ERDC TN-APCRP-EA-05
January 2002

Experimental Determination of the Impacts of Sediment Desiccation and Rewetting on Sediment Physical and Chemical Characteristics in Lawrence Lake, Pool 8, Upper Mississippi River

by William F. James, John W. Barko, and Harry L. Eakin

PURPOSE: This technical note examines changes in sediment physical and chemical characteristics due to dessication and rewetting in laboratory systems. Results from this study may be used to better understand how sediment changes as a result of a lake drawdown and has implications for macrophyte growth potential in shallow lake rehabilitation programs.

BACKGROUND: Evidence is increasing that submersed and emergent macrophyte communities are a very important component of shallow lake ecosystems (Scheffer 1998 and references therein). Macrophyte communities minimize sediment resuspension by dampening wave activity (James and Barko 1994, Barko and James 1998); they also provide refugia for fish and zooplankton populations (Timms and Moss 1984; Killgore, Hoover, and Morgan 1991; Dibble, Killgore, and Harrel 1997) and substrate for invertebrate communities (Beckett, Aartila, and Miller 1992). Shallow lakes that have lost macrophyte communities usually exhibit frequent periods of sediment resuspension, high turbidity levels, low light penetration, enhanced nutrient recycling, and the prevalence of nuisance cyanobacteria (Bengtsson and Hellström 1992; James and Barko 1994; Scheffer et al. 1997; Bachmann, Hoyer, and Canfield 2000). While it is desirable to promote macrophyte growth to improve water quality in shallow systems, it is usually difficult to reestablish macrophyte communities without some form of manipulation to improve light conditions (i.e., reduce resuspension) and to remove destructive benthivorous fish such as carp (*Cyprinus carpio*). One potential manipulation to improve conditions for macrophyte growth in lakes that have lost this component is a lake drawdown to promote sediment consolidation and reduction in resuspension potential (Scheffer 1998). Fish stocks can also be manipulated during the drawdown to remove bioturbation impacts to the water column. When resuspension is controlled via this lake management technique, macrophyte communities can be successfully reestablished either by planting desirable species (Doyle and Smart 1993, Smart and Dick 1999) or by allowing the native seed bank in the sediments to germinate (Madsen et al., in preparation).

More information is needed regarding impacts of changes in sediment characteristics as a result of the desiccation and rewetting process on shallow lake ecosystem dynamics. Sediment desiccation and rewetting may result in positive impacts such as reduction in resuspension potential, improvement of rooting medium (i.e., nutrients and sediment texture) for macrophyte growth, conversion of soluble nutrients to particulate forms, and reductions in organic matter concentrations. However, negative impacts include conversion of organic matter to soluble N and P, enhanced nutrient release rates from sediments, and potential stimulation of algal growth. Ultimately, these positive and negative impacts of sediment desiccation and rewetting need to be evaluated with respect to the potential for improved conditions for macrophyte growth in shallow systems.

Pool 8, a run-of-the-river impoundment of the Upper Mississippi River system, is currently being considered for water level drawdown to consolidate sediments and stimulate submersed and

emergent macrophyte growth. Surficial sediments collected from a backwater region of this pool (Lawrence Lake) were experimentally dried and rewetted in the laboratory. Changes in sediment physical-chemical characteristics as a result of the desiccation-rewetting process and the implications of these changes to macrophyte growth in this system are reported herein.

METHODS: Lawrence Lake is a relatively large backwater region located on the Minnesota side of Pool 8, Upper Mississippi River (Figure 1). It currently has a diverse macrophyte species assemblage surrounding the perimeter of the lake. In June 2000, over 50 intact replicate sediment cores were collected at a station (depth= 0.7 m) located near the entrance to Lawrence Lake (Figure 1). The sediment samples were collected with a Wildco KB sediment core sampler (Wildlife Supply Co., Saginaw, MI) equipped with a plastic core liner (7.6 cm in diameter and 20 cm long). In the laboratory, the upper 8 cm of sediment were carefully extruded into a core liner that was precut to exactly 10 cm. A rubber stopper was used to seal the bottom of each extruded sediment core. The sediment core incubation systems were desiccated in the laboratory under controlled temperature conditions (~20 ^{o}C) in a darkened room. During the initial stages of the dessication process, only the surface of the sediment core was directly exposed to air. Drying and consolidation of the sediment core over time resulted in exposure of the sides of the core, as well as its surface, to air.

The experimental dessication and reflooding process is shown in Table 1. For each treatment, six sediment cores were sacrificed for determination of physical and chemical characteristics. In order to estimate the loss of moisture from sediment cores over time, the 10-cm core liner and rubber stopper were weighed before and immediately after extrusion of sediment into them. The loss of mass as a result of dessication was calculated by difference and converted to a percentage of the initial mass. For treatments 5 and 6 (Table 2), sediments were rewetted with 100 mLs of lake water (collected from Lawrence Lake) that had been previously filtered through a glass fiber filter (Gelman A/E). When filled with water, sediment cores were completely covered so that direct rehydration could occur via exposure of the top and sides of the sediment core. The sediment core incubation systems containing lake water were covered with plastic film to minimize evaporative losses. The volume of water remaining (i.e., not soaked up by the sediment) prior to sediment analysis was measured and analyzed for total and soluble reactive phosphorus (P), total nitrogen (N), ammonium-N (NH_4-N), and nitrate-nitrite-N (NO_2-NO_3-N) using standard analytical techniques (American Public Health Association (APHA) 1992; see below).

During the rewater phase, water surrounding the sediment cores was analyzed for dissolved oxygen, pH, and eH at 1- to 3-day intervals over a 2-week period. Dissolved oxygen (YSI BOD polarographic probe 5010 series) and pH were precalibrated against Winkler titrations (APHA 1992) and known pH buffer solutions. eH was measured using a combination platinum probe with an AgCl reference electrode (Fisher Acumet) referenced against saturated quinhydrone buffer solutions (pH 4 and pH 7 at 20 ^{o}C).

For each treatment, sediment core dimensions (height and diameter) were measured and the sediment was gently homogenized prior to analysis. Fresh sediment was carefully placed in a crucible and dried at 105 ^{o}C to a constant weight for determination of moisture content and sediment density (Allen et al. 1974), then combusted at 550 ^{o}C in a muffle furnace for determination of particulate

