THE USE OF SEDIMENT REMOVAL TO REDUCE PHOSPHORUS LEVELS IN WETLAND SOILS AND THE DISTRIBUTION OF PLANT-AVAILABLE PHOSPHORUS IN WETLAND SOILS AND ITS POTENTIAL USE AS A METRIC IN WETLAND ASSESSMENT METHODS

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Title

The Use of Sediment Removal to Reduce Phosphorus Levels in Wetland Soils and the Distribution of Plant-Available Phosphorus in Wetland Soils and its Potential Use as a Metric in Wetland Assessment Methods

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The Supervisory Committee certifies that this ***disquisition*** complies with North Dakota State University's regulations and meets the accepted standards for the degree of

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ABSTRACT

Plant-available phosphorus (P) in wetland soils and its relationship with wetland communities and condition is somewhat unknown in the Prairie Pothole Region (PPR) in North America. Research objectives were to determine if 1) sediment removal reduced P in seasonal wetlands; 2) P could be used as an indicator in wetland condition assessments; 3) a gradient in P amount and wetland elevation existed; and 4) differences of sampling and extraction methods change Olsen P results as a metric in assessments. Soil samples from North Dakota wetlands were collected from two depths (0-15 and 15-30 cm) and analyzed for pH, electrical conductivity (EC), and P (Olsen and water-extractable (WEP)). Sediment removal does not reliably reduce P in the shallow marsh zone based on the variability within and between locations. Phosphorus should not be used in wetland assessments, although the shallow marsh zone typically had the most P of the three landscape positions.

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GENERAL INTRODUCTION

Wetlands can be found in many different regions throughout the United States. One of these regions, known as the Prairie Pothole Region (PPR), an area approximately 715,000 km^2, includes parts of Minnesota, Iowa, Montana, North and South Dakota as well as the Canadian provinces Alberta, Manitoba, and Saskatchewan (Euliss et al., 1999). However, the amount and quality of these wetlands have been decreasing which is likely due to multiple factors including the encroachment of agriculture, removal of land from Conservation Reserve Program (CRP) and Wetland Reserve Program (WRP) protection for economic purposes, and other environmental factors (Mitsch and Gosselink, 2007; Gleason et al., 2011). The loss of wetland quality and quantity can lead to increased amounts of pollution from agricultural run-off (sediments, nutrients, and pesticides) affecting other sensitive ecosystems. For instance, natural flow-through wetlands in agricultural regions help improve water quality by intercepting run-off from irrigated pastures and "reducing loads of total suspended sediments, nitrate, and *Escherichia coli* on average by 77, 60, and 68 percent, respectively" (Knox et.al, 2008). In addition, the tillage of wetland basins along with surrounding upland is considered to be the second most altering agricultural activity (drainage to increase production is the first) contributing to the degradation of wetland processes (Gleason et al., 2011).

Changes in wetland quality may also affect the composition of plant communities which may be reflected in other functions and services provided by the wetland. For example, clonal species such as cattails (*Typha*) typically contribute to waterfowl habitat degradation in wetlands by "choking off" other vegetation (Mitsch and Gosselink, 2007).

A difference in plant community structure would change the habitat and the wetland's suitability for other plant and animal populations. For example, in any given year, approximately 50 to 75 percent of waterfowl which originate in North America come from the PPR (Mitsch and Gosselink, 2007). Since waterfowl use wetlands for courtship, brood raising, fall migration, and as a water source in times of drought (Kirby et al., 2002), a change of waterfowl population and species as wetland function changes would be expected. Waterfowl hunting can also provide economic benefits to local communities. For example, during the 2011 waterfowl hunting season, waterfowl hunters spent roughly $17.5 million in rural areas of North Dakota (Taylor et al., 2013). Wetlands provide many other less noticeable functions and services as well.

Researchers, government officials, and land managers have shown interest in methods for determining the condition of a wetland to help with preservation and restoration efforts. To accomplish this, classification systems of wetlands specific to certain regions have been created. These systems are primarily based on characteristics of the plant community. However, there has been recent discussion that soil characteristics may be valuable to include in wetland classification systems as well (Rokosch et al., 2009).

Phosphorus (P) is a nutrient in the soil which is crucial to plant development. This nutrient plays an important role in plant processes such as photosynthesis, maturation, and nitrogen fixation (Brady and Weil, 2010). However, if too much plant-available P is introduced into a wetland (via run-off from agricultural activities), the plant community and wetland ecosystem may be affected as a result. In addition, eutrophication can occur in freshwater systems by additional P in a normally P-limited system (Brady and Weil, 2010).

Sediment removal is a restoration technique used to help improve a wetland's condition. The process initially appears to be successful with restoring plant communities (LaGrange et al., 2011; Smith, 2011). However, little information is available on the effectiveness of this technique in reducing concentrations of plant-available P or other changes it may affect in chemistry characteristics of wetland soils.

This thesis contains two separate manuscripts addressing P in wetland soils. The 'Literature Review' is a general review of literature relevant to past and current studies and issues related to P concentrations in soil, wetland assessments, and remediation techniques such as sediment removal. The 'Literature Review' is followed by 'Paper I' which contains a study on the effectiveness of sediment removal to reduce P in wetland soils. 'Paper II' follows with a study on the potential use of P distribution in wetland soils as a metric in wetland assessment methods. Both papers include a study- specific abstract, introduction, materials and methods, statistical analysis, results and discussions, and conclusions. 'General Conclusions' then discusses the relation of conclusions from both studies.

LITERATURE REVIEW

Wetlands and Agriculture

Wetland functions, such as habitat and maintenance of water quality, may be affected by the surrounding land use. In the PPR, wetlands are commonly surrounded by land used for agriculture or forage for cattle. Grazing of a wetland may cause disturbance which can increase plant diversity and provide other benefits to wetland habitats as long as the intensity of the grazing is controlled (Kirby et al., 2002). As a result, there is growing concern regarding the intensity of agricultural activities and their effects on wetlands and changes within the plant community. Less land which had been previously protected by contracts under Farm Bill programs is being renewed as conversion to other uses is being favored by economic incentives (Gleason et al., 2011). Agricultural practices which involve different intensities and frequencies of tillage show that herbicide levels and fertilizer use have a considerable effect on plant species composition in surrounding woodlots and hedgerows (Boutin and Jobin, 1998). A study by Knox et al., (2008) found that a channelized, degraded wetland in an agricultural landscape had significantly lower pollution load retention rates (except for soluble reactive P) than a reference wetland in the same setting. Adjacent land use activities may also impact nutrient enrichment and storage in plants and soil of temporary wetlands which may cause changes in their structures and functions (Gathumbi et al., 2005). One way to help manage and improve water quality of wetlands is to regulate inflow rates of run-off (Knox et al., 2008). By controlling run-off which may contain nutrients, sediments, and pathogens, negative effects which contribute to the degradation of areas that receive this inflow may be decreased (Knox et al., 2008).

Wetlands also provide many ecosystem functions in grazing areas. Some of these functions include providing grazers an area to cool themselves in warm weather, wildlife habitat, and production of high quality forage (Gathumbi et al., 2005). However, when native rangelands become pastures which are used more intensively, nutrient concentrations increase and production patterns of seasonal plants change within the surrounded wetlands (Gathumbi et al., 2005).

Phosphorus and Wetlands

The relationship between nutrients and plant communities in wetlands is a common topic in wetland research and nitrogen (N) and P have been known to greatly influence freshwater lake ecosystems (Moss et al., 1986). In some areas, wetland retention of P in natural and constructed wetlands is considered an important wetland function (Mitsch and Gosselink, 2007). For example, in the Florida Everglades, wetlands have been created to act as sinks for P originating from agricultural fields (Mitsch and Gosselink, 2007). However, some researchers have questioned the ability of wetland systems to serve as nutrient sinks (Gathumbi et al., 2005).

