Effect of ditch dredging on the fate of nutrients in deep drainage ditches of the Midwestern United States

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Abstract: Dredging of drainage ditches is necessary to ensure that agricultural fields are drained adequately. This study compared the potential impacts of dredging on water quality. Using a fluvarium (stream simulator), bed material collected from drainage ditches prior to dredging was better able to remove NO_3-N, NH_4-N, and soluble P from water than material collected from the bed of the ditches after dredging. Water column NH_4-N concentrations were reduced to 0 mg L^{-1} (0 ppm) earlier in pre-dredged bed material. Nutrient uptake rates were greater for the ditch bed materials collected prior to dredging. Dredging decreased the specific surface area of ditch bed sediments and removed some of the biota responsible for nutrient uptake by the bed sediments in these ditches. Resource managers should perform maintenance tasks, including ditch dredging, when nutrient loads are expected to be low, thus minimizing the potential water quality impacts.

Key words: ammonium—drainage ditch—dredging—phosphorus—nitrate—water quality

The soils of the Midwestern United States are some of the most potentially productive in the world. Nizeyimana et al. (2001) estimated that Iowa, Illinois and Indiana contain the greatest amount of potentially highly productive soils in the United States (figure 1a). The combination of the slowly permeable glacial till soils (most commonly alfisols and mollisols), and the humid environment require agricultural producers to artificially drain many of these soils. This drainage allows for trafficability and for crops to germinate and grow when otherwise, the soil conditions would be too wet. The greatest density of subsurface drainage occurs in the Midwestern United States (figure 1b), which coincides with the general region with the most potentially productive soils in the nation.

In this region, drainage water from fields is typically conveyed through a network of tile lines, generally located approximately 0.6 to 1.0 m (2.0 to 3.3 ft) below the soil surface, to managed drainage ditches. The ditches then convey the water to natural streams or rivers. One necessary management strategy in these systems, which is typically performed when it is perceived that drainage water is not efficiently removed from adjacent agricultural fields, is dredging. Dredging can occur as often as every 5 years or as infrequently as every 50 years. Typically, a local government entity such as a Country Drainage Board or the local Soil and Water Conservation District is responsible for planning and coordinating ditch maintenance and dredging.

Bed materials (i.e. sediments) are known to act as sources or sinks for phosphorus (P) and ammonium (NH_4-N) in the water column, especially in lower order stream networks (first, second, or third order streams) (McDowell and Sharpley 2003; Malecki et al. 2004; Storey et al. 2004; Merseburger et al. 2005; Zhou et al. 2005; Bernot et al. 2006). When wastewater treatment plant effluent with unregulated levels of soluble phosphorus (SP) in discharge was conveyed into streams, elevated SP concentrations in stream water and sediments were observed as far as 30 km (18.6 mi) downstream (Haggard

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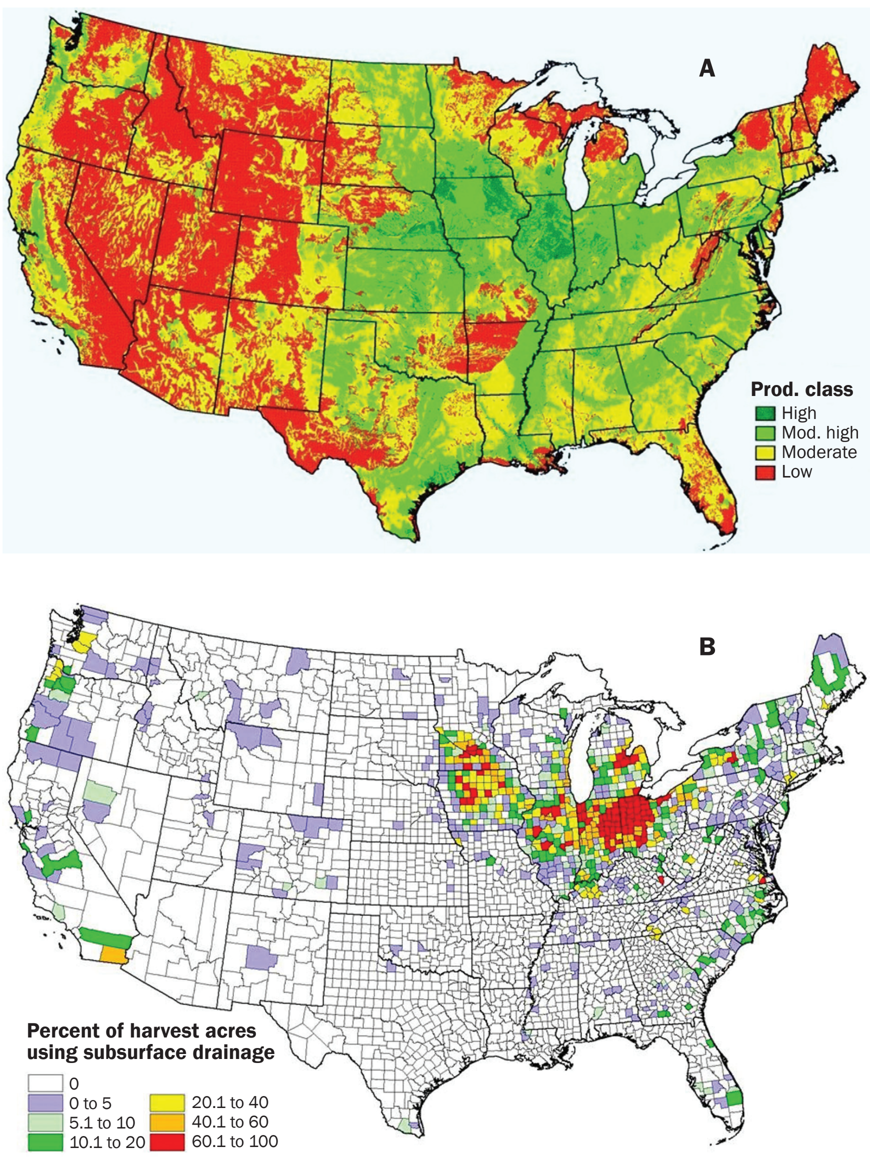
Figure 1
(A) Map of soil productivity in the United States (from Nizeyimana et al. 2001) and (B) density of subsurface drained cropland by county in the United States.