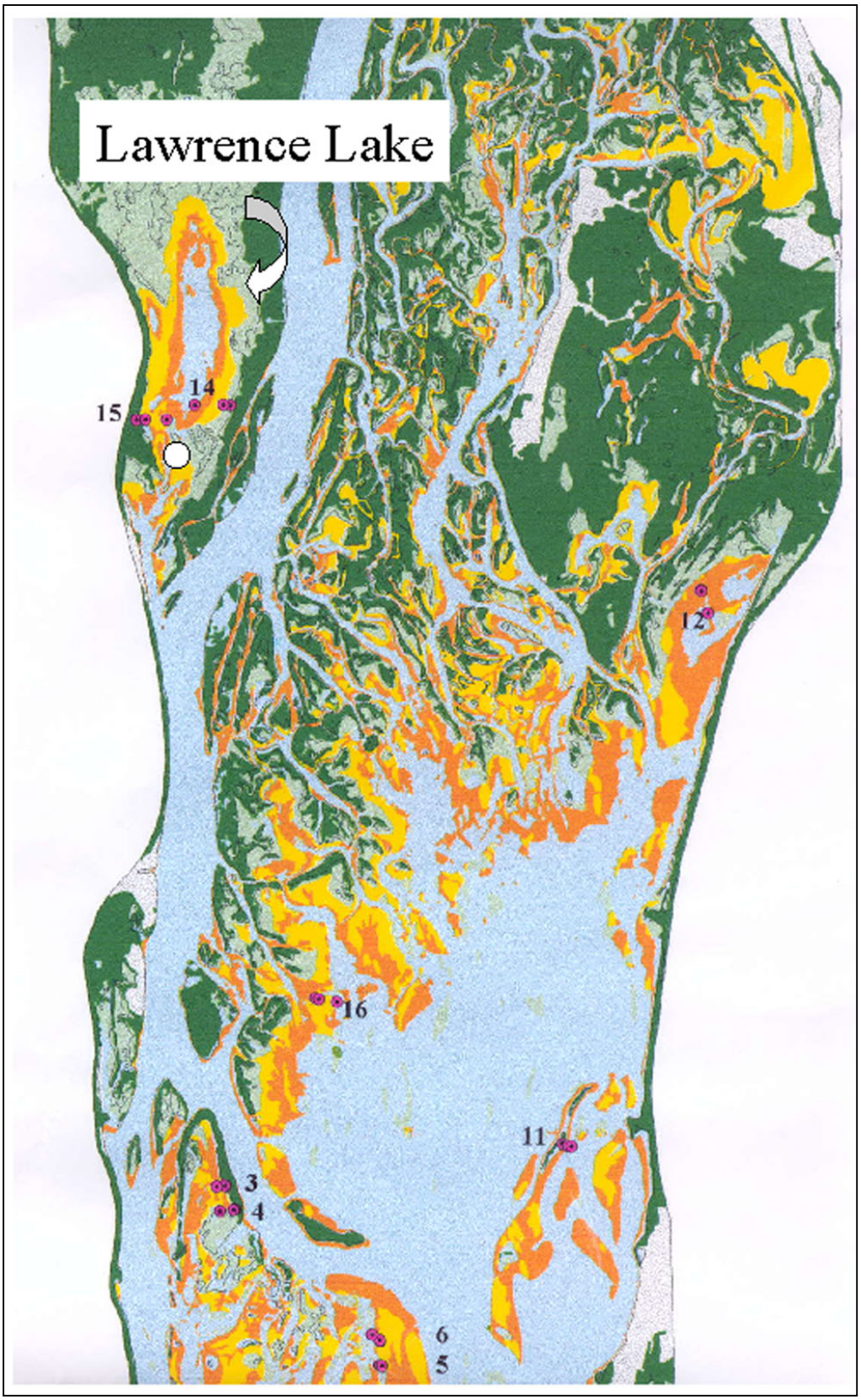

Figure 1. Sampling location in Lawrence Lake

Table 1
Mean[1] (1 S.E; n=6) Physical and Chemical Characteristics of Sediment Collected in Lawrence Lake

Variable	Concentration
Moisture Content, %	80.0 (0.2)
Sediment Density, g/mL	0.212 (0.002)
Particulate Organic Matter, %	12.4 (0.2)
Porewater Soluble Reactive P, mg/L	2.506 (0.219)
Sediment Loosely Bound P, mg/g	0.022 (0.002)
Sediment Iron-bound P, mg/g	0.665 (0.024)
Sediment Aluminum-bound P, mg/g	0.210 (0.036)
Sediment Calcium-bound P, mg/g	0.064 (0.014)
Sediment Labile Organic P, mg/g	0.281 (0.031)
Sediment Refractory Organic P, mg/g	0.208 (0.032)
Sediment Total P, mg/g	1.450 (0.017)
Porewater Ammonium-N, mg/L	1.536 (0.355)
Porewater Nitrate-Nitrite-N, mg/L	0
Sediment Exchangeable Ammonium-N, mg/g	0.072 (0.004)
Sediment Organic N, mg/g	
Sediment Total N, mg/g	6.518 (0.091)
P Release Under Oxic Conditions, mg m^{-2} d^{-1}	0.4 (0.1)
P Release Under Anoxic Conditions, mg m^{-2} d^{-1}	9.2 (1.2)
NH_4-N Release Under Oxic Conditions, mg m^{-2} d^{-1}	48.2 (8.4)
NH_4-N Release Under Anoxic Conditions, mg m^{-2} d^{-1}	71.2 (3.4)
NO_2NO_3-N Release Under Oxic Conditions, mg m^{-2} d^{-1}	11.2 (7.5)

[1] 1 S.E., n = 6.

Table 2
Description of Sediment Desiccation Timetable

Treatment #	Description
1	Initial conditions before the start of desiccation
2	15% dewatered (i.e., 22 days of dessication)
3	40% dewatered (i.e., 38 days of dessication)
4	100% dewatered (i.e., 70 days of dessication)
5	Rewetted for 1 day
6	Rewetted for 3 weeks

organic matter content (POM). Care was taken to minimize compaction of sediment in the crucible for density determination. Additional sediment was centrifuged at 4 ^{o}C at 3000 rpm for approximately 2 hr for separation of sediment porewater. The porewater was carefully decanted under a nitrogen atmosphere, filtered through a 0.45-µm filter (Gelman GN-6 Metricel), and analyzed for soluble reactive P, NH_4-N, and NO_2NO_3-N using methods described in APHA (1992). Total sediment N and P concentrations were analyzed colorimetrically using Lachat QuikChem procedures (Lachat Method 10-107-06-2-D for N and 13-115-06-1-B for P; Zellweger Analytics, Lachat Div., Milwaukee, WI) following digestion with sulfuric acid, potassium sulfate, and red mercuric oxide (Plumb 1981). Sequential fractionation of inorganic P in the sediments was conducted according to Hieltjes and Lijklema (1980) and Nürnberg (1988) for the determination of ammonium-chloride-extractable P (NH_4Cl-P; loosely bound and $CaCO_3$-adsorbed P), bicarbonate-dithionite-extractable P (BD-P; iron-bound P), sodium hydroxide-extractable P ($NaOH_{reactive}$-P; aluminum-bound P), and hydrochloric acid-extractable P (HCl-P; calcium-bound P). A subsample of the NaOH extract was digested with potassium persulfate to determine unreactive NaOH-extractable P ($NaOH_{nonreactive}$-P; Psenner and Puckso (1988)). Labile organic P was calculated as the difference between $NaOH_{reactive}$ -P and $NaOH_{nonreactive}$-P. Refractory organic P was calculated as total P minus labile organic P and inorganic P fractions. Each extraction was filtered through a 0.45-µm filter, adjusted to pH 7, and analyzed for soluble reactive P. The loosely bound P, iron-bound P, and labile organic P fractions in the sediments are classified as labile (i.e., exchangeable) sediment P, and the aluminum-bound P, calcium-bound P, and refractory organic P fractions as refractory (i.e., not exchangeable) sediment P (Penn et al. 1995). Exchangeable NH_4-N plus porewater NH_4-N were determined by cation exchange according to Bremner (1965). Exchangeable NH_4-N was corrected for porewater NH_4-N concentrations by difference. Sediment organic N was calculated as the difference between total N and the sum of exchangeable NH_4-N, porewater NH_4-N, and porewater NO_2NO_3-N.

N and P release from sediment under oxic and anoxic conditions was determined using additional sediment cores that were subjected to treatments 1 and 6. Experimental cores were transferred to a core liner (6.5-cm ID and 25-cm length) and 300 ml of filtered (Gelman A/E glass fiber) lake water were siphoned onto the sediment. The systems were sealed with rubber stoppers. For treatment 6,

consolidation of sediments resulted in a decrease in the sediment surface area and core length. Thus, it was much smaller than the diameter of the sediment incubation system. To prevent exposure of the sides of the core to overlying water during nutrient release analysis, the sides of the core were first gently wrapped in two layers of cellophane (leaving the top open), and then fine-grained sand was gently packed around the sides of the core after placement in the incubation system.

The redox environment in each system was controlled by bubbling the water with air (for oxic conditions) or nitrogen (for anoxic conditions). Sediment systems were incubated at 20 ^{o}C over a 1-week period to simulate in situ temperatures during summer. Samples were collected in the middle of the water column in each system on a daily basis for SRP and NH_4-N and NO_2NO_3-N analysis. Rates of N and P release from the sediment (mg m^{-2} d^{-1}) were calculated as the linear change in concentration in the overlying water divided by time and the area of the incubation core liner. Changes in the surface area of the sediment core due to desiccation were factored into the rate calculations.