In wetland soils, P can be found in organic and inorganic forms and soluble or insoluble complexes (Mitsch and Gosselink, 2007). Also known as orthophosphates, the presence of the three inorganic forms of P ($H_2PO_4^-$, HPO_4^{2-}, and PO_4^{3-}) largely relies on pH and may form complexes with Al, Ca, and Fe (Mitsch and Gosselink, 2007; Stevenson and Cole, 1999). The water-soluble orthophosphate primary and secondary ions ($H_2PO_4^-$ and HPO_4^{2}) are the most bio-available form of P in soil solution for plant uptake (Stevenson and Cole, 1999). Estimated concentrations of bioavailable P can be determined using a range of methods which rely on regional soil characteristics (such as the Olsen, Bray, and Mehlich III methods) and extraction types (such as ion exchange resin, NaOH, and NH_4F) (Sharpley, 2009).

Phosphorus in agricultural wetlands is not viewed as a limiting factor for plants since it is relatively available and biochemically stable (Mitsch and Gosselink, 2007). However, there have been some concerns regarding the effect of excess P on freshwater systems including eutrophication and changes in plant communities (Moss et al., 1986; Gathumbi et al., 2005; Mitsch and Gosselink, 2007). Most of the P in fertilizers (up to 90%) is retained in the soil as forms which are insoluble or fixed instead of being used by crops (Stevenson and Cole, 1999). Wetlands that are surrounded by land which had been fertilized in the past have shown more P in aboveground plant tissues and shallow soil layers than wetlands surrounded by semi-native pasture (Gathumbi et al., 2005). Since the main cause of P loss from the majority of agricultural land is erosion (Stevenson and Cole, 1999), the concern for potential accumulation of P in wetlands is valid.

Other soil factors can influence the availability of P as well. The availability of P to plants in a wetland is influenced by factors including pH, salinity, and the hydrolysis of Al and Fe phosphates (Mitsch and Gosselink, 2007). Salinity is known to reduce phosphate availability and uptake in plants (Grattan and Grieve, 1999).

Measuring the Quality of a Wetland

Since a variety of environmental factors greatly influence plant composition of a wetland (USACOE, 2010), a single classification system to determine wetland quality would be difficult to develop. As a result, there have been multiple attempts to develop accurate assessments for wetlands (Lillie et al., 2002; Mack 2007; Stoddard et al., 2008). However, wetland assessments lose credibility when applied over large areas where variability in factors affecting wetlands can be large. For instance, it has been shown that northern wetland communities should not be considered homogeneous for climate change models (Bridgham et al., 1998). In an attempt for higher reliability and accuracy, assessments with multiple parameters have been developed and/or modified for specific regions and wetland types. These include the Ohio Rapid Assessment Method (ORAM) (Rokosch et al., 2009), the California Rapid Assessment Method for Wetlands (CRAM) (California Wetlands Monitoring Workgroup, 2013) and the Oregon Rapid Wetland Assessment Protocol (ORWAP) (Adamus et al., 2010). The PPR is no exception (DeKeyser et al., 2000; DeKeyser et al., 2003; Hargiss et al. 2008; Hargiss 2009; Stasica 2012). Still, other researchers question the usefulness of such divisions in assessments (Euliss and Mushet, 2001; Mita et al., 2007).

Government response to discourage further loss of wetlands started from passing Swampbuster, which was introduced in the 1985 Farm Bill, and Section 404 of the Clean Water Act. A need for defining a wetland and determining its quality also rose as a result. Some states and regions have developed classification systems catered toward wetlands that share at least one similar trait (Lille et al. 2002; Rokosch et al., 2009). One of these methods is the hydrogeomorphic (HGM) wetland classification system developed by Brinson (1993) and expanded by Smith et al., (1995) for determining how well a wetland is functioning instead of its condition. This system has been used for wetland management, designing mitigation projects, and establishing wetland restoration guidelines (Gilbert et al., 2006). The HGM uses three wetland characteristics (hydrodynamics, geomorphic setting, and water source) to group similarly functioning wetlands.

Many wetland assessments are heavily based on hydrologic- or plant-based parameters (metrics) while soil parameters have not been as well-developed. As a result, more researchers are recognizing the need for studies relating soils and wetlands to vegetation (Galatowitsch and van der Valk, 1996). Many factors within soil can influence wetland vegetation and could be potentially reflected in a wetland's overall condition score. For example, the relationship between salinity and P uptake and accumulation in plants has shown mixed results which may be contributed to other simultaneously occurring nutrient interactions (Grattan and Grieve, 1999). Researchers have also encountered some difficulty in establishing a clear gradient in conditions with soil parameters (Freeland et al., 2009; Rokosch et al., 2009). Few studies have been done regarding the inclusion of bioavailable P as a soil parameter in wetland assessments (Rokosch et al., 2009). What information is available on P in wetlands and its relationship with wetland vegetation has shown conflicting results (Craft et al., 1995; Johnson and Rejmánková, 2005).

Disturbed and Undisturbed Wetlands

A common method used to determine the success of a wetland restoration or the accuracy of a wetland assessment method is to compare disturbed and/or restored sites to relatively undisturbed reference wetlands. When studying the correlation between wetland criteria (hydric soils, wetland hydrology, and hydrophytic vegetation) between disturbed and relatively undisturbed wetland sites, Janisch and Molstad (2004) found that undisturbed areas were significantly more likely to meet all three requirements than disturbed sites. Of the wetlands used in the study, 42% of the data points from undisturbed wetlands met all three criteria while only 22% of disturbed wetland data points met the same criteria (Janisch and Molstad, 2004). Another study involving the comparison of 10 natural and 10 restored wetlands in the PPR over three years after re-flooding resulted in sparse stands of emergent and wet meadow species in restored wetlands while in similar natural wetlands large stands of emergent species were predominant (Galatowitsch and van der Valk, 1996).

Determining the causes, effects and amounts of disturbance to the wetland community can be difficult. Similarity of vegetation between natural and restored wetlands has been shown to be reliant on the likeliness of wetland species' propagules spreading to the reflooded wetlands and the similarity of environmental conditions (Galatowitsch and van der Valk, 1996). To help further refine wetland criteria in relation to disturbed and less disturbed sites, more research is needed (Janisch and Molstad, 2004).

Cattails and Excess P

Land use practices strongly influence within-stand nutrient cycling as well as soil and plant nutrient content (Gathumbi et al., 2005). In agricultural areas where fertilizer is applied, this may affect wetland communities which are surrounded by cropland. The *Typha* species, more well-known as cattails, are a common sight in PPR wetlands usually as monotypic stands of the species. *Typha* have been shown to have 2-3 times higher net accumulations of P in their shoots compared to other species (Newman et al., 1996). *Typha* invasion of wetland communities may alter nutrient cycling of P, Nitrogen (N), and Carbon (C) in other plants (Meyers, 2013). Communities with *Typha latifolia* have expressed significantly higher available P in the A horizon (0.03 g kg^{-1}) than communities which were not dominated by *T. latifolia* (0.01 g kg^{-1}) (Drohan et al., 2006). Compared to some wetland species, *Typha* may have an advantage in disturbed wetlands. In a plant mixture of *Typha*, *Cladium*, and *Eleocharis*, *Typha* was the only species to respond positively to increased water depth and nutrients (Newman et al., 1996). The temporary increase in available resources after a fire is competitively used by *Typha* which tend to also temporarily increase in density after such events (Ponzio et al., 2004).

Restoring wetlands to avoid and discourage monotypic stands of *Typha* is challenging. Hydrologic restoration and decreasing surface water nutrients should be considered when making management decisions to control the spread of *Typha* (Newman et al., 1996). In some cases, disturbances such as grazing can be effective in altering stands of cattails to promote more diverse plant communities (Kirby et al., 2002). However, others suggest that cattle grazing in close proximity to wetlands likely increases nutrient loads of N and P into wetlands, partially contributing to *T. latifolia* encroachment (Drohan et al., 2006). Fire has also been used as a tool to counter *Typha* expansion. When used to study density changes in *Typha domingensis*, even though hydroperiods and soil nutrient levels were in range of supporting *Typha* expansion, no lasting changes were observed in *Typha* density (Ponzio et al., 2004).

