Nitrogen Fate in Drainage Ditches of the Coastal Plain after Dredging

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Drainage ditches are a key conduit of nitrogen (N) from agricultural fields to surface water. The effect of ditch dredging, a common practice to improve drainage, on the fate of N in ditch effluent is not well understood. This study evaluated the effect of dredging on N transport in drainage ditches of the Delmarva Peninsula. Sediments from two ditches draining a single field were collected (0–5 cm) to represent conditions before and after dredging. Sediments were packed in 10-m-long recirculating flumes and subjected to a three-phase experiment to assess the sediment’s role as a sink or source of ammonium (NH₄⁻) and nitrate (NO₃⁻). Under conditions of low initial NH₄⁻ and NO₃⁻ concentrations in flume water, sediment from the undredged ditch released 113 times more NO₃⁻ than water than did sediment from the dredged ditch. When flume water was spiked with NH₄⁻ and NO₃⁻ to simulate increases in N concentrations from drainage and runoff from adjacent fields, NO₃⁻ in flume water increased during 48 h compared with the initial spiked concentration, while NH₄⁻ decreased. These simultaneous changes were attributed to nitrification, with 23% more NO₃⁻ observed in flume water with undredged ditch sediment compared with dredged ditch sediment. Replacing the N-spiked water with deionized water resulted in two times more NO₃⁻ released from the undredged ditch sediment than the dredged ditch sediment. These results suggest that ditch sediments could represent significant stores of N and that dredging could greatly affect the ditch sediment’s ability to temporarily assimilate N input from field drainage.

NITROGEN (N) is strongly tied to eutrophication of brackish waters and is a key control of eutrophication in the Chesapeake Bay (Howarth et al., 2002; Boesch et al., 2001). Minimizing the loss of N from agricultural lands in the Chesapeake Bay watershed has therefore been a major water quality priority (Chesapeake Bay Program, 2008; USEPA, 2006), particularly from the Delmarva Peninsula, which abuts the eastern shore of the Bay. The Delmarva Peninsula produces approximately 600,000,000 broilers annually (Delmarva Poultry Industry, 2007), resulting in 750,000 tons of poultry manure. Much of that manure is applied to local agricultural soils that are extensively ditched due to poor inherent soil drainage. Vadas et al. (2007) reported that NO₃⁻ concentrations in 540 ground water samples obtained from fields adjacent to the drainage ditches ranged from 0.01 to 59.1 mg L⁻¹, with 54% of samples exceeding the United States Environmental Protection Agency limit (10 mg L⁻¹) for drinking water. Annual N losses from drainage ditches can exceed 20 kg ha⁻¹, with N largely deriving from subsurface flow in the form of NO₃⁻ (Schmidt et al., 2007).

Drainage ditches represent direct conduits between agricultural soils and downstream water bodies. Therefore, drainage ditch management is an important area of water quality concern. Perhaps the most common drainage ditch management practice is dredging, which is intended to optimize the flow of water and to ensure adequate capacity for drainage. On the Delmarva Peninsula, most ditches are dredged on a 10- to 30-yr cycle (Public Drainage Task Force, 2000). In addition to removing accumulated sediments, dredging removes biomass (standing biomass as well as the benthic biota associated with the sediments), and newly uncovered sediments typically contain different abundances, diversity, and species composition of organisms than that of removed sediments (Koel and Stevenson, 2002). Previous research has demonstrated that after dredging, sediments and their associated biota are no longer able to buffer nutrient concentrations as effectively as they were before dredging (Smith et al., 2006; Smith and Pappas, 2007; Shigaki et al., 2008).

