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USING NATURAL AND LANDSCAPED BUFFERS TO REDUCE POLLUTANT LOADING FROM AGRICULTURAL RUNOFF

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ABSTRACT

The use of buffers to reduce the impact of agricultural runoff on surface water is a well established practice. We tested typical grassed and wooded buffers in a Piedmont location and grassed and landscaped buffers in a Coastal Plain location for pollutant removal from runoff. At the Coastal Plain site, runoff from fields growing a variety of crops was directed to four 10 m long by 5 m wide buffer strips, two of which had the lower 5 m planted in juniper with pine bark mulch. Water was collected above and below each plot and analyzed for soluble nutrients, total suspended solids (TSS), and 11 pesticides and metabolites. The Piedmont site had grassed or wooded buffers of 4 m or 8 m widths and received water from a 0.3 ha field planted in corn. Chlorpyrifos was applied as Lorsban 15G applied in-furrow at the rate of 2.3 kg a.i./ha. Atrazine and acetochlor were broadcast applied as a tank mix of Aatrex 4L and Harness 7 EC at the rates of 2.2 kg a.i./ha and 1.7 kg a.i./ha. Water samples were taken after 4 natural events and 4 artificial runoff events and analyzed for soluble nutrients, TSS, and the three pesticides applied. Our study results continued to support the use of buffers to reduce sediment loads in runoff. Reductions ranging from 40-60% for the natural events were recorded for the 4 m grass and both forested buffers, with the 8 m grass buffer averaging 90% reduction. There were no differences in sediment reduction between the 10 m grass and a 5 m grass + 5 m landscaped buffers, both reducing sediment loads by approximately 60%. We found no evidence that the buffers we examined had any effect on reducing concentrations of dissolved nutrients or pesticides passing through the buffer. In fact, the grass buffers at the Piedmont site were often sources of nutrients rather than sinks. The reductions in loading which occurred in the forested buffers was proportional to the reductions in runoff volume, which averaged 52% in the 4 m plot and 90% in the 8m plot. Antecedent soil moisture conditions clearly controlled this effect, as there were several dry periods of 2-3 weeks at the Piedmont site. Several of the highest flows at the Coastal Plain site exceeded our flume design and therefore were not included in our analyses. It also is important to note that our focus was strictly on runoff and not on loading reductions that may be associated with processes occurring within the soil profile or ground water table.

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SUMMARY AND CONCLUSIONS

The use of buffers to reduce the impact of agricultural runoff on surface water is a well established practice. Our study results continued to support the use of buffers to reduce sediment loads in runoff. Reduction averages of up to 90% were recorded for an 8 m grass buffer in the Piedmont, with the other buffers averaging 40-60% for natural events. At a Coastal Plain site with somewhat steeper slopes (5% vs. 3%), there were no differences in sediment reduction between the 10 m grass and a 5 m grass + 5 m landscaped buffers, which reduced sediment concentrations by approximately 60%. There was no evidence that the use of a landscaped buffer of low-growing woody plants and pine bark mulch improved pesticide or nutrient removal, as was our hypothesis. The contact time between the more degraded mulch layer, which presumably would have greater adsorption capacity, and runoff was apparently not sufficient to impact pesticide concentrations in the runoff. There was clear evidence of the formation of channels within the mulch after major runoff events. This in turn would greatly reduce the ability of the buffer to reduce pollutant loads. We found no evidence that the buffers we examined had any effect on the concentrations of dissolved nutrients or pesticides passing through the buffer, except possibly by increasing it. Reductions in total amounts of nutrients and pesticides leaving the wooded buffers occurred primarily as the result of infiltration and the resulting loss in runoff volume. Reductions in loading occurred especially under dry conditions when substantial amounts of runoff were absorbed in the buffer, and declined with subsequent rainfall events.

This study was designed to determine how the various buffers affected water quality under a variety of runoff conditions. The wooded buffers were more effective in reducing runoff volume and as a result reduced all constituents proportionately. The grassed buffers were much more effective in sediment retention when the width was doubled. The landscaped buffers were just as effective as the grass buffers in reducing sediment loads. We expected the pesticide concentrations to be reduced after passing though the landscaped buffer but this was not evident. However, the one most commonly detected pesticide, metolachlor, was significantly reduced in total amounts leaving the landscaped buffer compared to the grass only buffer.

There are several limitations to this study. The Piedmont study only had one year of data and half of the eight events were artificially induced runoff with relatively low volumes compared to natural events. The two-month period following planting at that site was unusually dry and hence the buffers may have had an unusually high infiltration rate for runoff. Overall, this site had a lower field:buffer ratio than would be found in most agricultural settings, so "normal" buffers in similar situations may not perform as well. The Clinton study excluded the four largest events due to overtopping of the plot flumes and thereby missed the most important events in terms of pollutant loading. It is important to note that our focus was primarily on runoff water quality and not on processes occurring within the soil profile or ground water table.

RECOMMENDATIONS

There are many reasons for establishing buffers around bodies of water. Sediment, the single greatest threat to surface water quality, can be greatly reduced when runoff is dispersed in a buffer. The effectiveness is directly related to the ability of the buffer to slow the flow of runoff enough to allow sediment to settle out. As has been reported in many previous studies, a grassed buffer on a gentle slope is very effective in removing sediment and is the best buffer for runoff from typical agricultural operations. Because of their ability to hold sediment, grass buffers also require periodic maintenance to remove the accumulated material. The 8 m width was much more effective than the 4 m width for most storm events, in large part due to increased infiltration.

The buffers we tested do not reduce the concentration of dissolved nutrients and pesticides in runoff. The process of adsorption at the soil-water interface is apparently ineffective in the time scale of typical runoff events. Many of the runoff events that occurred at the landscaped buffer site clearly exceeded the ability of the mulch to maintain sheet flow. Further work on design specifications for mulched buffers would be needed before these could be recommended for buffering runoff. The grass areas which received identical flows were much more resilient and recovered quickly.

Some reductions in loading can be achieved strictly through the process of infiltration in the buffer. However, many buffers are poorly maintained and flow occurs largely in channels, resulting in little effect on runoff. Our recommendation would be to minimize the occurrence of nutrients and pesticides in runoff water through various practices in the field, if possible, because these pollutants will be largely untreated as the runoff passes through a buffer.

Future research should focus on buffer performance over a period of many years to determine the average and range of pollutant attenuation expected. Numerous replications are needed to understand the range of responses across a landscape. It is also suggested that the flow measuring system should be able to handle even the largest runoff events, even if it means sacrificing measurements from the smallest events.

INTRODUCTION

Agricultural runoff has been widely recognized as a major pollutant source (Welsch, 1991; US EPA, 1995). One approach to mitigating these impacts has been the use of buffer zones of various designs between fields and surface waters (Fennessy and Cronk, 1997; Lowrance et al., 1997). These have been shown to be very effective in removing up to 90% of sediment, particularly the larger particle sizes (Tim and Jolly, 1994; Meyer et al., 1995; Cooper et al., 1987; Muscutt et al., 1993; Daniels and Gilliam, 1996). Nutrient removal is a more complex issue due to the varying behaviors of N and P depending on their form and association with sediment. Phosphorus associated with the sediment is dependent on sediment trapping for removal, but dissolved P may pass through a buffer (Muscutt et al., 1993; Daniels and Gilliam, 1996). Nitrate removal is primarily dependent on infiltration and subsequent denitrification and plant uptake (Lowrance, 1992; Jordan et al, 1993; Haycock and Pinay, 1993; Ambus and Lowrance, 1991; Gold and Groffman; 1995; Schultz et al., 1995; Verchot et al., 1997). Pesticide removal in buffers has been less well studied, but it appears that significant removal is possible with the right combination of pesticide properties and buffer design (Poletika et al., 1996; Mickelson and Baker, 1993; Asmussen et al., 1977; Rhode et al., 1980).

The effectiveness of a buffer zone is a function of several factors. Phillips (1989a, b) evaluated these factors using a mathematical approach and determined that buffer width was the most important factor for dissolved pollutants while slope and soil hydraulic conductivity affected suspended pollutant reduction the most. He also evaluated the most common soils in Carteret County, North Carolina, for buffer effectiveness and determined that some soils would require a buffer width of over 100 m to reduce flow rates. No experimental data was provided, however. Field and laboratory studies have also demonstrated that the effectiveness of buffer strips for pollutant reduction increases with width (Pearce et al., 1997; Magette et al., 1989; Parsons et al., 1994a,b; Mickelson and Baker, 1993; Dillaha et al., 1989).

The type and condition of vegetation in the buffer zone may also impact its pollutant reduction function. Grass has been demonstrated to be effective in removing sediment but less effective for N removal compared to riparian forests (Daniels and Gilliam, 1996; Verchot et al., 1997; Pearce et al., 1997). This has been attributed to the deeper root systems of forest species which provide for both uptake and carbon contributions to the denitrification process (Jacobs and Gilliam, 1985; Lowrance, 1992; Nelson et al., 1995; Haycock and Pinay, 1993). However, the nitrate removal rates beneath the grass strips were still significant. A graphical approach to determining buffer width for grass has been developed which uses NRCS hydrologic soil group and grass condition to estimate the required area for infiltration (Edwards et al., 1996). The goal in that approach is to achieve complete infiltration of received runoff. Grass height and aerification were not found to affect pesticide and nutrient removal in bermudagrass (*Cynodon dactylon* (L.) Pers.), but greater antecedent soil moisture reduced effectiveness (Cole et al. 1997).

Retention of nitrate flowing in shallow aquifers in riparian zones is often over 90% of input with widths as small as 5 m (Hill, 1996). Daniels and Gilliam (1996) found that the majority of sediment and chemical load was reduced in the first 6 m of a grassed buffer, with the exception of soluble P. A 4.6 m grassed buffer was as effective for sediment removal as a 9.1 m buffer, both removing more than 70% and most of the removal occurring in the first 1 m (Mickelson and Baker, 1993). Dillaha et al. (1989) reported pollutant reductions of 54-70% for 4.6 m grassed strips compared to 73-84% for 9.1 m strips, also suggesting that the majority of the reduction occurs in the first 5 m of the filter strip.

The concept and function of buffer zones to protect water quality has been well established. Riparian forests function well in removing nitrogen, but there are many situations where a strip of trees is impractical due to space and shading. An alternative is the use of low-growing woody ornamental species to provide the deeper rooting needed for a carbon source for denitrification. If effective, this type of buffer could be used in many areas where trees are not desirable, including areas adjacent to crop areas to avoid shading, and in urban environments such as golf courses, parks, or schools.

Sediment, nutrients, and pesticides can be washed into surface waters during storm events when runoff from agricultural areas is carried in ditches directly from field to the water body. Grassed and forested buffers have been demonstrated to remove 50-80% of sediment and nitrogen in agricultural runoff. In a review of pesticide removal by buffers, loads were reduced from as little as 11% to as much as 100% of input (U.S.D.A., 2000). However, much of the reduction occurred as the result of infiltration, not adsorption.

Forested buffers are not available in many situations, so we also examined landscaped buffers of low-growing, woody plants as an alternative. If effective, this type of buffer could be used in many areas where trees are not desirable, including areas adjacent to crop areas to avoid shading, and in urban environments such as golf courses, parks, or schools. This approach may provide the deeper rooting systems needed to enhance denitrification as has been demonstrated for forests compared to grass. Grass was also included as it is superior for sediment removal. The utility of various buffer systems for reducing non-point pollution were examined in the Piedmont and Coastal Plain regions. The results will be used in Extension programs and publications as recommendations for landscaping near surface water in the region.

This study evaluated constructed and natural buffer systems for effectiveness in runoff pollutant reduction in Piedmont and Coastal Plain agricultural settings. The objectives were:

- Test the effectiveness of a low-maintenance, woody ornamental landscape as a buffer for pesticide, nutrient, and sediment removal in the Coastal Plain.
- Determine the removal capacity for grass and riparian forest buffers for pesticides, nutrients, and sediment in runoff from Piedmont agricultural fields.

METHODS

LANDSCAPED BUFFER EXPERIMENT (COASTAL PLAIN)

This site was established at the Horticultural Crops Research Station in Clinton, NC. The station is used for research on the production of horticultural crops, primarily vegetables. The soils are moderately fine to coarse textured Coastal Plain sediments. Nearly all of the drainage from approximately 30 ha of production areas flows to a central irrigation pond. All runoff from the surrounding fields is directed into the pond in ditches or grassed waterways. Grass is maintained around the pond for a distance of 15-50 m with slopes of approximately 5%. The pond subsequently flows into Beaverdam Swamp.

We directed the flow of drainage from one section of approximately 3 ha into a 0.6 m H-flume (Fig. 1). Flow was measured and samples taken using an Isco 6700 sampler with integral bubbler flow meter. After passing through the flume, the runoff entered a shallow distribution box constructed of PVC plastic. Water exited the box through four 10 cm plastic pipes, each of which was level with the other. The pipes sloped down to the top of four plots. Across the top of each plot a 5 x 15 cm board was inserted in the ground until approximately 5 cm was above ground. The board was leveled to allow a relatively even distribution of runoff water across the top of the plots. Two plots remained in grass with two mowings per year to reduce perennial weed competition. The other two plots had a 5 m strip of grass immediately below the spreader, followed by a 5 m strip of ground cover of low-growing juniper established in late 1998 with shreaded pine bark mulching to a depth of 8-10 cm. The plots were separated by 12 cm plastic dividers inserted into the soil. The nominal ratio of source area to buffer area is 600:1.

Runoff was recollected at the bottom of each plot using plastic barriers to direct the flows through a 15 cm flume with integral stilling well. Automatic samplers (Isco 3700) were triggered to initiate sampling by the sampler in the upper flume, and took samples periodically based on time once triggered. Flow was estimated using the height of water in the flume as measured in the stilling well by a float attached to a potentiometer. Potentiometer readings were recorded by a data logger (Campbell Scientific CR10).

Shallow wells (1.5 m) were installed in the top and bottom of each plot, screened from 0.5 – 1.5 m. They were sampled periodically using a peristaltic pump (Isco 150) after recording the water depth and purging the well of three volumes. Water samples were placed on ice during the sampling procedures and stored under refrigeration on campus the same day. Total suspended solids (TSS), nitrate, ammonia, orthophosphate, and selected pesticides were determined at NCSU. TSS were determined by filtration and soluble nutrients were determined colorimetrically using a Lachat Autoanalyzer.

Pesticide analyses by gas chromatography/mass spectrometry (Hewlett-Packard G1800A - GCD) was conducted after solid phase extraction as described in McLaughlin and Johnson (1997). Because the drainage area included a wide variety of crops, we did not have a list of pesticides applied. We selected a variety of compounds which may be used in horticultural crop production. These are listed in Table 1.

Monitoring was conducted in both 1999 and 2000. The number of storm events included in our analysis changed with each parameter due to sampler failures or difficulties in retrieving the samples. Flow was recorded for many events in which one or more samplers failed, resulting in incomplete data sets for those events.

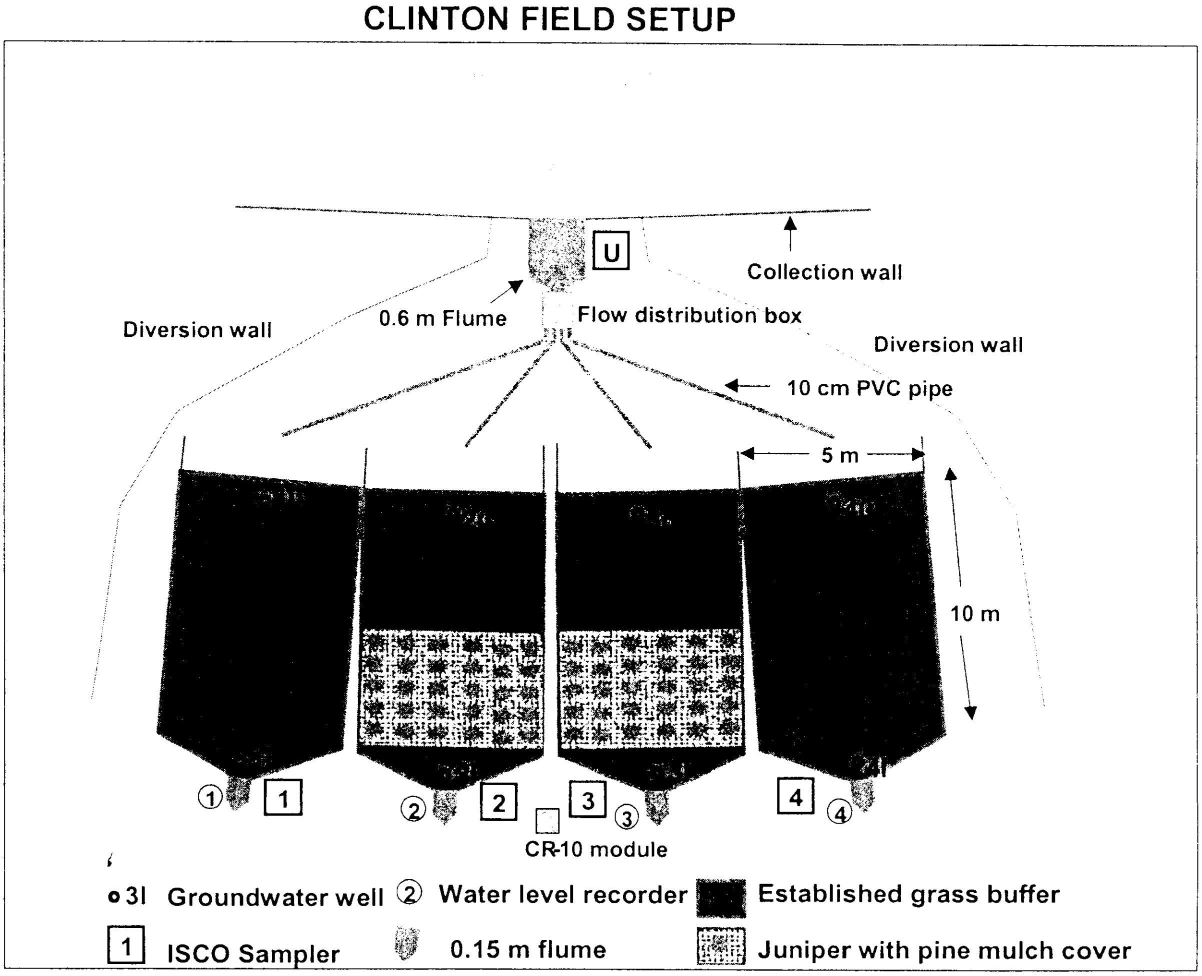


Figure 1. Design of the Landscaped Buffer site at the Horticultural Crops Research Station in Clinton, N.C.

Table 1. Pesticides and metabolites measured in this study.

Common Name	Chemical Formula	Use
atrazine	6-chloro-N-ethyl-N'-(1-methylethyl)-1,3,5-triazine-2,4-diamine	Herbicide
deisopropylatrazine	6-chloro-N-ethyl-1,3,5-triazine-2,4-diamine	atrazine metabolite
deethylatrazine	6-chloro-N-(1-methylethyl)-1,3,5-triazine-2,4-diamine	atrazine metabolite
acetochlor	2-chloro-N-(ethoxymethyl)-N-(2-ethyl-6-methylphenyl)acetamide	Herbicide
alachlor	2-chloro-N-(2,6-diethylphenyl)-N-(methoxymethyl)acetamide	Herbicide
ametryn	N-ethyl-N'-(1-methylethyl)-6-(methylthio)-1,3,5-triazine-2,4-diamine	Herbicide
metolachlor	2-chloro-N-(2-ethyl-6-methylphenyl)-N-(2-methoxy-1-methylethyl)acetamide	Herbicide
metalaxyl	N-(2,6-dimethylphenyl)-N-(methoxyacetyl)-DL-alanine methyl ester	Fungicide
chlorothalonil	Tetrachloroisophthalonitrile	Fungicide
triadimefon	1-(4-chlorophenoxy)-3,3,-dimethyl-1-(1H-1,2,4-triazol-1-yl)-2-butanone	Fungicide
triadimenol	*B*-(4-chlorophenoxy)-*a*-(1,1-dimethyethyl)-1H-1,2,4-triazole-1-ethanol	Triadimefon metabolite
chlorpyrifos	O,O-diethyl O-(3,5,6-trichloro-2-pyridinyl) phosphorothioate	Insecticide

GRASSED AND NATURAL RIPARIAN BUFFER EXPERIMENT (PIEDMONT)

The Piedmont site has been in use for riparian buffer research for nine years and has been the subject of several publications (Parsons et al., 1994a,b; Daniels and Gilliam, 1996). Details of the field setup can be found in Parsons et al. (1994a). It is located at the Lake

Wheeler Field Laboratory near Raleigh, NC. The site is typical for the Piedmont in that it has rolling topography with agricultural fields on the uplands and a wooded buffer along the streams. Buffer plots of either tall fescue (*Festuca arundinacea* Schreb.)) or natural woodland were established along a slope below a 0.4 ha field planted to corn (*Zea mays* L.) (Fig. 2). No runoff was recorded for several months after planting in 1999 so the study was repeated in 2000.