Restoration and Sediment Removal

Sedimentation in wetlands can be detrimental to the ecosystem and plant community. Sedimentation is a natural process which occurs in wetlands; however, this process is commonly accelerated in wetlands surrounded by agriculture since soil is more easily eroded from the surrounding upland area (LaGrange et al., 2011). In some cases, sediment burial depths as small as 0.25 cm have been shown to decrease hydrophyte emergence, germination, and species richness in wetlands (Jurik et al., 1994; Wang et al., 1994). Wetland surrounded by cropland have shown 2.7 to 6 times greater sedimentation rates compared to wetlands surrounded by native prairie (Preston et al., 2013). Leaving this sediment in place while creating or restoring a wetland can be problematic. Clay and P amounts in wetlands surrounded by cropland are higher than in wetlands surrounded by native prairie (Preston et al., 2013). If surface soil is not removed when creating a wetland in arable land, it has the potential of becoming a source of P instead of a sink (Liikanen et al., 2004). This could potentially have detrimental effects on the surrounding wetland ecosystem.

There have been multiple studies focused on the relationship between soil depth and nutrient concentrations in wetland sediments (Gathumbi et al., 2005; Liikanen et al., 2004). In a study by Gathumbi et al., (2005), soil nutrient concentrations in wetlands surrounded by pastures which had been fertilized in the past and semi-native pastures both decreased with depth. However, the method and depth of soil removal to encourage ideal plant communities for restoration on a regional basis have not been well established.

The use of sediment removal to improve wetland condition, lake condition and to promote plant diversity has had mixed results. It has been found that despite planting and seeding a wetland (species included cattail (*Typha latifolia* L.), common water plantain (*Alisma plantago-auquatica* L.), Meadowsweet [*Filipendula ulmaria* (L.) Maxim.], yellow flag (*Iris pseudacorus* L.), and compact rush (*Juncus conglomerates* L.)) which was created in arable land with the surface soil removed, cattail still became the dominant species within 3 years (Liikanen et al.., 2004). In a study by Moss et al., (1986), a non-isolated freshwater lake area that was under eutrophication recovered after sediment was removed while a lake with similar conditions was isolated and still periodically experienced eutrophication within the same time period. However, positive results from sediment removal in wetlands include an increase of waterfowl use and the development of plant communities similar to undisturbed wetlands (LaGrange et al., 2011; Smith, 2011). More research is needed to determine long-term effects of sediment removal and wetland restoration.

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PAPER 1. THE USE OF SEDIMENT REMOVAL TO REDUCE PHOSPHORUS LEVELS IN WETLAND SOILS

ABSTRACT

Sediment removal from wetlands may help control or avoid the growth of monotypic stands of hybrid cattail (*Typha x glauca*) by removing nutrients, including P, from the shallow marsh zone. In this study, sediment at a depth of 10 to 51 cm was removed from the shallow marsh zone of 18 wetlands in the Prairie Pothole Region of North Dakota four to nine years prior to collecting soil samples. Samples were collected from two depths (0-15 and 15-30 cm) from 38 wetlands that included excavated (sediment removed), converted cropland, and reference type wetlands for comparison. Samples from three clusters each consisting of all three wetland types were analyzed for pH, electrical conductivity (EC), and P (Olsen and water-extractable (WEP)). Olsen and WEP concentrations ranged from 6.8 to 47.5 and 0.01 to 8.1 mg/kg, respectively. Results in plant-available P (as well as EC and pH) varied unexpectedly within and between clusters, therefore suggesting that removing sediment is not necessarily a reliable way to reduce P in the shallow marsh zone.

INTRODUCTION

The PPR is regarded as one of the most important wetland regions on earth since this area consists of many shallow lakes and wetlands along with warm summers which make an ideal habitat for waterfowl (Mitsch and Gosselink, 2007). Wetlands in this region have been declining in number (Dahl, 2000) and there is an increasing concern regarding the quality of the wetlands which remain. Some believe this decrease is largely due to the prevalence of agriculture in this region (Mitsch and Gosselink, 2007). There is a higher risk of the loss and degradation of smaller wetlands (such as seasonal wetlands) in agricultural fields compared to other wetland types since they have the lowest recovery rate along with the highest impact from agricultural activity (Bartzen et al., 2010). Excess nutrients in run-off, such as P, from cultivated fields enter wetlands as agricultural pollutants/contaminants with sediment or in surface run-off (Neely and Baker, 1989). With an increase of nutrients, plant community composition can change. In the Florida Everglades, for example, the spread of *Typha domingensis* into conservation areas is believed to be a result of an increase in agricultural run-off (Mitsch and Gosselink, 2007). This stress can decrease the quality of a wetland by affecting hydrology and the plant community. Sediment, even in small amounts, will impact the functions of small depressional wetlands (Richardson et al., 2001).

In North Dakota, the hybrid cattail (*Typha x glauca*) is an invasive species which forms monotypic stands due to the species' inherited traits and clonal nature along with its ability to rapidly uptake nutrients (Woo and Zelder, 2002). Monotypic stands of clonal species such as *Typha* degrade the quality of habitat for waterfowl and inhibit other types of vegetation (Mitsch and Gosselink, 2007). Changes in wetland ecosystem functions and structure within the first ten years after restoration may be greatest in shallow soil depths (0-10 cm) (Meyer et al., 2008). This would support sediment removal as a viable option for restoration of low-quality wetlands. It is thought that by removing nutrient-rich soil which has accumulated in “cattail-choked” wetlands, native vegetation would also be able to reestablish. Past studies of excavating wetlands have yielded positive results. In Nebraska, removal of sediment from “cattail-choked” wetlands has resulted in an increase of waterfowl use and improvement of multiple wetland functions (LaGrange et al., 2011). In Florida, results from a study conducted by Dalrymple et al., (2003) showed completely removing soils from wetland sites as a promising solution to prevent recolonization of *Schinus terebinthifolius* monocultures. Since the study, this technique was to be applied to all wetlands in the Hole-In-The-Donut area of the Everglades National Park in Florida as a long-term restoration program. However, data regarding the success of excavated PPR wetlands in North Dakota is limited. A study by Smith (2011) on vegetation present at the same wetlands used in the following study found that excavated wetland vegetation was becoming increasingly similar to natural wetlands while unmanaged converted cropland wetlands were “cattail-choked”.

Information comparing soil levels of plant-available P in addition to vegetation data between excavated, natural, and converted cropland wetlands which are left unmanaged would be useful in determining the effectiveness on sediment removal’s reduction of nutrients. The objective of this study was to determine if the removal of sediment in the shallow marsh zone of seasonal prairie pothole wetlands in North Dakota decreased the amount of plant-available P.

MATERIALS AND METHODS

Sample collection for this study took place summer of 2012 and analysis was completed by spring of 2013. Soil samples were collected from three different wetland types: 1) Excavated, where 10 to 51 cm of sediment was removed from within the basin of the wetland, 2) Converted Cropland, which were unexcavated wetlands recovering from past tillage practices, and 3) Reference wetlands, which were in a natural state having not been greatly disturbed by humans and occurring in native prairie. All three wetland types were considered recharge wetlands located in the PPR of North Dakota, USA (Figure 1). Recharge wetlands are wetlands which have an outflow of groundwater since the wetland's ground or surface water is hydrologically higher than the surrounding water table (Mitsch and Gosselink, 2007). Sediment removal was performed on the Excavated wetlands which generally had greater than 25 cm of sediment above the A horizon (C. Dixon, personal communication, 2013).The surrounding landscape, including crop and rangeland, was the likely source of this sediment. After sediment removal was accomplished using excavating equipment and after which the upland area was seeded to grass and some wet meadow zones were planted with plugs of prairie cordgrass or *Carex athrodes* (C. Dixion, personal communication, 2013). Otherwise, no further actions were taken. A total of 18 Excavated, 11 Converted Cropland, and 9 Reference sites were sampled. Specific data on sample number and soils series present can be found in Table 1. Since previous P data does not exist for these sites, Converted Cropland sites were included to represent wetland conditions prior to sediment removal. Sites from all three wetland types were arranged in clusters, wetlands from a certain geographical location, as determined by