Source: 1992 NRI; 1992 Census of Agriculture.

et al. 2001). The authors further indicated that when SP discharges from the wastewater treatment plant were low (i.e. resultant SP concentrations in stream water were lower than equilibrium phosphorus concentrations [EPC_0]), SP temporarily sequestered in bed sediments could be released, thus elevating the stream water column SP concentrations throughout this 30 km (18.6 mi) stream reach. The physiochemical properties of sediments coupled with environmental conditions, such as pH, redox conditions, particle size distribution, inherent SP concentrations, and Fe and Al concentrations are known to influence the SP adsorption phenomenon of sediments (Koski-Vahala et al. 2001; Pant and Reddy 2001; Zhou et al. 2005; and Smith et al. 2006a).

Triska et al. (1994) observed that bed material and stream bank soils serve as transient storage pools for NH_4-N. Benthic sediments and the hyporheic zone are known to be active sites of N transformations, including nitrification, denitrification, and ammonification (Jones et al. 1995; Storey et al. 2004; Harrison et al. 2005). Nitrification tends to be the dominate process for controlling NO_3-N concentrations in N-limited ecosystems, whereas denitrification is often the predominant process in ecosystems with high levels of N, such as is often found in agricultural streams or ditches (Jones and Holmes 1996).

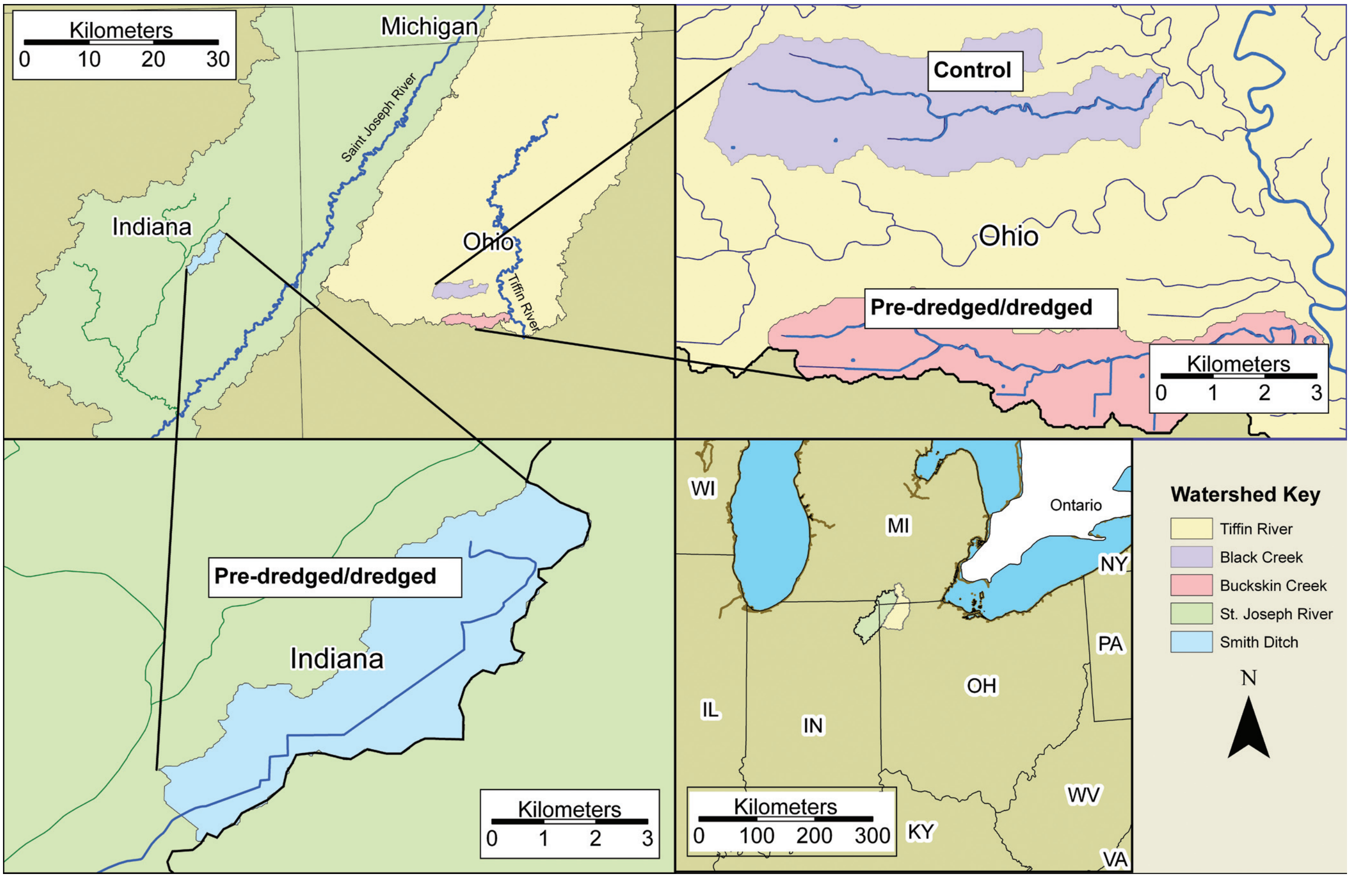
Most studies into in-stream contaminant transport have occurred within natural streams draining forested, agricultural or urban land uses. Agricultural drainage ditches are different from natural streams because management such as construction and dredging results in an alteration of the bed material. These artificial drains also short-circuit the hydrologic cycle via the sub-surface tiles resulting in altered loading of water, nutrients and pesticides to these systems. Recent work has shown that bed material in agricultural drainage ditches can act as SP sources or sinks to the water column (Smith et al. 2005). Other research has shown the dynamic nature of these systems, such as scouring and deposition of bed material, can alter SP dynamics in agricultural ditches (Smith et al. 2006a). However, little is known about how dredging of agricultural drainage ditches affects water quality. Therefore, the objective of this research was to evaluate how dredging alters the dynamic interactions between nutrients in the water column and bed material.

Materials and Methods

Three agricultural drainage ditches have been intensively studied in Northeast Indiana (Smith et al. 2005). One of these ditches, Ditch B, drains approximately 1,400 ha (3,460 ac) and was dredged in 2004. The land use in the sub-watershed drained by Ditch B is approximately 83% agriculture and 12% grass or pasture. The predominant soils in this sub-watershed include Blount silt loams (fine, illitic, mesic, Aeric Epiaqualfs), Pewamo silty clays (fine, mixed, active, mesic Typic Argiaquolls), and Glynwood loams (fine, illitic, mesic Aquic Hapludalfs).