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Abbreviations: CEC, cation exchange capacity; Sₙ, uptake length; V, mass transfer coefficient.
Although dredging affects nutrient dynamics in ditches, the exact nature of this effect is poorly understood (Smith et al., 2006). Most studies of N dynamics in flowing waters have focused on natural streams (Haggard et al., 2001; Chaubey et al., 2007). Agricultural drainage ditches are different from natural streams because management practices, such as construction and dredging, result in an alteration of the bed material. The research that exists on dredging drainage ditches has been limited to several studies from Indiana, which is highly context specific (i.e., unique landscape and soil characteristics). Smith and Pappas (2007) conducted hydraulic flume experiments with sediments representing conditions before and after dredging to evaluate N dynamics. They enriched flume water with NH$_4$ by adding NH$_4$Cl or adding NH$_4$ simultaneously with NO$_3$ by adding NH$_4$NO$_3$. They found that when only NH$_4$ was added to the flume water, as the level of NH$_4$ decreased the levels of NO$_3$ increased, likely due to nitrification. Similarly, when NH$_4$NO$_3$ was used as a N source, the absolute values of the nutrient uptake rate for NH$_4$ were lower than those calculated by solute injections in natural fluvial systems, also indicating that nitrification was the major process occurring within the flumes. Thus, it is clear that processes occurring within ditch sediments have the potential to affect N fate, but it is unclear how widely these processes vary, particularly when contrasted with sediments derived from marine deposits on the Atlantic Coastal Plain.

Streams have the potential to retain, transform, and remove organic and inorganic materials through in-stream biological processes that may mediate downstream fluxes of N (Peterson et al., 2001). The nutrient spiraling concept describes the downstream transport of nutrients as they cycle between forms retained in biomass and sediments and dissolved forms in the water column (Newbold et al., 1981; Newbold, 1992) and provides the appropriate framework and tools with which to investigate N retention in stream systems. The number of nutrient cycles within a given stream reach depends on the spiraling length (S), which is the downstream distance required for the nutrient to complete a nutrient cycle (Elwood et al., 1983). Spiraling length depends on uptake length (S$_U$) and turnover length (S$_T$) and indicates nutrient retention efficiency of the stream (Marti and Sabater, 1996). The uptake length is the distance that a nutrient molecule travels in dissolved inorganic form before it is taken up from the water column, and turnover length is the distance that a nutrient molecule travels in particulate form before it is released to the water column in an inorganic form. Nutrient uptake length is the largest component of the spiraling length (Newbold et al., 1981); it is generally >90% of S (Newbold et al., 1983) and is therefore a good index of the relative importance of nutrient utilization and transport.

Research on the effect of dredging on N transport has been limited to several studies from Indiana, results from which are limited by geographic scope. Given the paucity of information on N cycling in drainage ditches and the absence of findings related to dredging of drainage ditches on the Delmarva Peninsula, we sought to evaluate the effect of ditch dredging on sediment N transport in drainage ditches. Sediments, representing conditions before and after ditch dredging, were evaluated for N release and uptake characteristics under an array of antecedent N conditions intended to yield insight into sediment-related controls of N spiraling.

**Materials and Methods**

**Site Description and Sediment Sampling**

Two field ditches with a recent history of water quality monitoring were selected for evaluation. The ditches were located on the research farm at the University of Maryland Eastern Shore, Princess Anne, MD. Geographic and climate characteristics of the farm are described by Kleinman et al. (2007), and ditches selected for this study are identified as ditches 1 and 2 (Fig. 1A and 1B, respectively). These ditches have a combined contributing area of approximately 4.5 ha, and both drain fields were cropped to an annual rotation of corn (Zea mays L.) and soybean (Glycine max L.). The fields have a 25+ yr history of receiving poultry manure at rates often exceeding annual crop requirements such that N mineralization potential is high (Schmidt et al., 2007). Field soils belong to the poorly drained Othello series (fine-silty, mixed, active, mesic Typic Endoaquults), which have a 50- to 75-cm deep surface mantle of loess material with textures ranging from silty clay loam to sandy loam overlying a substrate of coarse, sandy marine sediments (Matthews and Hall, 1966).

Before the current dredging experiment, the two ditches had been under similar management, confirmed by similarities in sediment profiles as described by Vaughan et al. (2007). In addition, Schmidt et al. (2007) reported similar N content for soils immediately adjacent to the two ditches, with 2 mol L$^{-1}$ KCl-extractable NH$_4$–N ranging from 2.0 to 2.4 mg kg$^{-1}$ and NO$_3$–N ranging from 10.5 to 12.2 mg kg$^{-1}$. In July 2006, ditch 1 was dredged, and ditch 2 was left undredged (Fig. 1A and 1B). Dredging was conducted by backhoe and laser level engineered to provide a 0.2% slope while removing approximately 0.5 m of bed material from ditch 1 (Fig. 1A). Dredged material was hauled off-site so that it would not serve as a potential source of nutrients and sediment to field runoff and ditch flow.

