Journal of Water Resource and Protection, 2025, 17(8), 575-589
https://www.scirp.org/journal/jwarp
ISSN Online: 1945-3108
ISSN Print: 1945-3094

Effect of Sand Capping on Phosphorus Release from Phosphorus-Enriched Coastal Wetland Sediments: A Laboratory Study

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How to cite this paper: Steinman, A.D., Tyrrell, K. and Hassett, M.C. (2025) Effect of Sand Capping on Phosphorus Release from Phosphorus-Enriched Coastal Wetland Sediments: A Laboratory Study. *Journal of Water Resource and Protection*, **17**, 575-589.
https://doi.org/10.4236/jwarp.2025.178030

Received: July 24, 2025
Accepted: August 19, 2025
Published: August 22, 2025

Abstract

Restoration of coastal wetland habitat in the Laurentian Great Lakes often includes addressing excess soil nutrient levels because of prior land use. A wetland restoration project is underway in former celery fields located at the mouth of Mona Lake (MI), a drowned rivermouth system that drains directly into Lake Michigan. One approach proposed to address high phosphorus (P) levels in the sediment is sand capping. This study examined the effectiveness of two sand sources (within-pond and quarry) and two sand cap thicknesses (15 and 30 cm) in inhibiting P release from the native sediment using sediment cores incubated in the laboratory. Our results indicated that there was not a strong difference in sand type or capping depth on P release. Both sand sources resulted in an approximate total phosphorus (TP) reduction of 60% - 70% over time, although this still left TP concentrations well above desirable levels. The quarry sand did better than pond sand in reducing TP, whereas pond sand did better with respect to soluble reactive phosphorus (SRP). Given the relatively small differences in P reduction performance, pond sand is recommended because it is already present on site and therefore is less expensive and easier to move. In addition, pond sand had higher amounts of apatite than quarry sand, suggesting more of the attached P will remain in a stable form. Sand capping is a relatively inexpensive approach for reducing P levels in coastal wetlands, although it may need to be paired with other approaches to reduce nutrient concentrations to desired levels.

Keywords

Sand Capping, Coastal Wetlands, Restoration, Water Quality, Phosphorus

1. Introduction

Coastal wetlands in the Laurentian Great Lakes are under increasing pressure from development, climate change, and eutrophication [1] [2]. Despite providing critical ecosystem services, such as habitat for biota, recreational opportunities for humans, nutrient sequestration, and shoreline protection from floods and wave energy [3], these ecosystems remain under threat. As a consequence, coastal wetland restoration is receiving increasing attention as a means to both restore habitat and improve water quality [4]-[7].

In West Michigan, it was not uncommon for the coastal wetlands associated with drowned river mouths to be drained and converted to agricultural production by Dutch settlers in the late 19th and early 20th centuries. Celery production was a common practice in the region, including the Mona Lake celery flats, which are described in this study. These fields lost their profitability over time, and the celery fields were eventually abandoned. When the pumps were turned off, the muck fields became ponds that were filled with P-rich water from past fertilizer applications [7] [8]. Conversion of available ponds back to wetlands has been prioritized as a way to improve fish and wildlife habitat, as well as reduce phosphorus concentrations that otherwise were impacting downstream drowned river mouth lakes and the nearshore regions of Lake Michigan [9].

The Mona Lake celery flat ponds flow into Mona Lake (**Figure 1**). Steinman and Ogdahl [8] identified these ponds as the major source of phosphorus to Mona Lake, whose total phosphorus (TP) concentrations in the summer usually exceeded 50 μg/L in the surface water and often exceeded 100 μg/L in the hypolimnion. The ponds themselves have TP concentrations ranging from ~250 to 710 μg/L [10]. Recent funding from the Great Lakes Restoration Initiative (https://www.glri.us/) to the Muskegon County Office of the Water Resources Commissioner resulted in a feasibility study to address the high P concentrations in these former celery flats.

There are numerous approaches to reduce high P concentrations in enriched wetlands, including sediment removal, aeration, chemical addition (P-binding agents), and sand capping (see [11] for more details). Studies at the current study site have shown that TP concentrations in both the water column and sediment are extremely high [8] [12] (GEI, pers. comm.), indicating that sediment remediation will be necessary. Given the high cost of overall sediment removal by dredging, the limited capacity of the Muskegon County Wastewater Management Facility to handle the dewatering volume prior to dredging, and possible permit issues, the project team agreed to examine whether sand capping may be a viable alternative to prevent P release from sediment. Moving relatively clean sand from within the pond itself or from a nearby quarry that was available free of charge (excepting transportation costs) made sand capping a potentially attractive management option.