Dredging started at the confluence of Ditch B and the natural stream into which it drains (Cedar Creek) and continued upstream along Ditch B. This ditch was previously dredged approximately eight years prior to the current dredging. Dredging removed roughly 30 cm (12 in) of the surface of the submerged bed material. During dredging, ditch bed material was collected from the surface 2.5 to 5 cm (1 to 2 in), approxi-

Figure 2
Map of the St. Joseph River watershed with ditches in Indiana and Ohio.



mately 5 m (16.4 ft) upstream of dredging activity (hereafter referred to as pre-dredge) and approximately 5 m (16.4 ft) downstream of dredging activity (hereafter referred to as dredged). Approximately 130 L (34 gal) of bed material was collected from both sites.

Upon arrival at the laboratory, bed material was refrigerated at 4°C (39°F) until experiments could be performed. Pre-dredge and dredged bed materials were placed in separate troughs of a fluvarium, built based on the fluvarium described by McDowell and Sharpley (2003). Each trough of the fluvarium was 8 m (26 ft) long by 0.2 m (0.66 ft) wide by 0.2 m (0.66 ft) deep. The fluvarium was designed so that water for each trough continuously circulated over the bed material in that trough. Water was adjusted to 2.5 mM $CaCl_2$ prior to initiating adsorption or desorption experiments, in order to mimic the salt concentrations present in water found in these ditches (Smith et al. 2006b).

Experiments were conducted for several contaminants individually. For each constituent, adsorption experiments were performed by introducing water containing high levels of each nutrient (SP, NO_3-N, NH_4-N), followed by desorption experiments, in which the contaminated water was replaced with contaminant free water. The order of experiments were (1) SP adsorption (initial concentration of 17 mg L^{-1} (17 ppm) as KH_2PO_4) for 120 hours; (2) SP desorption for 24 hours; (3) NO_3-N and NH_4-N adsorption (initial concentrations of 22 mg L^{-1} NO_3-N and 4.4 mg L^{-1} NH_4-N) for 66 hours; and (4) NO_3-N and NH_4-N desorption for 66 hours. For SP adsorption experiments, water samples were collected hourly for the first 24 hours and every 4 hours thereafter. For SP desorption experiments, water samples were collected hourly for 24 hours. During the N adsorption and desorption experiments, samples were taken every 6 hours for 66 hours. The concentrations of contaminants in some cases were quite high on purpose in order to "stress" the system. A detailed discussion of the results of the SP adsorption and desorption experiments can be found in Smith et al. (2006b). Adsorption and desorption experiments were also conducted for pesticides, and further discussion of those experiments can be found in Pappas and Smith (2007).

Water samples for nutrients were filtered (0.45 μm), and acidified with HCl prior to analysis. Soluble P was analyzed in filtered, acidified water samples using inductively coupled plasma-optical emission spectrometry (ICP-OES). Nitrate-N was analyzed colorimetrically on filtered, acidified water samples using method APHA 4500-NO_3 H, and NH_4-N was analyzed on these samples using APHA method 4500-NH_3 F (American Public Health Association 1998).

Particle size distribution of bed materials was determined by the micro-pipette method (Miller and Miller 1987), and loss on ignition (400°C [752°F]) was used to determine organic matter (Nelson and Sommers 1982). Mehlich 3 extraction was used to determine the P, Al, and Fe concentrations in bed materials (Mehlich 1984) and to calculate a P saturation ratio (PSR; Maguire and Sims 2002)

As the bed materials collected from the ditch in Indiana demonstrate the potential effects on water quality immediately after dredging, a study has been conducted to evaluate the long-term (up to one year) effect of dredging on water quality. With this objective in mind, bed materials were collected from an adjacent watershed near Defiance, Ohio. For this portion of the research, two adjacent ditches were paired. A "control" ditch was selected that had no dredging history for the last 20 years. Dredging for the other ditch last occurred approximately 50 years prior to the current dredging project. This ditch was overgrown with trees and brush. Before dredging could occur in this ditch, all trees and brush were removed. Bed materials were collected in the same manner as described above, except the "pre-dredge" samples were collected after tree/brush removal. The control ditch was selected in order to evaluate temporal changes that may occur without any dredging activity. The "pre-dredge" conditions no longer exist in the dredged ditch. Temporal changes in the dredged ditch must therefore be compared to the control ditch so that we may determine the longevity of the potentially detrimental effects to water quality from the dredging activity.

These bed materials were used for a similar series of experiments. Sediments were placed in the fluvarium troughs, and contaminated water was used for 120 hours, removed, and then was replaced with contaminant-free water for 24 hours. For these studies the order was (1) SP adsorption (mean initial concentration was 16 mg SP L^{-1} as KH_2PO_4); (2) SP desorption; (3) NH_4-N adsorption (mean initial concentration was 12.9 mg NH_4-N L^{-1} as NH_4Cl); and (4) NH_4-N desorption. Samples were collected, processed and analyzed as described above. Since this is an on-going study, only preliminary results from the bed materials collected in Ohio will be presented in this manuscript.

Regression equations were fit to water column contaminant concentration data. For adsorption concentration data, logarithmic decay functions were found to best describe the decreases in NO_3-N and SP concentrations in the water column with time, and linear functions were found to best describe the zero-order kinetics governing decreases in NH_4-N concentrations. For concentration data from desorption experiments, an exponential increase to a maximum function was found to be the most appropriate equation to accurately describe the kinetics involved with nutrient release from the bed material to the water column.

Table 1

Particle size distribution, organic matter content, Mehlich 3 extractable Fe and Al concentrations, and P sorption ratio of bed materials collected from drainage ditches in Indiana and Ohio for use in fluvarium studies of nutrient dynamics.

	Indiana ditch		Ohio ditches		
	Pre-dredge	Dredged	Control	Pre-dredge	Dredged
Silt	17.1%	3.0%	11.4%	30.3%	39.3%
Clay	20.1%	14.6%	50.2%	33.3%	29.9%
Organic matter	4.9%	2.0%	4.1%	4.6%	2.1%
Al (mg kg^{-1})	313	90	966	854	316
Fe (mg kg^{-1})	2180	1190	442	3570	227
PSR (unitless)	0.077	0.119	0.0148	0.0078	0.0182

Note: PSR = phosphorus sorption ratio.