Smith (2011). All sites were in North Dakota, Cluster A wetlands in Benson County were located 6.5 km southwest of Leeds, and the range and date of sediment removal was between 20 to 30 cm and 2007, respectively. Cluster B wetlands in Towner County were located 12.6 km north of Cando and sediment was removed in 2008 and removal ranged from 25 to 51 cm. Between 10 to 41 cm of sediment was removed in 2003 from Cluster C wetlands in Wells and Eddy Counties which were located 31.8 km southwest and 34.9 km east of New Rockford. Cluster C reference wetlands in Eddy County were located on Camp Grafton South state land. At each site, three soil samples were randomly collected within a 10 m transect in the shallow marsh zone from two depths (0 to 15 cm and 15 to 30 cm). Samples were collected using a tiling-spade shovel or Dutch auger depending on the site conditions, stored in plastic bags, transported in iced coolers, and stored field-moist at 4 ^{o}C. Prior to P extraction, all samples were homogenized by hand and any rocks, large roots, and visible macrofauna were removed.

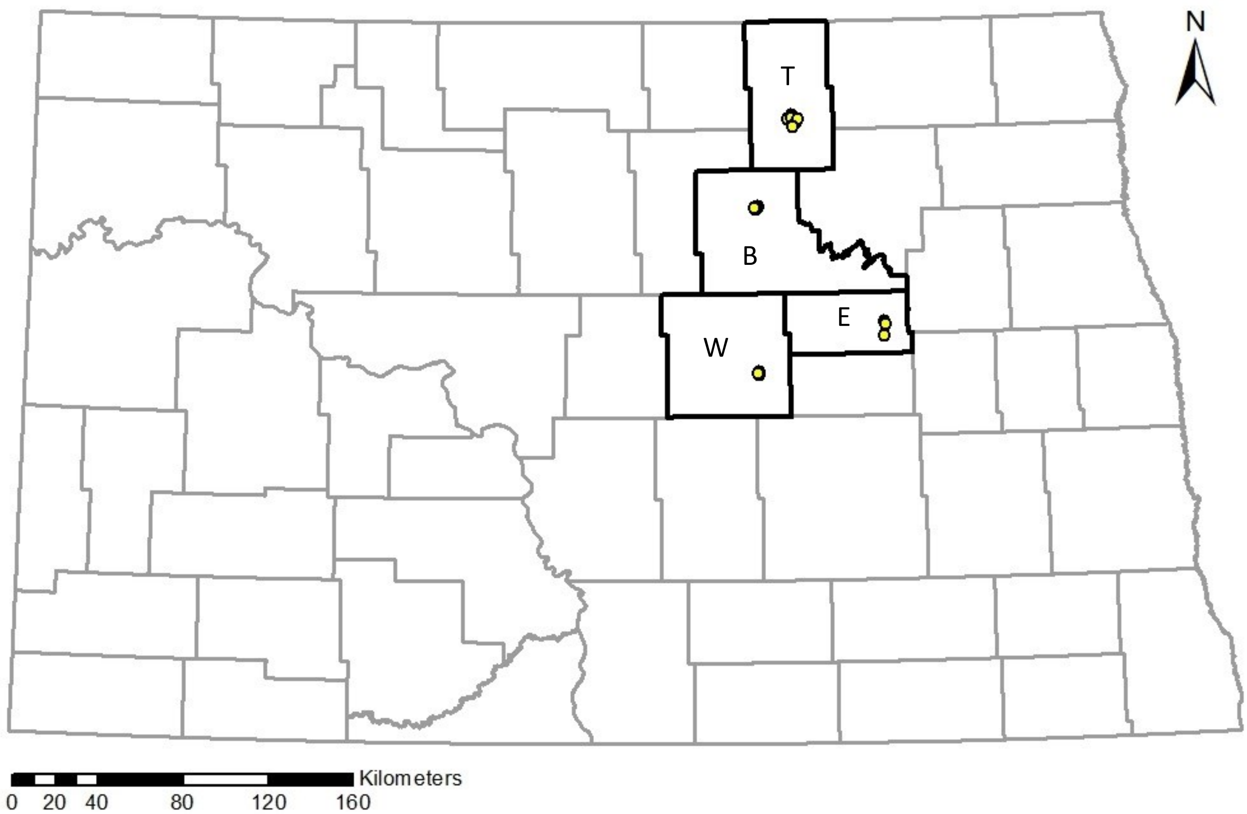

Figure 1. Sites from all three wetland types were arranged in cluster samples as determined by Smith (2011). Outlined counties include: Benson (B), Cluster A; Towner (T), Cluster B; and Wells (W) and Eddy (E), Cluster C.

Table 1. Number of wetland types sampled and soils present for each county cluster.

Cluster†	Wetland Type‡	N (sites)	Soils Present
A	Converted Cropland	2	Vallers loam, saline, 0 to 1 percent slopes Barnes-Svea loams 3 to 6 percent slopes
	Reference	3	Hamerly-Wyard loams 0 to 3 percent slopes Vallers loam, saline, 0 to 1 percent slopes
	Excavated	3	Vallers loam, saline, 0 to 1 percent slopes Svea-Cresbard loams 0 to 3 percent slopes Barnes-Cresbard loams 3 to 6 percent slopes
B	Converted Cropland	4	Vallers, saline-Parnell complex, 0 to 1 percent slopes Vallers-Hamerly loams, saline, 0 to 3 percent slopes
	Reference	3	Lowe-Fluvaquents, channeled complex, 0 to 2 percent slopes , frequently flooded Hamerly-Tonka-Parnell complex, 0 to 3 percent slopes
	Excavated	5	Hamerly-Tonka-Parnell complex, 0 to 3 percent slopes Barnes-Buse loams 3 to 6 percent slopes Vallers, saline-Parnell complex, 0 to 1 percent slopes
C	Converted Cropland	5	Heimdal-Emrick loams 0 to 3 percent slopes Fram-Wyard loams 0 to 3 percent slopes
	Reference	3	Southam silty clay loam 0 to 1percent slopes Parnell silty clay loam 0 to 1 percent slopes Heimdal-Esmond-Sisseton loams 9 to 15 percent slopes
	Excavated	10	Heimdal-Emrick loams 0 to 3 percent slopes Fram-Wyard loams 0 to 3 percent slopes

†Cluster A wetlands were located in Benson County, Cluster B wetlands were located in Towner County, and Cluster C wetlands were located in Wells and Eddy Counties.
‡ Wetland types include: Converted Croplands, wetlands recovering from past tillage practices; Reference, natural or native prairie wetlands; and Excavated, 10-51 cm of sediment removed from the shallow marsh zone.

The Olsen P extraction used in this study was a modification of the procedure recommended for the North Central Region of the United States of America (Frank et al., 1998). Two grams of the homogenized field-moist samples were weighed into 50 mL plastic centrifuge tubes (06-443-18, Fisher Scientific, Pittsburgh, PA) and 40 mL of sodium bicarbonate (0.5 M $NaHCO_3$, pH 8.5) extracting solution was added. Samples were shaken for 30 min at 280 oscillations/min on a reciprocal shaker followed by centrifugation for 20 min at a RCF of 804 x *g*. The supernatant was then filtered through Whatman No. 2 paper into 10 mL plastic vials. Olsen P extracts were analyzed the same day using a flow-injection analyzer (FIALab 2500, Bellevue, WA) at a wavelength of 880 nm. The FIA analysis of Olsen P extracts included the use of a 10 cm flowcell (for low P concentrations) and an Edmund Optics TS Longpass filter to avoid saturation and bleed over which can occur at 880 nm (FIALab® Instruments). For increased sensitivity, the mixed solution was passed through a plastic tube coil submerged in a water bath (BM100, Yamato Scientific America Inc., Santa Clara, CA) set for 45 ^{o}C. All Olsen P wet soil values were converted to oven-dry soil values for final concentration determination.