DeKalb 508 variety corn was planted at the rate of 64,000 seeds/ha on May 8, 2000, in furrows running up and down the slope. Chlorpyrifos was applied as Lorsban 15G applied in-furrow at the rate of 2.3 kg a.i./ha. Atrazine and acetochlor were broadcast applied as a tank mix of Aatrex 4L and Harness 7 EC at the rates of 2.2 kg a.i./ha and 1.7 kg a.i./ha. Starter fertilizer was applied as a dry blend at the rate of 27 kg N/ha, 36 kg P/ha, and 54 kg K/ha. No additional fertilizer was applied as the pesticide component of the system was our focus.

Runoff from the field was allowed to flow in either 4.3 m or 8.5 m grassed buffers or a gutter buried along the field edge. After passing through the buffers, runoff was also collected in buried gutters. The buffers were separated by 10 cm plastic barriers buried 5 cm into the ground. Buried pipes carried the water from the gutters downslope to two locations for flow measurement and sampling. Flow was measured either by means of a bubbler or a float/potentiometer system as described above. Sampling was accomplished by an array of automatic samplers. For the riparian plots, the runoff collected at the edge of the field was released into the top of the plots and recollected for flow measurements and sampling. The riparian plots were 1.3 m wide by either 4.3 or 8.5 m, each surrounded by a 10 cm metal barrier buried 5 cm to prevent water from entering or leaving the plots. There were two replications of each width for the grassed buffers but only one for the riparian buffers. The approximate ratios of source:buffer areas are 4.5:1, 9:1, 13.5:1, and 27:1 for the 4 m grass, 8 m grass, 4 m riparian, and 8 m riparian buffers, respectively.

There were no runoff events following planting and pesticide application in 1999. Rainfall during the period from May 1 until July 10 was less than 5 cm. We did not attempt to collect data after that point because we assumed a large portion of the applied pesticides had dissipated or leached far enough to be unavailable for runoff. Prior to planting and pesticide application in 2000, we generated an artificial runoff event in order to obtain background values for nutrients and pesticides. Following planting and pesticide application in early May 2000, there was no rainfall for nearly two weeks, and the first rainfall did not generate runoff. Because of our experience in 1999, we decided to generate artificial runoff in order to compare the buffer plots for pesticide removal. No irrigation lines are available at the site, so we brought in two nurse tanks and pumps to provide water.

The tanks held a total of 6800 L of water drawn from a well. The water was pumped into a system of hoses and oscillating lawn sprinklers to generate runoff from an area just uphill from plots. We irrigated an area of approximately 10 x 10 m of the field above three of six plots until runoff occurred. The upper 3 m of the grassed buffers was irrigated simultaneously. After a sufficient number of samples were taken, the sprinklers were moved to a similar area above the remaining three plots. We were not trying to simulate a storm event but merely attempting to generate runoff into the buffers for comparison. This artificial runoff testing was conducted four times prior to the occurrence of natural runoff events in June.

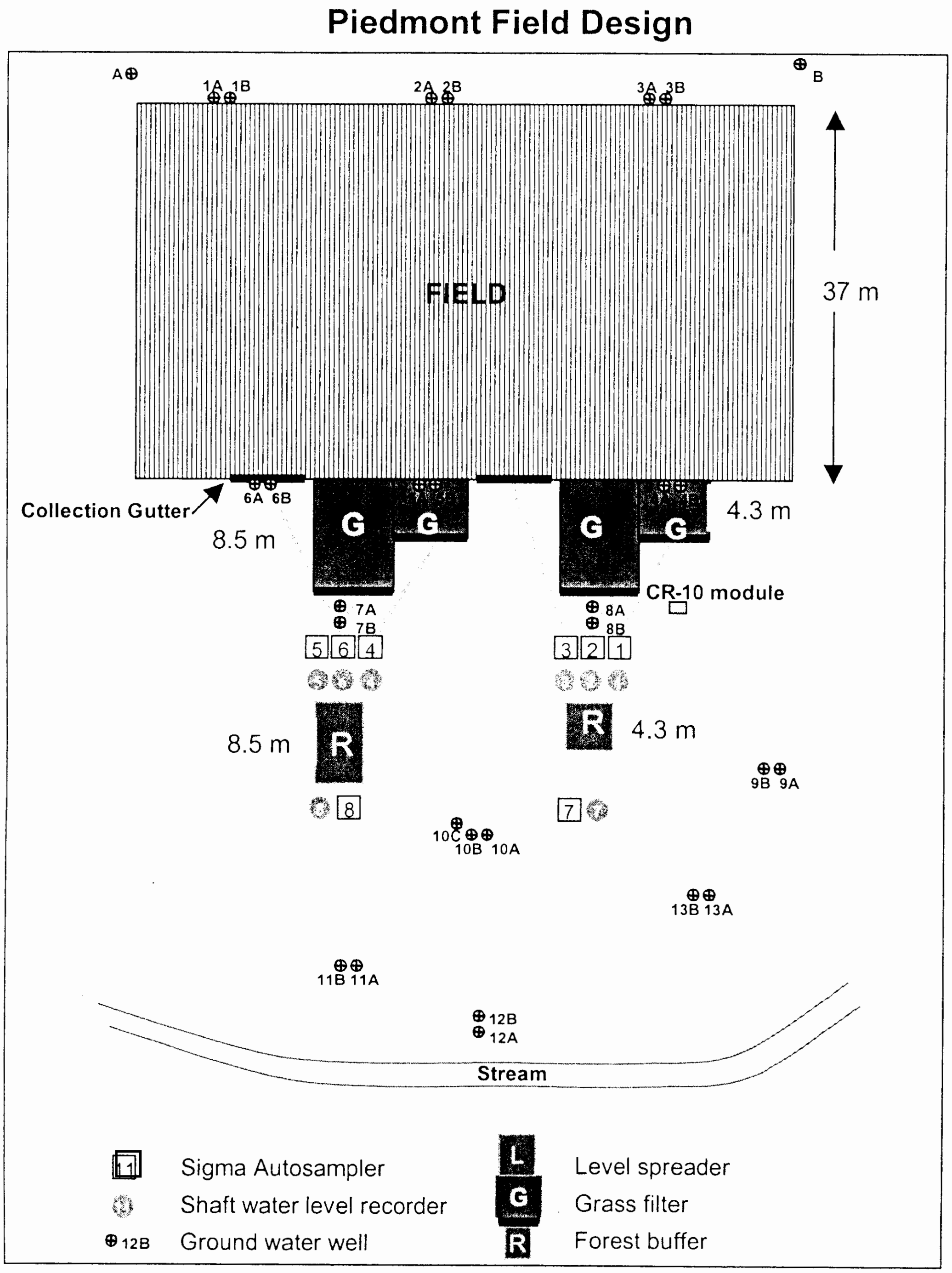


Figure 2. Design of the Grassed and Natural Riparian Buffer plots at the Lake Wheeler Road Field Laboratory in Raleigh, N.C.

Rainfall data was collected at the Lake Wheeler Road Field Laboratory office approximately 1 km east of the field site. Hourly data were not available for May and June 2000 due to an equipment failure, but daily rainfall was recorded manually. Analyses were conducted as described for the Coastal Plain site, except that only the applied pesticides were included in the laboratory analysis.

STATISTICAL ANALYSIS

The change in volume or concentration for runoff, nutrients, and pesticides was estimated by dividing the observations recorded at the outlet of each buffer plot by the observations recorded at the inlet of each buffer plot. Low values (< 1) of delivery ratio indicate the degree of effectiveness of the buffer plots in reducing the flow or pollutant concentration.

Descriptive statistical procedures were used to summarize the distribution characteristics of the delivery ratio. The distribution characteristics were examined by estimating their percentile values and coefficient of variations (CV). In this study, we first compared the plots by examining 50th percentile (second quartile or median) of observations. The 50th percentile is the value of the data such that 50% of the observations fall at or below it. The advantage of examining the median value over the arithmetic mean is that the median value is not affected by the outlier high values and hence it is more representative of expected values for a small sample size.

We compared the effect of the treatments on volume or concentration by performing univariate analysis at 5% significance level. The difference between the two treatments was accepted when the value of the significant level (p) was less than 0.05. The acceptance was further checked by estimating the coefficient of variation (CV - the ratio of standard deviation divided by mean) values. Higher CV values (> 1) suggests high variability in the observations. A study of the variability of data is important to compare the effectiveness of the plots, because the two plots could predict the same result in an average even though they are different naturally.

Total mass changes after the runoff passed through the buffers was estimated for events which had sufficient flow data and samples for analysis to characterize an event. Among the field and four plot sampling points, this was between 5-7 events at the landscaped buffer site in 2000. One-way analysis of variance was used to determine treatment effects on runoff volume and pollutant mass in the volume. For the grass and riparian buffer site, a series of paired t-tests were performed among the treatments. This allowed a comparison of the total mass passing through each treatment by storm or artificial runoff event. This was the most effective approach to the data as there were many events in which one or more plots did not produce samples or had an equipment malfunction, producing an incomplete set of data. The field and grass buffers were replicated and so averages are compared. The two forested buffers are compared to the data from the field replicate from which they received their runoff. Up to eight events were available for these comparisons, as indicated in Table 11.

RESULTS

LANDSCAPED BUFFER SITE

Runoff

During the course of early 1999, 10 runoff events were recorded with sufficient flow in all plots to obtain flow and runoff estimates (Fig. 3). Peak flow rates from the field over 2 m^3/min overtop our flumes, and this occurred twice on June 15 and 16. An additional 15 events were recorded in 2000, with the highest peak flow rate recorded for the study occurring on June 3 at over 5 m^3/min. Both the June 3 and 4 events exceeded the capacity of the plot flumes. An example of the characteristic pattern of a steady flow rate for the plot flumes while the upper flume flow was rising and falling is given in Figure 4 for the June 4 event. Clearly the plot flumes were at maximum capacity and were being overtopped. An example of a pattern of all flumes rising and falling together is shown for the March 16, 2000, event (Fig. 5). After eliminating the events in which the plot flumes were overtopped, flows from the plots ranged up to 0.42 m^3/min.

There were several events when runoff occurred as a result of irrigation instead of rainfall. We did not monitor when or how much irrigation was applied, but merely recorded the runoff as it occurred. Many irrigation events generated enough runoff to trigger the upper flume but not enough to produce runoff from all of the plots, so these events were not included in our analysis.

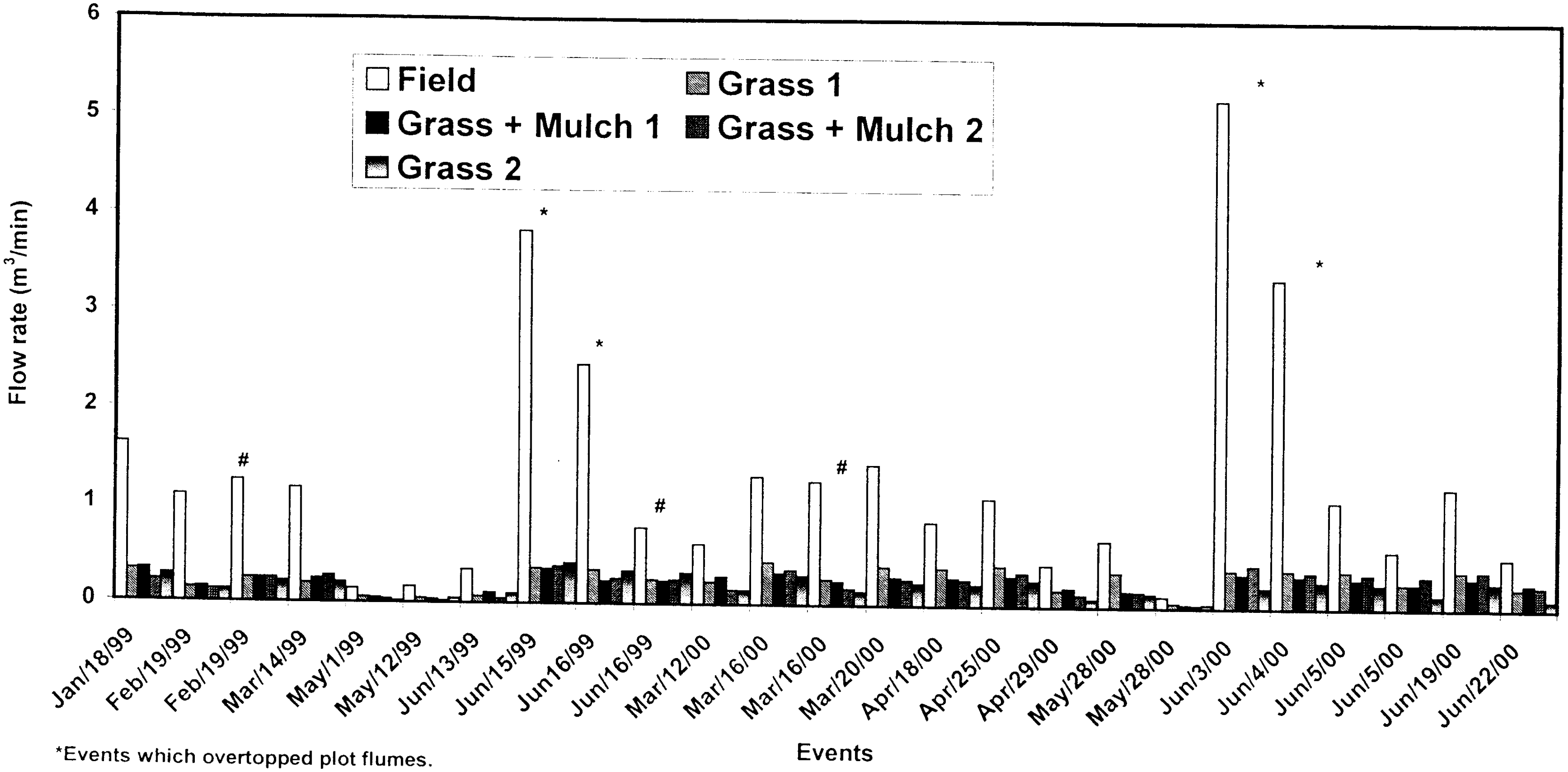
A gradient of soil moisture levels was evident across the four plots. The Grass 1 plot was on the wet end of the gradient and the Grass 2 plot was on the dry end. As a result, there was a predictable difference in the ability of the plots to retain runoff. This is illustrated by graphing the net loss of runoff water (infiltration) or the net gain (seepage). During 18 events where all four plots generated runoff data, the Grass 1 plot gained runoff in 10 events while Grass 2 lost runoff water in all but one (Fig. 6). The seepage was particularly evident on June 5, 2000, after the site had received heavy rains on June 3 (58 mm) and 4 (25 mm). Some of the net gain was the result of rainfall on the plots, but in most cases this accounted for less than 20% of the total flow.

The ground water monitoring wells had a gradient of about 0.3 m across the plots (Fig. 7). Water levels in the wells at the bottom of the plots were above ground level at times in the two wet plots, indicating that ground water was seeping out of the soil at that time. The gradient was present in the wells at the upper end of the plots as well, but water levels remained below ground level at all sample times (Fig. 8)

The median delivery ratio of water in the plots was also impacted by the relative wetness of each plot. Both the Grass 1 and the Grass + Mulch 1 plots delivered more water in runoff than entered the plots, while the Grass 2 and Grass + Mulch 2 plots removed water (Table 2). The end result was that the average delivery ratio for each treatment is approximately the same and very close to one. Only the Grass 2 plot, which was the driest, had evidence of removing runoff through infiltration with a delivery ratio of 0.75.

Sediment

The grassed plots reduced total suspended solids by an average of 52% compared to only 63% in the grass + mulch plots (Table 2). This difference was not statistically significant (Table 3) due to the high variability among events. Although the distribution of flows among the two years is similar (Fig. 3), the plots were much more effective at TSS removal during the second year. For the two events recorded for all four plots in 1999, TSS was higher in runoff from the grass + mulch plots than in the upper flume. This was possibly due to erosion occurring in the channels that were observed. There did not appear to be any relationship between flow and TSS concentrations, with correlations of 0.06 to –0.44 for the four plots. The reduction in sediment concentrations for both buffer types was evident for most runoff events in 2000 (Fig 9). TSS was usually highest in the first few samples and declined over the course of a runoff event, similar to the April 25, 2000, event shown in Figure 10.



*Events which overtopped plot flumes.

#Flows occurred at different times of the day.

Figure 3. Peak flow rates for the field and buffer plots at the Landscaped Buffer site

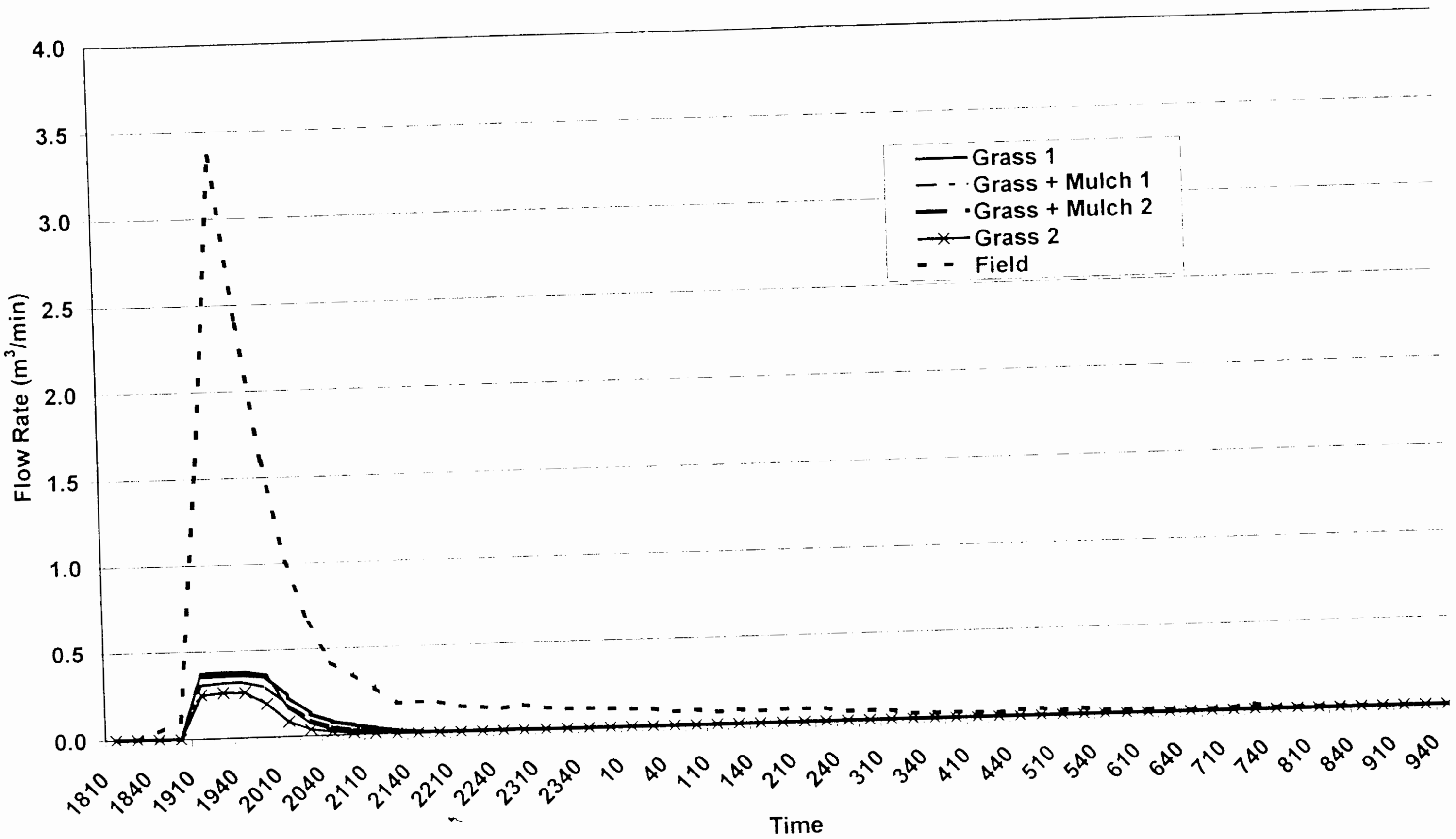


Figure 4. Sample hydrograph for a runoff event which overtopped the plot flumes at the Landscaped Buffer site. Event occurred on June 4, 2000.