Sediment sampling was conducted in October 2006, shortly after the dredging of ditch 1. Bulk samples of the upper 5 cm of sediment were collected from each ditch, placed into 110-L sealed plastic containers, mixed thoroughly (for each ditch), and stored at 4°C until use in the flume or incubation experiments.

**Flume Experiments**

To evaluate sediment controls on nutrient transport in ditch flow, a series of experiments was conducted for N and P simultaneously, using 10-m-long × 0.2-m-wide recirculating flumes as previously described by McDowell and Sharpley (2003). Two replicates of each ditch sediment (dredged ditch and undredged ditch) were placed into separate flumes to achieve a uniform depth of 5 cm. The reservoir of each flume was filled with 200 L of water, which was pumped over the sediment at
0.17 L s\(^{-1}\), equivalent to the average long-term base flow of the two ditches.

The flume experiments involved three phases, each lasting 48 h, as follows: (i) An initial “equilibration” phase, in which sediments were exposed to deionized water to assess initial NH\(_4\)–N and NO\(_3\)–N release. (ii) An “uptake” phase, in which water from the equilibration phase was removed and replaced with 200 L of simulated runoff water that was enriched with 13 mg L\(^{-1}\) of NH\(_4\)–N and 5 mg L\(^{-1}\) of NO\(_3\)–N. This enriched solution was recirculated within the flume to evaluate N uptake by sediments. (iii) A final “release” phase, in which the water from the uptake phase was removed and sediments were exposed to deionized water to assess their potential to retain NH\(_4\)–N and NO\(_3\)–N from the uptake phase.

Phosphorus equilibration, uptake, and release were also evaluated. Results from that part of the experiment can be found in Shigaki et al. (2008).

Water samples were collected at the lower end of the flume (just before reentry into the reservoir) during each of the three phases of the flume study (equilibration, uptake, and release). Automated water samplers (Model 900; American Sigma, Lakeland, CO) were programmed to sample flow with decreasing frequency over each 48-h phase: 5 min (0–15 min), 15 min (30–60 min), 30 min (90–120 min), 1 h (3–6 h), 2 h (8–16 h), and 4 h (20–48 h). Samples were immediately filtered (0.45 \(\mu\)m), acidiﬁed with H\(_2\)SO\(_4\) to pH <2, and stored at 4°C until analyzed.

Sediments representing initial conditions (before equilibration phase) and conditions at the end of each phase (equilibration, uptake, and release) were collected with a spoon (2-cm deep) at 2-m intervals along the 10-m length of the flume (i.e., ﬁve subsamples). Sediments were air dried and sieved to 2 mm for laboratory analyses.

**Incubation Experiments**

To elucidate processes occurring within the sediments during the flume experiments, additional batch experiments were conducted as follows. Samples of fresh sediment were incubated in the laboratory (25°C) and shaken with a series of solutions (sediment:solution ratio, 1 g:2.5 mL) that represented the same solutions and sediment:solution ratio used in the flume experiments. Sediment samples from ditches 1 and 2 were ﬁrst subjected to a 24-h period of NH\(_4\)–N and NO\(_3\)–N equilibration phase using a solution of 0.01 mol L\(^{-1}\) CaCl\(_2\). This step was used to emulate the NH\(_4\)–N and NO\(_3\)–N equilibration phase of the flume experiments. Samples were then centrifuged, and the solution and solid phases were separated by ﬁltration (Whatman #1). The extract was analyzed for NH\(_4\)–N and NO\(_3\)–N as described below. Sediment samples were then subjected to a 24-h uptake phase in a solution of 13 mg NH\(_4\)–N L\(^{-1}\) (as NH\(_4\)Cl). This step was used to provide a better insight of the magnitude of nitrification process when NH\(_4\)–N was the only N-added source to ditch sediments. After this uptake phase, sediment samples were submitted to a 24-h release period using 0.01 mol L\(^{-1}\) CaCl\(_2\) solution. This final step emulated the N release phase of the flume experiments. Solid–solution separation by ﬁltration and solution NH\(_4\)–N and NO\(_3\)–N determination steps were repeated as described previously.