To compare results from this study to those of in-situ stream studies, values for nutrient kinetic parameters commonly estimated by stream biogeochemists were calculated (Marti and Sabater 1996). The nutrient concentrations were regressed against the estimated distance the fluvarium water had been conveyed at the sampling times for the initial and final 24 h periods of the adsorption studies. The equation used was as follows:

$$C = C_o\, e^{(-K_L x)}, \qquad (1)$$

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

The nutrient processing length (S_{net}) is the inverse of the calculated coefficient with units of m. A mass transfer coefficient was calculated using the equation:

$$Vf = d\,(v/S_{net}) \times 3{,}600 \times 100, \qquad (2)$$

where Vf is the mass transfer coefficient (cm hr^{-1}), d is the mean depth of water in the fluvarium (m), v is the velocity of water in the fluvarium (m s^{-1}), 3,600 is used to convert from seconds to hours, and 100 is used to convert from m to cm.

The nutrient uptake rate (U; mg m^{-2} hr^{-1}) was also calculated for the nutrients in these experiments using the equation from Marti et al (1997):

$$U = [(C \times Q)/(Snet \times w)] \times 3{,}600, \qquad (3)$$

where Q is the fluvarium discharge (L sec^{-1}), w is the fluvarium width, and 3,600 is a conversion from seconds to hours.

These calculations were made with the understanding that if these experiments were carried out for an infinite time, the nutrient concentrations would not necessarily reach 0, but instead are used to estimate these parameters and compare them to what has been calculated in natural streams.

Results and Discussion

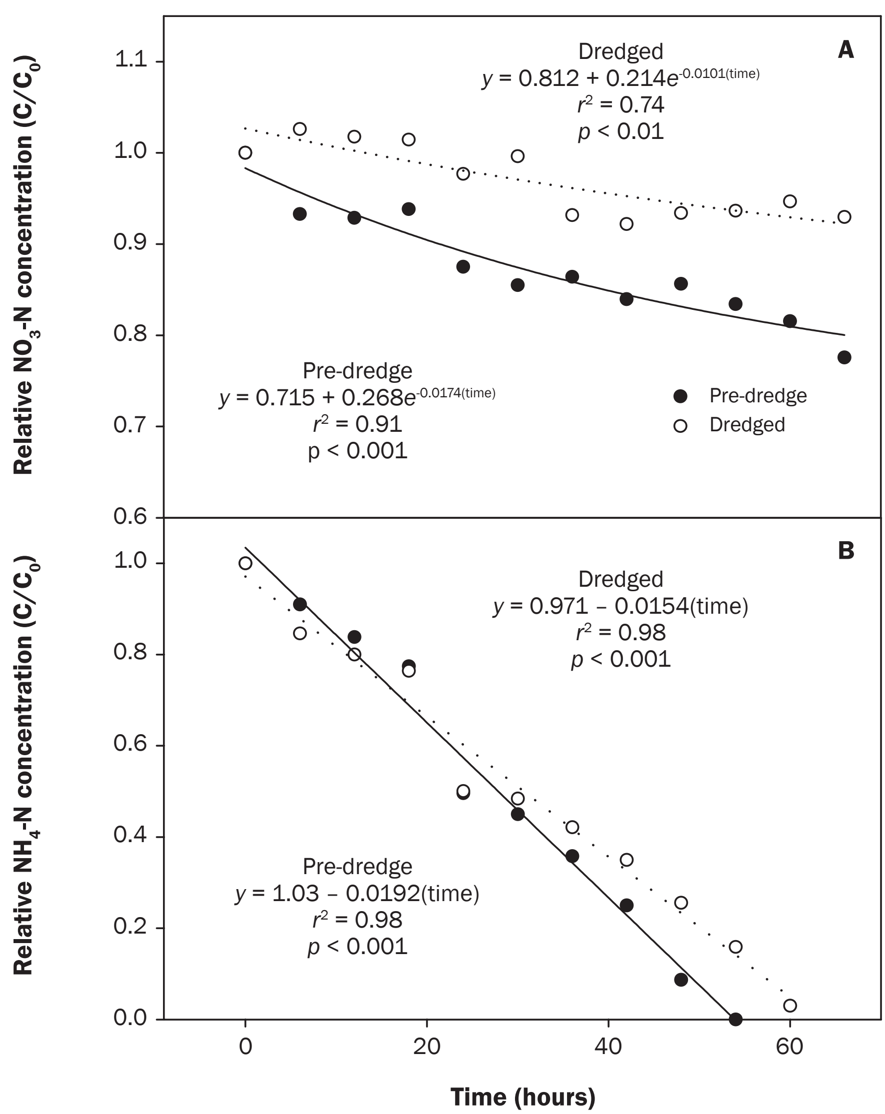
Prior to dredging, the bed materials were composed primarily of alluvial sediment deposits, and the buildup of organic matter resulting from decay of plants and algae that grew in the ditches. Deposition of "fresh" bed materials after storm events has been shown to alter some of the chemical parameters that affect SP adsorption such as decreasing the EPC_0 and increasing the P buffering capacity of the bed materials (Smith et al. 2006a). Dredging removed approximately 30 cm (12 in) of bed material and lowered the ditch bed to compact glacial till. There were lower levels of silt and clay in the ditch bed after dredging in the Indiana ditch (table 1). Removal of clay size particles likely resulted in lower concentrations of Al and Fe in the dredged bed material. Many of the surfaces exposed by dredging were gleyed, suggesting the pedogenic removal of Fe from this stratum also decreased Fe concentrations in the dredged bed material compared to the pre-dredge bed material.

The ditches in Ohio (0.6 to 1.2 m [2 to 4 ft]) were not as deep as the Indiana ditch (2.4 to 3.0 m [8 to 10 ft]). The Ohio ditches would be considered intermittent ditches, because they do not contain water year round, whereas the ditch in Indiana does typically contain water all year long. There was much more clay in the control ditch than the other Ohio ditch bed before or after dredging (table 1). During the collection of bed materials from the Ohio control ditch, gleyed surfaces were observed, indicating pedogenic loss of Fe from these bed materials. These observations were not made for the other ditch when collecting either pre-dredge or dredged bed materials. It would appear that at some point, the strata exposed by dredging had been under a reduced environment at some time, since there were much lower levels of Fe in the dredged bed materials than the pre-dredged bed materials, despite minimal differences in the percentage of clay (table 1).