RESULTS: The surface sediments at the sampling station in Lawrence Lake exhibited a high moisture content and low sediment density, indicative of flocculent, fine-grained particles (Table 1). Sediment inorganic P fractions (i.e., loosely bound P, iron-bound P, aluminum-bound P, and calcium-bound P) accounted for 66 percent of the sediment total P, while sediment organic P fractions (i.e., labile and refractory organic P) accounted for 34 percent of the sediment total P. Iron-bound P was the dominant sediment inorganic P fraction (69 percent), followed by aluminum-bound P (22 percent), and calcium-bound P (7 percent). Loosely bound P accounted for only 2 percent of the sediment inorganic P fraction. Sediment total N was dominated by the organic N fraction (98 percent; Table 1). Sediment exchangeable NH_4-N accounted for approximately 1 percent of the sediment total N. Porewater concentrations of NH_4-N and SRP, as well as rates of NH_4-N and P release under anoxic conditions, were high relative to other eutrophic aquatic systems (Barko and Smart 1986; Nurnberg et al. 1986).

During the dessication process (i.e., treatments 1-4), sediment cores lost water at a constant rate of 1.5 percent water mass d^{-1} (Figure 2). Complete dryness occurred in approximately 70 days. Sediment cores consolidated as a consequence of water loss, resulting in a nearly constant decline in the sediment core length from approximately 8.5 cm to less than 4 cm (Figure 2). Core diameter declined from approximately 6.5 cm for treatment 1 to 4.3 cm for treatment 6.

During the rewetting process, the lake water used to rehydrate the sediment cores exhibited detectable dissolved oxygen during the first 2 days of exposure (Figure 3). Dissolved oxygen declined to less than 0.5 mg/L after day 2 of exposure. The eH of the lake water was greater than 75 mV during the first 2 days of exposure to desiccated sediment. In conjunction with low dissolved oxygen, it declined to less than -100 mV after day 2 of the rewetting process. The pH of lake water remained nearly constant at about 7.1 throughout the rewetting procedure.

The actual mean moisture content (i.e., the percentage of moisture as a function of sediment fresh weight as opposed to the percentage of water lost from the cores) declined, while mean sediment density increased nearly 7 times during the dessication process (Figure 4). Moisture content increased, while sediment density declined, as a result of rewetting the sediment cores for 1 day and

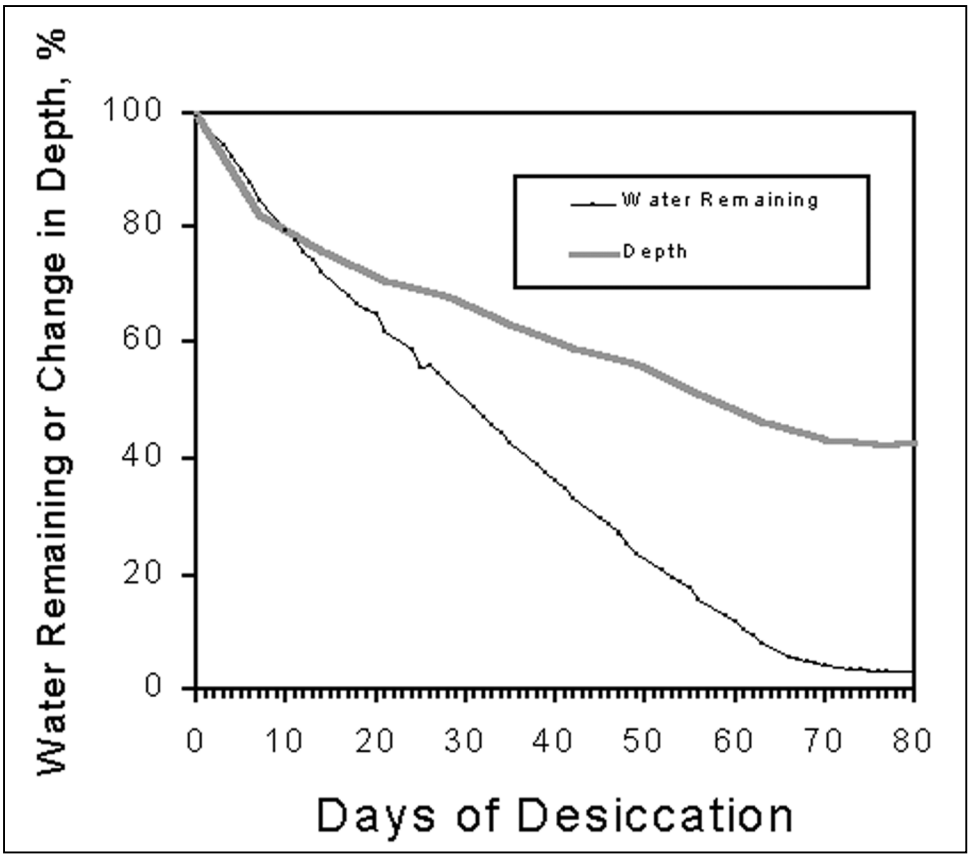

Figure 2. Changes in the percent water remaining and percent change in sediment core depth as a result of experimental desiccation of sediment cores measuring 8 cm long by 8 cm diameter

for 3 weeks (i.e., treatments 5 and 6). However, the mean moisture content of the sediment cores after the dessication-rewetting process was significantly lower (i.e., less than 40 percent; $p < 0.05$), compared to the initial mean moisture content of 80 percent (Table 1 and Figure 4). Similarly, mean sediment density was significantly higher ($p < 0.05$) after the dessication-rewetting process, compared to its initial mean concentration (Figure 4). Mean particulate organic matter content declined slightly ($p < 0.05$) from its initial mean concentration as a result of the dessication-rewetting process.

Sediment dessication and rewetting resulted in marked changes in sediment P characteristics (Figure 5). The mean mass of porewater P declined dramatically during the dessication process. It increased slightly as a result of rewetting the sediment cores; but there was an overall net loss of porewater P mass as a result of the dessication-rewetting process. The mean mass of loosely bound P increased substantially during the dessication process, then declined slightly as a result of rewetting the sediment cores. Overall, however, there was a net increase in loosely bound P as a result of the desiccation-rewetting process (i.e., treatment 1 versus treatment 6). The mean mass of aluminum-bound P and calcium-bound P remained nearly constant during the dessication process. However, rewetting of the sediment cores coincided with a significant increase in the mean mass of these constituents. In contrast, the mean mass of iron-bound P remained constant during the dessication