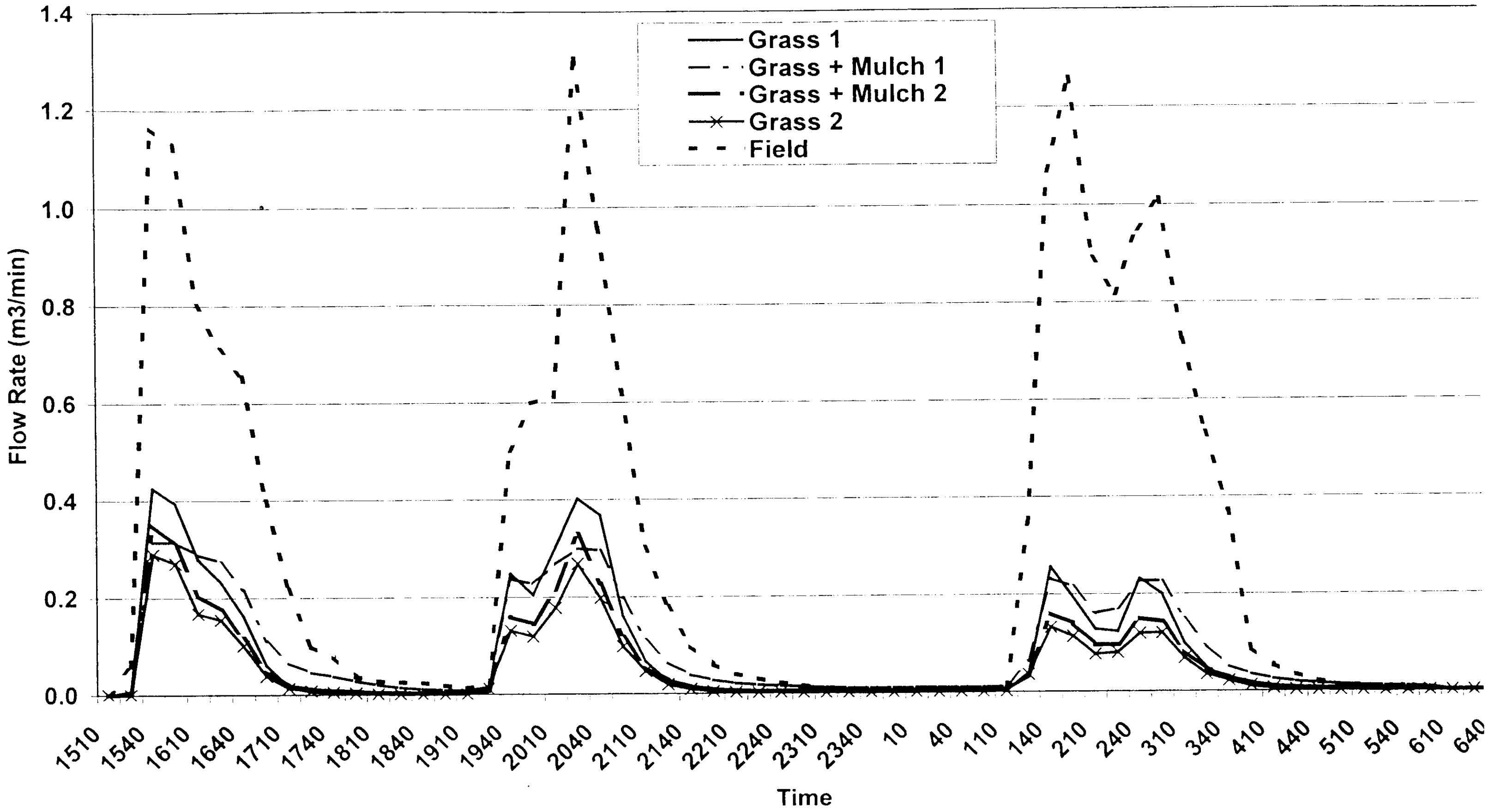


Figure 5. Sample hydrograph for a runoff event within the range of the plot flumes at the Landscaped Buffer site. Event occurred on March 16, 2000.

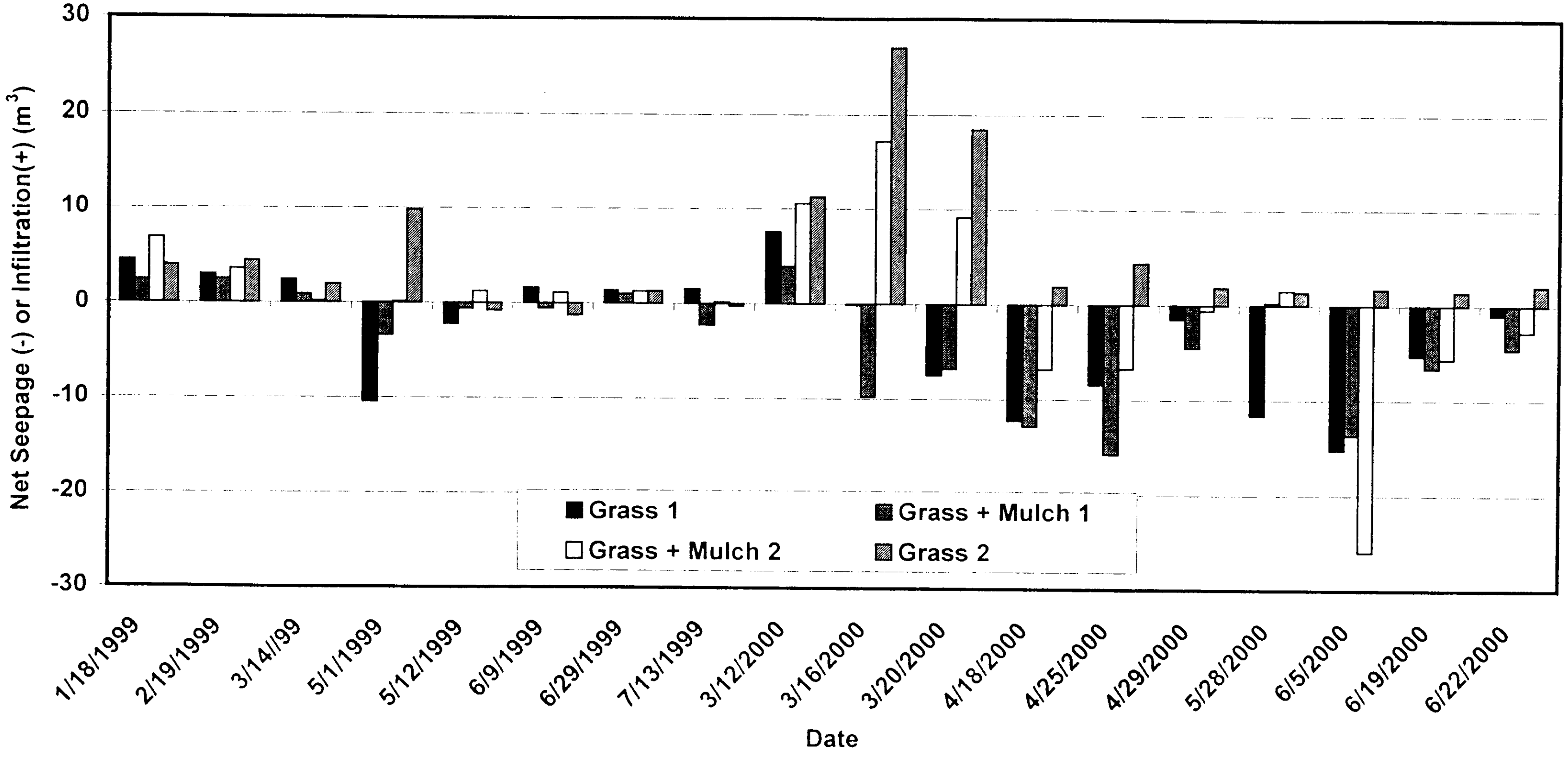


Figure 6. Hydrologic response of buffer plots at the Landscaped Buffer site. Net seepage indicates a net increase in flow over field inputs, and infiltration indicates a net decrease.

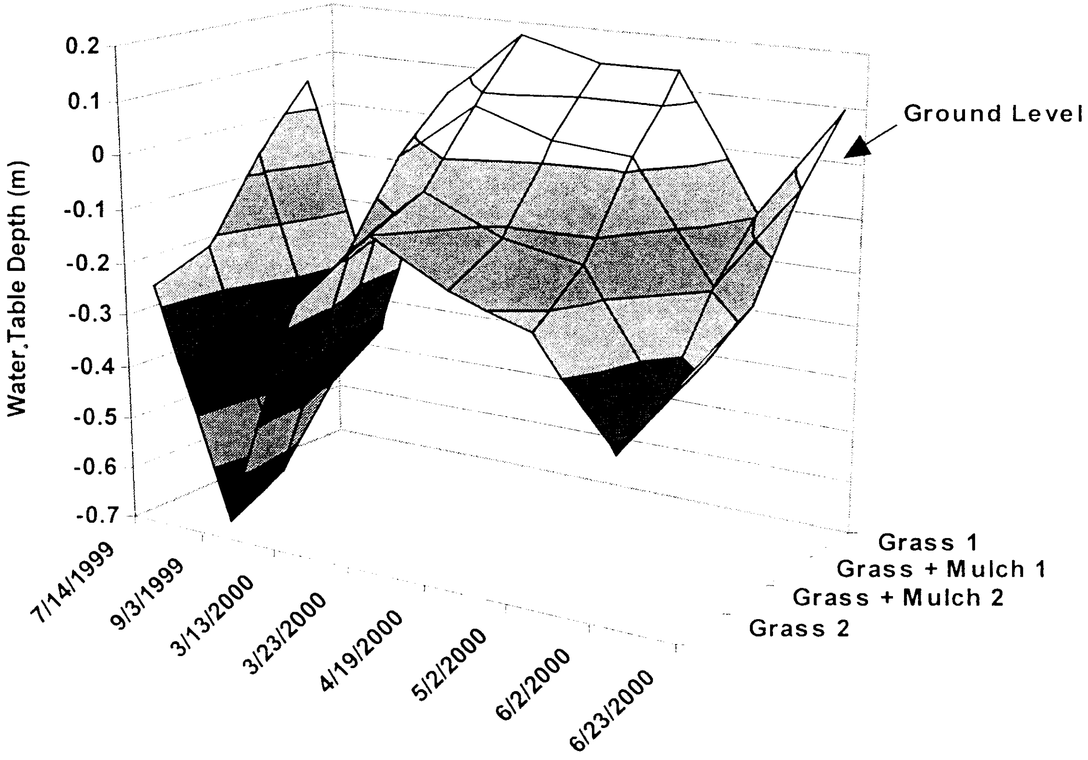


Figure 7. Ground water elevations relative to ground level at the lower end of the plots at the Landscaped Buffer site. Positive values indicate measurements in the casing above ground level.

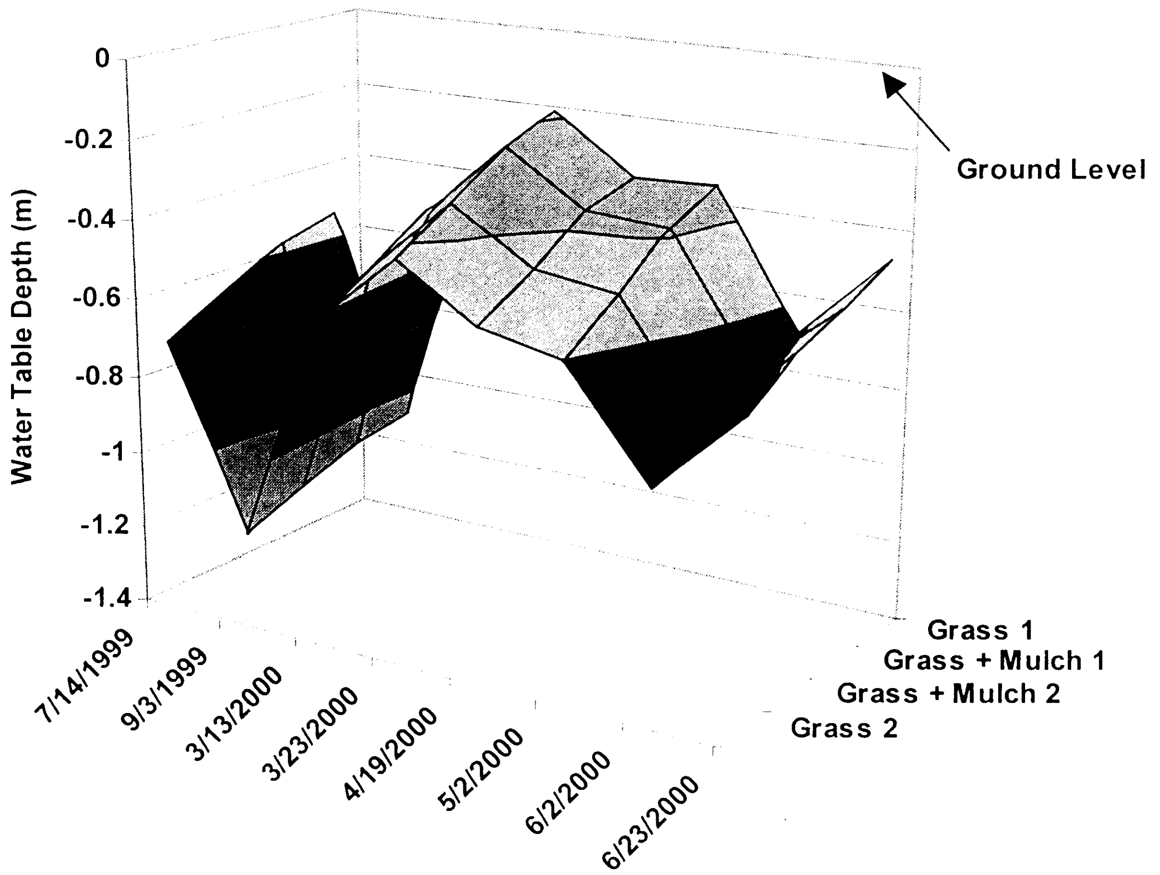


Figure 8. Ground water elevations relative to ground level at the upper end of the plots at the Landscaped Buffer site.

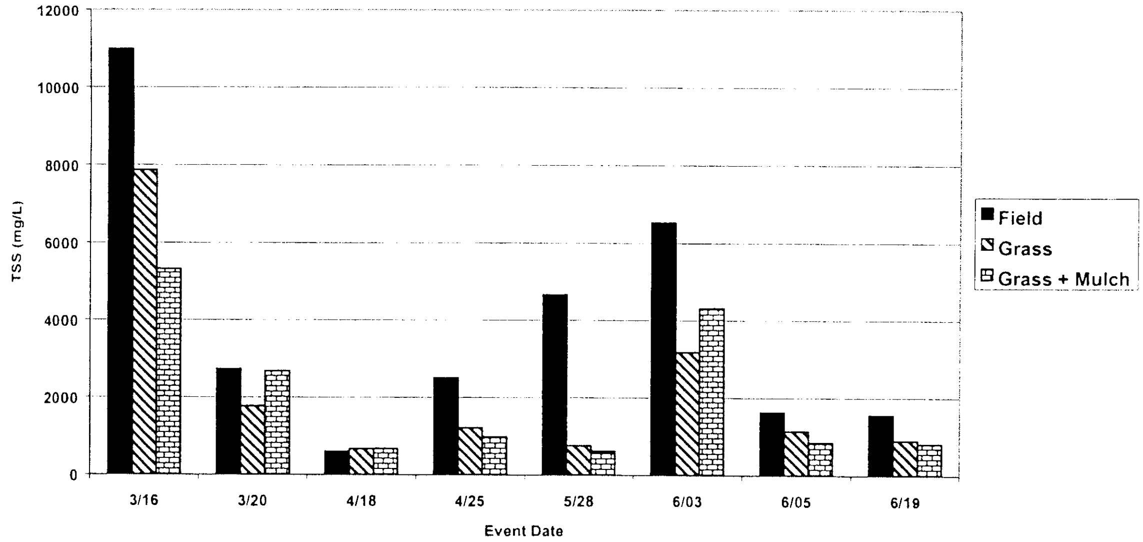


Figure 9. Weighted average concentration of total suspended solids for field, grass, and grass + mulch plots at the Landscaped Buffer site in 2000.

Table 2. The median values of the delivery ratio of volumes or concentrations and their coefficients of variations for the Clinton Experiment Site.

Variables	Sample Size	Median Value (Q2)	Coefficient of Variation
		Hydrology	
Runoff	18		
Grass 1		1.17	0.50
Grass 2		0.75	0.49
Grass + Mulch 1		1.22	0.34
Grass + Mulch 2		0.93	0.52
Average for grass		0.92	0.38
Average for grass with mulch		0.96	0.39
		Sediment	
TSS	11		
Grass 1		0.48	0.36
Grass 2		0.36	0.65
Grass + Mulch 1		0.41	0.75
Grass + Mulch 2		0.39	1.29
Average for grass		0.48	0.37
Average for grass with mulch		0.37	0.99
		Pesticide	
Metolachlor	8		
Grass 1		2.86	0.99
Grass 2		0.39	1.15
Grass + Mulch 1		2.19	0.83
Grass + Mulch 2		2.47	1.25
Average for grass		1.63	0.85
Average for grass with mulch		1.34	1.13
		Nutrients	
NH_3-N	9		
Grass 1		0.95	1.64
Grass 2		0.94	0.28
Grass + Mulch 1		0.66	0.70
Grass + Mulch 2		1.00	1.26
Average for grass		0.95	1.24
Average for grass with mulch		1.00	0.85
NO_3-N	8		
Grass 1		0.96	0.33
Grass 2		1.00	0.38
Grass + Mulch 1		1.20	0.15
Grass + Mulch 2		1.00	0.50
Average for grass		1.00	0.35
Average for grass with mulch		1.00	0.38
PO_4-P	9		
Grass 1		0.65	0.33
Grass 2		1.03	0.36
Grass + Mulch 1		0.86	0.19
Grass + Mulch 2		1.04	0.35
Average for grass		0.82	0.32
Average for grass with mulch		1.02	0.32

Table 3. Treatment comparisons between the buffer plots in Clinton Experiment Site. Numbers represent p>t for each comparison of average concentration in runoff.

Buffers	Runoff	TSS	Metolachlor	PO_4-P	NH_4-N	NO_3-N
	Individual Comparison					
Grass 1 & Grass + Mulch 1	0.11	0.69	0.32	0.20	0.88	**0.05**
Grass 1 & Grass + Mulch 2	**0.0003**	0.60	0.58	**0.02**	0.65	0.42
Grass 1 and Grass 2	**0.02**	0.93	**0.05**	**0.05**	0.29	0.43
Grass + Mulch 1 and Mulch 2	**0.001**	0.69	0.23	**0.08**	0.28	0.77
Grass 2 and Grass + Mulch 1	**0.0003**	0.60	**0.02**	0.25	0.52	0.19
Grass 2 and Grass + Mulch 2	0.11	0.39	**0.08**	**0.03**	0.32	0.75
	Treatment Comparison					
Grass vs Grass + Mulch	0.08	0.61	0.58	0.02	0.65	0.42

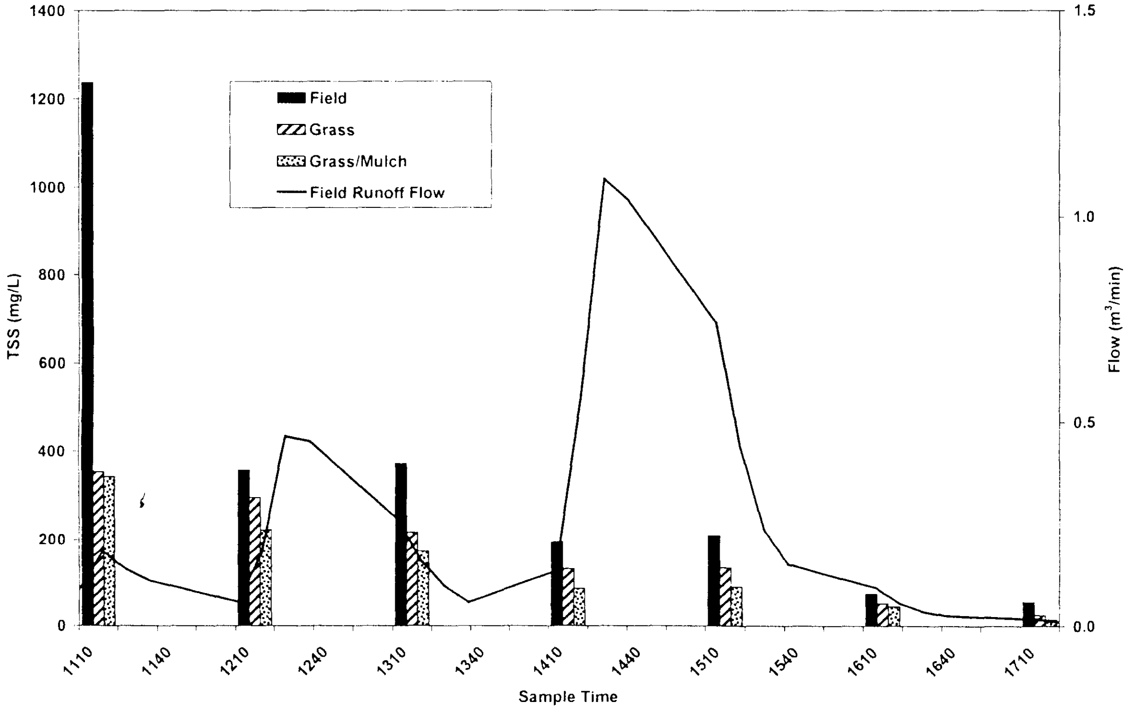


Figure 10. Total suspended sediment and hydrograph for the April 25, 2000, runoff event at the Landscaped Buffer site. Flow measurements are from the field runoff only.