**Laboratory Analyses**

Samples of sediment representing conditions at the initiation and at the end of each phase of the flume experiment (equilibration, uptake, and release) were analyzed for NH\(_4\)–N and NO\(_3\)–N using a 2 mol L\(^{-1}\) KCl extraction solution (solution:sediment = 5:1, 1-h extraction, solid separation by centrifugation followed by Whatman #1 paper ﬁltration). Ammonium-N and NO\(_3\)–N in the extract were determined by colorimetry (QuikChem Method 10–107–04–1-A and 10–117–07–1-C; Lachat Instruments, Loveland, CO).

Particle size distribution of sediments was determined by the hydrometer method (Day, 1965). Organic matter was determined by loss on ignition (360°C), and pH was determined in water (sediment:water = 1:1). Sediment cation exchange capacity (CEC) was determined with 1 mol L\(^{-1}\) NH\(_4\)OAc adjusted to pH 7.0 (Hendershot et al., 1993).
Data Analysis

Nitrogen spiraling parameters were estimated from the results of the flume experiments. Specifically, N concentrations were regressed against the estimated distance of the flume water at the sampling time for the initial and final 24-h periods of the uptake phase. The following equation was used (Smith and Pappas, 2007).

\[ C = C_o e^{-K_L x} \]  

where \( C \) is concentration, \( C_o \) is the initial concentration (mg L\(^{-1}\)), KL is the calculated coefficient for N uptake rate in units of L m\(^{-1}\), and \( x \) is the calculated distance of travel downstream for the stream water (m).

The N uptake length is an indicator of stream N retention efficiency—the shorter the length, the greater the efficiency of the reach to retain N, and vice versa. This parameter is estimated from the decline in N concentration with distance from the solute addition point, assuming that N additions are sufficiently low that resulting stream concentrations are below saturation (Haggard et al., 2005). The N uptake length \( (S_w) \) is the inverse of the N uptake rate per unit distance coefficient \((1/\delta)\), with units of meters. A mass transfer coefficient was calculated using the following equation:

\[ V_t = d(v/S_w) \times 3600 \times 100 \]  

where \( V_t \) is the mass transfer coefficient (cm h\(^{-1}\)), \( d \) is the mean depth of water in the flume (m), \( v \) is the velocity of water in the fluvium (m s\(^{-1}\)), 3600 is used to convert time from seconds to hours, and 100 is used to convert meters to centimeters.

The N uptake rate \((U; \text{mg m}^{-2} \text{h}^{-1})\) is the product of the mass transfer coefficient and ambient N dissolved concentration; uptake rates increase with increases in ambient concentration (Haggard and Sharpley, 2007). The N uptake rate was calculated using the equation from Marti et al. (1997):

\[ U = [(C \times Q)/(S_w \times w)] \times 3600 \]  

where \( Q \) is the flume discharge (L s\(^{-1}\)), \( w \) is the flume width, and 3600 is a conversion from seconds to hours.

Analysis of variance, Tukey (HSD), and Student’s \( t \) test were performed in Minitab v. 15 (Minitab Inc., 2001). Results discussed in the text were considered significant at \( \alpha = 0.05 \).