Prior studies have shown that sand capping can reduce P diffusion from sediments via physical trapping and/or adsorption if organic matter is present (e.g., [13] [14]). However, results from prior studies are variable and depended on both the composition of the sand material being deployed and the thickness of its ap-

plication.

This report provides results from a laboratory-based study evaluating the effectiveness of two sources of sand and two depths of sand cover in reducing P release from the Celery Flats sediment into the water column. We hypothesized that 1) the quarry sand would contain lower metal concentrations and thus be a less effective binding agent for P; and 2) the deep sand treatment (30 cm) would result in less P diffusion into the water column than the shallow sand treatment (15 cm), given the greater opportunity for P to bind.

2. Materials and Methods

2.1. Study Site

The Mona Lake celery flats are part of the Mona Lake watershed (200 km^2), located in West Michigan. These former wetlands, now flooded fields, drain into Mona Lake, which in turn connects directly to Lake Michigan through a dredged navigation channel (Figure 1). The flooded fields are bisected by the bermed Black Creek into the north and south fields. The fields were taken out of production in the mid-1980s and allowed to refill for conservation and recreational value. Most of the north pond was placed into the USDA WRP (Wetland Reserve Program) in the late 1990s and has never been actively managed. The south pond is privately owned by riparian homeowners. Private residences are located on both ponds, and their input has been solicited as part of the restoration planning process.

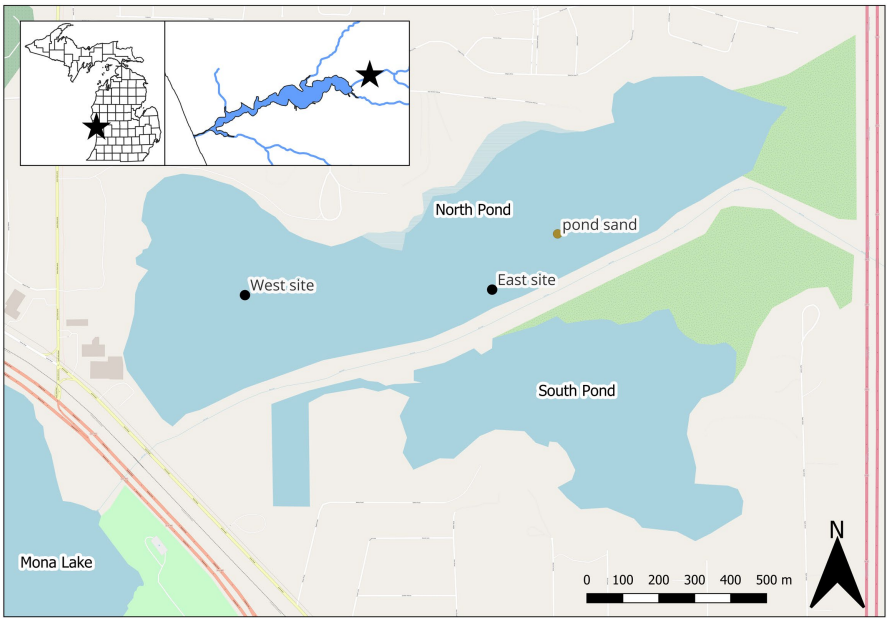

Figure 1. Map of Mona Shores Celery Flat ponds where sand cores and sediment cores were removed for testing. Inset left: star shows the location of Muskegon County in Michigan's Lower Peninsula; inset right: star shows the location of the ponds just upstream of Mona Lake, which flows into Lake Michigan (far left). Basemap data: OpenStreetMap (https://www.openstreetmap.org/copyright). Inset data: State of Michigan GIS Open Data (https://gis-michigan.opendata.arcgis.com/).

2.2. Site Selection

Water quality and sediment chemistry coring sites were selected by project partners from areas previously studied by GEI Consultants and the Steinman Lab at GVSU as a direct follow-up to a recent pre-restoration monitoring study in the north pond of the Mona Lake celery flats [10] [12]. Coring sites were selected based on their previously observed range of medium (west pond: 1900 mg/kg dry weight) and high (east pond: 3200 mg/kg dry wt.) sediment TP concentrations (**Figure 1**) [8] (GEI, unpubl. data). Likewise, the site selected for pond sand collection was chosen for its low TP concentration (75 mg/kg dry wt.; **Figure 1**).