Total sediment delivered from the fields to the plots averaged 55.6 kg per plot over the five storms for which a complete data set was available (Table 4). Retention within the two buffer types was similar at over 60%. The high variation in total sediment between the five events resulted in no significant differences between the treatments and the field input.

Table 4. Average total sediment delivered from the fields to the plots and average total sediment exiting the plots. Only storm events where data from all plots was available were included.

Source	Storm Events	Mean Total Sediment (kg)	Standard Deviation
Field	5	55.6	86.2
Grass	5	19.6	19.4
Grass/Mulch	5	18.0	16.3

Nutrients

As expected, neither treatment substantially reduced dissolved nutrients in runoff (Table 2). The grassed plots had less orthophosphate in runoff compared to the grass + mulch plots (Table 3), but the reduction was only about 18% of input. Orthophosphate was typically between 200 and 1,000 μg P/L, while neither NH_4 nor NO_3 were above 0.1 mg N/L in most runoff samples. The concentrations were often relatively constant during runoff events, as is shown in Figure 11. There were no significant differences between the total amount of the three measured nutrients entering the plots and exiting them (Table 5).

Table 5. Average total amount of nutrients in runoff from seven storm events in 2000.

Source	PO_4-P (mg)	NH_4-N (g)	NO_3-N (g)
Field	10.2	5.8	6.3
Grass	6.1	4.3	4.6
Grass/Mulch	10.8	4.2	7.3

Shallow ground water typically had nitrate concentrations of 6 - 12 mg NO_3-N/L, with a median of about 9 mg NO_3-N/L (Table 6). There was no treatment effect on shallow ground water, as expected, since the plots were relatively small and most of the ground water originated in the field. The concentration of ammonia was usually below or close to the detection limit of 0.1 mg NH_3-N/L. The exception was the upper well in the grass + mulch 1 plot, which consistently had 5-9 mg NH_3-N/L. Nitrate was never detected in this well. Two other wells, the upper well in grass 1 and the lower well in grass 2, exhibited NH_3-N occasionally with a concomitant drop in NO_3-N (Figures GW1 and GW2). There appeared to be a shift from NO_3-N to NH_3-N as water levels dropped, but there is not enough data to evaluate this relationship. Orthophosphate was not detected above 0.01 mg PO_4-P/L in most well samples.

There were no differences in the nutrient concentrations between the upper and lower wells (Table 7).

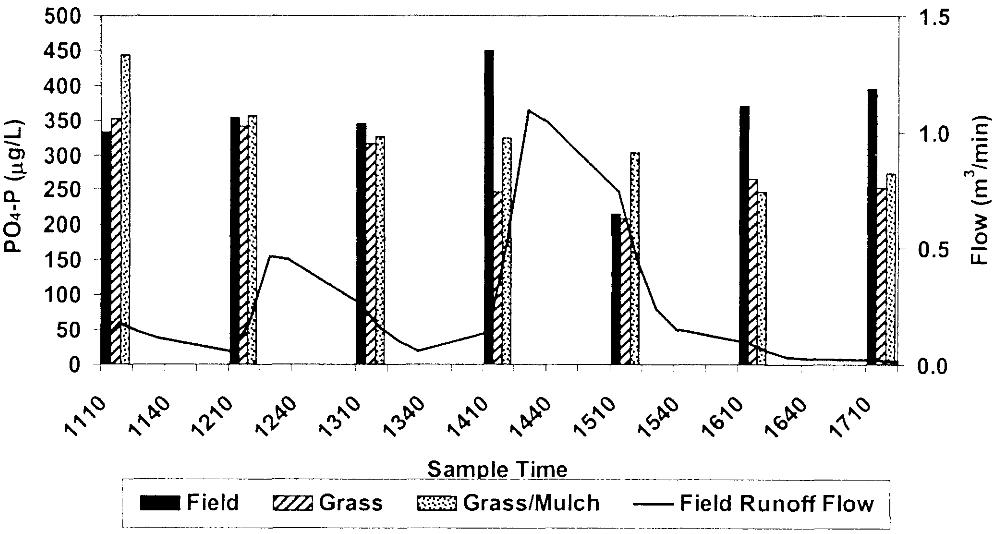


Figure 11. Orthophosphate concentrations in runoff and hydrograph from the field, grass, and grass/mulch plots for the April 25, 2000, runoff event.

Table 6. Nutrients in shallow ground water: all wells.

	Maximum	Median	% Detection
PO_4 (µg P/L)	27	10	21
NH_4 (mg N/L)	10	1.8	33
NO_3 (mg N/L)	12	8.9	85

Table 7. Well nutrients: upper and lower wells.

	Median	Maximum	% Detection
		Lower	
PO_4 (µg P/L)	10.4	16.0	23
NH_4 (mg N/L)	0.9	3.7	35
NO_3 (mg N/L)	8.9	12.2	83
		Upper	
PO_4 (µg P/L)	6.4	26.8	21
NH_4 (mg N/L)	0.6	7.1	38
NO_3 (mg N/L)	9.0	11.8	83

Pesticides

Of the 12 pesticides and metabolites in our analysis, eight were detected over the course of the study (Table 8). The rate of detection in samples ranged from 1% for atrazine in field runoff to 80% for metolachlor in runoff from the grass/juniper plots. The peak concentrations were nearly always higher in the buffer plots than in the field runoff. One possible explanation is that the first flush, which often has the highest concentrations of pesticides, may have passed through the large upper flume before the samples at that point were taken. Once the upper flume sampler was triggered, the samplers for each plot were triggered. Thus, the samplers were sampling at the same time and therefore were sampling different parts of the hydrograph. Another possibility is that flow out of the plots was insufficient for the samplers to obtain a sample during the initial flow. None of the detections exceeded drinking water standards. There was no evidence of either ground cover having any effect on the concentration of pesticides in runoff (Table 2). This can be seen in the pattern of metolachlor in runoff during an event on April 25, 2000 (Fig. 12). Metolachlor concentrations among field, grass, and grass/juniper samples show no evidence of any treatment effect but generally decline during the course of the runoff event in spite of increasing flow. Over seven events in 2000, the total amount of metolachlor exiting the plots averaged 7.0 mg and 2.7 mg for the mulched and grassed plots, respectively, which was significantly different ($p = 0.05$).

Six of the 12 pesticides and metabolites were detected in shallow ground water (Table 9). Detections were relatively rare, however, and the concentrations were well below drinking water standards. The most frequently detected compound was metalaxyl, a fungicide which is the most easily leached of all of the compounds. It is widely used in vegetable crops and has been in use for many years, so it is not surprising that it was present in shallow ground water. Chlorothalonil and triadimefon, two other fungicides widely used in vegetable production, are much less likely to leach. The former was never detected in ground water but was found in some runoff samples. The latter was frequently found in runoff but only detected in 4 of 35 ground water samples. Two herbicides, metolachlor and alachlor, are considered moderate leaching risks, and this is reflected in the comparison of their detections in runoff and ground water. Both were frequently detected in runoff but much less frequently found in ground water.

Table 8. Pesticides in runoff at Clinton. Concentrations in μg/L.

	Maximum	Minimum	Median	% Detection
	Field Runoff			
Atrazine	0.35	0.35	0.35	1
DEA	0	0	0	0
DIA	0	0	0	0
Acetochlor	0.20	0.15	0.16	4
Ametryn	0	0	0	0
Metolachlor	0.48	0.11	0.19	57
Simazine	0	0	0	0
Alachlor	0.41	0.10	0.24	24
Metalaxyl	1.09	0.11	0.74	8
Chlorothalonil	0	0	0	0
Treadimefon	0.99	0.19	0.61	38
Triadimenol	0	0	0	0

	Grass Plot Runoff			
Atrazine	1.45	0.11	0.18	4
DEA	0	0	0	0
DIA	0	0	0	0
Acetochlor	1.18	0.12	0.19	4
Ametryn	0	0	0	0
Metolachlor	1.79	0.10	0.20	44
Simazine	0	0	0	0
Alachlor	1.44	0.12	0.23	16
Metalaxyl	0.65	0.16	0.40	6
Chlorothalonil	0.35	0.30	0.33	3
Treadimefon	1.02	0.10	0.13	16
Triadimenol	0.33	0.33	0.33	2
	Grass/Mulch Plot Runoff			
Atrazine	1.45	0.10	0.14	7
DEA	0	0	0	0
DIA	0	0	0	0
Acetochlor	1.17	0.17	0.37	3
Ametryn	0	0	0	0
Metolachlor	1.62	0.10	0.24	80
Simazine	0	0	0	0
Alachlor	1.57	0.10	0.22	56
Metalaxyl	1.19	0.12	0.31	9
Chlorothalonil	2.96	0.16	0.69	6
Treadimefon	1.48	0.10	0.14	26
Triadimenol	0	0	0	0

Table 9. Pesticides in shallow wells at Clinton. All values in μg/L.

	Maximum	Minimum	Median	% Detection
Atrazine	0	0	0	0
DEA	0.29	0.29	0.29	2
DIA	0	0	0	0
Acetochlor	0.28	0.08	0.12	5
Ametryn	0	0	0	0
Metolachlor	0.14	0.07	0.13	7
Simazine	0	0	0	0
Alachlor	0.15	0.15	0.15	1
Metalaxyl	1.67	0.12	0.19	19
Chlorothalonil	0	0	0	0
Treadimefon	0.26	0.17	0.25	10
Triadimenol	0	0	0	0

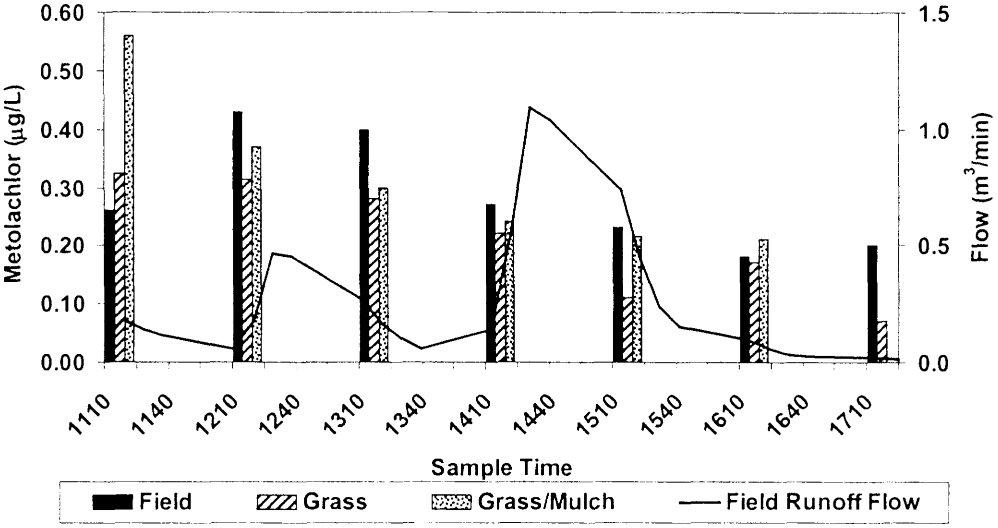


Figure 12. Metolachlor in runoff and hydrograph over time from the field and each plot for the April 25, 2000, runoff event at the Landscaped Buffer site.

RIPARIAN BUFFER SITE

Rainfall and Runoff

There were no runoff events in the three months after planting and pesticide application in 1999, so there is no data to report. In 2000, there was no rainfall during the two weeks following planting and pesticide application. Because of concern about a recurrence of the 1999 situation and the degradation losses of the pesticides prior to a runoff event, we decided to generate artificial runoff, as noted in the Methods section. We conducted three artificial events a few days after natural events that did not produce runoff in order to take advantage of the resulting moisture levels in the soil (Fig. 13). The rainfall event which occurred on June 17 was not recorded at the weather station because it was an isolated thunderstorm. We had completed an artificial runoff event on June 16 and reset the samplers that evening because we noticed the approaching storm on radar. We terminated the study after the last natural event in June because pesticide concentrations had declined rapidly to very low levels.

Runoff from the four artificial events ranged from < 0.5 m^3 to > 1 m^3 at the field edge (Fig. 14). On average, less than half of this made it through the 4 m grassed buffer and none was collected after 8 m of grass buffer. Even less runoff made it through the 4 m forested buffer and very little or none after 8 m of forest. This clearly indicates how dry this period was because so much of the runoff was being absorbed by the soil. Even though we were irrigating the first 3 m of the grass buffer, the water deficit in the soil was sufficient to soak up much or all of the artificial runoff. The corn was relatively small during this first month and did not appear to be greatly affected. The rains in June were sufficient for the corn to grow rapidly, so the sprinkler system had to be placed on

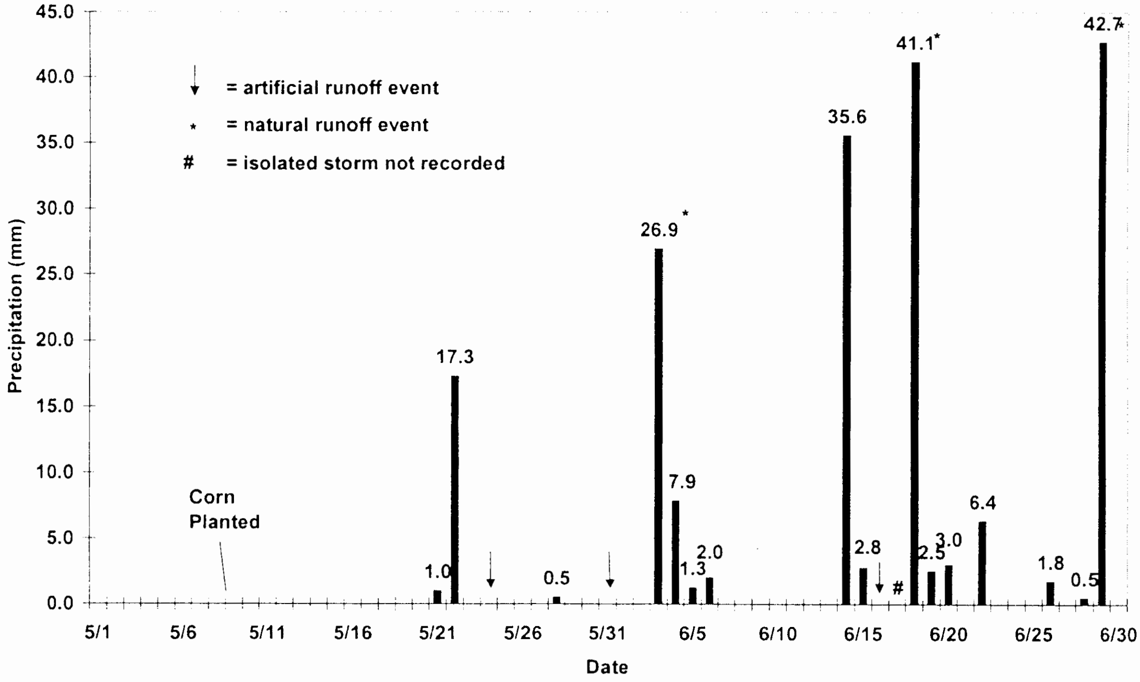


Figure 13. Precipitation and runoff events at the Grassed and Natural Riparian Buffer site in Raleigh, N.C., for May and June 2000.

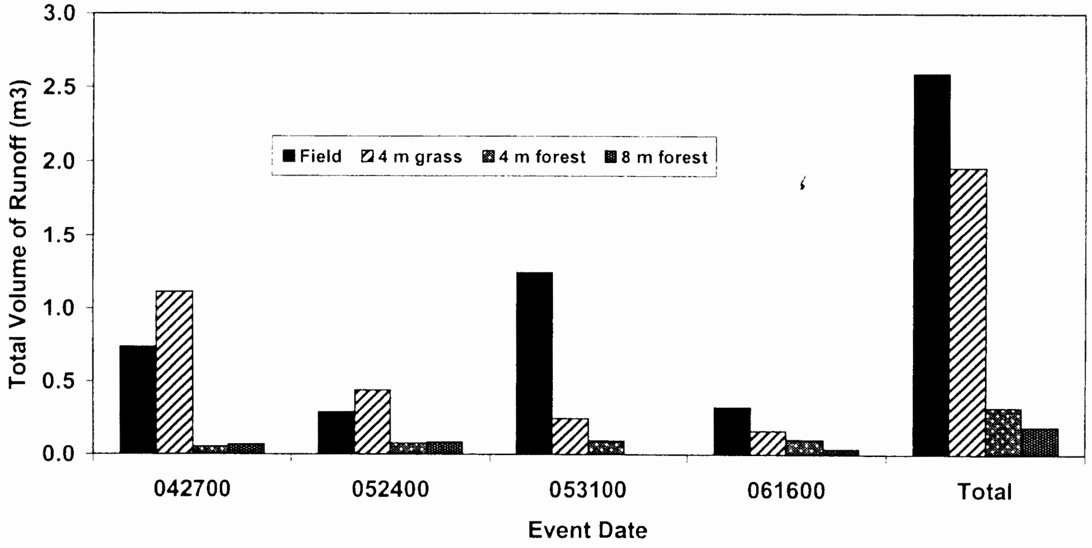


Figure 14. Runoff distribution for artificial events among field and buffer plots for artificial runoff events at the grass and forest buffer site.

stands 1.5 m above ground to obtain good distribution on the last artificial event June 16. Although the corn yield was not recorded, 2000 was reported to be a record yield year for corn in the area.

The volumes from the natural events were larger and some runoff was measured in the 8 m forest buffer from three of four events (Fig. 15). On average, the relative ranking of volume absorbed in the plots was similar to the artificial events: 8 m forest > 4 m forest or 8 m grass > 4 m grass. It is possible that the 8 m forested buffer plot had large macropores into which the runoff was draining. We detected one approximately 2 cm in diameter and plugged it with soil, but others may have been present. There was no evidence of water arising downslope, but water can run under the litter and escape detection.

Runoff was significantly greater ($p > 0.05$) for the 4 m grass buffer compared to the 4 m riparian and 8 m grass buffers (Table 10). There were no significant differences between the two riparian buffers in the amount of runoff collected, which is why we combined them in the graphs described below. Overall, the 8 m buffers both significantly reduced runoff volume.

The responses of the field and buffer to the different rainfall events provide good evidence of the influence of antecedent moisture levels on runoff volumes (Figs. 16-18). The June 3 event had been preceded by a very dry May, and as a result the volumes were relatively low and runoff was largely absorbed in all but the 4 m grass buffer. After nearly 38 mm of rainfall in the 3 days before the June 17 event, runoff passed through all three buffers but the volume was greatly reduced. Runoff the next day, June 18, largely passed through the 4 m grass buffer but was somewhat attenuated in the 8 m grass buffer and the wooded buffers. The large event (>40 mm) on June 29 was characterized by four distinct runoff periods corresponding to intense rainfall occurrences over the course of the day.

