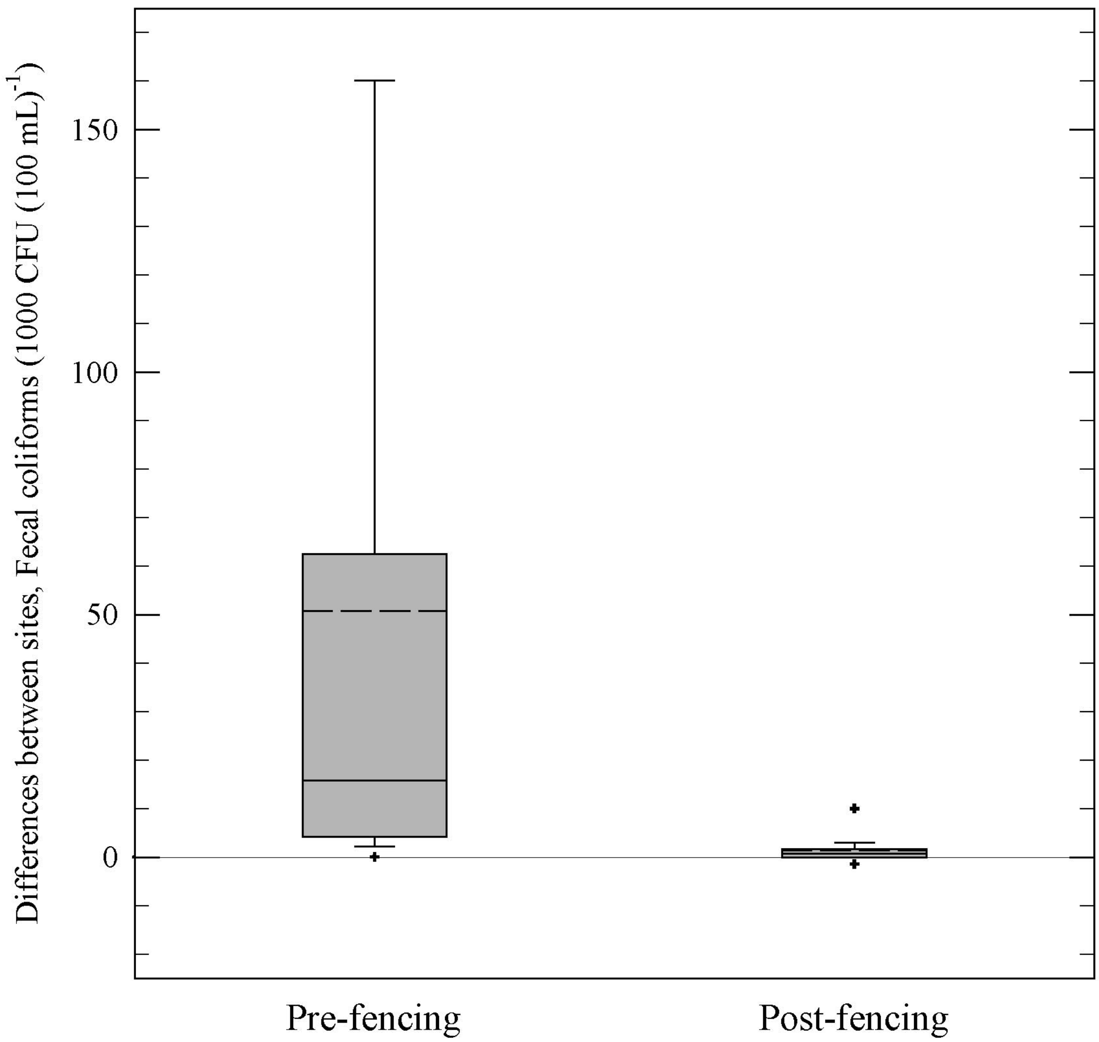

Figure 18. Box plots of differences in Fecal coliforms between pasture and reference sites. Difference computed as (Pasture - Reference). Dashed line indicates mean of differences between sites.

Figure 19. Pictures of the pasture site during and after the drought. Top photo was taken on 6/500. Bottom photo was taken one year later on 6/9/01.

Figure 20a. View of study stream within fenced area prior to closing fencing. Photo taken in May 2000.

Figure 20b. View of study stream within fenced area 4 months after fencing closed. Photo taken in Feb 2001.

Figure 20c. View of study stream within fenced area 9 months after fencing closed. Photo taken in June 2001.