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Effects of Site Preparation for Pine Forest/Switchgrass Intercropping on Water Quality

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Abstract

A study was initiated to investigate the sustainability effects of intercropping switchgrass (*Panicum virgatum* L.) in a loblolly pine (*Pinus taeda* L.) plantation. This forest-based biofuel system could possibly provide biomass from the perennial energy grass while maintaining the economics and environmental benefits of a forest managed for sawtimber. Operations necessary for successful switchgrass establishment and growth, such as site preparation, planting, fertilizing, mowing and baling, may affect hydrology and nutrient runoff. The objectives of this study were (i) to characterize the temporal effects of management on nutrient concentrations and loadings and (ii) to use pretreatment data to predict those treatment effects. The study watersheds (~25 ha each) in the North Carolina Atlantic Coastal Plain were a pine/switchgrass intercropped site (D1), a midrotation thinned pine site with natural understory (D2), and a switchgrass-only site (D3). Rainfall, drainage, water table elevation, nitrogen (total Kjedahl N, NH_4–N, and NO_3–N), and phosphate were monitored for the 2007–2008 pretreatment and the 2009–2012 treatment periods. From 2010 to 2011 in site D1, the average NO_3–N concentration effects decreased from 0.18 to −0.09 mg L^{-1}, and loads effects decreased from 0.86 to 0.49 kg ha^{-1}. During the same period in site D3, the average NO_3–N concentration effects increased from 0.03 to 0.09 mg L^{-1}, and loads effects increased from −0.26 to 1.24 kg ha^{-1}. This study shows the importance of considering water quality effects associated with intensive management operations required for switchgrass establishment or other novel forest-based biofuel systems.

Large areas of the US coastal plain and the gulf coast are covered by pine forests managed for timber production. Many studies have documented the effects of silvicultural operations such as clear-cutting, thinning, and site preparation on nitrogen (N), phosphorus (P), total organic carbon, and total suspended sediment (Amatya and Skaggs, 2011; Amatya et al., 2006b; Amatya et al., 1998; Lebo and Herrmann, 1998; McBroom et al., 2008; Aubertin and Patric, 1974; Van Lear et al., 1985). In some studies, clear-cutting and thinning have led to increases in nutrients in stream flow (Aubertin and Patric, 1974; Arthur et al., 1998; Amatya et al., 2006b; Grace et al., 2007). Bedding was reported to create a penetrable soil environment for young pine roots, to enhance microsite drainage for young pine trees, and to increase N availability (Eisenbies et al., 2005); however, the bedding operation can reduce effective hydraulic conductivity (Skaggs et al., 2006). An increase in nutrient loads was attributed to increases in stream flow on managed pine and natural forests in the Atlantic Coastal Plain (Amatya et al., 1998, 2007; Shelby et al., 2005) and increases in the availability of nutrients in moderately disturbed soil (Eisenbies et al., 2005; Kranabetter et al., 2006). The change in stream flow has been attributed to changes in the soil properties because of soil compaction (Grace et al., 2006, 2007) and changes in microtopography and effective hydraulic conductivity (Skaggs et al., 2006). The influence of operation/compaction on saturated hydraulic conductivity (K_{sat}) and the implications on drainage was explored in detail by Grace and Skaggs (2013). The authors found differences in K_{sat} between thinned and unthinned loblolly pine (*Pinus taeda* L.) plantations and used the changes in K_{sat} values to explain differences in hydrology of the watersheds. Other potential drivers of N and P movement from a landscape include precipitation, water table elevation (Tian et al., 2012), surface topography, and distribution of root systems in the soil profile. We hypothesize that the timing and nature of silvicultural management operations and stage of plant growth will affect nutrient levels in forest drainage water.

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Abbreviations: ROC, runoff coefficient; TKN, total Kjedahl nitrogen.

Planting switchgrass (*Panicum virgatum* L.) as a cellulosic biofuel crop between wide rows of *P. taeda* beds is hypothesized to be an economically viable option. Space, nutrients, and water between pine beds can possibly be used by switchgrass to increase overall biomass production of pine forest land. The pine/switchgrass intercropping practice is also hypothesized to increase site nutrient uptake, thereby improving water quality. The effects of switchgrass planting on hydrology and water quality in agricultural landscapes have been reported elsewhere (Brown et al., 2000; Blanco-Canqui, 2010). Brown et al. (2000) observed greater water use efficiency and reduction of soil erosion by switchgrass compared with other traditional crops (e.g., corn). Albaugh et al. (2012a) studied the plot-scale impact of intercropping switchgrass and loblolly pine in Lenoir County, North Carolina and reported that plants in a switchgrass-only stand grow taller than switchgrass in the intercropped pine/switchgrass system. However, watershed-scale effects of these treatments on hydrology and nutrient cycling have not been studied. Management operations for establishing pine/switchgrass intercropping and pure switchgrass systems will affect the soil hydraulic and chemical properties and thus may affect watershed hydrology and nutrient concentrations and loads.

The paired watershed approach has been widely used to quantify the effects of forest management operations on hydrology and water quality (Ssegane et al., 2013; Bren and McGuire, 2012; McBroom et al., 2008; Amatya et al., 1998). The control and treatment watersheds are selected such that both watersheds are similar in terms of soil properties, drainage type, topography, and location and are monitored at the same time (Amatya et al., 2000; Ssegane et al., 2013; Ssegane et al., 2015). In a companion study at the same site, Ssegane et al. (2015) hypothesized that silvicultural operational practices undertaken for pine and switchgrass establishment may act as external factors, potentially shifting hydrologic calibration relationships.