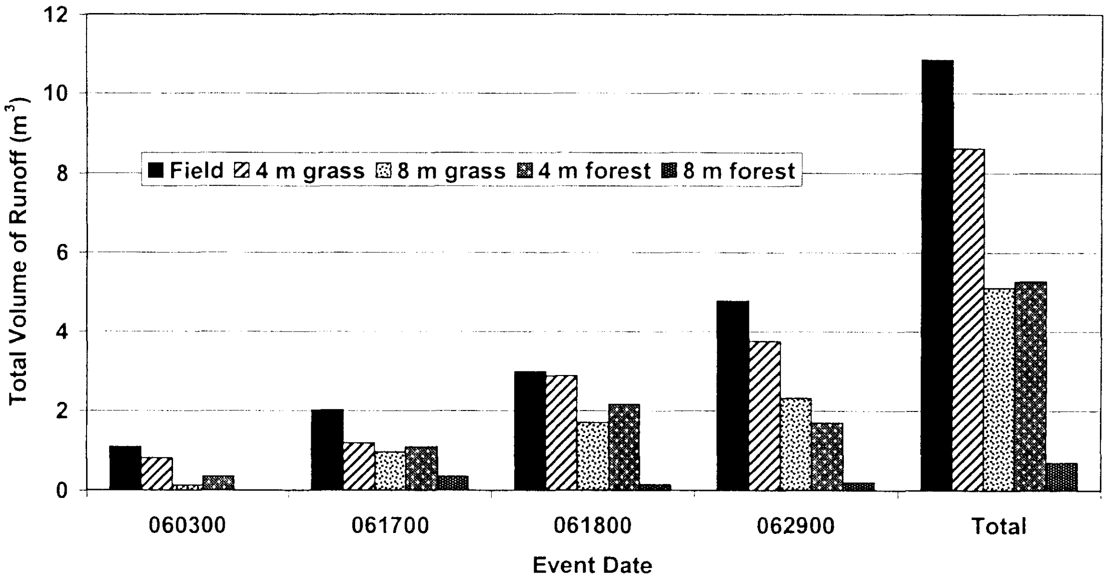


Figure 15. Runoff distribution for natural events among field and buffer plots at the Grassed and Natural Riparian Buffer site.

Table 10. Statistical tests of effects of buffers on runoff quantity and constituent concentrations. Highlighted are comparisons that are significant at $p \leq 0.10$.

	Runoff	TSS	Atrazine	Acetochlor	PO_4	NH_3	NO_3
	Buffer Comparisons						
4m and 8m Grasses	**0.01**	0.28	**0.02**	**0.03**	0.68	0.33	**0.01**
4m grass and 4m Riparian	**0.03**	0.17	0.30	0.44	**0.02**	0.24	0.33
8m grass and 8m Riparian	**0.06**	**0.05**	**0.07**	**0.04**	0.31	0.49	0.20
4m and 8m Riparian	0.22	0.57	0.8	0.29	0.18	0.7	**0.01**
	Net Change, by Buffer						
4 m Grass	**0.09**	**0.005**	.55	.30	**0.008**	.34	.17
8 m Grass	**0.005**	**0.003**	.23	.21	.34	.29	**0.09**
4 m Riparian	**0.10**	**0.09**	.26	.20	.92	.31	.67
8 m Riparian	**0.02**	.17	.24	.15	.67	.88	1.49

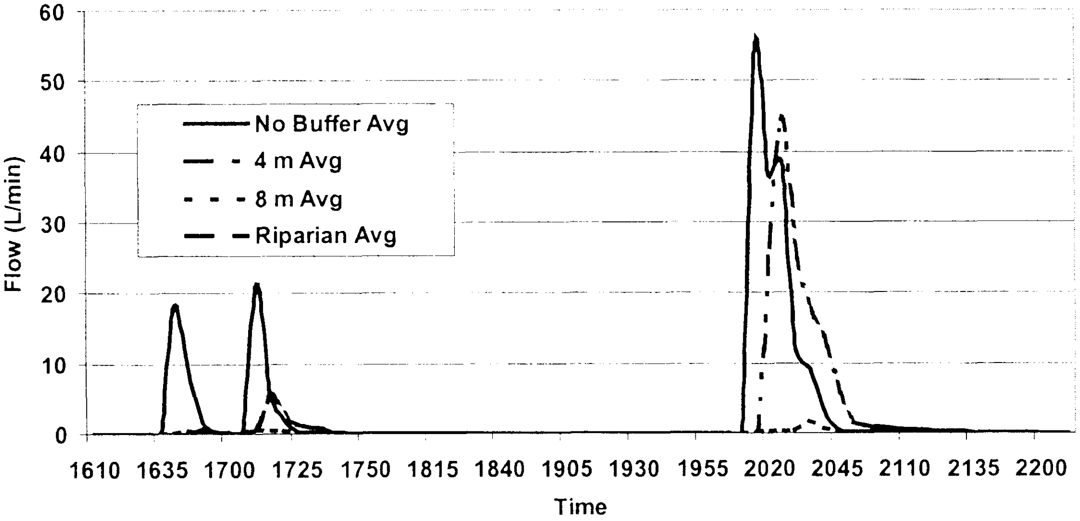


Figure 16. Hydrograph for different buffers during the June 3, 2000, rainfall event.

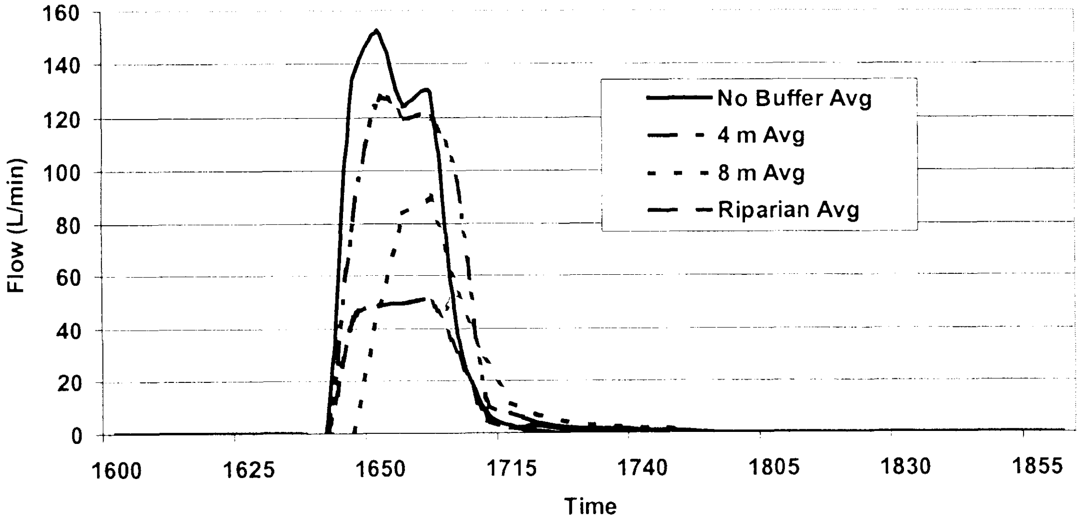


Figure 17. Hydrograph for different buffers during the June 18, 2000, rainfall event.

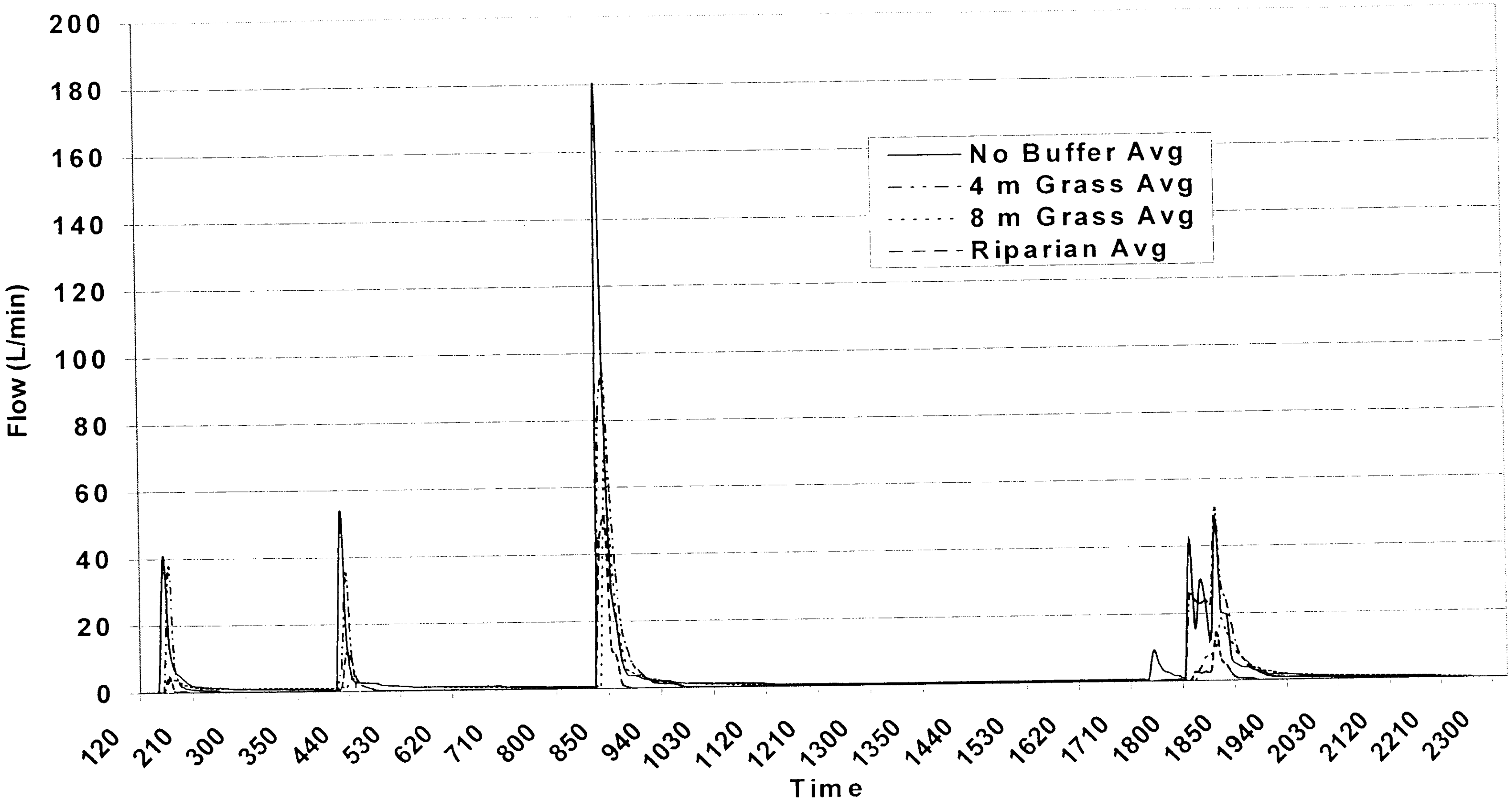


Figure 18. Hydrograph for different buffers during the June 29, 2000, rainfall event.

Sediment

The most dramatic effects that the buffers had on water quality was in reducing sediment. As expected from previous work, the grass buffers were nearly always more effective than the wooded buffers (Figs. 19-22). The 4 m and 8 m grass plots reduced TSS concentrations by up 64% and 95% for individual events. Maximum reductions of 42% and 37% occurred in the 4 m and 8 m forested buffers. Both the 4 m grass and the 8 m riparian buffers had statistically significant reductions (Table 11). The 8 m grass buffer absorbed a significant amount of water and in fact no runoff occurred during the artificial events, greatly reducing the statistical power of our comparisons for this treatment. The variability between events was large enough to statistically eliminate any differences which were occurring between the remaining treatments.

The total amount of sediment removed by the buffers was greater relative to the field inputs for the artificial events compared to the natural events (Table 12). This was most likely because of the much greater water retention during those events, as mentioned previously. The change in concentration was not as dramatically different between the artificial and natural runoff events compared to the total sediment. Overall, only TSS was reduced in most cases in greater proportion than the runoff itself. This clearly indicated that these buffers were most effective at removing sediment compared to other pollutants. The paired t-tests of all events showed significant reductions in the 4 m grass and the 8 m riparian buffers. The 8 m grass buffer averaged 90% reduction among the natural events and would likely have been significant if there had been more events where runoff was collected. In fact, there is some argument to be made that the lack of runoff through the buffer could be considered the effect, essentially resulting in a zero for statistical analysis. However, this is likely to be a relatively rare occurrence in nature so we did not use this approach in our analyses.

Table 11. Statistical tests of buffer effects on total amounts of runoff constituents. Highlighted are comparisons which are significant at $p \leq 0.10$ by a paired t-test. The number of events available for analysis is in parentheses.

	TSS	Atrazine	Acetochlor	PO_4	NH_4	NO_3
Field and 4m Grass (8)	**0.05**	0.79	0.40	0.11	0.11	0.17
Field and 8m Grass (4)	0.16	0.23	0.37	0.93	0.92	0.63
Field and 4m Riparian (8)	0.20	0.11	0.13	0.86	0.85	**0.07**
Field and 8m Riparian (5)	**0.05**	0.24	0.14	**0.10**	0.11	0.11
4m and 8m Grasses (4)	0.21	0.25	0.63	0.34	0.32	0.58
4m grass and 4m Riparian (8)	0.31	0.18	0.23	0.23	0.25	**0.07**
8m grass and 8m Riparian (3)	0.46	0.27	0.24	**0.07**	0.20	0.26
4m and 8m Riparian (5)	0.22	0.08	0.08	0.34	0.34	0.36

Table 12. Ratios of total runoff and total amounts of constituents leaving the buffers compared to the field edge. Number of events recorded is in parentheses.

	Volume	TSS	Atrazine	Acetochlor	PO_4	NH_4	NO_3
Artificial Events							
4m Grass	0.67 (4)	0.38(4)	0.87 (3)	0.59 (3)	4.9 (4)	5.1 (4)	1.2 (4)
8m Grass	0.41 (1)	0.16 (1)	(0)	(0)	4.5 (1)	2.1 (1)	0.98 (1)
4m Riparian	0.15 (3)	0.12 (4)	2.7 (3)	0.18 (3)	0.22 (4)	0.16 (4)	0.20 (4)
8m Riparian	0.12 (3)	0.06 (3)	0.09 (2)	0.06 (2)	0.17 (3)	3.5 (2)	0.12 (3)
Natural Events							
4m grass	0.82 (4)	0.58 (4)	1.0 (4)	0.74 (4)	2.6 (4)	3.4 (3)	0.88 (4)
8m grass	0.93 (3)	0.10 (3)	0.58 (3)	0.90 (3)	1.2 (3)	1.2 (2)	1.5 (3)
4m Riparian	0.48 (4)	0.38 (4)	0.69 (4)	0.79 (4)	1.3 (4)	0.11 (3)	0.55 (4)
8m Riparian	0.11 (2)	0.35 (2)	0.12 (2)	0.15 (2)	0.06 (2)	0 (2)	0.15 (2)

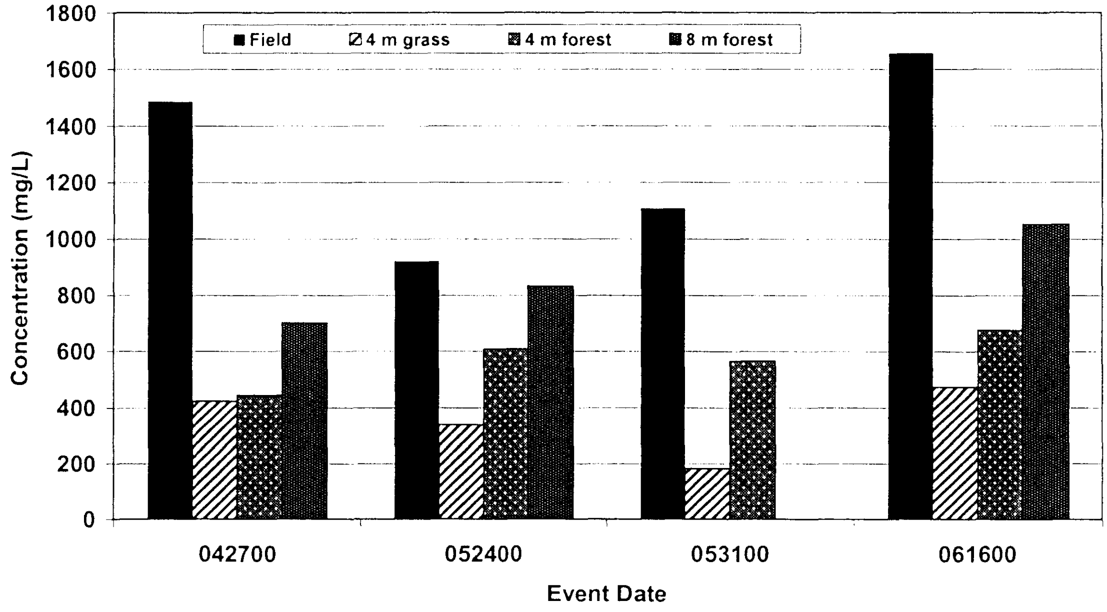


Figure 19. Sediment concentrations in runoff for four different buffer types at the Piedmont site during artificial runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

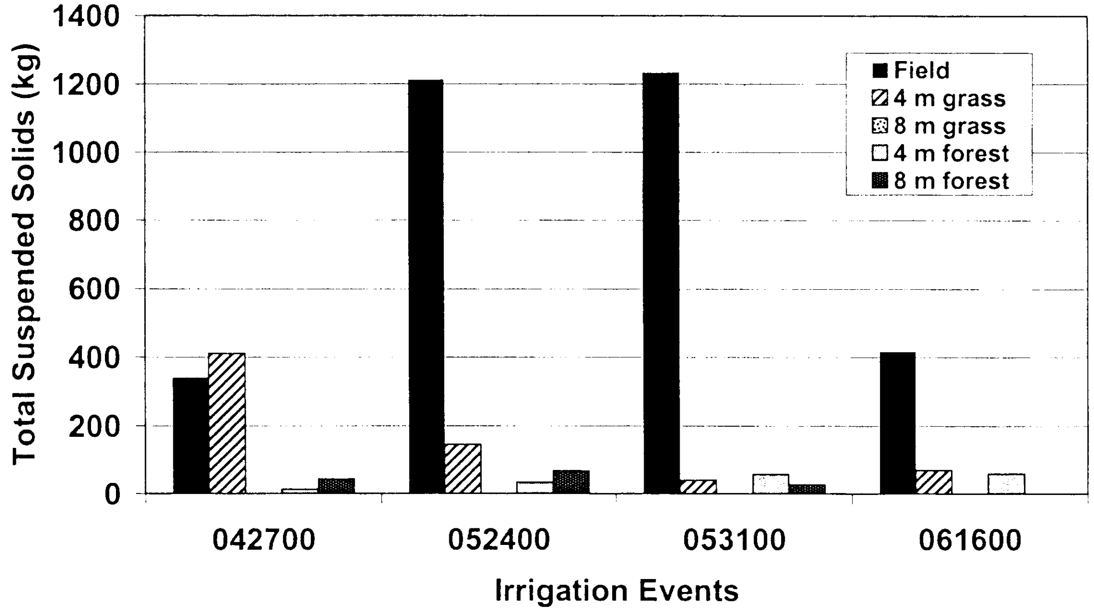


Figure 20. Total suspended sediment in runoff for artificial runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

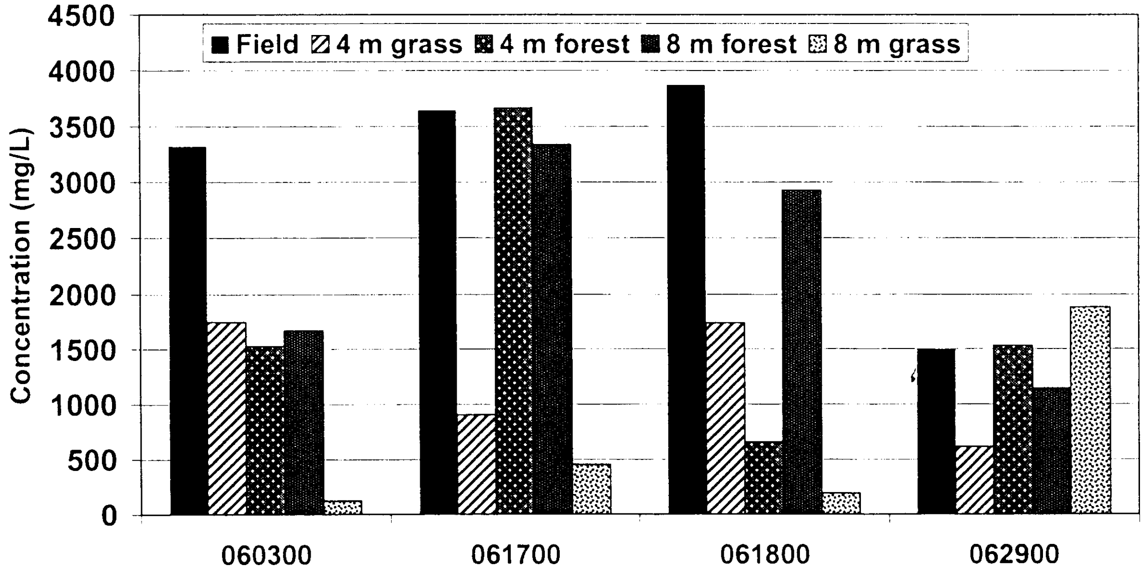


Figure 21. Sediment concentrations in runoff for four different buffer types at the Piedmont site during natural runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