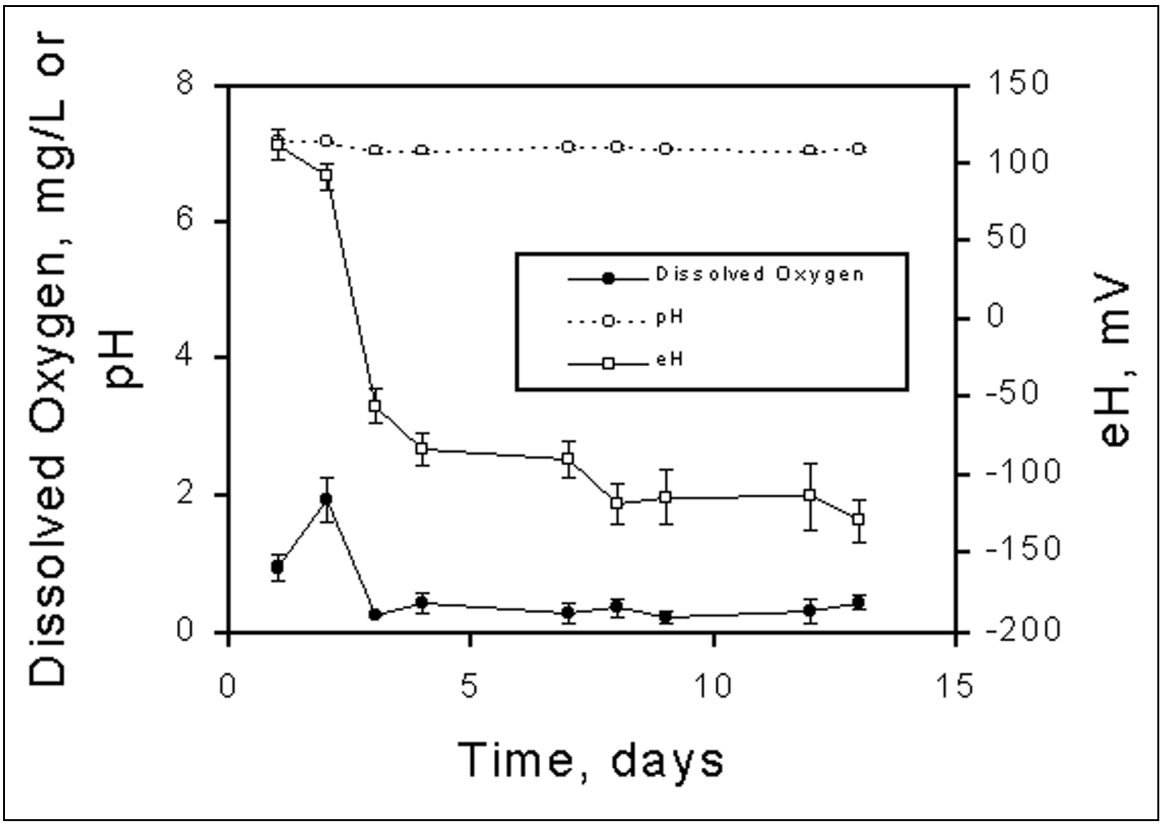

Figure 3. Variations in dissolved oxygen, pH, and eH of filtered lake water used to rehydrate desiccated sediment cores

process and declined significantly ($p < 0.05$) as a result of the rewetting process. The mean mass of sediment organic P (both labile and refractory organic P fractions) appeared to remain approximately constant throughout the dessication-rewetting process.

Overall net changes in sediment P fractions (i.e., treatment 1 versus treatment 6) as a result of the dessication-rewetting process are shown in Table 3. In general, sediment total P and organic P mass did not change significantly ($p > 0.05$) as a result of the dessication-rewetting process (Table 3). In contrast, loosely bound P, aluminum-bound P, and calcium-bound P exhibited a net increase in mass ($p < 0.05$) while iron-bound P exhibited a nearly equivalent net decrease in mass ($p < 0.05$). Less than 0.25 mg total P was measured in the excess water that was not soaked up by the sediment during the rewetting process, suggesting that loss of P mass via diffusion and desorption out of the core was minor in relation to exchanges of P between the various sediment P fractions.

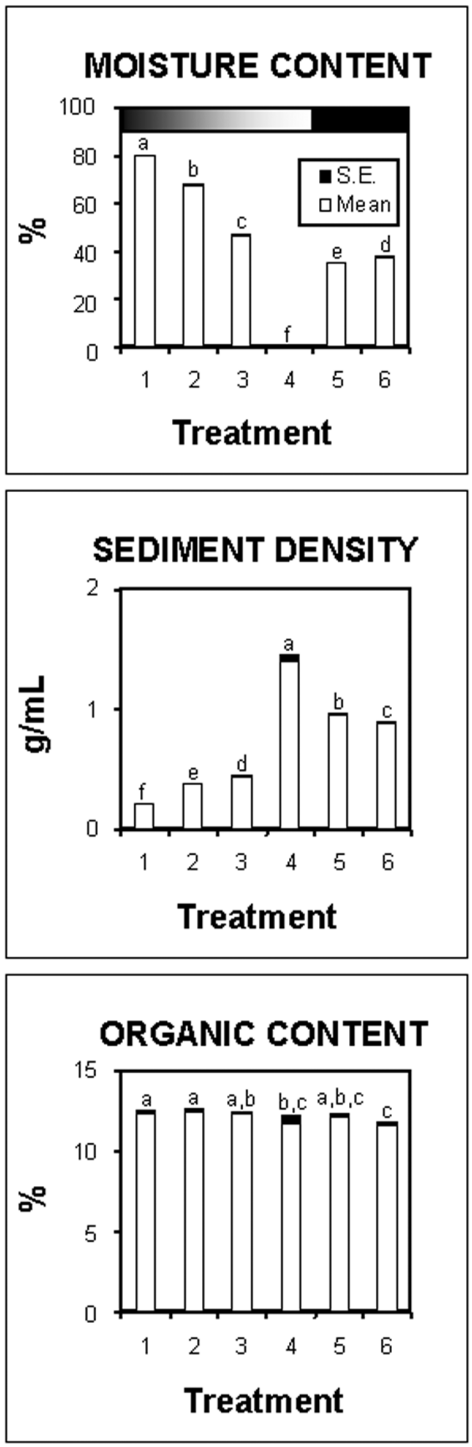

Figure 4. Variations in mean and standard error (SE; n=6) sediment moisture content, sediment density, and organic matter content of the sediment as a function of treatment. Treatment 1 = initial conditions, 2 = 15% dewatered, 3 = 40% dewatered, 4 = 100% dewatered, 5 = rewetted for 1 day, and 6 = rewetted for 3 weeks. Shaded bar indicates relative dryness of sediment cores (black = wet; white = dry). Different letters indicate significant differences in means ($p<0.05$) based on Duncan's Multiple Range Analysis (ANOVA; SAS 1994)