This study was conducted on the same research watersheds in Carteret County, North Carolina that had been previously used for the paired watershed research described above. In this study, data collected in 2007–2008 (Amatya and Skaggs, 2011; Tian et al., 2012) were used for pairwise pretreatment calibration. Watersheds D1 (pine intercropped with switchgrass) and D3 (switchgrass only) were 35-yr-old pine stands before harvest (2009), site preparation, and treatment; D1 was then planted in intercropped pine and switchgrass and D3 in switchgrass only. The control watershed (D2) was a midrotation mature pine forest with natural understory that was thinned in 2009 at the age of 12 yr. Although D2 was younger than D1 and D3 during the calibration period, all watersheds were planted at the same density and reached canopy closure, allowing the relationships to be used as predictive parameters. Thinning on D2 was assumed to have minimal effect on nutrient dynamics (Gurlevik et al., 2004; Eisenbies et al., 2005; Amatya and Skaggs, 2008) in comparison to the intensive harvest and site preparation on D1 and D3. The objectives of this study were to examine possible intercropping and energy crop treatments by (i) characterizing the temporal effects of management operation treatments on nutrients and (ii) quantifying those treatment effects using predictive equations developed with pretreatment data following a paired watershed approach.

Materials and Methods

Site Description

The experimental watersheds for this study are located on Weyerhaeuser Company land in Carteret County, NC (34°49′ N, 76°40′ W). The watersheds D1, D2, and D3 are artificially drained by parallel lateral ditches (1.4–1.8 m deep) with a spacing of 100 m. The drainage areas for D1, D2, and D3 are 26.3, 25.9, and 27.1 ha, respectively. The north, south, and western sides are forest dominated, and the east side is agricultural. Surface flow between adjacent watersheds is prevented by raised beds (∼0.4 m) parallel to watershed boundaries. The sites' topography is characterized by flat coastal plain, 0.1% gradient, and a 3-m elevation above sea level (McCarthy et al., 1991). The sites' soils (Deloss fine sandy loam soil, which is a fine-loamy, mixed, thermic Typic Umbraquult) have poor drainage with shallow water tables and a pH range of 3.5 to 4.5 (acidic) (Amatya et al., 1998; Beltran et al., 2010). The texture is fine sandy loam from 0 to 50 cm and clay loam at depths below 50 cm, with average hydraulic conductivity of 3.9 m d^{-1}, a drainable porosity of 0.05 m m^{-1}, a saturated water content of 0.43 m^3 m^{-3}, and a wilting point water content of 0.22 m^3 m^{-3} (Beltran et al., 2010).

Silvicultural Management Operations for Switchgrass Establishment

Silvicultural management operations, including harvesting, shearing, bedding, and raking, and the dates when they were implemented on D1 and D3 are shown in Table 1. Site D1 was harvested and then underwent shearing and bedding. Shearing clears the rows for bedding and was performed with a V shear blade mounted on the front of a crawler tractor, and bedding was performed by a special harrow pulled behind a crawler tractor to create elevated ridges or beds that provide good drainage conditions for pine seedlings. Pine seedlings were planted by hand at a density of 1087 trees ha^{-1} in rows 6.1 m apart. About 10 mo after planting, the areas between the rows of trees were sheared in preparation for switchgrass establishment. Broadleaf herbicide was applied 2 wk before switchgrass seeds were planted in a 3.05-m-wide area centered between the edges of pine beds using a modified LandPride planter. Poor germination rates occurred at the site due to heavy rains from Hurricane Irene less than 2 wk after planting. Broadleaf herbicide was again applied at the end of 2011, and switchgrass was replanted in April 2012.

The trees at D3 were harvested later than at D1, with 85% harvested 6 mo later than D1 and the final 15% harvested after an additional 5-mo delay due to excessive wet conditions. The entire field area was then sheared, and the debris was raked into windrows using a root rake attached to the front of a crawler tractor. Switchgrass seeds were planted over the field using the modified LandPride planter 2 wk after a broadleaf herbicide application. As at D1, poor germination rates at D3 occurred due to heavy rains from Hurricane Irene, and broadleaf herbicide was again applied at the end of 2011. Switchgrass was replanted in April 2012.

The only operation performed on D2 was thinning at the end of 2008. The trees were thinned from a density of 1087 to 346 trees ha^{-1} to reduce competition for water and nutrient resources for the remaining trees and thus achieve the optimum growth.

Although thinning can increase flow and water table elevation, soil disturbance is minimal, and no materials are incorporated into the soil. The effect of thinning at D2 on nutrients is very likely minimal compared with the effect of significant disturbances that occurred on D1 and D3. Site D2 was assumed to be a control even though thinning had occurred at D2 a year before major operations began to occur at D1 and D3.

Measurements of Rainfall, Water Table, and Flow

Rainfall was measured using tipping-bucket rain gauges (HOBO) backed up by manual gauges, and daily totals were computed. Water table elevations were recorded hourly by stage recorders (In situ Level TROLL 500) in shallow wells at the front and rear of the field plots, and daily average was used as the representative water table elevation. Drainage flow values were calculated from upstream and downstream stages measured by recorders (In situ Level TROLL 500) and recorded by a data logger (CR200, Campbell Scientific) at 12-min intervals at a 120° V-notch weir (Amatya and Skaggs, 2011), and daily totals were computed. A downstream pump was activated under high water conditions to minimize weir submergence (Amatya et al., 1996).