Water-extractable P (WEP) was determined using a modified procedure of Self-Davis et al., (2009). Here, 4 g of the homogenized samples were weighed into a 50 mL plastic centrifuge tube (Fisher Scientific, Pittsburgh, PA) and 40 mL of deionized water was added, shaken for 60 min at 280 oscillations/min using a reciprocal shaker, centrifuged for 90 min at a RCF of 647 x *g*, and supernatant filtered through a 47 mm, 0.45 µm filter (097191B, Fisher Scientific, Pittsburgh, PA) using a 47 mm Telfon filter apparatus (1-47, 47-6, and 47, Savillex Corp., Minnetonka, MN). Filtered extracts were acidified by adding two drops of hydrochloric acid (pH 2.0) to deter phosphate compound precipitation, transferred to Wheaton plastic scintillation vials(16300-219, VWR International LLC., Batavia, IL), and frozen at -10 ^{o}C until analysis. Analysis was done using a flow-injection analyzer (FIALab 2500) that was configured as above but the wavelength was set at 860 nm and the water bath at 40 ^{o}C. All WEP wet soil values were converted to oven-dry soil values for final concentration determination.

Electrical conductivity (EC) was analyzed for each sample followed by pH. Both were performed on 10 g of air-dried soil ground to pass through a 1 mm sieve. Electrical conductivity was determined using the 1:1 soil -to-deionized water method described by Whitney (1998) with an EC probe (SenseION378, Swedesboro, NJ) which was then followed by determination of pH as described by Watson and Brown (1998) and a pH electrode (Accumet AB15, Pittsburgh, PA). Due to high organic contents, some individual samples from both depths from Converted Cropland sites in clusters A and B, and for the 0 to 15 cm Reference samples from cluster A, were analyzed with a 1:2 soil to water ratio instead of 1:1 due to a lack of measureable solution. Conversions to a 1:1 ratio for EC were applied to these samples (Al-Mustafa and Al-Omaran, 1990).

STATISTICAL ANALYSIS

Statistics were completed using Microsoft Excel and JMP ver. 8 (ver. 8.0 SAS Institute Inc., Cary, North Carolina). Analysis of variance (ANOVA) and Tukey-Kramer HSD were used to test among the different wetland types, soil depths, and clusters using a $p \leq 0.05$ significance level.

RESULTS AND DISCUSSIONS

Phosphorus

Across all wetland types and soil depths, average Olsen P and WEP values ranged from 6.8 to 47.5 and 0.01 to 8.1 mg/kg, respectively (Table 2). As expected, Olsen P values were greater than WEP across all samples.

Table 2. Average soil Olsen P and water-extractable P (WEP) concentrations for each wetland type and cluster for the shallow marsh zone.

		Sampling Depth (cm)			
		0-15		15-30	
		Extraction Method		Extraction Method	
Cluster	Wetland Type†	Olsen P	WEP	Olsen P	WEP
		mg/kg			
A	Converted Cropland	16.0 (1.76)‡b§	0.22 (0.25)b	16.7 (3.21)a	0.73 (0.49)a
	Reference	13.0 (8.21)b	0.44 (0.37)b	16.7 (10.8)a	1.79 (1.28)a
	Excavated	47.5 (12.5)a	6.36 (1.49)a	36.1 (27.7)a	8.10 (4.52)a
B	Converted Cropland	6.80 (3.90)a	0.65 (0.59)a	5.78 (3.22)a	0.78 (0.85)a
	Reference	6.39 (2.67)a	0.01 (0.01)a	3.83 (1.67)a	0.89 (0.57)a
	Excavated	10.6 (12.9)a	1.12 (1.01)a	9.87 (15.7)a	2.23 (2.08)a
C	Converted Cropland	12.0 (7.33)b	2.66 (2.25)a	12.7 (8.8)b	3.89 (2.91)ab
	Reference	39.3 (10.5)a	4.59 (3.47)a	42.9 (15.1)a	5.77 (3.52)a
	Excavated	12.8 (13.0)b	1.60 (1.66)a	6.97 (6.33)b	1.21 (1.44)b

†Wetland types include: Converted Croplands, wetlands recovering from past tillage practices; Reference, natural or native prairie wetlands; and Excavated, 10-51 cm of sediment removed from the shallow marsh zone.
‡Numbers in parenthesis indicate standard deviation.
§ Different letters within extraction method depth separated by cluster indicate significant difference at $p \leq 0.05$.

Sediment removal appears to be ineffective in reducing the amount of available Olsen or water-extractable P from soil within the shallow marsh zone of a prairie pothole wetland. In the case of Cluster A, the P for Excavated wetlands was significantly greater than Reference or Converted Cropland wetlands. The significantly greatest average

Olsen P concentration (47.5 mg/kg; 0-15 cm) was observed in the Cluster A treatment wetlands while the WEP concentration (6.36 mg/kg) was also significantly greatest at this depth (Table 2). Since the Converted Cropland wetlands are included in this study to represent the condition of Excavated wetlands prior to sediment removal, the Cluster A results would suggest that sediment removal would not only be ineffective, but it would potentially create a more P-rich scenario than if the wetlands were left undisturbed. However, no significant difference in P for Cluster A at a depth of 15-30 cm might suggest that if more sediment had been removed, the amount of P at a depth of 0-15 cm may not have been significantly greater. There is also a possibility that the plant communities of Excavated wetlands, which had been disturbed during sediment removal, had not recovered completely, leaving more P in the soil than plant communities of Reference and Converted Cropland wetlands, which would have more P tied up in established plant communities.

Clusters B and C had no significant difference in P between Excavated and Converted Croplands. Initially, the amount of soil removed or the date of its removal may be suspected as the cause for the difference observed in Cluster A excavated wetlands. However, Cluster A Excavated wetlands fit between Clusters B and C with amount of sediment removed (20-30, 25-51, and 10-41 cm, respectively) and year excavation took place (2007, 2008, and 2003, respectively). This shows that the amount of P may not be influenced by sediment removed and the recovery time of the wetland. Meyer et al., (2008) suggests that the shallow depths of a restored wetland experiences the most changes regarding the functions and structure of its ecosystem, along with variable recovery rates, during the first ten years after restoration. Since the Excavated wetlands have been restored within ten years of sampling, this could be a likely contributor to the variability in results.

No differences were observed for Cluster B wetlands for either extraction or depth. Cluster B reference and converted cropland mean values for Olsen P (6.39 and 6.8 mg/kg, respectively) fall between Olsen P values found by Freeland et al., (1999) for a semi-permanent wetland surrounded by grassland and a semi-permanent wetland surrounded on three sides by grassland (4.8 and 9.0 respectively) in the shallow marsh zone of prairie pothole wetlands in the Cottonwood Lake Research Area, North Dakota. Based on this comparison, Cluster B likely represents the ideal wetland parameters for this study.

In Cluster C, both Olsen P depths and the 15-30 cm WEP extractions from the Reference wetlands were significantly greater than at least one of the other wetland types, by as much as six times greater (Table 2). Reference wetlands from Cluster C were surprisingly higher in Olsen P than either converted cropland or sediment removed wetlands. It is possible that cattle grazing of these sites may have contributed to excess P, but reference sites included in Clusters A and B were also subject to grazing. In a study by Knox et al., (2008), no significance was found in total P and soluble reactive P loads in runoff from pastures with active cattle grazing during a runoff event compared to their absence, further supporting grazing as an unlikely cause in P differences between clusters. Another possibility would be that the parent material in the reference area naturally has the mineral apatite which would provide higher amounts of P in the soil. Since this area consists of glacial outwash, end moraines, and ground moraines (Bryce et al., 1998), higher amounts of P could have been brought in, although high concentrations of P are not typical for soils in North Dakota. No NRCS soil characterization data was available for the soil series in question located in Eddy County, although data from one of the series (Parnell) located in Ottertail, MN did show trace amounts of P-bearing apatite at a depth of 145-180 cm (Soil Survey Staff pedon 93P0763).