In addition to removal of sediments, dredging also would be expected to remove the biotic communities in the upper 30 cm (12 in) of the profile. While this was not explicitly measured, it could be reflected in the reduction in organic matter observed following dredging from sediments collected in Indiana and Ohio (table 1). These biotic communities would include aquatic plants, algae (including algal mats), and the associated microbial communities. The sediment-water interface was very different in terms of biogeochemical properties when comparing the pre-dredge and dredged bed materials.

Processing of Water Column Nitrogen by Ditch Bed Materials. During the adsorption experiments conducted with ditch bed materials collected from the Indiana ditches, NO_3-N concentrations in the water column decreased with time (figure 3A). The pre-dredged bed material appeared to remove NO_3-N from the water column at a greater rate than the dredged bed material. After 66 hours, the water flowing over the pre-dredged bed material contained 16.8 mg NO_3-N L^{-1}, and the water in the dredged treatment contained 20 mg NO_3-N L^{-1}. During the same adsorption experiments, removal of NH_4-N from the water column appeared to be best represented by zero order kinetics, as the regression equation was linear to 0 mg L^{-1} (figure 3B). Ammonium-N concentrations in the water exposed to the pre-dredged bed material reached 0 mg L^{-1} 12 hours earlier than the water exposed to the dredged bed material. At the end of desorption experiments, concentrations of NO_3-N in the water column were roughly 0.62 mg L^{-1} and 0.73 mg L^{-1} for the pre-dredge and dredged bed material, respectively (data not shown). Ammonium-nitrogen concentrations during desorption experiments were all below 0.03 mg L^{-1} and appeared to be negligible for both treatments.

Figure 3
Relative concentrations of NO_3-N (A) and NH_4-N (B) during adsorption experiments with bed material from drainage ditches collected before dredging (pre-dredge) and after dredging (dredged) (Indiana).



There are several potential reasons for the differences observed in N adsorption data between the pre-dredged and dredged bed material. The NO_3-N was most likely removed by aquatic microbes and plants or via denitrification (Howard-Williams et al. 1982; Wollheim et al. 1999; Perez et al. 1999; Harrison et al. 2005; Bernot et al. 2006). At the end of N experiments, we observed that more plants had germinated and were growing in the pre-dredged bed material. Dredging would remove benthic species and aquatic plants present in the upper 30 cm (12 in) of bed material. The apparent linear rate of removal for NH_4-N from the water column was most likely due to biochemical transformations, such as nitrification. Harrison et al (2005) observed nitrification and denitrification occurring simultaneously in a subtropical stream. Our results indicate that the rate of decrease in NH_4-N concentrations from the water column was limited only by the amount of NH_4-N in the system (i.e. mass of NH_4-N in the water column). In other agricultural catchments, NH_4-N uptake did not appear to have exceeded the biological uptake capacity of the streams (Bernot et al. 2006)

The slow removal of NO_3-N from the water column by sediments is reflected in the long nutrient processing length, the low mass transfer coefficient, and nutrient uptake rate values during the initial 24 hours of adsorption experiments (table 2). Calculation of these variables in the fluvarium may not be the most accurate method to estimate these kinetic parameters since the water in the fluvarium is recirculated in the fluvarium, unlike what occurs in a natural stream or ditch. However, calculation of these parameters from the data gathered in our experiments may allow us to compare the treatments to one another, and comparison to field data can be helpful in evaluating the kinetics of nutrient fate and transformation in these systems to what has been observed in natural systems. There is typically a wide range for the nutrient processing length, mass transfer coefficient and nutrient uptake rate when evaluated using in-situ stream injection techniques (table 3). There was a similar wide range in these parameters for NO_3-N kinetics for bed materials taken from the Indiana ditch when calculated using fluvarium data (table 2). For the pre-dredge bed material, there was a slight decrease for the mass transfer coefficient and nutrient uptake rate values for NO_3-N during the final 24 hours compared to the initial 24 hours. During this same period, the nutrient processing length changed from more than 23 km (14.3 mi) to approximately -55 km (-34.2 mi) for NO_3-N from the dredged bed material. The change in sediment kinetics from a slight sink to a slight source for NO_3-N in the water column actually represents little change, as these values indicate there is relatively little net change in the NO_3-N concentrations in the water column during either period. Ditch or stream bed materials are commonly observed to be a source of NO_3-N to the water column (Valet et al. 1996; Haggard et al. 2005; Mersburger et al. 2005). In other agricultural systems, biological uptake of NO_3-N appeared to have been saturated (Bernot et al. 2006), which appears to have been what occurred in our experiment. Other studies have observed that scouring of sediments during flood events removes benthic algae, thereby increasing the nutrient processing length for NO_3-N immediately after the flood (Marti et al. 1997). In their study, Marti et al. (1997)

Table 2

Nutrient processing length (S_{net}), mass transfer coefficient (Vf), and nutrient uptake rate (U) calculated for P, ammonium (NH_4-N) and nitrate (NO_3-N) during the initial and final 24 hours of fluvariam adsorption experiments using ditch bed material collected before and after dredging.

	Pre-dredge			Dredged		
	S_{net} (m)	Vf (cm hr^{-1})	U (mg m^{-2} hr^{-1})	S_{net} (m)	Vf (cm hr^{-1})	U (mg m^{-2} hr^{-1})
Initial						
NH_4-N	757	0.248	3.96	807	0.232	3.72
NO_3-N	4,100	0.046	0.73	23,500	0.008	0.13
P	640	0.293	4.69	1,050	0.179	2.87
Final						
NH_4-N	164	1.15	18.3	282	0.665	10.6
NO_3-N	6,170	0.030	0.49	−55,000	−0.003	−0.05
P	1,320	0.142	2.27	1,910	0.098	1.57

Table 3

Nutrient uptake length (S_{net}), nutrient uptake velocity (Vf), and nutrient uptake rate (U) for P, ammonium (NH_4-N), and nitrate (NO_3-N) as reported in the literature.