Water Quality Sampling and Analysis

An automatic sampler (SIGMA 900) was located at each flow monitoring outlet. The stage data recorders calculated cumulative flow and collected 150 mL of drainage water for every 200 m^3 over the 120° V-notch weir, equivalent to about 0.8 mm of watershed area–based water depth. The flow proportional–composite samples were collected approximately every 2 wk from November 2009 to March 2012. Acidic conditions (water pH of 4 for this study) inhibit nitrification and therefore preserved the collected samples before analysis. During the summer months, no-flow and low-flow periods were common, during which there was not enough sample for analysis. Grab samples were also collected when the sampler was serviced. All the samples were analyzed for NH_4–N, NO_3–N, total Kjedahl N (TKN), and phosphate. Total Kjedahl N was analyzed using acid digestion method (USEPA standard method 4500 Norg B [USEPA, 1998]), NO_3–N was measured using cadmium reduction (USEPA standard method 4500 NO3-E [USEPA, 1998]), and NH_4–N was measured using ammonium salicylate method (USEPA standard method-4500 NH_3G [USEPA, 1998]). Phosphate was measured using ascorbic acid method (USEPA standard method-4500 P-F [USEPA, 1998]). A Bran Luebbe Autoanalyzer II (SEAL Analytical Inc.) was used for N analysis. The detection limits for NH_4–N, NO_3–N, TKN, and phosphate were 0.01, 0.01, 0.04, and 0.01 mg L^{-1}, respectively.

Statistical Analyses and Paired Watershed Approach

Total N was calculated as the sum of TKN and NO_3–N. Organic N was calculated as TKN minus NH_4–N. Inorganic N was calculated as NH_4–N plus NO_3–N. Because flow proportional composite sampling was used, the concentration of the composite sample was multiplied by the flow volume to calculate load over the service interval, and annual loads were totaled. Site D2 was missing flow data from 25 Sept. 2010 to 20 Oct. 2010; missing data were filled in using linear regression with D1 data (R^2 = 0.90). Because a paired watershed approach was used and nutrients were monitored at the same time, a paired *t* test (MS-Excel Software) was used to identify significant differences ($\alpha = 0.05$) in annual nutrient concentrations between treatment and control watersheds and between two treatments.

A paired watershed approach was used to develop calibration equations between each of the treatment and control watershed pairs. A bootstrap geometric regression that is nonparametric (Zhu and Jing, 2010; Plotnick, 1989; Ricker, 1973; Hardle and Bowman, 1988) was used using 2007–2008 pretreatment data. The calibration equations were used to calculate expected concentrations and loads for the treatment period (Nov. 2009–Mar. 2012) based on measured data for D2. Treatment effect was calculated as the difference between measured and expected concentrations and loads. The effects were also plotted over the treatment period. Biweekly composite nutrient concentrations and loads were used for bootstrap regression to capture variability on a biweekly basis. Dates without measured concentrations but

Table 1. Silvicultural management operations undertaken on watersheds from 2009 to 2012.

Watershed†	Silvicultural management practice	Dates
D1	harvesting pine trees (clear cut harvest)	5 Jan.–1 Apr. 2009
	shearing and bedding	13 June–30 July 2009
	pine planting (1087 trees ha^{-1})	18–20 Jan. 2010
	shearing within the pine rows	16–17 Dec. 2010
	broad leaf weed control	1 Aug. 2011
	switchgrass planting (seeds were broadcasted)	15 Aug. 2011
	broad leaf weed control within pine beds	late 2011
	switchgrass reseeding (seeds were broadcasted)	9 Apr. 2012
D2	thinning from 1087 trees ha^{-1} to 346 trees ha^{-1}	18 Dec. 2008–9 Jan. 2009
D3	85% harvesting pine trees (clear cut harvest)	19 Oct.–30 Nov. 2009
	100% harvesting pine trees	30 Apr.–10 May 2010
	shearing and raking	12–19 Apr. 11
	raking to wind rows	20 Apr.–2 May 2011
	broad leaf weed control	1 Aug. 2011
	switchgrass planting (seeds were broadcasted)	15 Aug. 2011
	broad leaf weed control	late 2011
	switchgrass reseeding (seeds were broadcasted)	9 Apr. 2012

† D1, pine/switchgrass intercropped; D2, midrotation thinned pine with natural understory; D3, switchgrass only.

with measured flow were filled with the measured concentration in the subsequent sample. Only observations with samples from treatment and control watersheds were used for bootstrap geometric linear regression. A few observations assumed to be the outliers, particularly at D3, were removed for the regression. To identify outliers in a calibration data set, lower 25% percentile and upper 75% interquartile range values of the data were calculated first. Values below or above the upper percentile value plus three times the difference of the interquartile range were outliers (NIST/SEMATECH, 2014). With this assumption, nutrient samples used for bootstrap geometric linear regression were 28 and 30 for the D1-D2 and the D3-D2 watershed pairs, respectively. General linear models and ANCOVA procedures available in SAS 9.4 version (SAS Institute, 2012–2013) were used to identify significant differences between calibration and treatment equations and their parameters as performed by Beltran et al. (2010) and Bishop et al. (2005). In ANCOVA, the response variable from D2 was the covariate, and the change in treatment watershed response was determined.

Results and Discussion

Characterizing Temporal Effects of Silvicultural Management Operations on Hydrologic Values

Because nutrient loads are driven by the drainage flow as a result of rainfall response, we examined the pattern of runoff coefficient (ROC) defined as flow volume divided by the rainfall amount. The measured annual rainfall, flow, and average water table elevation and the ROC values for 2010 and 2011 are presented in Table 2. Although there may be some uncertainties in flow estimates from measured stages (upstream and downstream), particularly during weir submergence as noted by Amatya and Skaggs (2011), the ROC values for D2 reduced and the values for D3 increased from 2010 to 2011. The ROC for D1 was the same in 2010 and 2011. For all the sites, water table elevation decreased from 2010 to 2011 (Table 2).