2.3. Field Sampling

Grab samples of surface water at each site were collected from a kayak on May 13, 2024, in acid-washed carboys to be used to maintain core tube water levels throughout the incubation study. Additional samples were taken in acid-washed 250 mL plastic bottles for characterization of initial water conditions (henceforth referred to as Day-1) and were returned to the lab to be measured for TP and SRP as described below. Carboy water was filtered twice in the laboratory using a peristaltic pump with a series of WaterTEC filters (Graver Technologies, Glasgow, DE, USA), first targeting particulates in the 1 - 10 μm range and again in the 0.2 - 10 μm range.

Wet pond sand was collected from one location in the north pond via shovels and 5-gallon buckets. Dry quarry sand was collected, with permission, from a nearby off-site sand quarry in Egelston Township (Muskegon County, MI), also via a shovel and a 5-gallon bucket. For both sand amendment sources, no major effort was made to dry or filter out organic debris or small rocks, mimicking how pond sand might be used in a future restoration scenario. In the lab, sand was gently poured into the open tops of the core tubes and allowed to filter through the water column before settling on top of core sediments. Separate samples from each sand amendment type were collected in Ziploc bags for future sand characterization. Additionally, the two field characterization sediment cores were carefully dewatered via a peristaltic pump, and the top 10 cm of sediment was collected. All bags were then refrigerated until further analysis.

Thirteen sediment cores were collected at each of the east and west sites on May 14, 2024, capturing the top 25 cm of the benthos and 25 cm of overlying water using modified piston cores [15]. Twelve of the 13 cores from each site were designated for the incubation study, and the remaining one core per site was used to characterize field sediments.

2.4. Experimental Design

The 12 experimental sediment cores from each site were evenly divided into two treatment groups that received either quarry-sourced or pond-sourced sand. Each treatment group was further divided into two groups that received depths of either 15 or 30 cm of sand, resulting in a total of n = 3 cores for each combination of

sand source and sand amendment depth per site. Hence, the experimental design was a 2 × 2 full factorial design with factors of sand type (quarry vs. pond) and sand depth (15 vs. 30 cm) at 2 sites, with 3 replicates of each treatment. We used data from [10] to provide data for a control treatment (no sand added), which was conducted using the same experimental design (see below) but was conducted three weeks later than the current study. Ideally, the controls would have been conducted at the same time, but logistically it was not possible. Nonetheless, environmental conditions in the field were similar on both sampling dates (water temperature: 25˚C vs 23˚C; SRP: 13 µg/L vs. 7 µg/L; control vs. treatment, respectively), and the experimental set-ups in the lab were identical. As a consequence, we believe the data from [10] can be applied as a control, with appropriate caution.

2.5. Phosphorus Release Experiment

After field collection of sediment, the core tubes were sealed with an acid-washed rubber cap in the field and transported back to the laboratory for analysis. After settling, water columns in the core tubes were adjusted to a uniform height of 25 cm. Cores were then re-sealed with plastic knockout caps that were pre-drilled with a small hole and installed into racks in a temperature-controlled incubator chamber set to mimic pond water temperature conditions. Gas lines were plumbed through each core's knockout cap and were bubbled with atmospheric gas to oxygenate the core water columns. Additional sampling port tubes were inserted through the knockout caps to allow for easier water quality sampling. Cores were allowed to continue settling for 24 hours before Day 1 water quality samples were taken (described below), and replacement filtered water was added to maintain the 25 cm water column height. Water quality samples were taken twice more over the duration of the study: mid-incubation on Day 10 (May 24, 2024) and at the end of the incubation on Day 20 (June 3, 2024).

2.6. Analytical Methods

Of the water collected throughout the incubation, 20 mL was sampled for TP, and a separate 20 mL was sampled and filtered through an acid-washed 0.45 µm nylon filter for SRP analysis. TP and SRP samples were refrigerated at 4˚C until being analyzed within two weeks on a Seal AQ2 discrete autoanalyzer [16]. Following each sampling event, an equivalent volume of filtered water was slowly syringed back into each core to maintain a constant water volume.