Nutrient	S_{net} (m)	Vf (cm hr^{-1})	U (mg m^{-2} hr^{-1})	Reference
NH_4-N	10 to 463		5.57 to 12.6	Marti and Sabater 1996
	−366 to 2,870	−20.4 to 73.0		Merseburger et al. 2005
	200 to 1,100	0.04 to 0.2		Bernot et al. 2006
	5 to 270	0.05 to 0.65		Hall et al. 2002
	65			Tank et al. 2000
	400 to 1,400	5.04 to 35.3	270 to 1,170	Haggard et al. 2005
NO_3-N	61 to 799		1.3 to 19.9	Marti et al. 1997
	−7,470 to −322	−31.6 to −0.66		Merseburger et al. 2005
	−2,550 to 1,727			Valett et al. 1996
	199			Tank et al. 2000
	−4,600 to −1,600	−6.12 to −2.30	−256 to −97.2	Haggard et al. 2005
P	4 to 241		2.08 to 15.6	Marti and Sabater 1996
	−1,850 to 1,390	−96.4 to 6.78		Merseburger et al. 2005
	200 to 750	0.06 to 0.2		Bernot et al. 2006
	2 to 85	0.12 to 0.70		Hall et al. 2002
	151			Tank et al. 2000
	9,000 to 31,000			Haggard et al. 2001
	4,100 to 370,000	0.10 to 6.48	0.14 to 6.84	Doyle et al. 2003
	−13,000 to 13,000	−1.37 to 2.70	−32.4 to 104.4	Haggard et al. 2005

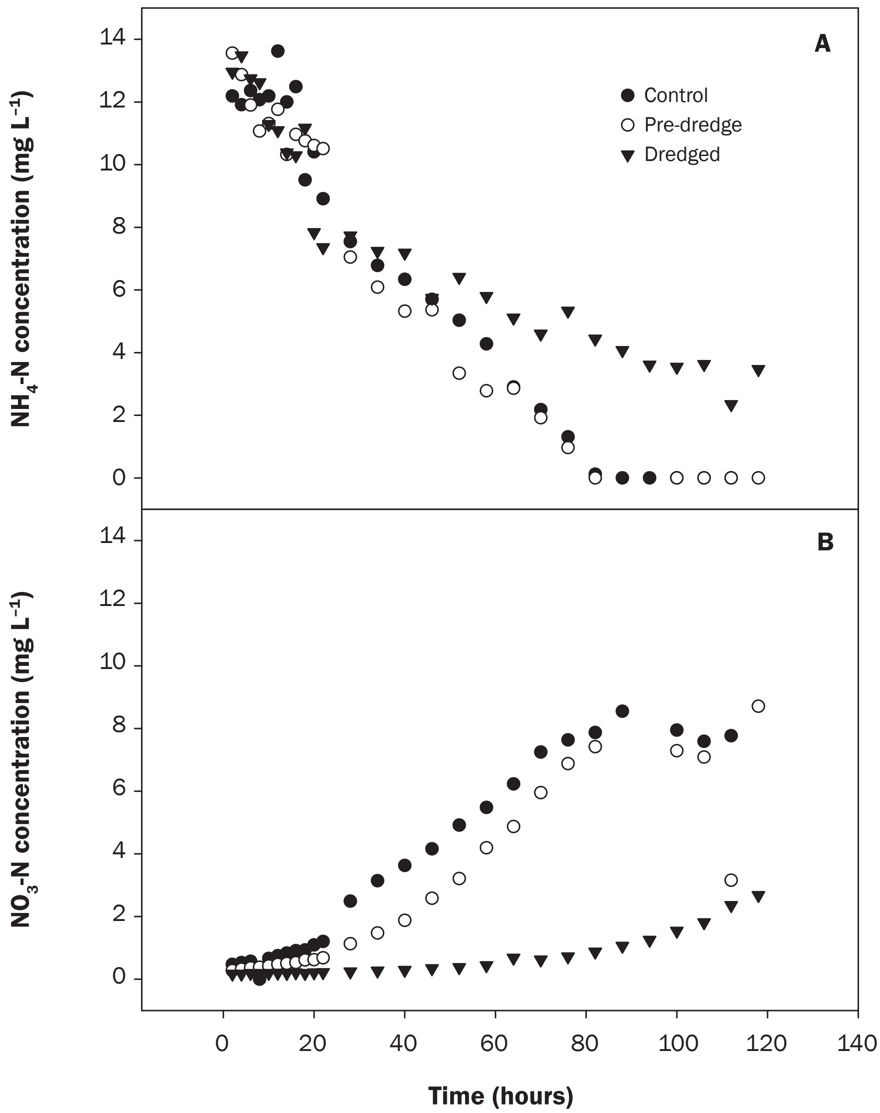
observed recovery of ecosystem function (i.e. return of the nutrient processing length to "normal") 30 days after a flood with a 3-year return period.

The absolute values of the nutrient uptake rate are lower in this study (table 2) than those calculated by solute injection experiments in natural fluvial systems (table 3). A potential reason for this difference is that in the present study, NH_4NO_3 was used as the N source, whereas in most stream injection studies, researchers conduct experiments with NH_4 and NO_3 separately. Thus our nutrient uptake rate values for NO_3 could potentially be lower than expected due to nitrification of NH_4 added from our N source. Nitrification and denitrification have been observed to occur simultaneously within the same system (Harrison et al. 2005), which was very likely what was occurring during this fluvarium experiment.

As part of an ongoing study, only NH_4-N was used in the contaminated water for a fluvarium adsorption experiment for sediments from two ditches in Ohio. In the control and pre-dredge bed materials, NH_4-N removal from the water column appeared to occur via zero-order kinetics (figure 4A), whereas NH_4-N removal for the dredged bed material appeared to follow first-order kinetics. With only NH_4-N added to the water column, as the levels of NH_4-N decreased, the levels of NO_3-N increased in the water column (figure 4B), indicating that nitrification was indeed occurring. The rates of increase for NO_3-N in the water column for the control and pre-dredge bed materials were much greater than those observed for the dredged bed material. This supports the hypotheses that nitrification was unintentionally elevating the NO_3-N concentrations in the previous experiments, and that nitrification was likely one of the main pathways for removal of NH_4-N from the water column.

Values for the nutrient processing length, mass transfer coefficient, and nutrient uptake rate were similar during the first 24 hours of the adsorption experiments for NH_4-N for the bed materials collected before and after dredging in Indiana (table 2). Nutrient processing length decreased with time during the adsorption experiments, while mass transfer coefficient and nutrient uptake rate for NH_4-N increased for both bed materials between the first and final 24 hours of the experiments (table 2). Values for these

Figure 4
Ammonium-N (A) and nitrate-N (B) concentrations in water column during adsorption experiment with the addition of NH_4Cl as the sole N source (Ohio).