Results and Discussion

Ditch Sediment Properties

Due to the vertical stratification of sediment in these ditches, dredging of ditch 1 exposed sediment that differed substantially from those at the surface of the undredged ditch (Table 1). Dredging removed finer-textured sediment (410 g kg\(^{-1}\) clay, 190 g kg\(^{-1}\) sand) that was high in organic matter (85 g kg\(^{-1}\)) and exposed coarse-textured sediment (680 g kg\(^{-1}\) sand, 150 g kg\(^{-1}\) clay) with low organic matter content (2.2 g kg\(^{-1}\)). The differences in organic matter content and texture corresponded with differences in the CEC of these sediments, which was three times lower in the dredged ditch sediment than in the undredged ditch sediment. The pH of the sediment from both ditches was <5.0, with lower pH observed in the sediment from the undredged ditch (Table 1). Higher KCl-extractable NO\(_3\)–N and NH\(_4\)–N concentrations were observed in the undredged ditch sediment (15 and 25 mg kg\(^{-1}\), respectively), reflecting the accumulation of N over time, such as through biological uptake or the deposition of N-enriched sediment (Table 1).

Flume Experiments

Equilibration Phase

The undredged ditch sediment released 110 times more NO\(_3\)–N to the flume water during the equilibration phase, with a mean final NO\(_3\)–N concentration of 33.0 mg L\(^{-1}\) at the end of 48 h compared with 0.3 mg L\(^{-1}\) of NO\(_3\)–N for the dredged ditch sediment (Fig. 2A). In contrast, the undredged ditch sediment released significantly lower NH\(_4\)–N during the same period, with a mean final NH\(_4\)–N concentration of <0.1 mg L\(^{-1}\), compared with 2.5 mg L\(^{-1}\) for the dredged ditch sediment (Fig. 2A). Differences in concentrations translated into significant differences in mass loadings of N to the flume water from the two sediments, with the undredged ditch sediment releasing 6604 mg NO\(_3\)–N and the dredged ditch sediment releasing 58 mg NO\(_3\)–N. Loss of NH\(_4\)–N was 492 mg from the dredged ditch sediment and 6 mg NH\(_4\)–N from the undredged ditch sediment.

Differences in initial and final NH\(_4\)–N and NO\(_3\)–N transfers to flume water during the equilibration phase corresponded with observed differences in NH\(_4\)–N and NO\(_3\)–N concentrations in the ditch sediments between the beginning and end of this phase (Table 2). During the equilibration phase, NH\(_4\)–N concentration decreased from 25.3 to 4.1 mg kg\(^{-1}\), and NO\(_3\)–N concentration increased from 13.2 to 26.3 mg kg\(^{-1}\) in the undredged ditch sediment (Table 2). These results suggest that nitrification within the undredged ditch sediment contributed to the increase in NO\(_3\)–N concentrations of the flume water. The magnitude of nitrification occurring within the undredged ditch sediment was likely influenced by the high organic matter content of this sediment.

In contrast to the results for the undredged ditch sediment, NH\(_4\)–N and NO\(_3\)–N concentrations in the dredged ditch sediment were lower at the start of the equilibration phase (Table 2), and NO\(_3\)–N in the flume water remained unchanged during the equilibration phase. Flume water NH\(_4\)–N increased significantly (to 2.6 mg L\(^{-1}\); Fig. 2A) during the equilibration phase. The different trends observed between dredged and undredged ditch sediments in NH\(_4\)–N and NO\(_3\)–N release imply that sediment removal by dredging significantly affected the nitrification processes.

There have been some arguments regarding the biological regulatory factors controlling nitrification in aquatic sediments; however, the detailed process of overall organic transformation, including seasonal variation in marine sediments, has not been well investigated (Kim et al., 1997). Generally, sediment nitrification is regulated by the availability of dissolved oxygen and NH\(_4\) (Triska et al., 1990). Our results suggest that NH\(_4\)–N...
availability in the sediment plays an important role, showing that the sediment from the undredged ditch has greater nitrification potential than the dredged ditch sediment, probably due to the greater NH$_4$–N concentration and greater organic matter content (Table 1) in the undredged ditch sediment.

Many studies have identified organic matter as a mineralizable source of NH$_4$–N for nitrification (Duff and Triska, 2000).