After the Day 20 water-quality sampling, cores were carefully dewatered via peristaltic pump, and the top 10 cm of core overlying sand and sediment was collected into Ziploc bags and refrigerated. These sediment bags, as well as the field characterization and sand amendment samples, were later separately homogenized and subsampled for sediment organic matter, ash-free dry mass, sediment TP, phosphorus fractionation analyses, and metals analyses.

Phosphorus fractionation [17] [18] was conducted on 2-g subsamples from incubation cores, field characterization cores, and raw sand samples. When present,

sediment porewater volumes within samples were minimal and thus were not independently analyzed prior to fractionation. Samples were sequentially fractionated using the following reagents to measure bound sediment phosphorus fractions: 1) ammonium chloride (NH_4Cl) extracts labile and loosely sorbed P; 2) buffered dithionite (BD) extracts reductant-soluble P; 3) sodium hydroxide (NaOH) extracts Fe- and Al-bound P; and 4) hydrochloric acid (HCl) extracts Ca- and Mg-bound P.

Metal analysis was conducted at Trace Analytical Laboratories, Inc. (Muskegon, MI). Aluminum, calcium, and iron were measured using EPA method 6010D, and manganese was measured using EP method 6020B. Metals data from the pond and quarry sand are based on single cores and unreplicated.

2.7. Data Analysis

All statistical analyses were done using R statistical software (v4.4.1; [19]). Comparisons for incubation and sediment characterization data were done using blocked 2-way ANOVA, with sand type and sand depth as independent variables and site as a blocking factor. Comparisons for fractionation data were done via aligned rank transformation ANOVA using the *ARTool* package in R [20] [21]. Inferential statistics were not applied to the metals data due to a lack of replicates.

3. Results and Discussion

3.1. Sand Characteristics

The pre-incubation percent organic matter (<1%) and TP concentration (~35 mg/kg dry wt.) of the two sand sources were very low compared with those of the muck sediment in the north pond (OM: 37% and TP: 3,150 and 2,410 mg/kg dry wt. at the east and west sites, respectively (Table 1). At the end of the 20-day incubation, TP concentrations in the top 10 cm of the sand caps ranged from 51 - 183 mg/kg dry wt. (Table 1), suggesting limited P accumulation. Even with this increase, TP levels remained orders of magnitude below those of the original sediment.

Table 1. Chemical and metal data of the two sand sources and north pond sediment (top 10 cm from 3 sites) used in the capping study. Observation numbers: pond and quarry sand = 1; north pond sediment = 2.

Substrate Type	Organic Matter (%)	TP-P (mg/kg DW)		Metals Data (mg/g DW)			
		Day −1	Day 20	Al	Ca	Fe	Mn
Pond Sand	0.4	35	51	610	220	870	9.6
Quarry Sand	0.2	37	183	1200	260	1300	21
North Pond sediment	34 - 51	2410 - 3150		No data	9400 - 80,000	6100 - 45,000	140 - 1300

The concentrations of all metals, except calcium, were approximately twice as high in the quarry sand as in the pond sand (Table 1). The metal concentrations in the muck sediment from the north pond, while variable depending on site, were

1 - 2 orders of magnitude greater than in the sand (Table 1). The higher concentrations of putative P-binding metals in the quarry sand compared to the pond sand were unexpected and, as a consequence, indicated that our hypothesis that pond sand would be more effective at sequestering P than quarry sand was not supported.

3.2. Incubation

Sand capping reduced initial water column TP concentration throughout the period of the study; reductions were substantial over the first 10 days and then slowed between Days 10 and 20 (Figure 2). In contrast to the sand cap treatment results, TP concentrations in control sediment cores (collected from the same site but without a sand cap from a study conducted three weeks after the current study [10]) increased over time from an initial concentration of 40 µg/L on Day 1 to a peak of 520 µg/L on Day 10, followed by a decline on Day 20 to 220 µg/L [10].