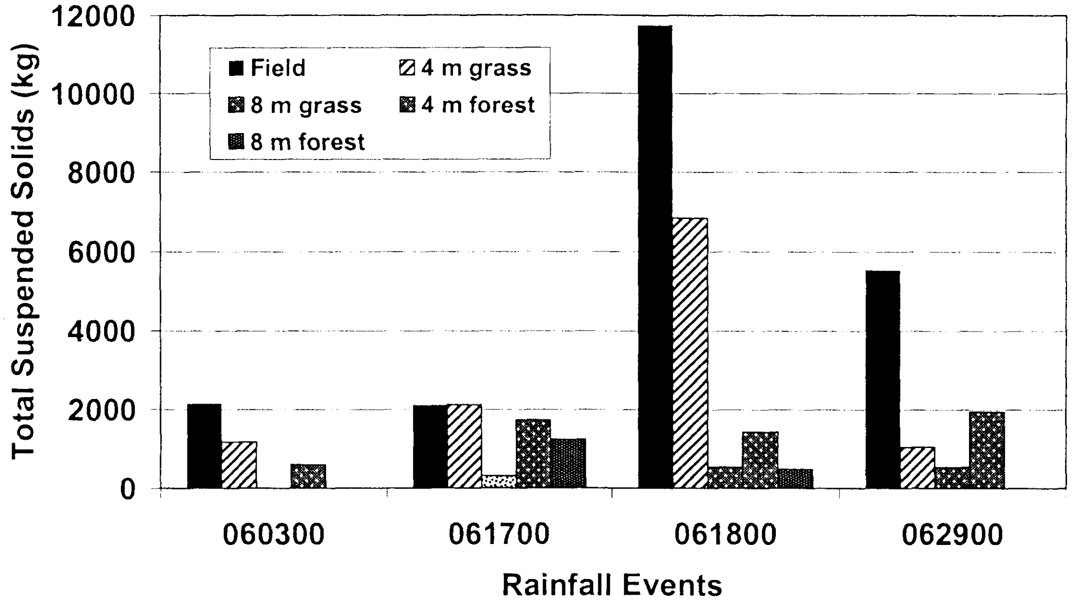


Figure 22. Total suspended sediment in runoff for natural runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

Nutrients

The natural buffers did not significantly reduce the soluble nutrients in runoff, actually increasing the concentrations in many cases. The only statistically significant effect was in the 4 m grass buffer, where PO_4-P increased significantly (Table 11). Among the treatments, the 4 m riparian buffer was significantly better than the 4 m grass buffer in reducing PO_4-P concentrations, and the 8 m riparian buffer was significantly better than the 4 m riparian buffer in reducing NO_3-N concentrations. However, since none of the treatments resulted in reduced nutrient levels the differences within treatments are not very meaningful.

The concentrations of PO_4-P generally declined over the course of the study but no clear pattern was evident for all plots in either artificial or natural runoff events (Figs. 23-26). Total soluble P actually increased in the several of the buffers for several events and in general was not reduced. Among the treatments, only the 8 riparian buffer reliably retaining soluble P, with significant reductions compared to the field and the 8 m grass buffer (Table 11). An outlier of over 3,000 μg PO_4-P/L occurred on June 3 for the 8 m grass buffer, but most averages fell in the 100-500 μg PO_4-P/L range. The grass buffers appeared to be a P source for most events, with more P leaving the buffer than was leaving the field (Table 12). The forested buffers did not contribute P to the runoff but generally reduced P in close proportion to reductions in runoff volume.

Nitrate concentrations during the first runoff event (artificial) after planting rose from 1-2 mg N/L to over 10 mg N/L at the field edge (Fig. 27). The buffers were relatively effective in retaining NO_3-N as a result of the greater infiltration during the artificial events (Fig. 28). During the natural events there were few differences in the concentrations or totals for nitrate N (Figs. 29-30). Nitrate N concentrations in runoff had dropped to below 0.3 mg NO_3-N/L by the end of June when we terminated the study. Paired t-tests suggested that the riparian buffers were somewhat more effective than the grass buffers (Table 11). The effects of the buffers on total nitrate in runoff was essentially the same as for orthophosphate, with little attenuation and possible some additions occurring (Table 12).

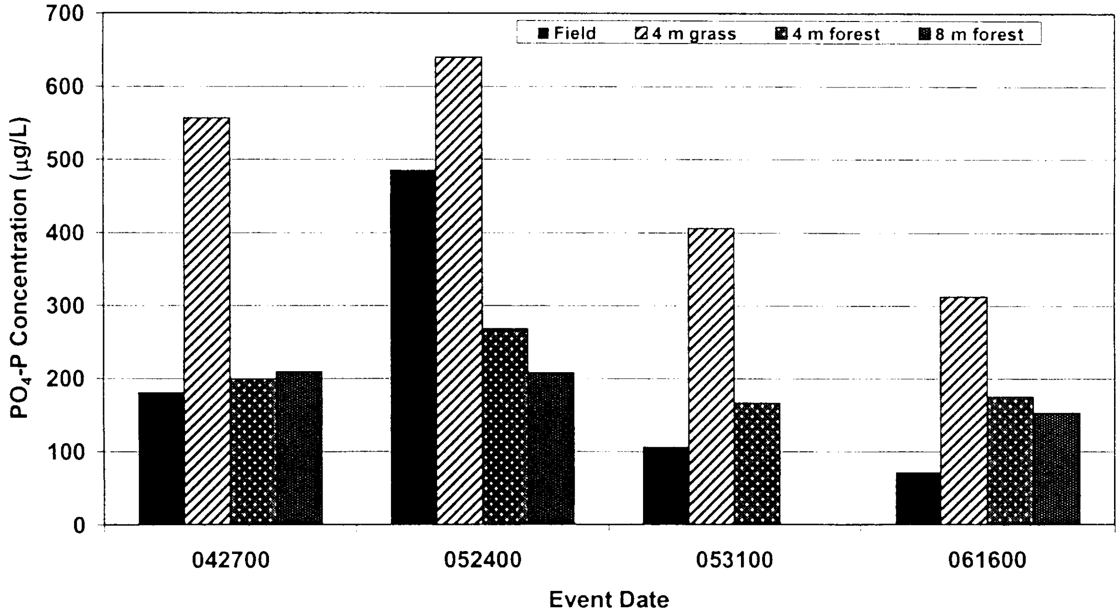


Figure 23. Dissolved phosphate concentrations in runoff for four different buffer types at the Piedmont site during artificial runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

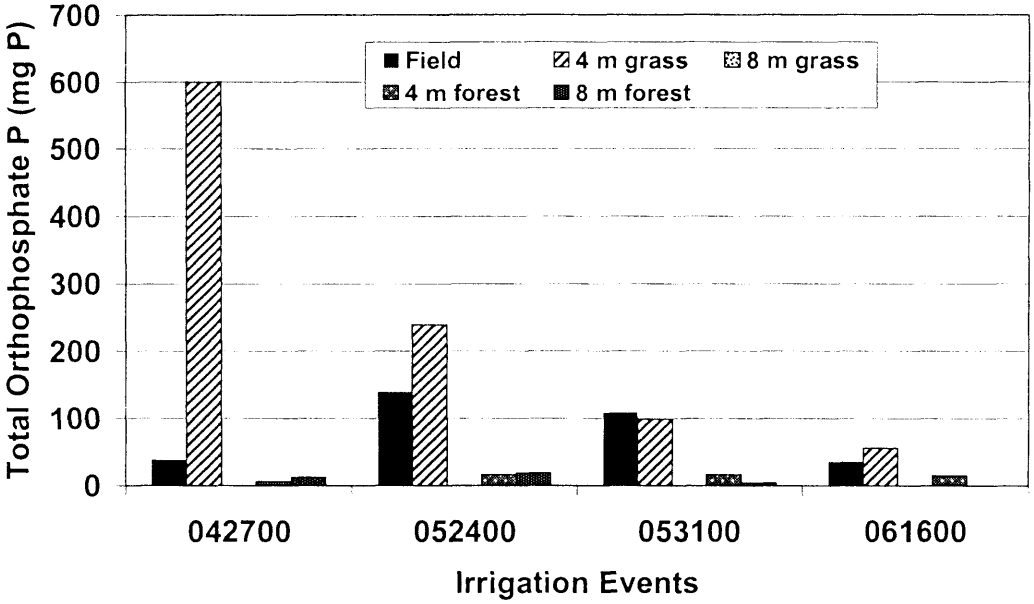


Figure 24. Total dissolved phosphate P in runoff for four different buffer types at the Piedmont site during artificial runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

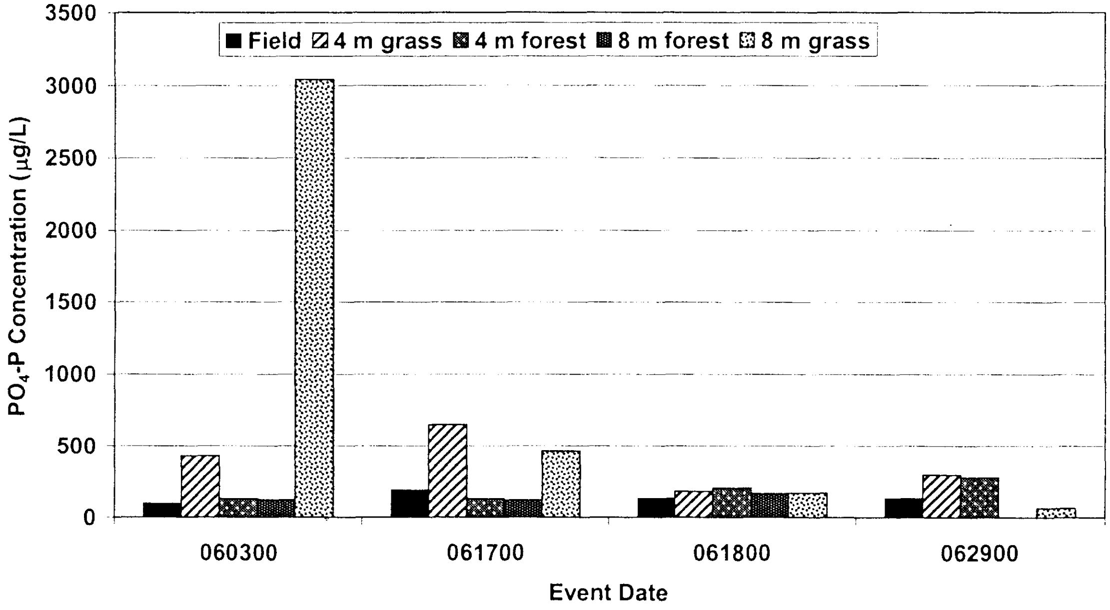


Figure 25. Dissolved phosphate concentrations in runoff for four different buffer types at the Piedmont site during natural runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

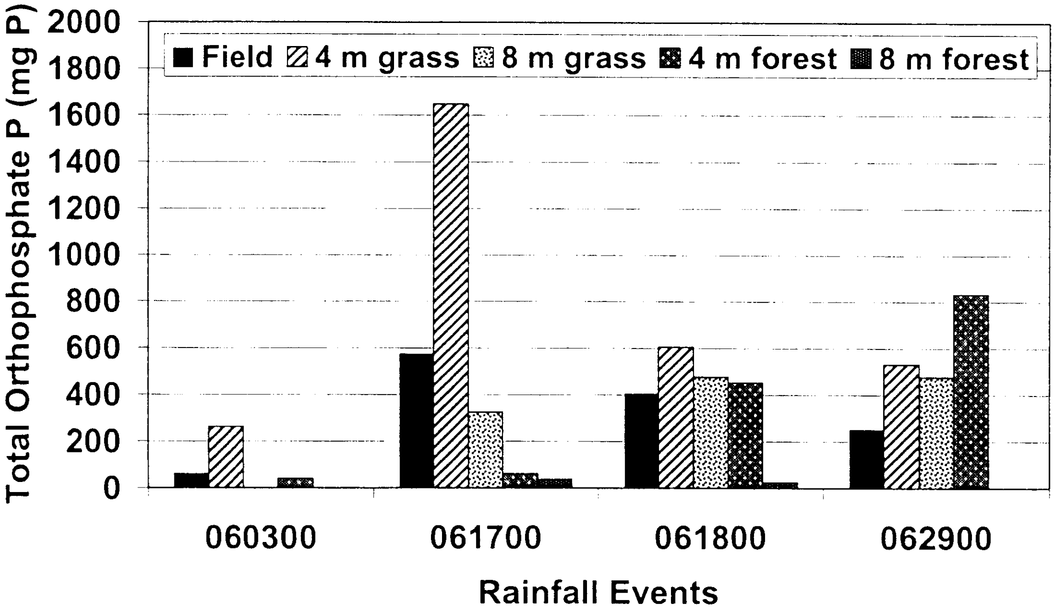


Figure 26. Total dissolved phosphate P in runoff for four different buffer types at the Piedmont site during natural runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

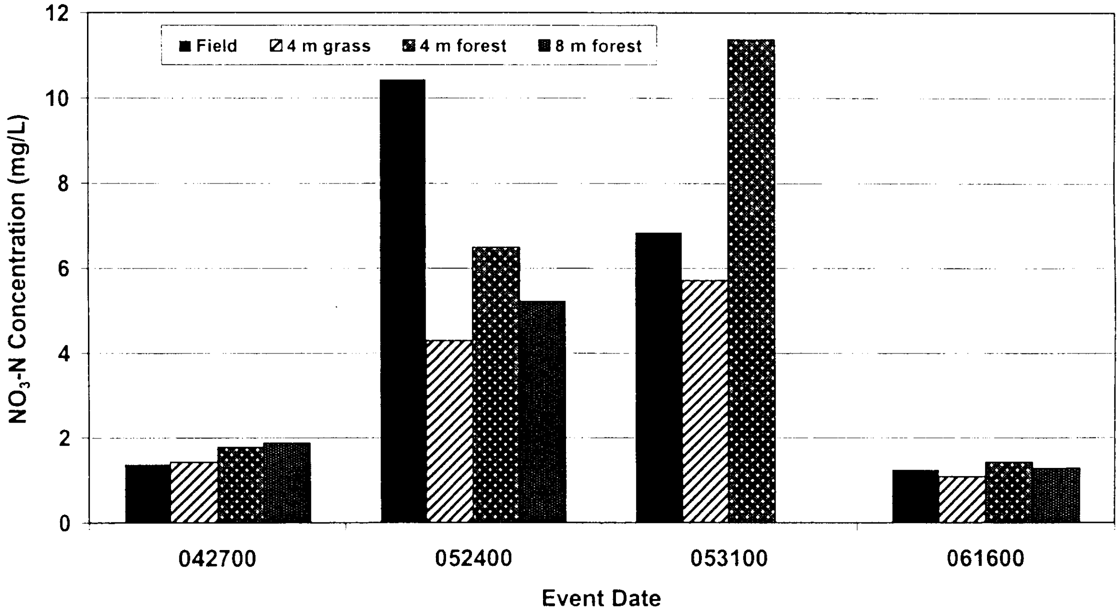


Figure 27. Nitrate concentrations in runoff for four different buffer types at the Piedmont site during artificial runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

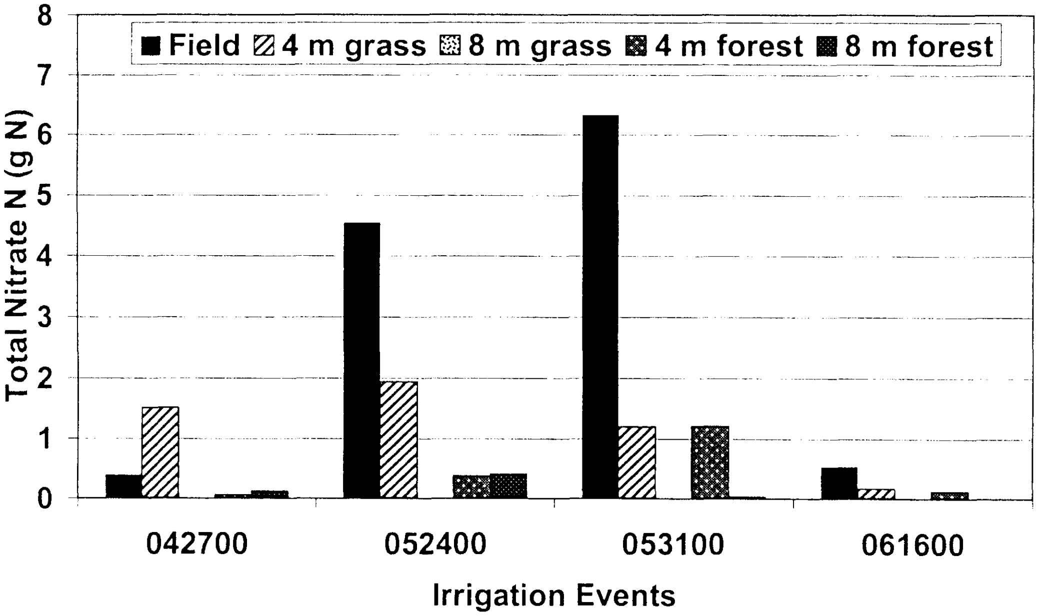


Figure 28. Total nitrate N in runoff for four different buffer types at the Piedmont site during artificial runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

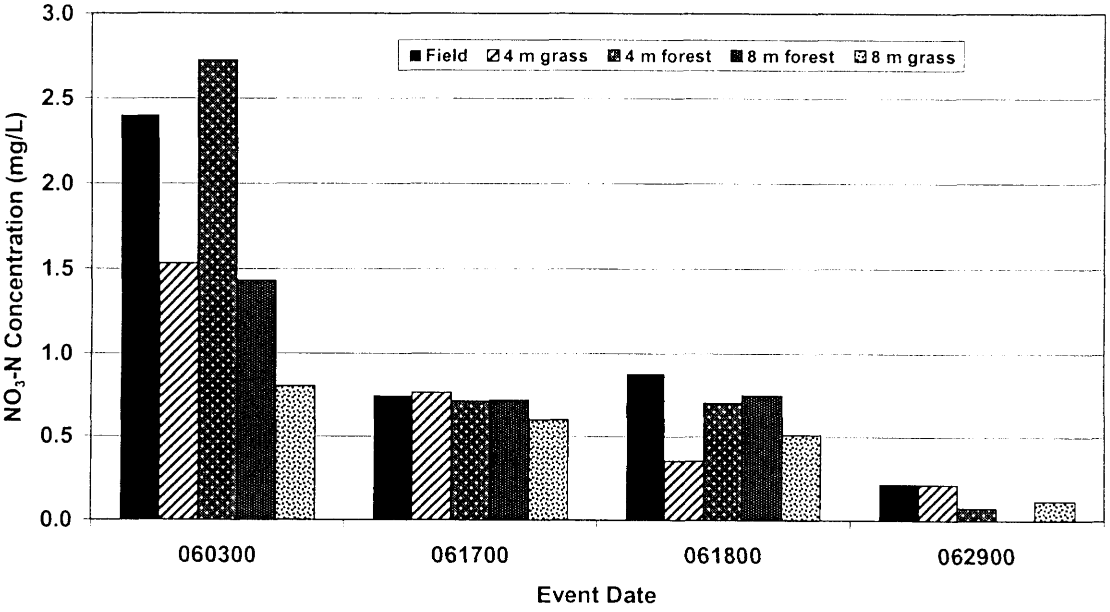


Figure 29. Nitrate concentrations in runoff for four different buffer types at the Piedmont site during natural runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

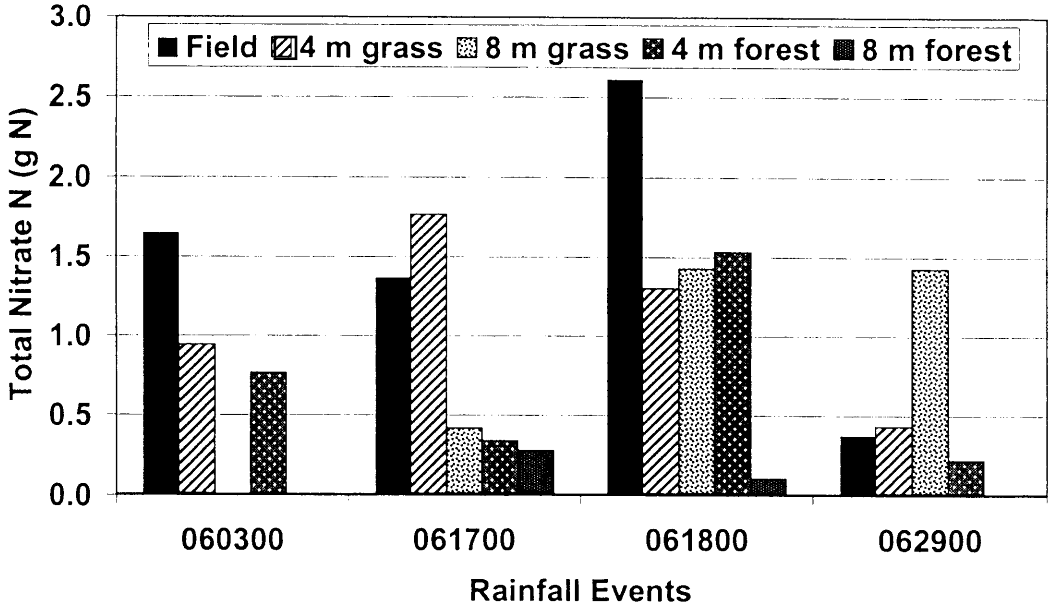


Figure 30. Total nitrate N in runoff for four different buffer types at the Piedmont site during natural runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

Ammonium N essentially followed the same pattern as orthophosphate, with little evidence of consistent treatment by any of the buffer systems (Figs. 31-34). As with the soluble P measurements, the buffers were sometimes sources rather than sinks for NH_4-N (Table 12). The concentrations were much lower than nitrate, indicating that it was not a significant source of N in this type of corn management (Table 11).