**Uptake Phase**

The effect of ditch sediments on simulated runoff water (i.e., spiked with 5 mg L$^{-1}$ of NO$_3$–N and 13 mg L$^{-1}$ of NH$_4$–N)

<table>
<thead>
<tr>
<th>Sediments</th>
<th>NO$_3$–N mg kg$^{-1}$</th>
<th>NH$_4$–N mg kg$^{-1}$</th>
<th>CEC† meq kg$^{-1}$</th>
<th>pH</th>
<th>OM g kg$^{-1}$</th>
<th>Sand g kg$^{-1}$</th>
<th>Silt g kg$^{-1}$</th>
<th>Clay g kg$^{-1}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dredged</td>
<td>1.0</td>
<td>11.0</td>
<td>32</td>
<td>4.9</td>
<td>2.2</td>
<td>680</td>
<td>170</td>
<td>150</td>
</tr>
<tr>
<td>Undredged</td>
<td>13.2</td>
<td>25.3</td>
<td>96</td>
<td>4.2</td>
<td>85</td>
<td>190</td>
<td>340</td>
<td>410</td>
</tr>
</tbody>
</table>

† CEC, cation exchange capacity; OM, organic matter.

Fig. 2. (A) Ammonium and nitrate equilibration phase, (B) uptake phase, and (C) release phase for ditch sediments during a 48-h flume experiment.
Nitrogen spiraling parameters were calculated for 0- to 24-h and 24- to 48-h periods to better illustrate that the major changes in \( \text{NO}_3^-\)-N and \( \text{NH}_4^+\)-N concentrations in flume water occurred during the first 24 h of the uptake phase (Fig. 2B), where we focus our discussion. Significant differences in calculated uptake rates were observed for \( \text{NH}_4^+\)-N and \( \text{NO}_3^-\)-N between the two sediments. For the undredged ditch sediment, \( U \) for \( \text{NH}_4^+\)-N was 9.4 mg m\(^{-2}\) h\(^{-1}\), and the uptake rate for \( \text{NO}_3^-\)-N was negative (\(-0.29\) mg m\(^{-2}\) h\(^{-1}\)). A more negative value for processing length \( (S_w = -250,000\, \text{m}) \) and a smaller \( V_f \) value (\(-0.006\) cm h\(^{-1}\)) were observed for \( \text{NO}_3^-\)-N (Table 3) in the flume water from the undredged ditch sediment.

The negative values observed for \( \text{NO}_3^-\)-N spiraling parameters for the undredged ditch sediment reflect the increasing \( \text{NO}_3^-\)-N concentration in the flume water during the first 24 h of the uptake phase (Fig. 2B). In natural fluvial (stream) systems, \( \text{NO}_3^-\)-N concentration tends to decrease in the downstream direction (Smith and Pappas, 2007). A potential reason for the difference in the present study is that \( \text{NO}_3^-\)-N concentration in the undredged ditch sediment at the beginning of the uptake phase was quite high at 26.3 mg kg\(^{-1}\) (Table 3). Release of \( \text{NO}_3^-\)-N to the flume water indicates that sediments from these undredged Coastal Plain ditches can be a source of \( \text{NO}_3^-\)-N in runoff waters. Values for the N process length \( (S_w) \), which reflects the efficiency of nutrient retention in stream systems, indicated that \( \text{NH}_4^+\)-N was removed more efficiently from the undredged ditch sediment \( (S_w = 20,000\, \text{m}) \) than from the dredged ditch sediment \( (S_w = 166,667\, \text{m}) \). This difference can probably be attributed to greater nitrification rates induced by the biologically more active undredged ditch sediment or to greater absorption associated with greater clay and organic matter content.

For the dredged ditch sediment, \( U \) for \( \text{NH}_4^+\)-N (1.12 mg m\(^{-2}\) h\(^{-1}\)) was eight times smaller than \( U \) for the undredged ditch sediment (Table 3), indicating that dredged ditch sediment may have few absorption sites for \( \text{NH}_4^+\)-N or that it is nearly biologically inactive. Overall, the values for processing length \( (S_w) \) and \( V_f \) for \( \text{NH}_4^+\)-N were greater for dredged than for undredged ditch sediment, pointing to a lower efficiency of \( \text{NH}_4^+\)-N retention by the dredged ditch sediment (Table 3).