Site D2, which was thinned in 2009, showed a decrease in annual ROC from 2010 to 2011 with pine and understory regrowth. Lynch and Corbett (1990) observed a similar increase in water yield for the first year after thinning and reductions in the following years due to understory regrowth. This was consistent with the pattern of flow increase then decrease to prethinned flow levels shown after the 2002 thinning on D3 (Amatya and Skaggs, 2008). The ROC for D3 increased from 2010 to 2011. This could be due to the effects of site preparation and cultural activities subsequent to the fallow period (May 2010–Apr. 2011) followed by the poor germination of the switchgrass seeds broadcasted in August 2011. Amatya et al. (2006b) reported greater ROC for a harvested site (D2) compared with the control (D1) in the year of harvest and the year after harvest. Increases in water table elevation after pine harvesting in the Atlantic Coastal Plain sites have been reported elsewhere (Amatya et al., 2006b; Xu et al., 2002; Sun et al., 2000).

Characterizing and Quantifying Temporal Effects of Silvicultural Management Operations on Nutrients

The timing of NO_3–N concentrations and loads differed among the treatments (Fig. 1). High NO_3–N concentrations and loadings occurred in late 2009 and early 2010 on D1, reflecting the occurrence of clear-cut harvesting and site preparation. The trend was different at D3, where lower NO_3–N concentrations and loads occurred in late 2009 and early 2010 and then increased through late 2010 and 2011. The trend at D3 reflects the later and prolonged tree harvest (late 2009 through May 2010) and site preparation in April 2010. Nitrate nitrogen concentrations and loads observed at D2 were low over the entire study period and were reduced to the levels observed at D2 by late 2011 at D1 and D3. Similar trends were not observed for TKN (Fig. 2) and phosphate concentrations (graphs not shown).

The annual mean TKN and NO_3–N concentrations for D1 were significantly ($p < 0.05$) greater than D2 values in 2010 and 2011 (Tables 3 and 4). The annual mean TKN concentration for D3 was significantly ($p < 0.05$) greater than D1 in 2011 (Tables 3 and 4). Phosphate annual mean concentrations for D2 were significantly ($p = 0.05$) greater than D3 concentration in 2011 (Tables 3 and 4). Other nutrient relationships were not significantly different (Tables 3 and 4). The organic N, inorganic N, and total N loads followed the trend D1 > D3 > D2 in 2010 and D3 > D1 > D2 in 2011 (Table 5). Very large rain events occurred on 30 Sept. 2010 (150 mm) and 27 Aug. 2011 (180 mm), resulting in export of large nutrient loads (Fig. 1 and 2), consistent with Shelby et al. (2005) for another coastal North Carolina site. Summer 2010 (June–Aug.) received relatively lower (321.1 mm) rainfall than summer 2011 (420.0 mm).

The slopes for NO_3–N concentration for the D1-D2 and D3-D2 treatment relationships (Eq. [1] and [3]) were significantly ($p < 0.05$) greater than for calibration (Eq. [2] and [4]).

$$\text{Treatment: } D1 = 11.87D2 + 0.06\ (R^2 = 0.83) \quad [1]$$

$$\text{Calibration: } D1 = 1.80D2 + 0.12\ (R^2 = 0.56) \quad [2]$$

Table 2. Annual rain, average water table elevation, flow, runoff coefficient, and percent of flow time occurred.

Watershed†	Year	Rain	WTE‡	Flow	ROC§	Flow time
		mm	m	mm		%
D1	2010	1484.7	1.53	379.3	0.26	39
D2	2010	1412.7	1.35	434.4	0.31	41
D3	2010	1420.3	1.51	239.0	0.17	47
D1	2011	1219.3	1.48	313.8	0.26	53
D2	2011	1177.5	1.29	263.9	0.22	53
D3	2011	1117.8	1.41	249.8	0.22	61

† D1, pine/switchgrass intercropped; D2, midrotation thinned pine with natural understory; D3, switchgrass only.

‡ Water table elevation.

§ Runoff coefficient.