Note: Bed material for this experiment was collected from a ditch before dredging (pre-dredge) and after dredging (dredged) and from an adjacent ditch that had no recent history of dredging (control).

variables calculated from data collected from the fluvarium were comparable to those determined in natural streams as reported in the available literature (table 3). Uptake of NH_4-N was not saturated when nutrient injections were made to agricultural streams in Indiana and Michigan (Bernot et al. 2006). Our data may indicate that NH_4-N in the water actually stimulated the nitrifiers, since the rate of removal was greater at the end of the experiments than at the beginning.

The nutrient processing length, mass transfer coefficient, and nutrient uptake rate values for the bed materials collected from Ohio when NH_4-N was applied were also within the range of values observed by field experiments (tables 3 and 4) and were also similar to those observed in the Indiana ditch (table 2). For the Ohio bed materials, the values for these parameters appeared to be more similar between the control and pre-dredge bed materials than either one of those compared to the dredged bed material (table 4). The dredged bed material may have become saturated with respect to NH_4-N (figure 4A). Similar observations have been made in

Table 4

Nutrient processing length (S_{net}), mass transfer coefficient (Vf), and nutrient uptake rate (U) calculated for ammonium (NH_4-N) and soluble P (SP) during adsorption experiments using ditch bed material collected from a control ditch, and an adjacent ditch before and after dredging.

	Control			Pre-dredge			Dredged		
	S_{net} (m)	Vf (cm hr^{-1})	U (mg m^{-2} hr^{-1})	S_{net} (m)	Vf (cm hr^{-1})	U (mg m^{-2} hr^{-1})	S_{net} (m)	Vf (cm hr^{-1})	U (mg m^{-2} hr^{-1})
NH_4-N	909	0.206	3.3	769	0.244	3.9	1,250	0.150	2.4
SP	204	0.919	14.7	417	0.450	7.2	526	0.365	5.7

streams receiving wastewater treatment plant effluent (Marti et al. 2004).

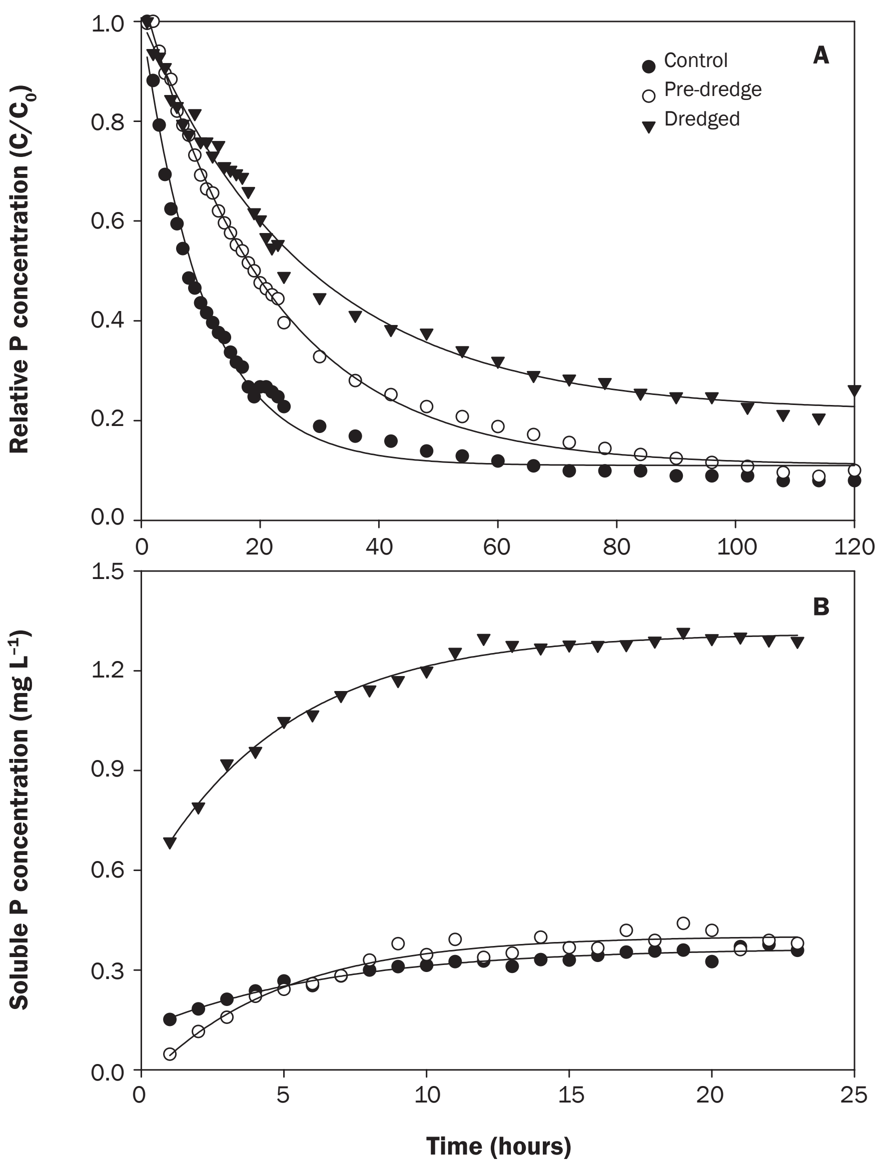
Processing of Water Column Phosphorus by Ditch Bed Materials. It was previously reported that during adsorption experiments with SP, the pre-dredge bed material was able to remove SP from the water column much more quickly than the dredged bed material (Smith et al. 2006b). This observation has also been made by subsequent research using the three bed materials collected from Ohio (figure 5A). The rates of SP removal were greater from the control and pre-dredge bed materials than the dredged bed materials. At equilibrium, the relative SP concentrations in the water column with the dredged bed materials were nearly three times greater than the relative concentrations observed with the control or pre-dredged bed materials.

Previous reports have also shown that SP concentrations in the water column from desorption experiments indicate that the dredged bed material released SP to the water column at a greater rate than the pre-dredge bed material (Smith et al. 2006b). Control and pre-dredge bed materials released SP back to the water column at very similar rates, which were much lower than those observed for dredged bed materials (figure 5B). Despite adsorbing greater quantities of P from the water column, the SP concentrations after 24 hours from the control and pre-dredged bed materials were less than 30% of the concentrations observed at the same time with the dredged bed material.