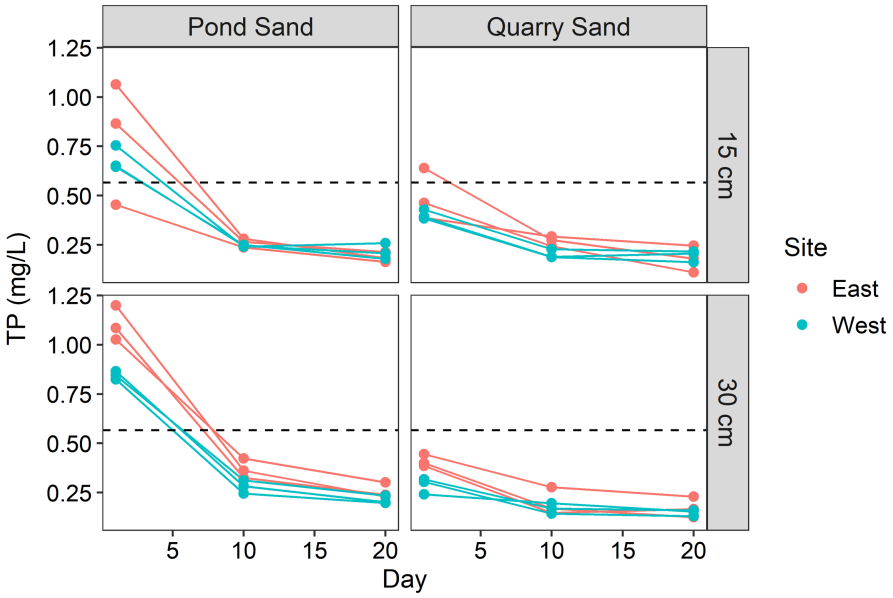

Figure 2. TP concentration in the water column of core tubes treated with two different sand sources (pond vs. quarry) and at two different depths (15 vs. 30 cm). Colored lines represent replicate core tubes (red: east site; green: west site). The dashed line represents the mean TP concentration in ponds on Day-1.

By the conclusion of the 20-day incubation period, TP concentrations in the sediment cores from the current study had decreased below the *in situ* level measured in the ponds on Day-1 (Figure 2). Comparison of the Day 20 measurements using a blocked 2-way ANOVA showed a minor but significant (p-value = 0.014; 95% CI = [0.04, 1.00]) difference between pond and quarry sand sources, with the water column TP being slightly lower in quarry sand core tubes; this difference was more pronounced in the cores with a sand depth of 30 cm, presumably because of greater opportunity for metal binding. No significant differences were observed between the east and west sites within the north pond.

In contrast to TP, both sand sources showed a net release of SRP compared to the low ambient concentration measured on Day-1 (Figure 3). We theorize that the accumulation of SRP was because these experimental cores contained few primary producers, which otherwise would take up the bioavailable P from the water column, convert it into particulate P, and result in an increase in TP. The SRP concentrations reached maxima of ~150 µg/L (Figure 3) in the current study, which have management implications given the bioavailable nature of SRP. However, these concentrations are much lower than those measured in the control treatment (no sand cap; [10]), where SRP concentrations in the water column of sediment core tubes reached a maximum of ~620 µg/L. The lower TP and SRP concentrations in treatments with sand caps vs. the control treatments show that both sand cap types were effective in reducing P release compared to sediments without sand caps, although insufficient to prevent algal blooms.

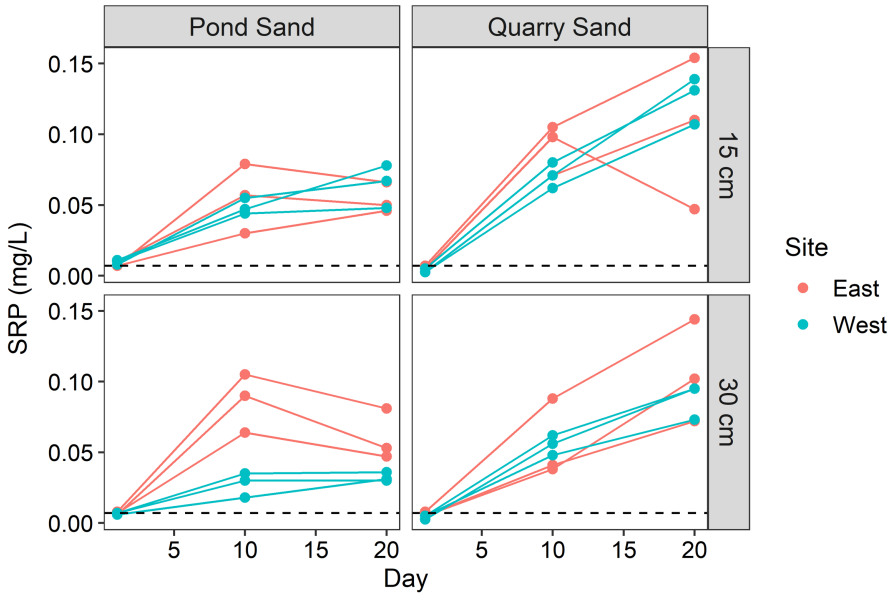

Figure 3. SRP concentration in the water column of core tubes treated with two different sand sources (pond vs. quarry) and at two different depths (15 vs. 30 cm). Colored lines represent replicate core tubes (red: east site; green: west site). The dashed line represents the mean SRP concentration in ponds on Day-1.