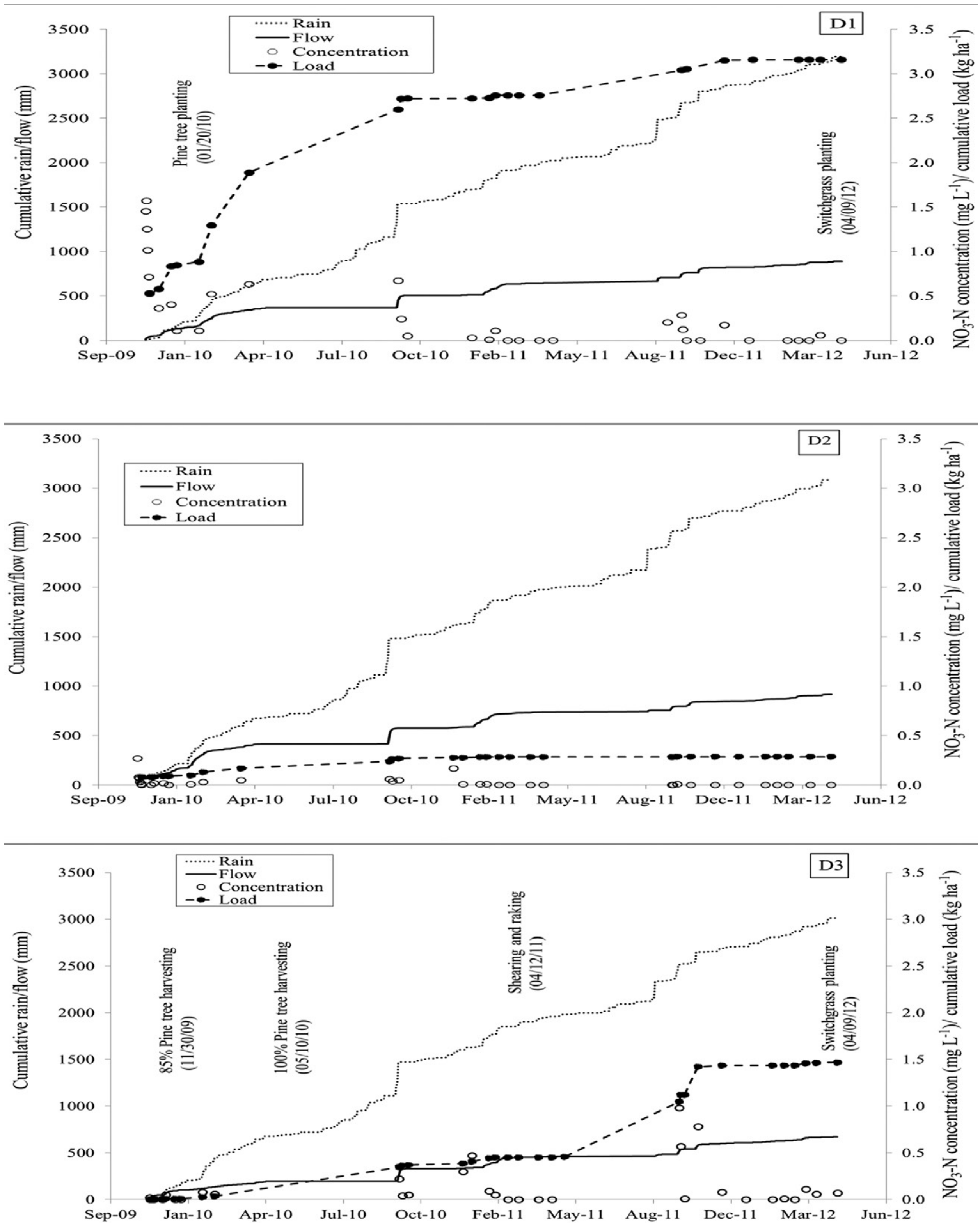

Fig. 1. Nitrate nitrogen concentration, cumulative load, cumulative rain, and cumulative flow as a function of time for D1 (pine/switchgrass intercropped site), D2 (midrotation thinned pine with natural understory), and D3 (switchgrass only) during the treatment period with management operations.

$$\text{Treatment: D3} = 2.64\text{D2} + 0.03\ (R^2 = 0.70) \quad [3]$$

$$\text{Calibration: D3} = 0.69\text{D2} + 0.05\ (R^2 = 0.75) \quad [4]$$

The nitrate loads slopes for the D1-D2 and D3-D2 treatment relationships (Eq. [5] and [7]) were significantly ($p < 0.05$) greater and lower, respectively, than calibration (Eq. [6] and [8]).