Removal of N and SP from the water column likely occurred due to different mechanisms. Reductions in NO_3-N most likely occurred due to biological activity (Marti et al. 1997; Harrison et al. 2005). However NH_4-N and SP were likely removed via a combination of biological and chemical processes (Triska et al. 1994; McDowell and Sharpley 2003; Marti et al. 2004; Jarvie et al. 2005). The initial concentrations of NO_3-N and SP in the water column were similar (22 and 17 mg L^{-1} respectively) for the studies

Figure 5

(A) Relative concentration of soluble phosphorus in water column during adsorption experiments and (B) soluble phosphorus concentrations in water column during desorption experiments for bed material collected before dredging (pre-dredge) and after dredging (dredged) and an adjacent ditch that had no recent history of dredging (control) (Ohio).



conducted with bed materials from Indiana. However, after 60 hours, SP concentrations were reduced by 63% to 74%, whereas NO_3-N concentrations were reduced by only 7% to 23%. The SP experiments occurred prior to the NO_3-N experiments, thus the potential for biological removal of SP from the water column could have been limited by N (N concentrations of sediments were not calculated for SP adsorption or desorption experiments). It is unlikely that biological NO_3-N uptake would have been SP limited, since SP concentrations were elevated due to the preceding adsorption experiments. Removal of NH_4-N from the water column appeared to follow zero order kinetics. In fact, NH_4-N removal rates increased as is evident in table 2. It is likely that nitrification was transforming NH_4-N to NO_3-N, thereby misleadingly elevating the NO_3-N concentrations above what would have been observed had NO_3-N been used alone.

There are several potential reasons for the differences observed in SP adsorption and release mechanisms when comparing the pre-dredge and dredged bed materials. The pre-dredge and dredged bed materials had very different particle size distributions and organic matter contents (table 1), which would result in very different specific surface areas of the two bed materials (Smith et al. 2006b). The Fe and Al concentrations in pre-dredge bed material were also much greater than those observed in the dredged bed material (table 1). Even with 50% greater inherent P concentrations in the pre-dredged bed material (121 mg P kg^{-1}) than in the dredged bed material (82 mg P kg^{-1}), the pre-dredged bed material had a lower PSR (table 1; Smith et al. 2006b). The PSR is an assessment of the availability of sites for adsorption based on the P, Fe, and Al concentrations in the sediments, such that higher values suggest relatively greater saturation of these sites with respect to P (Maguire and Sims 2002). It is not known from these studies what might occur if the alluvial sediments removed by dredging were coarser than the materials exposed by dredging. If surface adsorption was the predominant mechanism for nutrient removal from the water column, and not biological uptake, then it is very possible this hypothetical scenario would result in greater rates of nutrient removal from the water column after dredging.

The nutrient processing length for SP during the initial 24 hours of the adsorption experiments indicate that the pre-dredged bed material had 60% lower uptake length than the dredged bed material (table 2). The nutrient processing length is a term used to describe the processes involved in removal and/or release of nutrients from the water column. The observed nutrient processing length from the experiments using Indiana ditch bed materials were at the high end of the values reported in literature (table 3). When comparing the nutrient processing length results for the final 24 h to those of the initial 24 h, there was approximately a 100% increase in the values for both the pre-dredge and dredged bed materials from the Indiana ditch (table 2). This indicates that the sediments were becoming saturated with respect to SP and that the biogeochemical processes involved in removing SP from the water column were, therefore, no longer as effective.

The mass transfer coefficients for SP from the pre-dredged bed material were quite rapid compared to those of the dredged bed material (table 2). The mass transfer coefficient values for the initial 24 h period were also greater than those calculated for the final 24 h period. The mass transfer coefficient values for SP calculated from these experiments were within the range of values reported in literature for natural streams (table 3). Soluble P uptake rates were 63% greater during the first 24 hours of adsorption experiments, and 45% greater during the final 24 hours of adsorption experiments for the pre-dredge bed material compared to the dredged bed material (table 2). These values were comparable to those observed in natural systems (table 3).

Despite greater apparent rates of uptake and much lower P concentrations at 120 hours after the initiation of the adsorption experiments from the control and pre-dredge sediments compared to the dredged sediments collected in Ohio, the nutrient processing length, mass transfer coefficient, and nutrient uptake rate values calculated for the pre-dredge bed materials were more similar to the dredged bed materials than the control bed materials (table 4). This likely shows the bias of the initial 24-hour period on making these calculations. The values for nutrient processing length are lower for the Ohio bed materials, while the mass transfer coefficient and the nutrient uptake rate are greater for the Ohio bed materials (table 4) than the material collected from the Indiana ditch (table 2). In agricultural streams, Bernot et al. (2006) observed nutrient processing length values of 200 to 750 m with instream nutrient injections, which were very similar to what we observed in all of the Ohio bed materials and the Indiana pre-dredge ditch bed materials.

Summary and Conclusions

Dredging of the deep drainage ditches of the Midwestern United States is necessary to ensure adequate removal of water from agricultural fields. However, this practice may degrade water quality. When nutrient-rich water flowed over drainage ditch bed material that was collected before or after dredging, the pre-dredge bed material was able to remove NO_3-N, NH_4-N and SP quicker than the bed material present after dredging. Furthermore, when the water with high nutrient levels was removed and replaced with nutrient-free water, the release of SP from the dredged bed material to the water column resulted in greater nutrient concentrations in the water than the bed material collected prior to dredging. This was due to a decrease in the specific surface area of the bed materials present after dredging, as well as the removal of organic matter and biota during the dredging process. These experiments show the dynamic nature of nutrient transport within these ditches and demonstrate how the bed material can act as a source or a sink for contaminants. Results from this study represent the immediate impacts of dredging on water quality. These results are also taken from ditches in a fairly small geographic region, and as such may not represent all ditches across the United States. For example, if the ditches are composed of a sandy alluvium over a residuum with higher levels of clay and silt size fractions, then dredging could result in greater rates of nutrient adsorption and transformation compared to a pre-dredged condition. The time for the dredged ditch bed to recover from dredging and the downstream ecological impacts are not yet understood and warrant further investigation. Resource managers should consider these findings when planning dredging maintenance activities. To minimize water quality impacts, it is suggested that resource managers conduct maintenance on agricultural drainage ditches (i.e., dredging) during periods of the year when contaminant loads are expected to be low and work with producers to minimize fertilizer applications during and immediately after dredging.

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