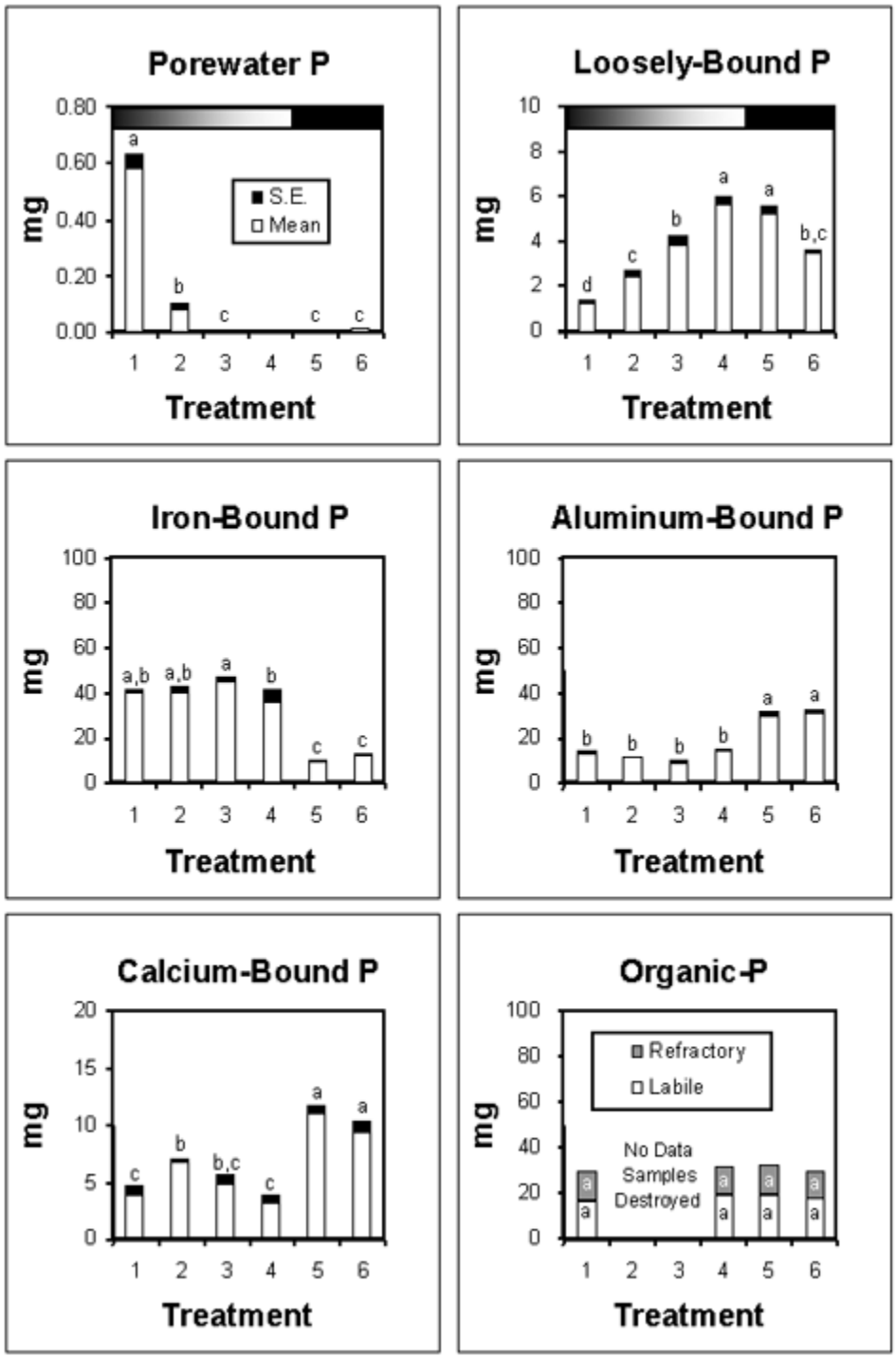

Figure 5. Variations in mean and standard error (SE; n=6) porewater phosphorus (P), loosely bound P, iron-bound P, aluminum-bound P, calcium-bound P, and organic P of the sediment as a function of treatment. Treatment 1 = initial conditions, 2 = 15% dewatered, 3 = 40% dewatered, 4 = 100% dewatered, 5 = rewetted for 1 day, and 6 = rewetted for 3 weeks. Shaded bar indicates relative dryness of sediment cores (black = wet; white = dry). Different letters indicate significant differences in means ($p<0.05$) based on Duncan's Multiple Range Analysis (ANOVA; SAS 1994)

Table 3
Changes in the Mean[1] (1 S.E.) Mass Sediment P Constituents as a Result of the Desiccation-Rewetting Process[2]

Variable	Net Change (mg)
Porewater P	-0.6 (±0.1)*
Loosely bound P	2.2 (±0.4)*
Iron-bound P	-27.8 (±1.7)*
Aluminum-bound P	19.0 (±3.7)*
Calcium-bound P	5.5 (±1.7)*
Labile Organic P	0.6 (±2.6)
Refractory Organic P	-0.2 (±4.4)
Total Sediment P	-0.7 (±2.0)

[1] Treatment 1 versus treatment 6.
[2] Asterisks indicate significant changes ($p < 0.05$) mass (t-test, SAS (1994)).

Changes in sediment N fractions during the dessication and rewetting process are shown in Figure 6. The mean mass of porewater NH_4-N and sediment exchangeable NH_4-N declined significantly ($p < 0.05$) during dessication. However, the mean mass of these constituents increased substantially as a result of rewetting (Figure 6). Although there was no detectable porewater NO_2NO_3-N during the dessication process, NO_2NO_3-N in the porewater exhibited a slight increase as a result of the rewetting process. The mean mass of sediment organic N generally declined throughout the dessication/rewetting process.

Overall, mean sediment total N exhibited a statistically significant net loss of mass as a result of the dessication-rewetting process which represented 17 percent of the original mean sediment N mass (Table 4). Total N mass measured in the excess water removed from the cores after the rewetting process (i.e., 1.3 mg) did not account for this net loss in sediment total N. Coincident with this pattern was a net decline in the mean mass of sediment organic N. There was no net change in the mean mass of porewater NO_2NO_3-N over the dessication-rewetting process. Although porewater NH_4-N and sediment exchangeable NH_4-N exhibited a net increase in mean mass as a result of the dessication-rewetting process, these constituents did not account for losses observed in sediment total and organic N.

As an apparent result of the dessication-rewetting process, mean rates of P release from sediments under anoxic conditions, measured in laboratory incubation systems, declined significantly ($p < 0.05$) over mean rates measured before treatment (Figure 7). This pattern coincided with declines in porewater concentrations of P (Figure 5). Mean rates of P release from sediments were greater under anoxic conditions than oxic conditions both before and after the dessication-rewetting process. The mean rate of P release from sediments under oxic conditions did not change as a result of the

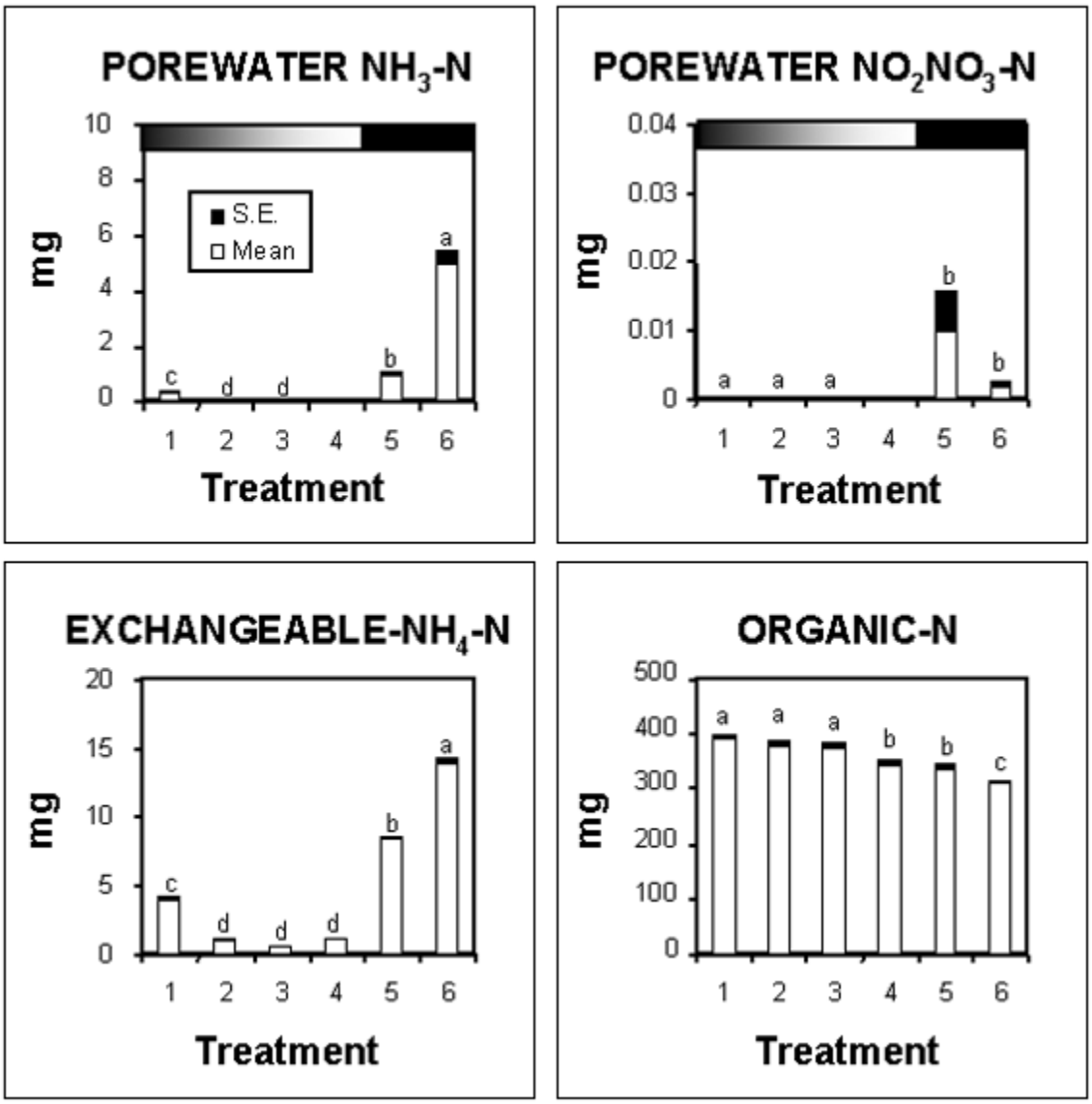

Figure 6. Variations in mean and standard error (SE; n=6) porewater ammonium (NH4-N), porewater nitrate-nitrite (NO_2NO_3-N), Exchangeable NH4-N, and organic nitrogen (N) of the sediment as a function of treatment. Treatment 1 = initial conditions, 2 = 15% dewatered, 3 = 40% dewatered, 4 = 100% dewatered, 5 = rewetted for 1 day, and 6 = rewetted for 3 weeks. Shaded bar indicates relative dryness of sediment cores (black = wet; white = dry). Different letters indicate significant differences in means ($p<0.05$) based on Duncan's Multiple Range Analysis (ANOVA; SAS 1994)

dessication-rewetting process. Mean rates of NH_4-N release from sediment increased substantially under both oxic and anoxic conditions after the dessication-rewetting process, compared to mean rates of N release from sediments measured before treatment (Figure 7). Mean rates of NO_2NO_3-N release under oxic conditions also increased substantially after the dessication-rewetting process (Figure 7). These patterns coincided with increases in concentrations of porewater and exchangeable NH_4-N and porewater NO_2NO_3-N after rewetting (Figure 6).