In the cores capped with pond sand, SRP accumulation appeared to level out between the Day 10 and Day 20 sampling events, while the quarry sand cores continued to experience increasing SRP concentrations through the end of the incubation (Figure 3). Comparison between the Day 20 values using a blocked 2-way ANOVA revealed that final SRP concentrations were significantly ($p < 0.001$; 95% CI = [0.29, 1.00]) lower in pond sand cores compared to quarry sand cores. No significant differences were observed between coring sites or sand cap depths.

Despite the higher metal concentrations in the quarry sand compared to pond sands, the absence of a major difference in P accumulation in the overlying water column between quarry and pond sands suggests that the higher metal concentra-

tions had limited or no effect on P binding. In addition, sand depth had no significant effect on P concentrations in the water column, counter to our expectations; these combined results indicate that neither the sand sources nor the sand depths applied in this study would make a substantial difference in reducing TP concentrations in these muck sediments.

3.3. Sand Fractionation

The loosely sorbed P (NH_4Cl-P) fraction was undetectable in all sand extractions, as was the BD-P-extracted fraction in both types of pre-incubation sand; however, the latter was measured in relatively low concentrations in the treatment pond and quarry sands (Figure 4). There were clear differences in the other two fractions based on sand origin, with pond sand containing greater concentrations of HCl-extractable P (representing Ca- and Mg-bound P) and quarry sand cores containing greater concentrations of NaOH-extractable P (Fe- and Al-bound P; Figure 4). Variation in P fractions based on sand depth was minimal.

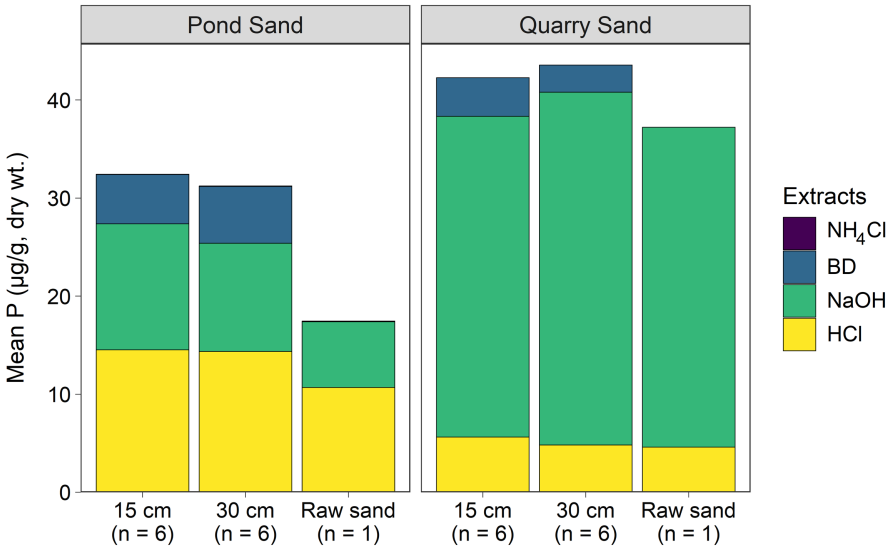

Figure 4. P fractionation results by sand origin at Day 20 of incubation (both 15- and 30-cm depths) and in raw sand samples.

Comparisons between fractions in the raw sand (pre-incubation) and following the 20-day incubation show relative changes in P over time (Figure 5). The only reduction in a P fraction was in loosely sorbed pond sand, but the absolute concentrations are so small as to have no ecological impact. In contrast, all the other fractions exhibited a mean increase over the twenty days, irrespective of sand type, as the P in the underlying sediment core was converted to either mobile (BD and organic P portion of the NaOH extracts) or stable (aluminum-P portion of the NaOH extraction and HCl extract) forms. Unfortunately, we cannot separate the mobile vs. stable portions of the NaOH extraction, but the stable HCl fraction (calcium-bound) was significantly greater in the pond sand treatment compared to the quarry sand treatment (p-value < 0.001; 95% CI = [0.25, 1.00]), despite the

generally similar Ca concentrations in both sand types (Table 1). Hence, the pond sand, despite a slightly lower effectiveness in reducing TP release compared to quarry sand, may be a preferable substrate to use because it is converting more of the TP to a stable fraction that is unlikely to diffuse out of the sand cap and into the water column.