Electrical Conductivity and pH

Average EC and pH values across all wetland types ranged from 0.32 to 4.9 dS/m and 5.2 to 7.5, respectively (Table 3). With the exception of Cluster C Reference wetlands, all clusters and wetland types showed a decrease in EC with an increase in depth. There were no differences within Cluster B at either depth and Cluster A at the 15-30 cm depth. Among all of the clusters, the average EC was lowest in both depths for Reference sites in Cluster C (0.32 dS/m). At the 0-15 cm depth, significant differences were observed within Cluster A between Converted Cropland and Excavated wetlands and the Cluster C Reference wetlands were lower than the other two wetland types. Also, the 15-30 cm depth in the Cluster C excavated wetlands was significantly greater than the Reference. Average pH was lowest in both depths for Reference sites in cluster C (pH of 5.2 and 5.3 for 0-15 and 15-30 cm, respectively), while no significant differences were noted for either depth in Clusters A and B.

Table 3. Average soil electrical conductivity (EC) and pH for each wetland type and cluster for the shallow marsh zone.

		Sampling Depth (cm)			
		0-15		15-30	
Cluster†	Wetland Type‡	EC	pH	EC	pH
		----dS/m----		-----dS/m----	
A	Converted Cropland	2.6 (0.96)§a*	7.2 (0.2)a	1.6 (0.84)a	7.1 (0.1)a
	Reference	0.95 (0.63)ab	6.2 (0.8)a	0.45 (0.19)a	6.5 (0.6)a
	Excavated	0.62 (0.37)b	6.4 (0.4)a	0.54 (0.34)a	6.7 (0.5)a
B	Converted Cropland	4.9 (4.8)a	7.0 (0.6)a	3.0 (1.9)a	7.1 (0.3)a
	Reference	4.4 (2.5)a	6.9 (0.2)a	1.5 (0.61)a	7.5 (0.1)a
	Excavated	2.1 (0.49)a	7.4 (0.1)a	1.7 (0.33)a	7.5 (0.2)a
C	Converted Cropland	1.2 (0.28)a	7.2 (0.4)a	0.83 (0.14)ab	7.4 (0.3)a
	Reference	0.32 (0.10)b	5.2 (0.2)b	0.32 (0.06)b	5.3 (0.2)b
	Excavated	1.3 (0.52)a	7.1 (0.5)a	0.94 (0.45)a	7.2 (0.4)a

† Significance determined within each column within each cluster.
‡Wetland types include: Converted Croplands, wetlands recovering from past tillage practices; Reference, natural or native prairie wetlands; and Excavated, 10-51 cm of sediment removed from the shallow marsh zone.
§Numbers in parenthesis indicate standard deviation.
*Different letters within depth for EC or pH test separated by cluster indicate significant difference at p≤ 0.05.

EC and pH values, just as with P concentrations, draw interest to the peculiarity of Cluster C Reference wetlands. The Reference soil pH was strongly acid (5.1≤pH≤5.5) and the Converted Cropland and Excavated wetlands were both neutral (6.6≤pH≤7.3), all three wetland types fell within the pH range of the best plant availability for P in the soil (Schoeneberger et al., 2012). Cluster C soils were all non-saline (dS/m≤1.4) as well (Soil Survey Staff, 1993). These non-saline, strongly acid Reference wetlands may actually be flow-through or discharge wetlands rather than the recharge wetlands which were the target of this study. If a wetland experiences changes in hydrology, the water chemistry may change as well. For example, a recharge wetland which receives low amounts of dissolved solids via precipitation and runoff may receive more dissolved solids should the source of water supply shift to groundwater characteristic of discharge wetlands (Richardson et al., 2001). This change in hydrology could have contributed to higher P and pH values. Another possible explanation was that military exercises in the area may have been using products containing P which would contribute to the overall higher value in P and may also account for the low pH, but base officials verified that these products have not been used in or near the location of the Reference wetlands. In a study of restored and natural wetlands in the Platte River Valley of Nebraska, restored wetlands which had been contoured and reseeded with local natural wetland native species had pH averages of 7.4 to 7.7, while averages for comparable natural wetlands ranged from 5.7 to 5.8 (Meyer et al., 2008). This similarity suggests that lower pH values for natural (or reference) wetlands compared to restored (or excavated) wetlands might be something to be expected.

CONCLUSION

The unexpected variation in plant-available P within and between clusters suggests that sediment removal may not be a reliable way to reduce P in the shallow marsh zone of prairie pothole wetlands. Initially, the expected outcome would have resulted in the Converted Cropland wetlands having significantly higher P than Excavated and Reference wetlands within each cluster. This would have indicated that sediment removal had decreased P in the shallow marsh zone of the Excavated wetlands to P concentrations similar to nearby Reference wetlands. However, Converted Cropland wetlands resulted in (some cases significantly) less P than Reference and/or Excavated wetlands for both extraction methods and depths within each cluster. Phosphorus values using the Olsen extraction method were higher than WEP extraction method values which was not surprising since the Olsen method uses sodium bicarbonate extracting solution to allow an increase in calcium phosphate solubility (Kuo, 1996).

All clusters and wetland types showed a decrease in EC with an increase in depth and no significant difference in pH except for Cluster C Reference wetlands. These wetlands could have flow-through instead of recharge hydrology which may contribute to higher amounts of P and different EC and pH values in comparison to Reference wetlands from Clusters A and B. Overland flow may have also contributed to higher P. Multiple factors may have contributed to the unexpected results and variance observed in this study. The depth of excavation should be carefully considered based on site characteristics since this may have contributed to the higher amount of P in Excavated wetlands in Cluster A. The time of recovery for plant communities after excavation may influence the amount of P in wetland soils. Further analysis of hydraulic connections regarding wetland proximity to agricultural areas would also be useful in determining if higher P in wetlands is due to spring runoff from these areas. Other factors including light grazing, products used during military exercises, and naturally occurring apatite probably had a minimal influence, if any, in the results of this study.

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PAPER II. DISTRIBUTION OF PLANT-AVAILABLE PHOSPHORUS IN WETLAND SOILS AND ITS POTENTIAL USE AS A METRIC IN WETLAND ASSESSMENT METHODS

ABSTRACT

Wetland conditions have historically been assessed using plant community and surficial characteristics while soil nutrients are seldom used. As part of the Environmental Protection Agency's (EPA) 2011 National Wetland Condition Assessment (NWCA), 53 wetlands (temporary, seasonal, semi-permanent, or permanent wetlands) were randomly selected across North Dakota for this study. Samples were collected from a depth of 0-15 and 15-30 cm from the shallow marsh, wet meadow, and upland areas and analyzed for pH, electrical conductivity (EC), and extractable P (Olsen and water-extractable). When arranged by condition categories pre-determined by the IPCI and NDRAM, a gradient showing significant difference in extractable P, EC, and pH between each category was not present at either depth. However, some categories were significantly different than others. High variability in extractable P was encountered in some categories. While there was not a significant gradient in extractable P based on landscape position, the shallow marsh had the greatest Olsen P levels (16.1 and 9.46 mg/kg, respectively) and EC. A comparison of Olsen P results and samples taken for the NWCA during this study were significantly different, likely due to a variance in sampling methods and extraction modifications. Overall, these results suggest that extractable P in soil (along with soil EC and pH) may not be reliable metrics for the IPCI and NDRAM.

INTRODUCTION

Wetlands in the Prairie Pothole Region (PPR) are decreasing in quantity and quality (Mitsch and Gosselink, 2007). To determine the quality or condition of a wetland and its disturbance level, researchers have been developing classification systems based on multiple wetland characteristics. Since multiple factors such as climate, elevation, topography, and soils greatly influence plant composition of a wetland (USACOE, 2010), a single classification system to determine wetland quality would be inaccurate and unreliable. An example of this would be the classification system developed by Stewart and Kantrud (1971) for wetlands in the northern Great Plains. Although this system addresses the influence of water depth and salinity on wetland vegetation and species composition, there are many instances which portray it as an archaic and inappropriate system (DeKeyser 2000).