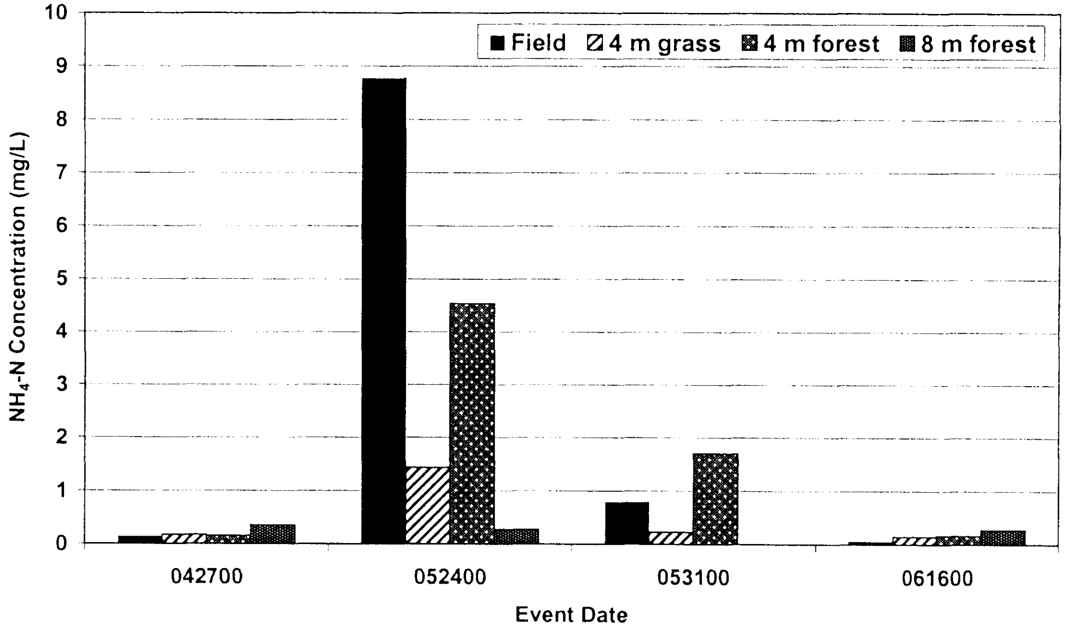


Figure 31. Ammonium concentrations in runoff for four different buffer types at the Piedmont site during artificial runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

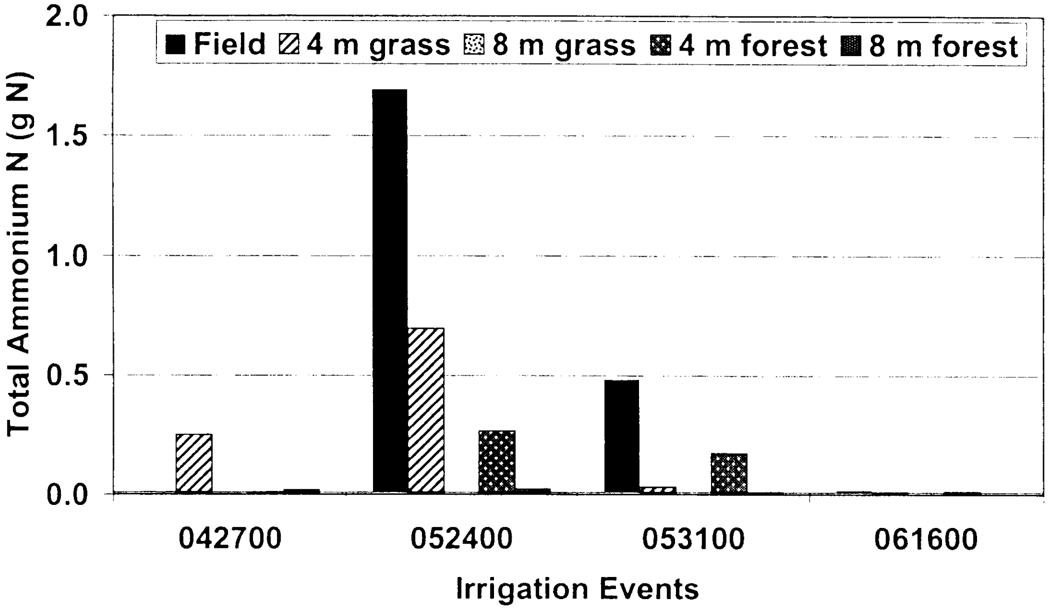


Figure 32. Total ammonium N in runoff for four different buffer types at the Piedmont site during artificial runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

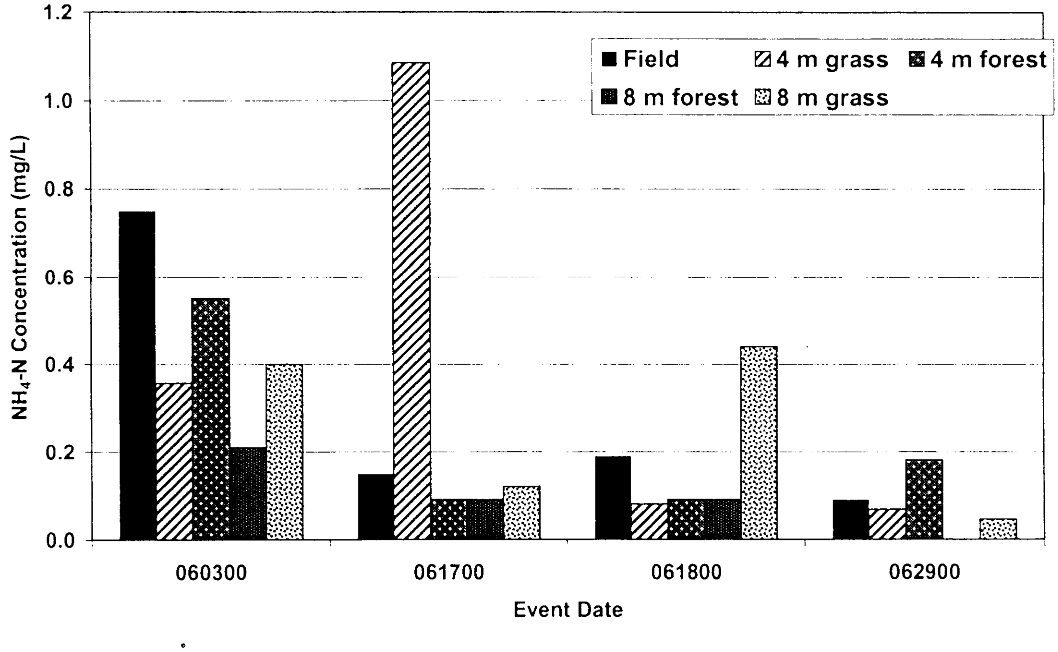


Figure 33. Ammonium N concentrations in runoff for four different buffer types at the Piedmont site during natural runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

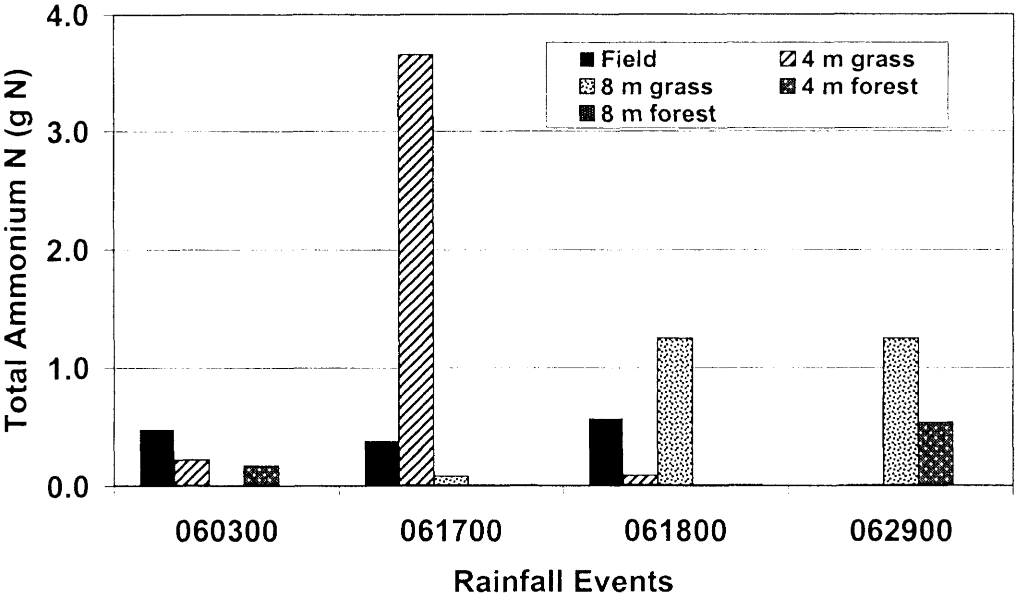


Figure 34. Total ammonium N in runoff for four different buffer types at the Piedmont site during natural runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

Pesticides

Chlorpyrifos was applied in-furrow at the time of planting, but it was never detected in runoff samples during the study. This is likely a combination of the relatively strong binding mechanisms in soil for that compound and the fact that it was applied below the soil surface. The remaining two pesticides, atrazine and acetochlor, were broadcast applied and they exhibited typical patterns of concentrations in runoff as the study progressed: high concentrations initially followed by an exponential decline.

A few individual samples had traces of atrazine during the pre-plant runoff event, but the average was essentially zero. The first runoff event on May 24 produced concentrations well over 100 μg/L at the field edge (Fig. 35), but these rapidly dropped to < 1 μg/L at the end of the study (Fig. 37). The 8 m forest buffer substantially reduced the concentrations during the first event after pesticide application, from over 100 μg/L to 25 μg/L, but proved less effective in subsequent runoff events. The 8 m grass buffer also appeared to be effective, but the lack of samples during the artificial events for both 8 m buffers reduced the statistical power of these comparisons. The total amount of atrazine in runoff was also relatively unchanged after passing through the buffers (Figs. 36,38), except where the volume was largely reduced. The only significant difference among the treatments was between the 4 m and 8 m riparian buffers (Table 11).

Acetochlor followed patterns similar to atrazine. Somewhat lower concentrations were evident in the initial runoff on May 24 (Fig. 39), but the application rates were also 23% lower. The 8 m grass buffer was very effective during the first natural runoff event on

June 3, but became less effective as the rainfall became more frequent toward the end of June (Fig. 41). The concentrations in runoff dropped to < 1 μg/L at the end of the study, and residues were likely to dissipate completely as indicated by the lack of detections in the April 27 event, even though acetochlor was applied in 1999. The buffers were inconsistent in reducing total amounts of acetochlor and none reduced totals significantly (Figs. 42,44; Table 11).

Ground Water

The ground water at this site originates from a large area and is not likely to be very responsive to treatments in the 0.4 ha plot that was established for this runoff study. However, we did sample the wells which were already established at the site in order to characterize ground water during the study. In particular, we were interested in determining if pesticides were present in the ground water since the previous studies had focused on nutrients. The shallow wells, installed approximately 4 m below the surface, were often dry and this reduces our ability to compare the results from these wells to those from the 7 m wells.

Of the nutrients, NH_3-N was not detected in the wells above 0.10 mg/L except in one sample, which was right at the detection limit (Table 12). Orthophosphate was detected above 5.0 μg PO_4-P/L in about half of the samples and ranged up to 104 μg PO_4-P/L, although most of the time the concentrations were far below that. In general, wells in the field had lower levels of PO_4-P than those closer to the creek. The opposite trend was evident for NO_3-N, which was generally highest in and adjacent to the field (Fig. 31). Concentrations in the riparian buffer were usually less than half of that in the field.

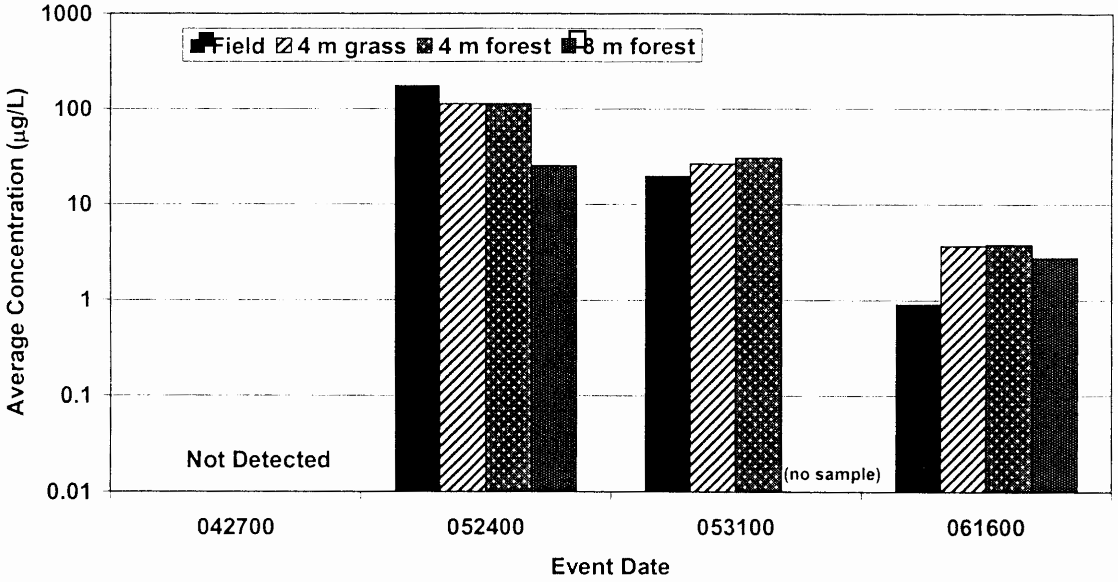


Figure 35. Atrazine concentrations in runoff for four different buffer types at the Piedmont site during artificial runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

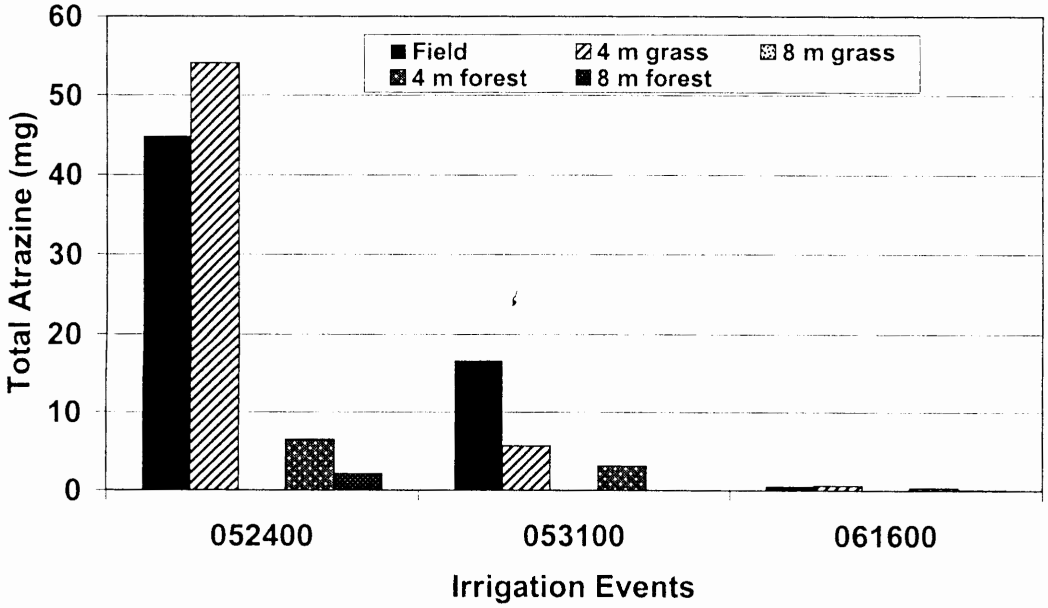


Figure 36. Total atrazine in runoff for four different buffer types at the Piedmont site during artificial runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

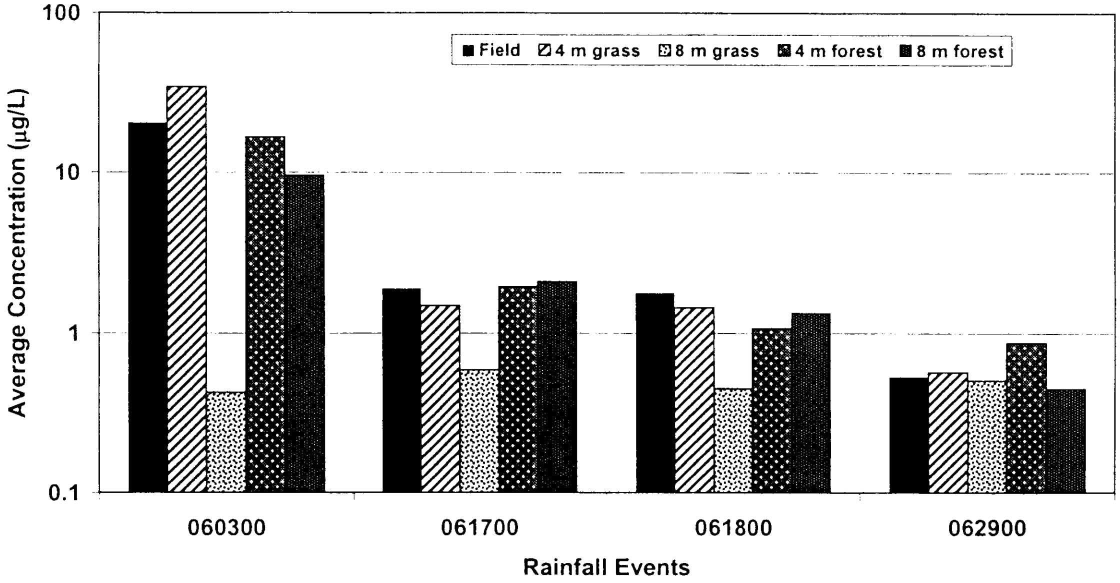


Figure 37. Atrazine concentrations in runoff for four different buffer types at the Piedmont site during natural runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