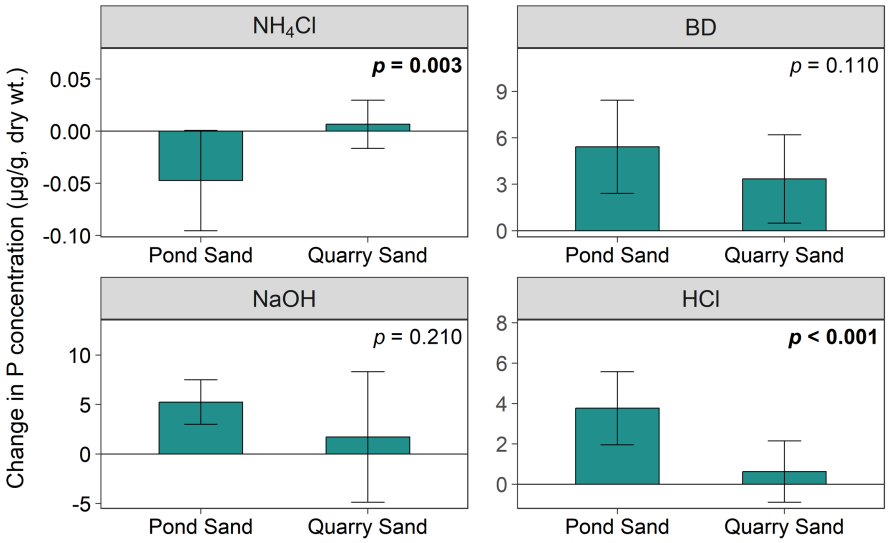

Figure 5. Mean changes in P fraction concentration from raw sand to the end of incubation (n = 12). Error bars represent one SD from the mean. *p*-values represent differences between pond and quarry sand according to a ranked ANOVA with coring site as a blocking factor. Bold indicates a significant difference ($p < 0.05$). Note the differing y-axis scales.

Overall, sand capping had mixed results on P release from the Mona Lake Celery Flats sediments. Both sand sources resulted in a reduction of TP release over time from the sediments, with the quarry sand having a statistically significantly better performance than the pond sand, although the absolute concentration difference was small. It is likely that both sand types acted as physical barriers, as well as possible chemical barriers, to P release and reduced P to a much greater absolute degree than sediment incubations without a sand cap [10]. Regardless of mechanism, the resulting 60% - 70% reductions in TP, while impressive, still resulted in TP concentrations of 161 - 234 µg/L (depending on sand source and sand depth). These concentrations are still substantially greater than what is desirable in Great Lakes coastal wetlands [22] and more than sufficient to stimulate algal blooms. Interestingly, sand cap depth had contrasting effects on TP release depending on sand type; the 30 cm depth resulted in less TP compared to 15 cm in the quarry sand, but the opposite result occurred with pond sand. Hence, it is possible that increasing sand cap depth would lead to lower TP if quarry sand were used.

The effect of sand capping on bioavailable SRP was less encouraging than for TP, as the SRP concentrations unexpectedly increased from their original values during incubations. We speculate that this increase may have been due to several

factors: 1) for the pond sand, there may have been dissolved P in the porewater that was displaced as part of the coring process; 2) the reduction in primary producers in the core tube setup may have reduced the demand for P, allowing the SRP to accumulate; 3) primary producers may have died over the 20-day incubation in darkness, allowing bound organic-P to be released and become bioavailable; and/or 4) there may have been desorption of inorganic P attached to sand grains. Without further testing, it is impossible to know which of these factors, if any, is responsible for the SRP increase, but we believe this would be a short-term phenomenon. The effect of sand cap depth had little impact on SRP release, regardless of sand type.