Other methods used in the PPR of North Dakota include the Index of Plant Community Integrity (IPCI) and the North Dakota Rapid Assessment Method (NDRAM). The IPCI is a vegetative version of Karr's (1981) Index of Biological Integrity (IBI) developed by DeKeyser et al., (2003). This method is based on multiple community attributes and consists of five condition categories for seasonal wetlands (Very Poor, Poor, Fair, Good, Very Good) and three for semi-permanent and temporary wetlands (Poor, Fair, Good) (Hargiss et al., 2008). The NDRAM was developed by Hargiss (2009) as a rapid assessment to determine depressional wetland condition in the PPR. Metrics for this method use Best Professional Judgment to assess hydric soils, hydrophytic vegetation, and hydrology on-site (Stasica, 2012).

While the IPCI and NDRAM primarily rely on vegetative characteristics, the analysis of soil nutrients such as P has been generally excluded. Mixed results have shown the amount of P and other available nutrients in the soil may influence the species composition of a wetland. In a study of N and P additions to three wetland plant communities in the Florida Everglades, no significant difference was noticed in macrophyte species diversity by Craft et al., (1995). However, Johnson and Rejmánková (2005) found a negative correlation between soil P and species richness of common plants to unimpacted marshes in Belize. This could indicate that plant-available P may be possibly contributing more to the determination of wetland condition than originally thought.

Due to the conflicting thoughts on how P impacts wetlands, determining if the amount or location of plant-available P is reflected in the plant community thorough condition scores could lead to the inclusion/exclusion of other soil nutrients as metrics for classification systems. The objectives of this study were to determine if: 1) the amount of plant-available P could be used as a reliable indicator of wetland quality with the IPCI and NDRAM; 2) a gradient in elevation of plant-available P amounts within a PPR wetland existed; and 3) if a difference in sampling and extraction methods of Olsen P significantly altered results and its potential use as a IPCI or NDRAM indicator.

MATERIALS AND METHODS

As part of the Environmental Protection Agency's (EPA) 2011 National Wetland Condition Assessment, 53 wetlands across North Dakota were used for this study (Figure 2). Wetlands were randomly selected by the EPA using the United States Fish and Wildlife Service (USFWS) Status and Trends plot database. The wetlands selected were either temporary (Class II), seasonal (Class III), semi-permanent (Class IV), or permanent (Class V) according to the classification system by Stewart and Kantrud (1971). A wetland's class is determined by which vegetative zone occupies at least five percent of the total wetland area and which also occurs in the central/deepest part of the wetland (Stewart and Kantrud, 1971). While the previously listed Classes have different vegetative zones, this study focused on the shallow marsh zone, wet meadow zone, and upland position. During the emergent phase of the shallow marsh zone, the presence of grass and grass-like species will be taller than plants located in the wet meadow zone. Plant species common in the wet meadow zone during the emergent phase include sedges, rushes, and fine-textured grasses of relatively small height (Steward and Kantrud, 1971). The upland position for this study was determined from the wet meadow as being the nearest position having a 1 m increase change within 50 m. By classification, all wetlands selected had at least a low-prairie and wet meadow zone. Also by classification, a seasonal wetland includes a shallow marsh zone, a semi-permanent wetland includes a shallow marsh and deep marsh zone, and a permanent wetland includes all of the previous zones plus an open water zone and possible fen zones (Stewart and Kantrud, 1971). In this study, wetland type will refer to wetland classes as defined by Stewart and Kantrud (1971) and vegetative zones will be collectively referred to as landscape positions.

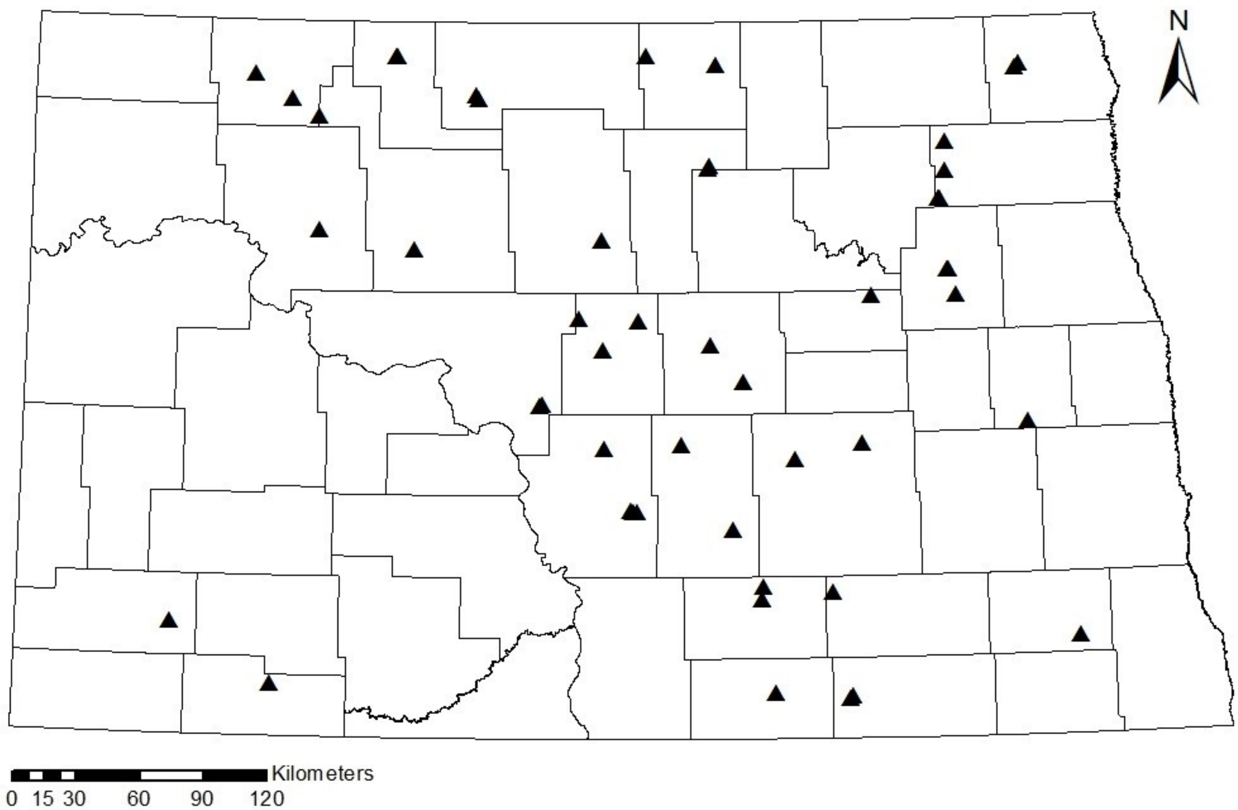

Figure 2. Locations of wetlands sampled in North Dakota.

Table 4. Number of wetlands per assessment, wetland type, and category. The same wetlands were used for both assessments.

Assessment†	Wetland Type‡	Total Number of Wetlands	Category	Number of Wetlands
IPCI	Seasonal	23	Very Poor	4
			Poor	3
			Fair	10
			Good	2
			Very Good	4
	Semi-Permanent/ Permanent	27		
			Poor	12
			Fair	7
			Good	8
NDRAM	Not Available	52 (54)§	Poor	7
			Fair Low	14 (15)
			Fair High	16
			Good	15 (16)

†Assessment used were the Index of Plant Community Integrity (IPCI) and the North Dakota Rapid Assessment Method (NDRAM).
‡ The wetland type temporary for the IPCI assessment type was excluded from this study.
§ Numbers in parenthesis indicate number of wetlands that included the wet meadow zone only.

The assessment's vegetative plot locations were centered around a pre-determined EPA mandated "assessment area" location that also included four standard soil pit locations in the wet meadow zone, one of which soil samples were collected for physical and chemical characteristics, including Olsen P. Within the assessment area, after the vegetative surveys were completed, soil samples were taken (0-15 and 15-30 cm) from three random locations along a 10 m transect within the shallow marsh, wet meadow, and upland positions (Figure 3). Samples were taken using a tiling-spade shovel or Dutch auger depending on sampling conditions and stored in plastic bags.