$$\text{Treatment: D1} = 7.51\text{D2} + 0.02\ (R^2 = 0.83) \quad [5]$$

$$\text{Calibration: D1} = 5.73\text{D2} - 0.02\ (R^2 = 0.43) \quad [6]$$

$$\text{Treatment: D3} = 2.31\text{D2} + 0.001\ (R^2 = 0.86) \quad [7]$$

$$\text{Calibration: D3} = 5.08\text{D2} - 0.02\ (R^2 = 0.71) \quad [8]$$

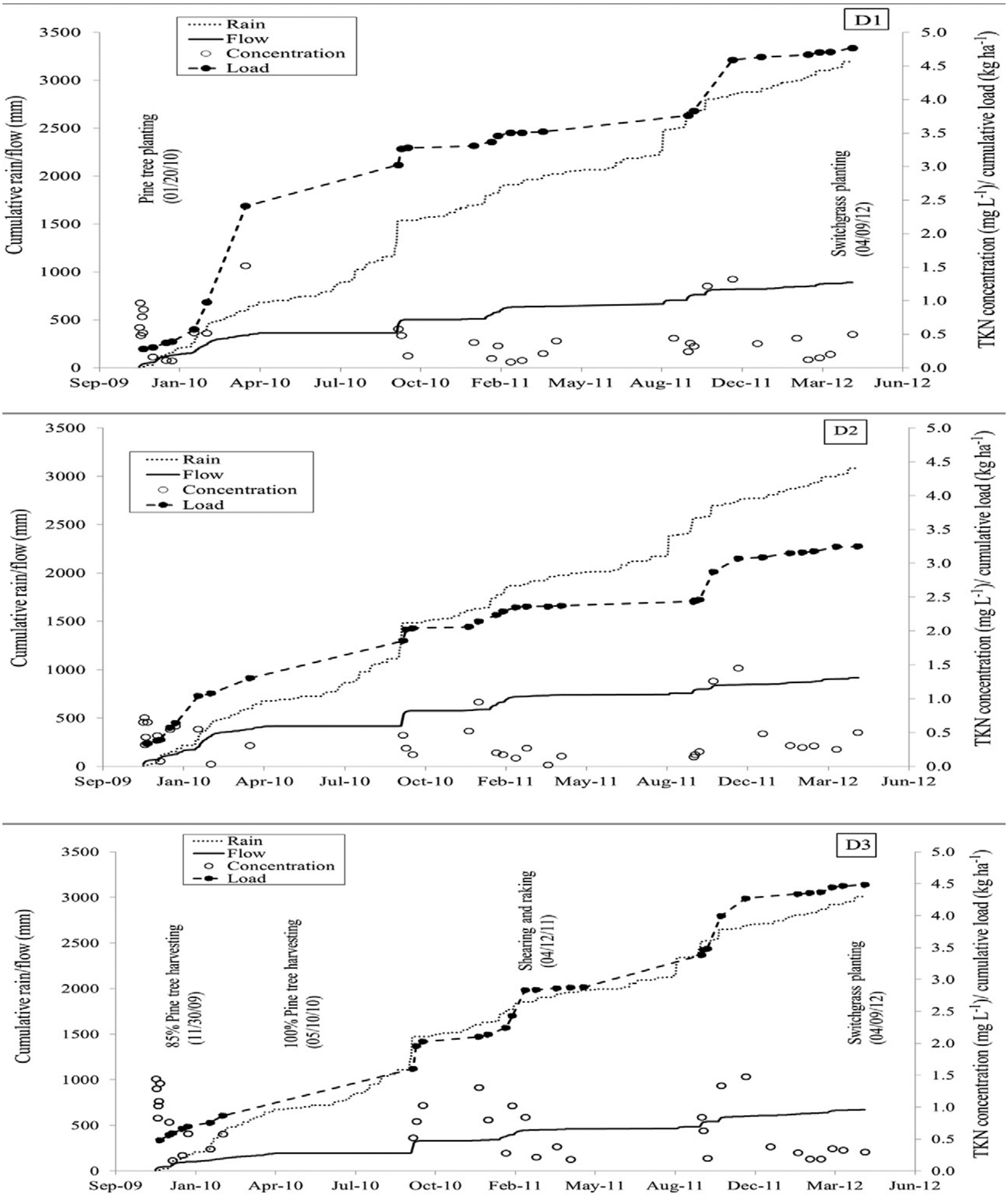

Fig. 2. Total Kjedahl nitrogen (TKN) concentration, cumulative load, cumulative rain, and cumulative flow as a function of time for sites D1 (pine/switchgrass intercropped), D2 (midrotation thinned pine with natural understory), and D3 (switchgrass only) during the treatment period with management operations.

The D1-D2 and D3-D2 phosphate loads relationship slopes for treatment (Eq. [9] and [11]), however, were significantly ($p < 0.05$) lower than calibration (Eq. [10] and [12]).

Treatment: $D1 = 0.81D2 + 0.001$ ($R^2 = 0.84$) [9]

Calibration: $D1 = 2.52D2 - 0.001$ ($R^2 = 0.55$) [10]

Treatment: $D3 = 0.65D2$ ($R^2 = 0.86$) [11]

Calibration: $D3 = 2.88D2 - 0.001$ ($R^2 = 0.68$) [12]

The slopes and intercepts for TKN concentrations and loads and phosphate concentrations were not significantly ($p > 0.05$) different from calibration slopes (equations not shown) for D1-D2 or D3-D2. Figure 3 shows the distribution of NO_3–N and phosphate effects with average and total effects for individual years (2010 and 2011) and for the entire treatment period. The average TKN concentration effects were 0.06 ± 0.07 mg L^{-1}

Table 3. Annual mean concentrations and ranges for nitrogen and phosphorus.

Year	Watershed†	TKN‡	NH_4–N	NO_3–N	PO_4–P	Organic N	Inorganic N	Total N
		mg L^{-1}						
2010	D1	0.63 (0.18–1.52)§	0.02 (0.01–0.05)	0.370 (0.05–0.67)	0.04 (0.00–0.10)	0.61 (0.17–1.51)	0.39 (0.06–0.68)	1.00 (0.23–2.15)
2010	D2	0.33 (0.03–0.55)	0.02 (0.00–0.05)	0.060 (0.01–0.17)	0.05 (0.00–0.10)	0.31 (0.00–0.52)	0.08 (0.05–0.17)	0.39 (0.06–0.69)
2010	D3	0.76 (0.35–1.31)	0.01 (0.00–0.02)	0.130 (0.04–0.30)	0.04 (0.00–0.08)	0.75 (0.34–1.31)	0.14 (0.06–0.30)	0.89 (0.43–1.61)
2011	D1	0.46 (0.09–1.32)	0.02 (0.00–0.18)	0.030 (0.00–0.17)	0.09 (0.07–0.11)	0.44 (0.09–1.30)	0.05 (0.00–0.19)	0.49 (0.09–1.49)
2011	D2	0.51 (0.02–1.45)	0.01 (0.00–0.06)	0.003 (0.00–0.01)	0.10 (0.07–0.15)	0.50 (0.01–1.45)	0.01 (0.00–0.06)	0.51 (0.02–1.45)
2011	D3	0.69 (0.18–1.48)	0.01 (0.00–0.04)	0.150 (0.00–0.78)	0.08 (0.05–0.11)	0.68 (0.18–1.44)	0.16 (0.00–0.80)	0.84 (0.18–2.12)

† D1, pine/switchgrass intercropped; D2, midrotation thinned pine with natural understory; D3, switchgrass only.

‡ Total Kjedahl nitrogen.

§ Values are means with ranges in parentheses.

Table 4. Comparisons for nitrogen and phosphorus concentrations.

Comparisons (watersheds)†	Year	NH_4–N	NO_3–N	TKN‡	Phosphate
		mg L^{-1}			
D1 vs. D2	2010	NS§	S¶	S	NS
D1 vs. D3	2010	NS	NS	NS	NS
D2 vs. D3	2010	NS	NS	NS	NS
D1 vs. D2	2011	NS	S	S	NS
D1 vs. D3	2011	NS	NS	S	NS
D2 vs. D3	2011	NS	NS	NS	S

† D1, pine/switchgrass intercropped; D2, midrotation thinned pine with natural understory; D3, switchgrass only.