Prior studies also have shown that sand capping is effective at reducing P release from sediments. Mechanistically, sand capping can reduce P release from sediments mainly via physical blocking and P adsorption, especially if organic matter is mixed in with the sand. *In situ* capping is reviewed in [23], where Liu *et al*. describe the various types of capping materials used to prevent P release. [24] found that a 35-cm-thick layer of clean, coarse sand was effective at reducing vertical flux of trace elements in Hamilton Harbour (Lake Ontario). Another study [25] reported that a 6-cm cap of sand, with a high concentration of HCl-extractant P, completely prevented the release of dissolved inorganic P (and ammonium), and [26] found that a clean sand cap of 4 cm was effective at reducing P release from sediments in a reservoir, and that the thicker the sand cap, the more P release was prevented. A 5 - 10-cm cap of clean sand was very effective in reducing ammonium and SRP flux from Lake Taihu (China) sediments, but after one year, the binding sites became saturated and lost their efficacy [27]. Overall, the results indicate that sand can serve as an effective barrier to P release, but the efficacy will be affected by sand composition, sand cap depth, time of deployment, and characteristics of both the underlying sediment and overlying water.

There are alternative approaches to phosphorus management in enriched wetlands and lakes, such as chemical inactivation, dredging, and phytoremediation [23]. The use of P-binding agents, such as alum and lanthanum, is also being considered in the current project, but there are concerns regarding cost and sediment resuspension caused by a very abundant population of common carp [28] [29]. If the carp can be removed at a reasonable cost, then the application of chemical inactivants will be evaluated.

Sediment dredging was shown to be highly effective in a nearby former celery field that was also abandoned and resulted in highly P-enriched ponds [4]. However, a study that simulated dredging in sediment cores from the Mona Lake celery ponds found dredging to result in P increases compared to controls due to the exposure of high P concentrations at depth [10]; as a consequence, dredging was removed as a management option for these ponds. Phytoremediation was ruled out as a primary P management option in these ponds because of the unstable sediment structure, although plantings are being considered as part of the final design plan once the sediments become more compacted (cf. [30]).

Short-term laboratory studies have the advantage of being able to manipulate environmental factors. However, there are distinct limitations; in our study, this includes the inability of our sediment cores to address sediment migration, bioturbation, resuspension, and consolidation, each of which can affect P release and retention. Nonetheless, the results clearly indicate that sand capping alone is insufficient to reduce phosphorus levels to a non-impairment state.

4. Conclusion

Both sand types were effective in reducing P release from sediment in the Mona Lake celery ponds. There was not a strong difference in the performance of the sand sources. It is likely that either type will perform well (60%-70% reduction in TP), although still resulting in a TP concentration well above what is considered desirable (20 to 30 μg/L). In contrast, SRP concentrations increased over time in both sand types, but were much lower than those in sediment cores incubated without sand (150 vs. 620 μg/L, respectively). The quarry sand did somewhat better than pond sand in reducing TP, whereas pond sand did better with respect to SRP. Given the relatively small differences in P reduction performance, we recommend the use of pond sand because it is already present in the pond and hence will be less expensive and easier to move. In addition, the greater tendency of pond sand to form apatite, a stable form of attached P in the sediment, favors its use over quarry sand. There was little difference in performance based on the 15 cm vs. 30 cm sand depths. Assuming there is little difference in cost or logistics in applying the pond sand, we recommend a deeper depth wherever possible; this provides a margin of safety. These preliminary experiments will provide guidance on the ultimate restoration plan, with implementation contingent on funding availability.

Data Availability

All data are available directly from the authors.

Acknowledgements

We gratefully acknowledge the logistical support of Brenda Moore and Dallas Goldberg from the Muskegon County Office of Water Resources, and Brian Majka's (GEI, Inc.) sharing of technical information. Lab and field support at AWRI was provided by Brian Scull, Kate Lucas, Alexis Porter, and Noah Tucker (AWRI). Additional field support was provided by Joe Vander Yacht (GEI). We are also grateful to Kate Lucas for sharing her preliminary thesis results, Steve Franklin for providing access to the site for sampling, and two reviewers for their constructive comments.

Authors' Contributions

ADS: conceptualization, writing and editing, fund acquisition; KT: study implementation, editing; MCH: study implementation, editing.

Funding

Funding for this work was provided by NOAA through a contract to the Muskegon County Office of Water Resources and by the Allen and Helen Hunting Innovation and Research Fund at AWRI.

Conflicts of Interest

The authors declare no conflicts of interest.

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