Table 4
Changes in the Mean (1 S.E.) Mass[1] Sediment N Constituents as a Result of the Desiccation-Rewetting Process[2]

Variable	Net Change (mg)
Porewater NH_4-N	4.6 (±0.4)*
Porewater NO_2NO_3-N	< 1
Exchangeable NH4-N	9.9 (±0.6)*
Organic N	-82.1 (±12.1)*
Total Sediment N	-67.6 (±10.8)*

[1] Treatment 1 versus Treatment 6.
[2] Asterisks indicate significant changes ($p<0.05$) mass (t-test, SAS (1994)).

DISCUSSION: The experimental dessication process resulted in substantial sediment consolidation, as the percent moisture content and organic matter content declined, while sediment density increased after the rewetting process. Similarly, James et al. (2001) found that moisture and organic matter content decreased, while sediment density increased, following drawdown of Big Muskego Lake for purposes of sediment desiccation and consolidation. These changes in sediment characteristics need to be considered within the context of efforts to stimulate macrophyte growth in shallow aquatic systems. In particular, Barko and Smart (1986) demonstrated experimentally that low sediment density and high organic matter content suppressed macrophyte growth, while higher density sediment with lower organic matter content were associated with greater macrophyte growth response. Higher density sediments may provide a better plant rooting medium for macrophytes, resulting in increased root contact with sediment particles and associated nutrients (Barko and Smart 1986).

Sediment consolidation via desiccation may also indirectly enhance macrophyte growth potential by increasing the critical shear stress required to resuspend sediments (Scheffer 1998). Frequent resuspension of flocculent sediments and resulting high turbidity in shallow, wind-exposed lakes often leads to light-limiting conditions for macrophyte growth (Blom, Van Diun, and Vermaat 1994). For instance, using macrophyte growth models, Best, Boyd, and James (2001) suggested that the growth of *Vallisneria americana* and *Potemogeton pectinatus* becomes limiting at turbidity values in excess of 15-18 NTU in shallow Peoria Lake, IL. Because resuspension is frequent in this lake due to a low critical shear stress of sediments in relation to wind-generated shear stress (James, Best, and Barko, in preparation), turbidity usually exceeds the threshold required for macrophyte growth in this system. Thus, the reproductive survival and persistence of submersed species can decline substantially as a result of high turbidity (Doyle and Smart 2001; Kimber, Korschgen, and Van der Valk 1995), leading to population extinction. By increasing the critical shear stress required to resuspend sediments via desiccation and compaction to lessen the frequency of resuspension and periods of low light conditions, light conditions may improve substantially for successful macrophyte growth.

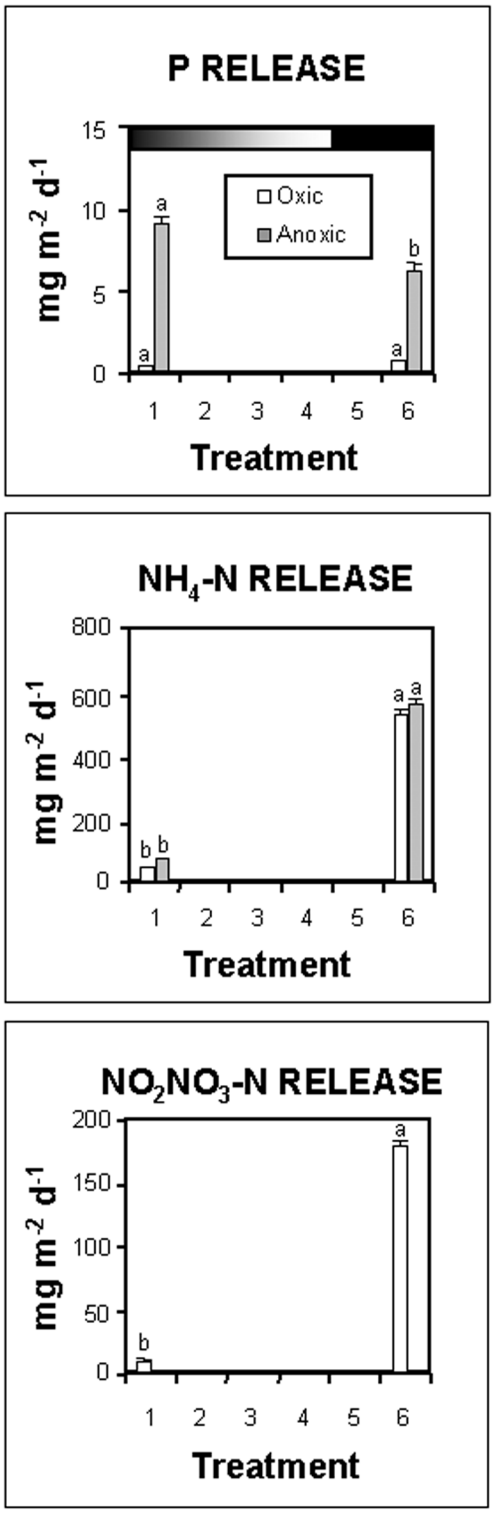

Figure 7. Variations in mean and standard error (SE; n=6) rates of phosphorus (P), ammonium-N (NH4-N), and nitrate-nitrite-N (NO_2NO_3-N) for treatments 1 and 6. Treatment 1 = initial conditions and 6 = rewetted for 3 weeks. Shaded bar indicates relative dryness of sediment cores (black = wet; white = dry). Different letters indicate significant differences in means ($p < 0.05$) based on Duncan's Multiple Range Analysis (ANOVA; SAS 1994).

Dynamic changes occurred in P associated with sediment as a result of sediment dessication and rewetting. It appeared that the loosely bound, aluminum-bound, and calcium-bound P fractions increased in conjunction with declines in the iron-bound P fraction after dessication and rewetting. In particular, 84 percent of the change in the iron-bound P fraction could be accounted for by increases in the more refractory aluminum- and calcium-bound P fractions, suggesting that dessication and rewetting of these sediments was associated with a net loss of sediment P from the labile P pool. Consistent with this change in the labile P pool of the sediments were declines in porewater concentrations of P and the rate of P release from the sediments under anoxic conditions. These patterns suggested that potential P flux from the sediments via diffusive processes was suppressed due to irreversible binding of P to aluminum and calcium (i.e., apatite formation) during the dessication-rewetting process.

In contrast, others have found that sediment desiccation resulted in an increase in iron-bound P. De Groot and Van Wijck (1993a, 1993b) attributed this pattern to oxidation of iron and sorption of P onto these particles during the desiccation process. They also suggested that some of the P mineralized from the organic P fraction was scavenged by iron and calcium in the sediment. In particular, they observed a decrease in both refractory and labile organic P fractions as a result of the desiccation process, suggesting enhanced microbial activity as a result of oxidation of sediments. Surprisingly, a statistically significant decline was not observed in either the labile or refractory sediment organic P fraction, nor was there an increase in the iron-bound P fraction as a result of desiccation and rewetting. The experimental drying process may have inhibited microbial activity and breakdown of organic P, compared to in situ desiccation of sediments in the De Groot and Van Wijck (1993a, 1993b) study.

Reasons for the marked increases in the aluminum and calcium-bound P fractions as a result of desiccation and rewetting of sediments from Lawrence Lake are not completely known. Perhaps the experimental drying and compaction process induced an early diagenesis of some of the sediment P (Penn et al. 1995), resulting in the conversion of some labile P into more refractory P (i.e., P that is not exchangeable and eventually buried; aluminum- and calcium-bound P). Whereas dessication of in situ lake and river sediments is usually subjected to temporary periods of rewetting due to precipitation, higher water levels, etc., sediments in this study were completely dried and compacted in an uninterrupted manner. This more complete desiccation pattern perhaps led to conversion of a portion of the labile pool to refractory P.