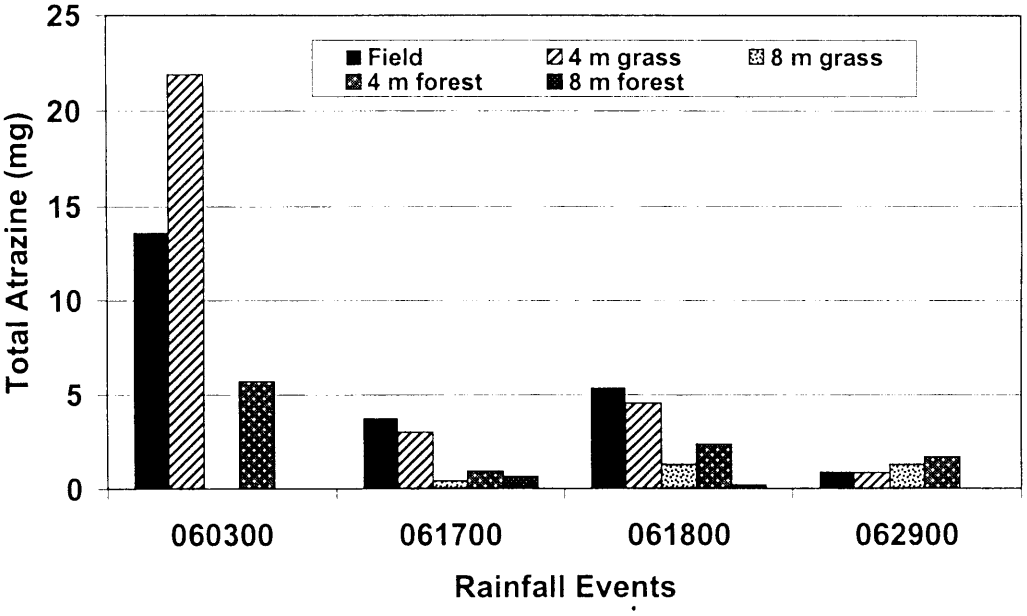


Figure 38. Total atrazine in runoff for four different buffer types at the Piedmont site during natural runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

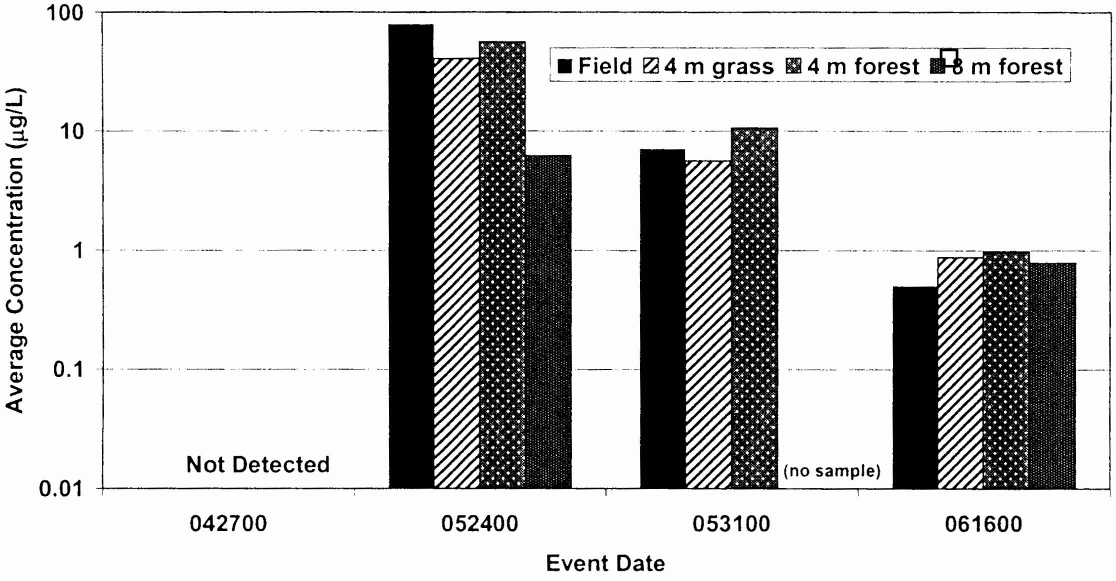


Figure 39. Acetochlor concentrations in runoff for four different buffer types at the Piedmont site during artificial runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

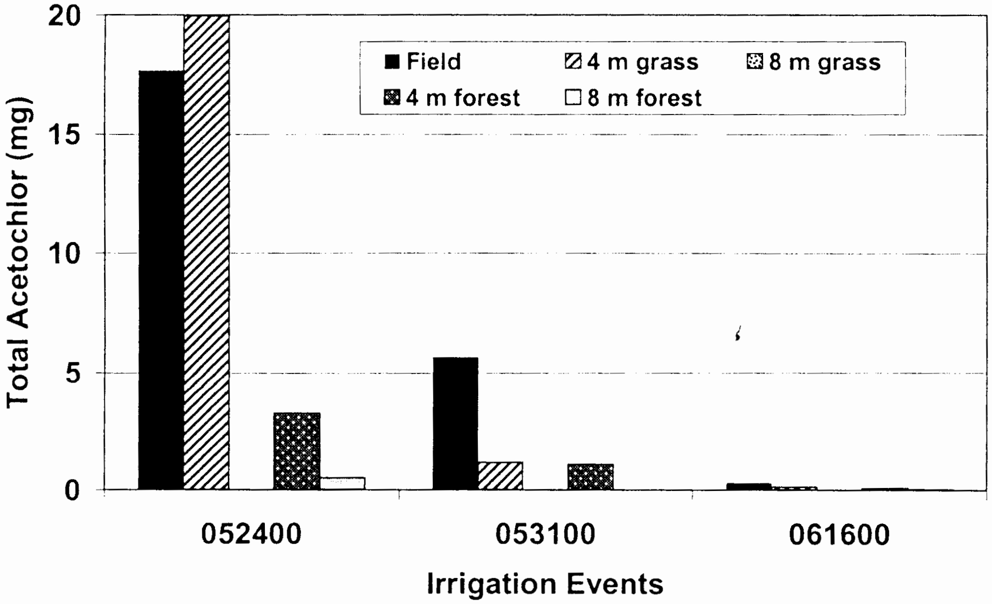


Figure 40. Total acetochlor in runoff for four different buffer types at the Piedmont site during artificial runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

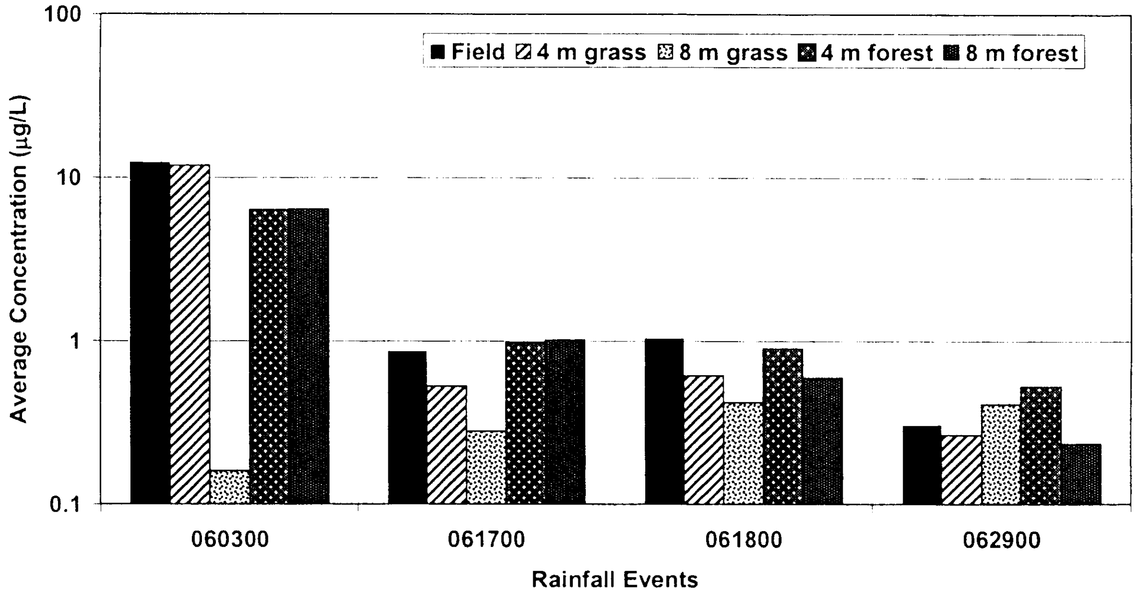


Figure 41. Acetochlor concentrations in runoff for four different buffer types at the Piedmont site during natural runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

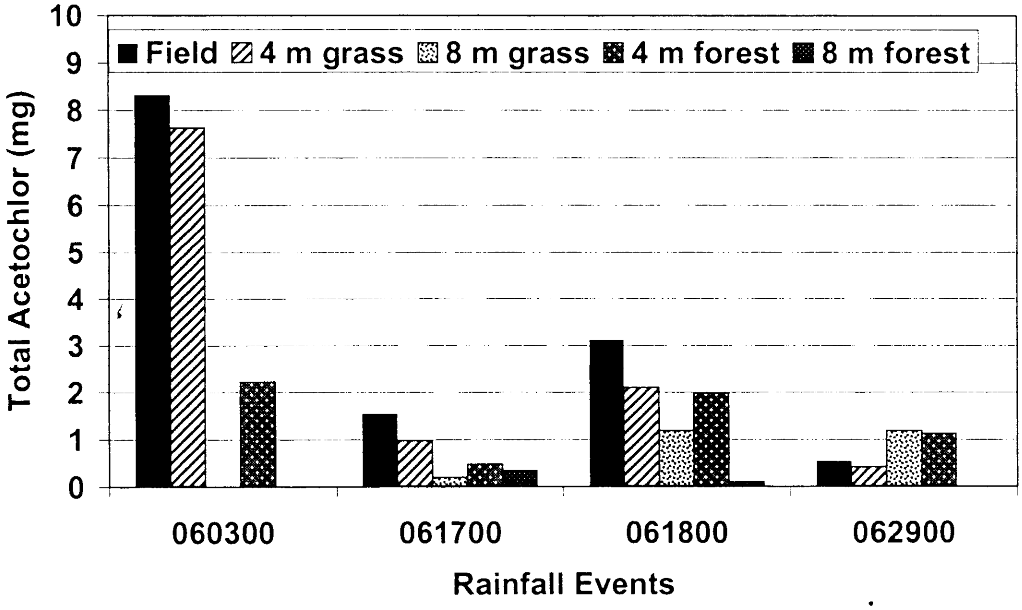


Figure 42. Total acetochlor in runoff for four different buffer types at the Piedmont site during natural runoff events. Values are averages of two plots for field and grass plots and are single plots for the forested plots.

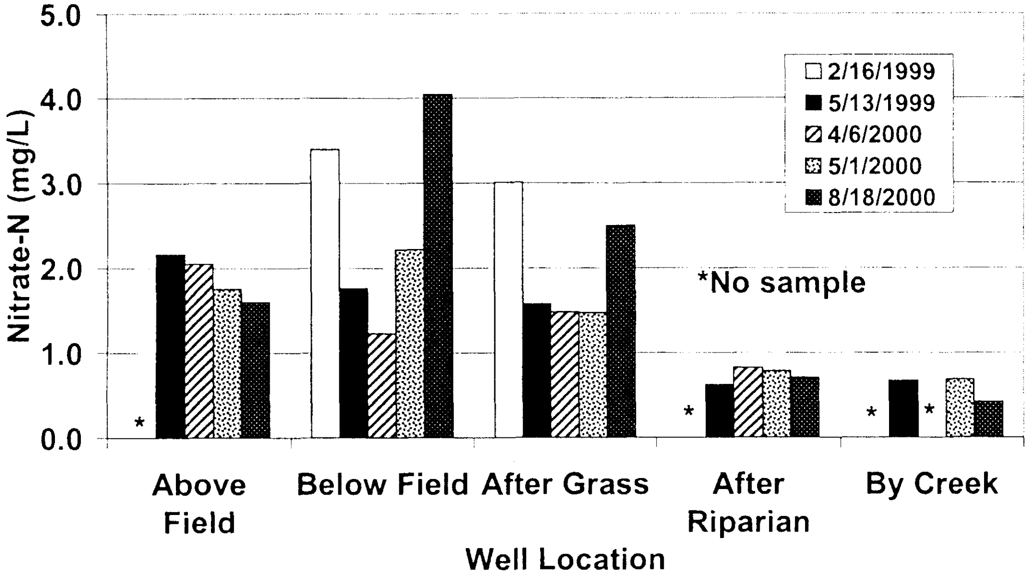


Figure 43. Nitrate in wells along a transect from above the field to the creek below the riparian buffer. Values are averages of 1-3 wells for each sample date.

Table 13. Chemical constituents in ground water samples: Piedmont site.

	Well	atrazine (μg/L)	acetochlor (μg/L)	PO$_4$–P (μg/L)	NH$_4$-N (mg/L)	NO$_3$-N (mg/L)
				Shallow Wells		
2/16/1999	AF	<0.10	<0.10	25.97	<0.05	4.00
2/16/1999	BF					
2/16/1999	AG					
2/16/1999	AR	<0.10	<0.10	50.62	<0.05	0.07
2/16/1999	CK	<0.10	<0.10	14.56	<0.05	<0.05
5/13/1999	AF					
5/13/1999	BF					
5/13/1999	AG					
5/13/1999	AR	0.16	<0.10	<5.0	<0.05	<0.05
5/13/1999	CK	0.18	<0.10	86.85	<0.05	<0.05
4/6/2000	AF	<0.10	<0.10	<5.0	<0.05	5.18
4/6/2000	BF					
4/6/2000	AG					
4/6/2000	AR	<0.10	<0.10	15.36	<0.05	0.13
4/6/2000	CK	<0.10	<0.10	61.25	<0.05	0.11
5/1/2000	AF	<0.10	<0.10	<5.0	<0.05	4.68
5/1/2000	BF					
5/1/2000	AG					
5/1/2000	AR	<0.10	<0.10	7.68	<0.05	0.15
5/1/2000	CK	<0.10	<0.10	78.82	<0.05	0.17

Table 13. Chemical constituents in ground water samples: Piedmont site.

	Well	atrazine (μg/L)	acetochlor (μg/L)	PO_4–P (μg/L)	NH_4-N (mg/L)	NO_3-N (mg/L)
8/18/2000	AF					
8/18/2000	BF					
8/18/2000	AG					
8/18/2000	AR	<0.10	0.03	12.14	<0.05	0.48
8/18/2000	CK	0.73	0.59			
			Deep Wells			
2/16/1999	AF					
2/16/1999	BF	0.05	<0.10	17.88	<0.05	3.40
2/16/1999	AG	0.05	<0.10	17.88	<0.05	3.01
2/16/1999	AR					
2/16/1999	CK					
5/13/1999	AF	0.16	<0.10	<5.0	<0.05	2.16
5/13/1999	BF	0.13	<0.10	24.09	0.21	1.76
5/13/1999	AG	0.22	<0.10	<5.0	<0.05	1.58
5/13/1999	AR	0.04	<0.10	10.14	<0.05	0.62
5/13/1999	CK	<0.10	<0.10	75.13	<0.05	0.67
4/6/2000	AF	<0.10	<0.10	<5.0	<0.05	2.05
4/6/2000	BF	<0.10	<0.10	15.67	<0.05	1.23
4/6/2000	AG	<0.10	<0.10	<5.0	<0.05	1.48
4/6/2000	AR	<0.10	<0.10	18.12	<0.05	0.82
4/6/2000	CK	<0.10	<0.10			
5/1/2000	AF	<0.10	<0.10	<5.0	<0.05	1.75
5/1/2000	BF	<0.10	<0.10	16.01	0.07	2.22
5/1/2000	AG	<0.10	<0.10	<5.0	<0.05	1.47
5/1/2000	AR	<0.10	<0.10	14.80	<0.05	0.78
5/1/2000	CK	<0.10	<0.10	70.42	<0.05	0.68
8/18/2000	AF	0.92	0.45	<5.0	<0.05	1.60
8/18/2000	BF	<0.10	<0.10	<5.0	<0.05	4.04
8/18/2000	AG	<0.10	<0.10	<5.0	<0.05	2.50
8/18/2000	AR	<0.10	0.03	9.76	<0.05	0.71
8/18/2000	CK	<0.10	<0.10	40.20	<0.05	0.42

AF=Above Field (Wells 1-3)
BF= Below Field (Wells 4-6)
AG= After Grass (Wells7,8)
AR= After Riparian (Wells 9,10,13)
CK= Adjacent to Creek (Wells 11,12)

Atrazine was detected in the wells in both 1999 and 2000 (Fig. 44), with the highest concentration of 0.92 μg/L in an above-field well in August 2000. The same sample also had the highest acetochlor detection at 0.45 μg/L (Fig. 45). No acetochlor was detected in 1999. The atrazine detections are well below the lifetime drinking water limit of 3.0 μg/L set by the US EPA.

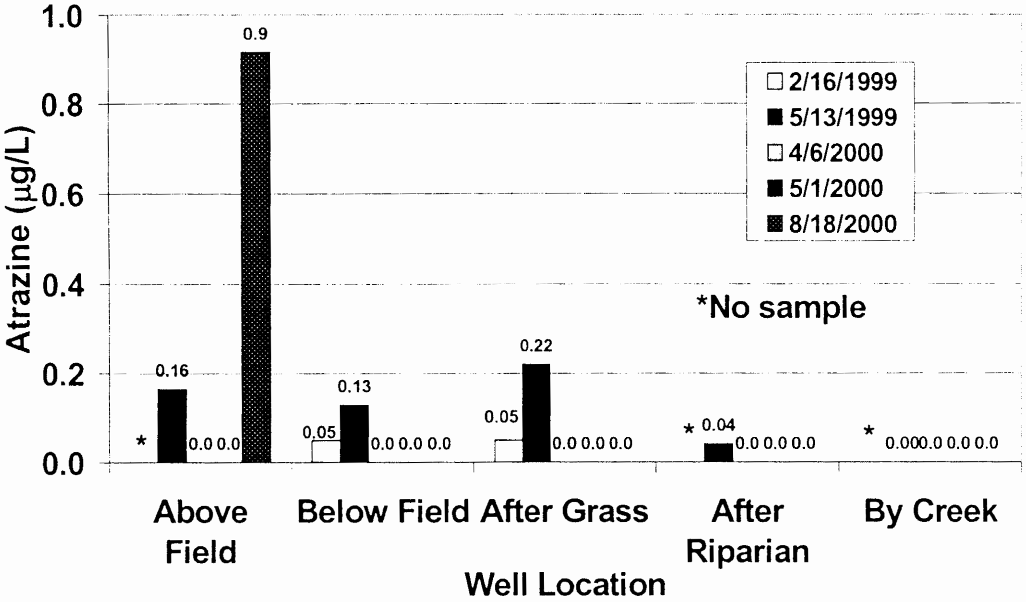


Figure 44. Atrazine in wells along a transect from above the field to the creek below the riparian buffer. Values are averages of 1-3 wells for each sample date. Zeros represent non-detections for that date.

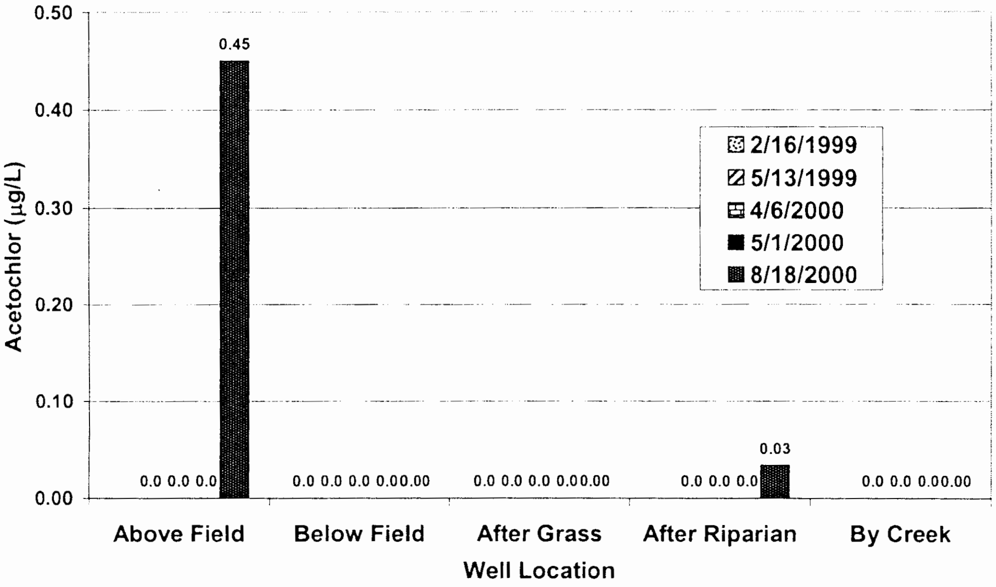


Figure 45. Acetochlor in wells along a transect from above the field to the creek below the riparian buffer. Values are averages of 1-3 wells for each sample date. Zeros represent non-detections for that date.

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