When comparing the uptake results for the final 24-h period to those of the initial 24-h period, there was an approximately 100% increase in \( \text{NO}_3^-\)-N uptake values for dredged and undredged sediments. This shows a bias from the initial 24-h period on making these calculations.

While evaluating \( \text{NO}_3^-\)-N retention during an 8-mo study in a third-order stream of the Ozark Plateau in Arkansas, Haggard et al. (2005) observed negative values for \( S_w, V_f, \) and \( U \), with \( S_w \) ranging from \(-4,600\) to \(-1,600\, \text{m}, V_f \) ranging from \(-5.04\) to \(-2.3\) cm h\(^{-1}\), and \( U \) ranging from \(-255\) to \(-0.97\) mg m\(^{-2}\) h\(^{-1}\). Nitrate-nitrogen release from these Coastal Plain ditches, particularly the undredged ditch, reflects the greater \( \text{NO}_3^-\)-N source in the sediments; consequently, uptake lengths \( (S_w) \) were more negative than Haggard et al.’s (2005) observations.

Release Phase

Replacement of enriched flume water from the uptake phase with deionized water for the release phase continued to reveal differences in the ability of the two sediments to buffer the quality of flowing water. For the dredged ditch sediment, approximately four times more \( \text{NH}_4^+\)-N (380 mg) was released than was removed from the flume water during the uptake phase (Fig. 2C), whereas for the undredged sediment, four times less \( \text{NH}_4^+\)-N was released to the flume water than was

### Table 2. Nitrate and ammonium concentrations in ditch sediments at different phases during the flume experiment.

| Phases          | Dredged |  | Undredged |  |
|-----------------|---------|  |-----------|---|
|                 | \( \text{NO}_3^-\)-N | \( \text{NH}_4^+\)-N | \( \text{NO}_3^-\)-N | \( \text{NH}_4^+\)-N |
| Before equilibration | 0.9 (0.0)† | 10.9 (1.02) | 13.1 (6.78) | 25.3 (1.4) |
| End equilibration | 0.8 (0.14) | 13.5 (1.73) | 26.3 (3.90) | 4.1 (0.3) |
| End uptake      | 0.9 (0.0) | 13.6 (0.48) | 10.1 (1.34) | 24.7 (2.2) |
| End release     | 1.3 (0.17) | 7.2 (1.09) | 12.3 (2.79) | 16.1 (0.6) |

† Standard deviations are in parentheses.
removed from the flume water during the uptake phase. Even so, NH$_4$–N concentrations in the flume water at the end of the release phase were similar for the two sediments (1.9 mg L$^{-1}$ for the dredged ditch and 1.5 mg L$^{-1}$ for the undredged ditch). Differences in NH$_4$–N release correspond with a decline in the KCl-extractable NH$_4$–N of the dredged and undredged sediments between the uptake and release phase—a 6.3 mg kg$^{-1}$ and 8.6 mg kg$^{-1}$ decrease, respectively (Table 2). Differences in NH$_4$–N release are also consistent with differences in CEC of the two sediments, resulting from organic matter and clay removal by dredging (Table 1). The lower CEC for the dredged ditch sediment corresponded with a lower capacity of this sediment to retain NH$_4$–N.

Final NO$_3$–N concentrations in the flume water at the end of the release phase were 0.75 mg L$^{-1}$ for the dredged ditch sediment and 0.25 mg L$^{-1}$ for the dredged ditch sediment. The slight difference in NO$_3$–N concentration observed in the flume water for the dredged sediment compared with the undredged ditch sediment might be explained by differences in NO$_3$–N concentration in the sediments. At the end of the uptake phase (Fig. 2B), NO$_3$–N in the undredged sediment was 10.2 mg kg$^{-1}$, more than 10 times greater than observed in the dredged sediment (Table 2). These results suggest that nitrification during the release phase could be contributing to a decrease in NH$_4$–N concentration in the sediment, which occurred concurrently with an increase in sediment NO$_3$–N concentration. Nitrate-nitrogen concentration in the flume water at the end of the uptake phase (5.7 mg L$^{-1}$) was 23% lower for the dredged sediment when compared with the final NO$_3$–N concentration of the flume water for the undredged ditch sediment (7.3 mg L$^{-1}$). These findings suggest that dredging limited the capacity of drainage ditch sediments to retain NO$_3$–N.