Changes in sediment P pools as a result of sediment desiccation appeared to provide both negative and positive impacts to macrophyte growth potential. Although P is generally not limiting to macrophyte growth, macrophytes derive most of their nutritional needs for this nutrient via root uptake from the sediments. Barko and Smart (1980) demonstrated that both porewater and labile P pools are accessed via root uptake. Since sediments from Lawrence Lake were rich in P, declines in the labile sediment P pool as a result of desiccation and rewetting would likely not impact macrophyte growth in this system, since concentrations are still high relative to growth needs. However, in other aquatic systems, declines in the available sediment P pool to limiting quantities could negatively impact macrophyte growth.

Positive impacts of a net decline in the labile sediment P pool and flux of P from the sediment after desiccation include a reduction in the flux of sediment-derived P to the water column for algal uptake and growth. Reduction in P flux from the sediment may impart a competitive advantage to submersed macrophytes for successful growth via increasing light penetration due to less algal biomass (and turbidity) in the water column. In contrast to patterns for sediments in Lawrence Lake, others have found that dessication can lead to substantial increases in the flux of P from sediments. In Big Muskego Lake, for instance, P flux from the sediment increased by one order of magnitude after desiccation and rewetting (James et al. 2001). Reasons for differences in the response of sediment P flux to the desiccation process for different sediments are not known and need to be investigated. In particular, changes in equilibrium processes (i.e., adsorption and desorption; De Groot and Van Wijck (1993b)) need to be examined in relation to changes in sediment P pools for different sediment types.

Sediment dessication and rewetting resulted in significant increases in sediment exchangeable NH_4-N, porewater NH_4-N and NO_2NO_3-N, and rates of NH_4-N and NO_2NO_3-N release from sediments. These changes are of ecological significance to macrophyte growth potential since root uptake is the principal means of N acquisition and macrophytes are usually limited by the availability of sediment N (Gerloff and Krombholz 1966; Barko et al. 1988; Barko, Gunnison, and Carpenter 1991; Rogers, McFarland, and Barko 1995). In particular, NH_4-N and exchangeable NH_4-N are both readily taken up via root systems. Anaerobic decomposition of organic N to NH_4-N was likely, because although dissolved oxygen in the water used to rewet the sediment was high (i.e., 1-2 mg/L) during the first day of rewetting, it subsequently declined to near zero as it soaked into the sediment, indicating that anoxic conditions were occurring during most of the rewetting process.

Initial rewetting (i.e., treatment 5) resulted in a small, but significant, increase in porewater NO_2NO_3-N, suggesting some oxidation of NH_4-N (i.e., nitrification) during the initial stages of the rewetting process. The initial introduction of oxygen via rewetting would provide a mechanism for conversion of NH_4-N to NO_2NO_3-N in the porewater via bacterial nitrification. The subsequent decline in NO_2NO_3-N and increase in NH4-N in the porewater after 3 weeks of rewetting was likely the result of dissimilatory reduction of nitrate. Declines in porewater NO_2NO_3-N were attributed to denitrification, since introduced water was anoxic after the second day. eH declined to between zero and -125 mV during this time period, which was within the range for nitrate reduction (Tomaszek 1995).

When rewetted sediments were subjected to an oxic environment in laboratory incubation systems for analysis of N and P release from sediments (i.e., overlying water was added to the rewetted sediments and gently bubbled with air), rates of both NH_4-N and NO_2NO_3-N release from sediments increased substantially over initial rates observed prior to the desiccation process. The substantial increases in porewater NH_4-N and exchangeable NH_4-N observed in the sediment as a result of rewetting, and anaerobic metabolism of organic N, perhaps caused a shift in NH_4-N equilibrium toward flux out of the sediment and into the water column of the sediment incubation systems subjected to an oxic environment. Increases in the rate of NO_2NO_3-N release from rewetted sediments under an oxic environment were likely due to nitrification of some of the NH_4-N.

The most unusual feature of sediment N dynamics during the desiccation-rewetting process was an unexplained net loss of sediment N, primarily in the form of organic N, that could not be accounted for by conversion to exchangeable NH_4-N in the sediments or to NH_4-N or NO_2NO_3-N in the pore water. While more research is needed to identify and quantify the transformations that may be resulting in this net loss of N during desiccation-rewetting, possibilities include nitrification followed by denitrification (for NO_2NO_3-N; Mitchell and Baldwin (1999)) and/or volatilization of NH_4-N to NH_3. The pH of water used to rewet the sediment was near 7.0, suggesting that conversion to NH_3-N was minor. Downward shifts in the eH of water and the occurrence of anoxia during the rewetting process implicated the possibility of N loss via denitrification. However, the processes resulting in loss of sediment N need to be identified because if desiccation is leading to net N loss from the sediment, pool elevation drawdowns may have management implications for N regulation in the Upper Mississippi River system.

SUMMARY: Experimental desiccation and rewetting of sediments from Lawrence Lake in Pool 8 of the Upper Mississippi River resulted in a decline in moisture content and organic matter, and an increase in sediment density, indicating consolidation of sediment. Pore water and iron-bound P, and rates of P release under anoxic conditions declined, while aluminum- and calcium-bound P increased as a result of desiccation-rewetting, resulting in a net increase in the sediment refractory P pool. Labile and refractory organic P in the sediment did not change as a result of the desiccation-rewetting process. Pore water NH_4-N and NO_2NO_3-N and exchangeable NH_4-N increased substantially as a result of the desiccation-rewetting process. Coincident with these constituent increases after rewetting were markedly elevated rates of NH_4-N and NO_2NO_3-N release from the sediment. However, there was an overall net loss of organic N as a result of the desiccation-rewetting process that could not be accounted for by increases in other N fractions. This pattern suggested that N was being lost to the atmosphere via denitrification as a result of the desiccation-rewetting process. Increases in available N, coupled with consolidation of flocculent sediments, suggested that desiccation of sediment in Lawrence Lake will likely result in improved conditions for macrophyte growth.

ACKNOWLEDGMENTS: This research was funded by the Aquatic Plant Control Research Program Work Unit 33309. Additional funding was provided by the U.S. Army Engineer District, St. Paul. We gratefully acknowledge Mses. Laura Blegen, Alyssa Boock, and Susan Fox (American Scientists Corp.); Mses. Emily Gilles and Stephanie Sweeney, and Mr. Dale Dressel (American Scientists Corp.); and Messrs. Alan Lamphere and Matthew Pommier of the Eau Galle Aquatic Ecosystem Research Facility, Spring Valley, WI, for field sampling and chemical analysis. We also appreciate the review of this manuscript by Dr. William Richardson, U.S. Geological Survey, Upper Midwest Environmental Management Center, LaCrosse, WI.

POINTS OF CONTACT: This technical note was written by Messrs. William F. James and Harry L. Eakin of the Eau Galle Aquatic Ecosystem Research Facility, Environmental Laboratory (EL), Engineer Research and Development Center (ERDC) and Dr. John W. Barko, EL, ERDC. For additional information, contact the managers of the Aquatic Plant Control Research Program, Dr. Barko (601-634-3654, *John.W.Barko@erdc.usace.army.mil*), or Mr. Robert C. Gunkel (601-634-3722, *Robert.C.Gunkel@erdc.usace.army.mil*). This technical note should be cited as follows:

James, W. F., Eakin, H. L., and Barko, J. W. (2001). "Experimental determination of the impacts of sediment desiccation and rewetting on sediment physical and chemical characteristics in Lawrence Lake, Pool 8, Upper Mississippi River," *APCRP Technical Notes Collection* (ERDC TN-APCRP-EA-05), U.S. Army Engineer Research and Development Center, Vicksburg, MS. *www.wes.army.mil/el/aqua*

REFERENCES

Allen, S. E., Grimshaw, H. M., Parkinson, J. A., and Quarmby, C. (1974). "Chemical analysis of ecological materials," Wiley Press, New York.

American Public Health Association (APHA). (1992). "Standard methods for the examination if water and wastewater," 18th ed., Washington, DC.