‡ Total Kjedahl nitrogen.

§ No significant difference ($\alpha = 0.05$; paired *t* test).

¶ Significant difference ($\alpha = 0.05$; paired *t* test).

Table 5. Annual loads per unit area for nitrogen and phosphorus.

Year	Watershed†	TKN‡	NH_4–N	NO_3–N	PO_4–P	Organic N	Inorganic N	Total N
		kg ha^{-1}						
2010	D1	2.92	0.09	1.88	0.19	2.83	1.97	4.80
2010	D2	1.34	0.08	0.18	0.22	1.26	0.26	1.52
2010	D3	1.43	0.04	0.39	0.03	1.39	0.43	1.82
2011	D1	1.33	0.04	0.43	0.28	1.29	0.47	1.76
2011	D2	0.97	0.03	0.01	0.26	0.94	0.04	0.98
2011	D3	2.17	0.17	1.03	0.19	2.00	1.20	3.20

† D1, pine/switchgrass intercropped; D2, midrotation thinned pine with natural understory; D3, switchgrass only.

‡ Total Kjedahl nitrogen.

on D1 and 0.33 ± 0.06 mg L^{-1} on D3, and total TKN loads effects were 0.55 ± 0.50 kg ha^{-1} on D1 and 0.71 ± 2.42 kg ha^{-1} on D3 during site preparation. The average phosphate concentration effects during site preparation period were 0.03 ± 0.0004 mg L^{-1} on D1 and −0.02 ± 0.003 mg L^{-1} on D3. Nitrogen components, unlike phosphate, resulted in a positive treatment effect. There were significant ($p < 0.05$) increases in D1 NO_3–N concentrations and loads and D3 TKN loads effects and significant ($p < 0.05$) decreases in phosphate loads for D1 and D3.

Greater TKN and NO_3–N concentrations were recorded after long dry periods probably due to the first flush phenomenon associated with the first rainfall events, as reported elsewhere (Amatya et al., 1998; David et al., 2003). Greater flow volumes were recorded for D1 compared with D2 in 2011 and for D1 compared with D3 in 2010 and 2011. Because D1 and D2 had greater flow volumes than D3 in 2010, greater NH_4–N loads were recorded on D1 and D2. Site D3 had the lowest flow volume in 2011, and the plant materials for site D2 from the 2009 thinning were not incorporated into soil. Seasonal variations in rainfall, flow, flush effect, and plant development were reported to influence the temporal changes of nutrients in past studies (Albaugh et al., 2012b; David et al., 2003; Arheimer et al., 1996; Beltran et al., 2010; Gurlevik et al., 2004; Li et al., 2003; Amatya et al., 2006a, 2006b).

The greatest NO_3–N concentration was observed in November 2009 on D1 (Fig. 1) likely due to high amounts of residual plant materials from the early 2009 harvest and bedding. Bedding influence was also a probable reason for greater NH_4–N loads on D1 than on D3 and D2 in 2010 (Table 5). Greater slopes for D1–D2 NO_3–N concentration and load relationships than for D3–D2 relationships show that the changes of NO_3–N over time compared with D2 are greater for D1 than D3. Eisenbies et al. (2005) reported that bedding increases N in soil and tree productivity and enhances microsite drainage. MacKenzie et al. (2005) also associated greater total N with bedding practices