The N status of a watercourse largely depends on N storage, uptake, release, and exchange by abiotic and biotic processes within and between sediments and the water column and subsequent transport longitudinally downstream (Bernot and Dodds, 2005). Results of the release phase demonstrate the variable N dynamics and sediment interactions with flowing water that can influence the potential of drainage ditch sediments to act as sinks or sources of N to ditch flow depending on adopted management practices.

The Role of Nitrification in Ditch Sediments

A laboratory simulation of the three-phase flume experiment (equilibration, uptake, and release) was conducted to assess the role of N nitrification in the observed changes of NH$_4$–N and NO$_3$–N during the flume experiments. To better elucidate the magnitude of nitrification occurring within the ditch sediments during the laboratory uptake phase, NH$_4$ was added as the only N source to the mixing solution, with the same concentration as used in the flume experiment (13 mg of NH$_4$–N L$^{-1}$).

Results from the equilibration phase indicated that NO$_3$–N was the predominant form of inorganic N within the undredged ditch sediment, with a NO$_3$–N concentration in solution of 33.0 mg L$^{-1}$ (Fig. 3A). During the uptake phase, NH$_4$–N in solution decreased in the undredged ditch sediment by 28% (compared with the initial 13 mg L$^{-1}$ in solution), whereas the NO$_3$–N concentration remained high (9.1 mg L$^{-1}$) by the end of the 24-h experiment (Fig. 3B). In the dredged sediment, NH$_4$–N decreased by 8%, probably representing the amount of NH$_4$–N associated with cation exchange sites (Fig. 3B). Nitrate-nitrogen in solution after the uptake phase was almost negligible from the dredged sediment. The greater NO$_3$–N concentration in solution for the undredged sediment suggests that nitrification played a role in converting NH$_4$–N to NO$_3$–N during the uptake phase of the laboratory simulation. At the end of the release phase, 30% of the applied NH$_4$–N (13.0 mg L$^{-1}$) was released back to solution by the undredged ditch sediment, which was comparable to the 28% taken up during the uptake phase. The dredged ditch sediment released back 18% of the 13.0 mg L$^{-1}$ of NH$_4$–N added during the uptake phase, which was comparable to the amount taken up (8%). During the laboratory simulation study, NH$_4$–N appeared to be conserved through the uptake and release phases (Fig. 3B, 3C); however, a net gain of NO$_3$–N suggests that nitrification was ongoing, perhaps based on an NH$_4$–N source of removal from the cation exchange sites.

Smith and Pappas (2007) found similar results when evaluating N uptake in two ditches in Ohio. In the control (pre-dredged sediments), they found higher NH$_4$–N removal from the water column than from the dredged ditch sediment. When only NH$_4$–N was added to the water column, as the levels of NH$_4$–N decreased, the levels of NO$_3$–N increased by 84% for the pre-dredged sediment compared with 22% for the water column of the dredged sediment, indicating that nitrification in the water column was occurring for pre-dredged sediment.

These results provide some insight supporting the hypotheses that nitrification is a contributing pathway for NH$_4$–N removal from the flume water, particularly for the undredged ditch sediment.

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Conclusions

We examined the effect of ditch dredging on N cycling in agriculture drainage ditches of the Atlantic Coastal Plain. Sediments were severely stratified so that dredging exposed materials that were substantially different from those that had accumulated within the ditch since the last episode of dredging. Consequently, dredging removed finer-textured sediments rich in organic matter and nutrients and exposed coarse-textured materials. These coarser-textured materials had a lower capacity to remove NH$_4$–N from flowing water. On the whole, NO$_3$–N
transport in simulated ditch flow was relatively unaffected by sediment, although nitrification played a key role in N cycling of sediments from an undredged ditch. Our findings confirm the profound impact of dredging on N cycling, highlighting the need for prudence in conducting dredging of ditches by minimizing the removal of material that may serve as a net nutrient sink or by minimizing the frequency of dredging.

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References