Bachmann, R. W., Hoyer, M. V., and Canfield, D. E. (2000). "The potential for wave disturbance in shallow Florida lakes," *Lake Res. Manage.* 16, 281-291.

Barko, J. W., and James, W. F. (1998). "Effects of submerged aquatic macrophytes on nutrient dynamics, sedimentation, and resuspension." *The structuring role of submerged macrophytes in lakes.* E. Jeppesen, Martin Søndergaard, Morten Søndergaard, K. Christoffersen, eds., Springer-Verlag, New York.

Barko, J. W., and Smart, R. M. (1980). "Mobilization of sediment phosphorus by submersed freshwater macrophytes," *Freshwat. Biol.* 10, 229-238.

_____ . (1986). "Sediment-related mechanisms of growth limitation in submersed macrophytes," *Ecology* 67, 1328-1340.

Barko, J. W., Gunnison, D., and Carpenter, S. R. (1991). "Sediment interactions with submersed macrophyte growth and community dynamics," *Aquat. Bot.* 41, 41-65.

Barko, J. W., Smart, R. M., McFarland, D. G., and Chen, R. L. (1988). "Interrelationships between the growth of *Hydrilla verticillata* (L.f.) Royle and sediment nutrient availability," *Aquat. Bot.* 32, 205-216.

Beckett, D. C., Aartila, T. P., and Miller, A. C. (1992). "Contrasts in density of benthic invertebrates between macrophyte beds and open littoral patches in Eau Galle Lake, Wisconsin," *Am. Midl. Nat.* 127, 77-90.

Bengtsson, L., and Hellström, T. (1992). "Wind induced resuspension in a small shallow lake," *Hydrobiologia* 241, 163-172.

Best, E. P. H., Boyd, W. A., and James, W. F. (2001). "Simulation of potential persistence of Sago Pondweed and American Wild Celery in Peoria Lake, Illinois using recent and historical data on light availability and suspended sediments," Abstract presented at the 62nd Midwest Fish and Wildlife Conference, December 3-6, 2000, Minneapolis, MN.

Blom, G., Van Diun, E. H. S., Vermaat, J. E. (1994). "Factors contributing to light attenuation in Lake Veluwe." *Lake Veluwe, a macrophyte-dominated system under eutrophication stress.* Van Vierssen, ed., Kluwer Academic Publishers, Amsterdam, The Netherlands, 158-174.

Bremner, J. M. (1965). "Inorganic forms of nitrogen." *Methods of soil analysis, Part 2: Chemical and microbiological properties.* Am. Soc. Agron., Madison, WI, 1179-1237.

De Groot, C. J., and Van Wijck, C. (1993a). "The impact of desiccation of a freshwater marsh (Garcines Nord, Camargue, France) on sediment-water vegetation interactions. Part 1: The sediment chemistry," *Hydrobiologia* 252, 83-94.

_____ . (1993b). "The impact of desiccation of a freshwater marsh (Garcines Nord, Camargue, France) on sediment-water vegetation interactions. Part 3: The fractional composition and the phosphate adsorption characteristics of the sediment," *Hydrobiologia* 252, 105-116.

Dibble, E. D., Killgore, J. J., and Harrel, S. L. (1997) " Assessment of fish-plant interactions," Miscellaneous Paper A-97-6, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Doyle, R. D., and Smart, R. M. (1993). "Potential use of native aquatic plants for long-term control of problem aquatic plants in Guntersville Reservoir, Alabama: Report 1, Establishing native plants," Technical Report A-93-6, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

_____ . (2001). "Impacts of water column turbidity on the survival and growth of *Vallisneria americana* winter buds and seedlings," *Lake Res. Manage.* 17(1), 17-28.

Gerloff, G. C., and Krombholz, P. H. (1966). "Tissue analysis as a measure of nutrient availability for the growth of angiosperm aquatic plants," *Limnol. Oceanogr.* 11, 529-537.

Hieltjes, A., and Lijklema, L. (1980). "Fractionation of inorganic phosphates in calcareous sediment," *J. Environ. Qual.* 9, 405-407.

James, W. F., and Barko, J. W. (1994). "Macrophyte influences on sediment resuspension and export in a shallow impoundment," *Lake Res. Manage.* 10, 95-102.

James, W. F., Barko, J. W., Eakin, H. L., and Helsel, D. R. (2001). "Changes in sediment characteristics following drawdown of Big Muskego Lake, Wisconsin," *Arch. Hydrobiol.* 151, 459-474.

James, W. F., Best, E. P., and Barko, J. W. "Suspended sediment dynamics and light attenuation characteristics in Peoria Lake, Illinois: Can submersed macrophyte communities improve water quality in this shallow system?" in prearpation, U.S. Army Engineer Research and Development Center, Vicksburg, MS.

Killgore, K. J., Hoover, J. J., and Morgan, R. P. (1991). "Habitat value of aquatic plants for fishes," Technical Report A-91-5, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Kimber, A, Korschgen, C. E., and Van der Valk, A. G. (1995). "The distribution of *Vallisneria americana* seeds and seedling light requirements in the upper Mississippi River," *Can. J. Bot.* 73,1966-1973.

Madsen, J. D., James, W. F., Eakin, H. L., Stewart, R. M., Barko, J. W., and Helsel, D. R. "Restoration of Big Muskego Lake," Letter report submitted to the Wisconsin Department of Natural Resources, in preparation, U.S. Army Engineer Research and Development Center, Vicksburg, MS.

Mitchell, A. M., and Baldwin, D. S. (1999). "The effects of sediment desiccation on the potential for nitrification, denitrification, and methanogenesis in an Australian reservoir," *Hydrobiologia* 392, 3-11.

Nürnberg, G. K. (1988). "Prediction of phosphorus release rates from total and reductant soluble phosphorus in anoxic lake sediments," *Can. J. Fish. Aquatic. Sci.* 44, 960-966.

Nürnberg, G. K., Shaw, M., Dillon, P. J., and McQueen, D. J. (1986). "Internal phosphorus loading in an oligotrophic precambrian shield lake with an anoxic hypolimnion," *Can. J. Fish. Aquat. Sci.* 44, 960-966.

Penn, M. R., Auer, M. T., Van Orman, E. L., and Korienek, J. J. (1995). "Phosphorus diagenesis in lake sediments: Investigations using fractionation techniques," *Mar. Freshwat. Res.* 46, 89-99.

Plumb, R. H. (1981). "Procedures for handling and chemical analysis of sediment and water samples," Technical Report EPA/CE-81-1, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Psenner, R., and Puckso, R. (1988). "Phosphorus fractionation; Advantages and limits of the method for the study of sediment P origins and interactions," *Arch. Hydrobiol. Bieh. Erg. Limnol.* 30, 43-59.

Rogers, S. J., McFarland, D. G., and Barko, J. W. (1995). " Evaluation of the growth of *Vallisneria americana* Michx. in relation to sediment nutrient availability," *Lake Res. Manage.* 11, 57-66.

Scheffer, M. (1998). *Ecology of shallow lakes.* Chapman and Hall, London.

Scheffer, M., Rinaldi, S., Gragnani, A., Mur, L. R., and van Nes, E. H. (1997). "On the dominance of filamentous cyanobacteria in shallow, turbid lakes," *Ecology* 78, 272-282.

Smart, R. M., and Dick, G. O. (1999). "Propagation and establishment of aquatic plants: A handbook for ecosystem restoration projects," Technical Report A-99-4, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Statistical Analysis System. (1994). "SAS/STAT user's guide, Version 6, 4th ed.," SAS Institute, Cary, NC.

Timms, R. M., and Moss, B. (1984) "Prevention of growth of potentially dense phytoplankton populations by zooplankton grazing in the presence of zooplanktivorous fish in a shallow wetland ecosystem," *Limnol. Oceanogr.* 29, 472-486.

Tomaszek, J. A. (1995) "Relationship between denitrification and redox potential in two sediment-water systems," *Mar. Freshwat. Res.* 46, 27-32.

NOTE: *The contents of this technical note are not to be used for advertising, publication, or promotional purposes. Citation of trade names does not constitute an official endorsement or approval of the use of such products.*