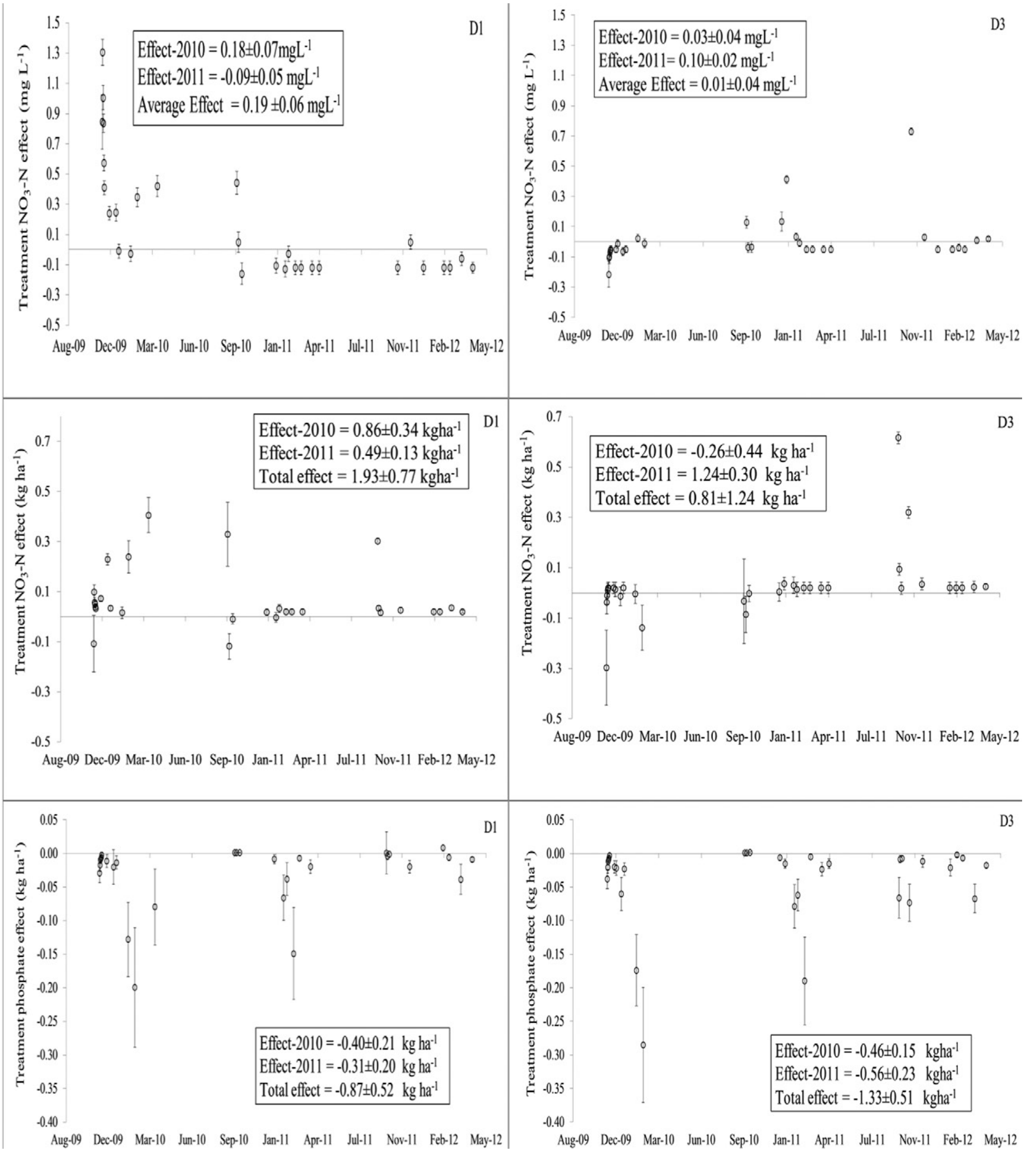

Fig. 3. Treatment effects on nitrate nitrogen concentrations, nitrate nitrogen loads, and phosphate loads. Error bars represent SE.

that incorporated organic matter into the soil in comparison to a 10-yr-old control stand. Increases in N concentrations after harvesting were attributed to the mineralization of materials from harvested plants (i.e., the “assart” effect). The assart effect and increases in temperature coupled with decreases in soil moisture associated with reducing vegetation have been documented elsewhere (Li et al., 2003; Lynch and Corbett, 1990; Arthur et al., 1998). Optimization of conditions (temperature and moisture) for plant material decomposition after clear cutting were also reported by Blair and Crossley (1988). The greater NO_3–N and NH_4–N loads in 2011 relative to 2010 for D3 were attributed to the gradual harvesting, which may have delayed the assart effect. Treatment effects with increased N in 2009 and 2010 on D1 could be attributed to greater decomposition rates than plant uptake rates; the reverse is true for negative treatment effects with reduced N in late 2010 and 2011. Lynch and Corbett (1990) also reported increased stream water nutrients after harvesting due to rates of decomposition exceeding plant uptake and soil exchange capacity and subsequently reduced due to dilution effects and plant uptake.

Decreased NO_3–N concentrations from 2009 to 2011 on D1 and D2 (Table 3; Fig. 1) and decreasing NO_3–N concentration treatment effects for D1 from 2009 to 2012 were also attributed to increased plant uptake and reduction in the quantity of mineralizable plant materials. Other research found increased plant uptake (Li et al., 2003) and N immobilization (Gurlevik et al., 2004) associated with increases in loblolly pine stand development. Reduced plant uptake might have led to greater NO_3–N concentrations on D3 due to site preparation for switchgrass establishment in 2011. Flavel and Murphy (2006) reported that age, lignin, and cellulose content of plant materials influence mineralization rates, with aged materials being of lower quality than young materials. In an earlier study at this study site with D1 as control and D2 as treatment, Amatya et al. (2006b) observed an increase in NO_3–N and TKN concentrations on D2 soon after harvest and the following year. The authors also reported that after the harvest, NO_3–N on the treatment was greater than on the mature pine control watershed. It is also important to note that Amatya et al. (1998, 2006b) reported that mean concentrations of NO_3–N (D1 = 0.93 mg L^{-1}; D2 = 0.60 mg L^{-1}; D3 = 0.23 mg L^{-1}) and TKN (D1 = 1.34 mg L^{-1}; D2 = 0.91 mg L^{-1}; D3 = 0.55 mg L^{-1}) concentrations were greater for D1 than for D2 and D3 during the 1988–1990 pretreatment period when all watersheds were on 14- to 15-yr-old pine stands. Phosphate temporal trends seem to have not been affected by type and timing of management operations (Fig. 3) compared with N temporal patterns. This observation may be explained by the fact that soils for all sites were acidic and that fixation of P by iron and aluminum was active such that there was a slow release of P for plant uptake. Fixation of P by aluminum was reported by Fox et al. (1990).

Summary and Conclusions

Management operations, changes in vegetation cover through time, flow dynamics, and large storms during the switchgrass establishment period determined the temporal variations of nutrient loads. The greatest loads were recorded during large summer and fall storm events of the study period, which occurred after long dry periods. The trend of high NO_3–N concentrations and loadings in late 2009 and early 2010 at D1 reflected the occurrence of clear-cut harvesting and site preparation in 2009. The trend of NO_3–N levels at D3 reflects the gradual tree harvest (late 2009 through May 2010) and site preparation in April 2010. Nitrate nitrogen levels in late 2011 and early 2012 were reduced at D1 and D3. However, P treatment effects showed reduced concentrations and loads in drainage water. Although management operations (e.g., harvesting, shearing between pine rows, raking, and bedding) implemented for pine and switchgrass establishments can lead to increases of N, the magnitude of increased values was lower than those usually observed on agricultural drainage waters in the region and also did not exceed the drinking water standards. The results of this study may have implications on evaluating the effects of intensive bioenergy practices that could be implemented on the Atlantic Coastal Plain.

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