Samples were transported in iced coolers and stored field-moist at 4 °C. In a select few sites, samples from all landscape positions were unable to be collected due to environmental conditions or absence of a specific position, but completed positions for these wetlands were still included in this study. Sampling occurred in 2011 from mid-June to late August. Prior to P extraction, all samples were homogenized by hand and any rocks, large roots, and visible macrofauna were removed.

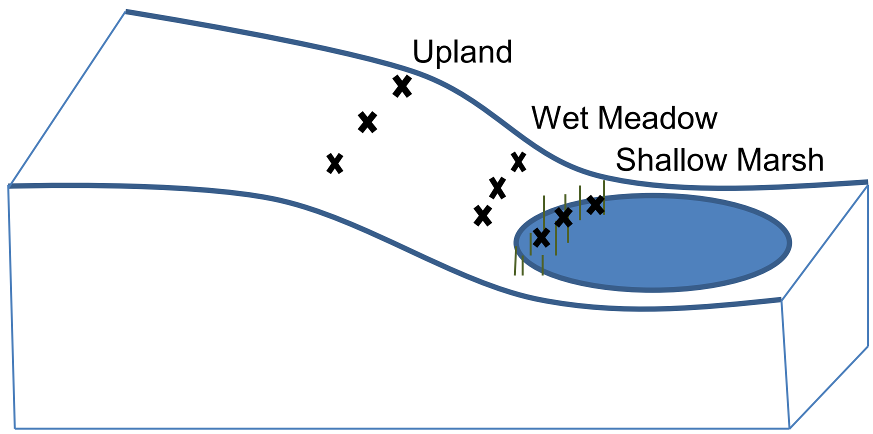

Figure 3. Example of approximate locations of landscape positions sampled for this study.

The Olsen P extraction used in this study was a modification of the procedure recommended for the North Central Region of the United States of America (Frank et al., 1998). Two grams of the homogenized field-moist samples were weighed into 50 mL plastic centrifuge tubes (06-443-18, Fisher Scientific, Pittsburgh, PA) and 40 mL of sodium bicarbonate (0.5 M $NaHCO_3$, pH 8.5) extracting solution was added. Samples were shaken for 30 min at 280 oscillations/min on a reciprocal shaker followed by centrifugation for 20 min at a RCF of 804 x *g*. The supernatant was then filtered through

Whatman No. 2 paper into 10 mL plastic vials. Olsen P extracts were analyzed the same day using a flow-injection analyzer (FIALab 2500, Bellevue, WA) at a wavelength of 880 nm. The FIA analysis of Olsen P extracts included the use of a 10 cm flowcell (for low P concentrations) and an Edmund Optics TS Longpass filter to avoid saturation and bleed over which can occur at 880 nm (FIALab® Instruments). For increased sensitivity, the mixed solution was passed through a plastic tube coil submerged in a water bath (BM100, Yamato Scientific America Inc., Santa Clara, CA) set for 45 ^{o}C. NWCA samples were analyzed for Olsen P at the NDSU Soil Testing Laboratory, Fargo, ND.

Water-extractable P (WEP) was determined using a modified procedure of Self-Davis et al., (2009). Here, 4 g of the homogenized samples were weighed into a 50 mL plastic centrifuge tube (Fisher Scientific, Pittsburgh, PA) and 40 mL of deionized water was added, shaken for 60 min at 280 oscillations/min using a reciprocal shaker, centrifuged for 90 min at a RCF of 647 x *g*, and supernatant filtered through a 47 mm, 0.45 µm filter (097191B, Fisher Scientific, Pittsburgh, PA) using a 47 mm Telfon filter apparatus (1-47, 47-6, and 47, Savillex Corp., Minnetonka, MN). Filtered extracts were acidified by adding two drops of hydrochloric acid (pH 2.0) to deter phosphate compound precipitation, transferred to Wheaton plastic scintillation vials (16300-219, VWR International LLC. , Batavia, IL), and frozen at -10 ^{o}C until analysis. Analysis was done using a flow-injection analyzer (FIALab2500) that was configured as above but the wavelength was set at 860 nm and the water bath at 40 ^{o}C.

Electrical conductivity (EC) was analyzed for each sample followed by pH. Both were performed on 10 g of ground, air-dried soil that was passed through a 1 mm sieve.

Electrical conductivity was determined using the 1:1 soil -to-deionized water method described by Whitney (1998) with an EC probe (SenseION378, Swedesboro, NJ) which was then followed by determination of pH as described by Watson and Brown (1998) using a pH electrode (Accumet AB15, Pittsburgh, PA). Due to high organic matter content, some individual samples from one wetland were analyzed with a 1:5 soil to water ratio instead of 1:1 due to a lack of measureable solution. Conversions to a 1:1 ratio for EC were applied to these samples (Sonmez et al., 2008).

STATISTICAL ANALYSIS

Basic statistics were completed using Microsoft Excel and JMP ver. 8 (ver. 8.0 SAS Institute Inc., Cary, North Carolina). Analysis of variance (ANOVA) and Tukey-Kramer HSD were used to find statistical differences at an alpha of 0.05. For IPCI and NDRAM condition category statistics, Multi-Response Permutation Procedures (MRPP) PC-ORD (ver. 6.0, MjM Software, Gleneden Beach, Oregon; McCune and Grace, 2002) was used to establish statistical significance and the p values for the paired comparisons were adjusted in SAS (ver. 9.3, SAS Inc., Cary, NC) using the Adaptive Hochberg procedure (Sarkar et al., 2012). Averages of the wet meadow and shallow marsh zones only were used for IPCI and NDRAM statistics. Condition categories were considered significantly different at a probability of $p \leq 0.05$. Analysis of samples occurred in 2012 from April to December.

RESULTS AND DISCUSSION

IPCI

Across all wetland types and categories at a depth of 0-15 cm, average Olsen P and WEP values ranged from 5.27 to 51.9 and 0.35 to 7.73 mg/kg, respectively (Table 5). Likewise at a depth of 15-30 cm, average Olsen P and WEP values ranged from 1.31 to 33 and 0.24 to 3.97 mg/kg, respectively. Across all categories, depth, and extraction methods, no differences were found in semi-permanent/permanent wetlands. The significantly greatest average Olsen P concentration (51.9 mg/kg; 0-15 cm) and WEP concentration (7.73 mg/kg; 0-15 cm) were observed in the Very Poor IPCI category for seasonal wetlands. With Olsen P, the Very Poor wetlands were significantly different than the Poor, Fair, and Very Good and with the WEP the Very Poor, Poor, and Good were different from the Very Good. The Very Poor category was also significantly greater than all categories but Good for Olsen P, 15-30 cm, and the Poor was different than the Very Good.

The high variability present in the IPCI results along with the low sample size per category likely reduced the ability to differentiate between different IPCI categories using average P concentrations. It is also possible that the amount of categories (5) under the seasonal wetland type could be masking the difference between categories. For example, the WEP 15-30 cm P value for Very Good in Table 5 is not significantly different from any other category but Poor even though it has the lowest mean value of all the categories. In addition to low sample size (four samples for Very Good and three for Poor), this result is likely due to the smaller spread in values of the Very Good and Poor categories (Figure 4) compared to the larger spread in the remaining categories which had reduced the ability to determine additional categorical differences. Mita et al., (2007) used known IPCI assessment scores to compare with a variety of landscape metrics (mean patch size, patch richness density, etc.) on seasonal wetlands in the PPR. Based on a Tukey HSD pairwise comparison test, none of the nine metrics showed significant difference between all categories. Mita et al., (2007) also found that when determining the distribution of wetlands using the IPCI model and the Landscape Wetland Condition Analysis Model (LWCAM), the association of distribution was stronger when the categories were limited to two (poor, good). Based on this study, it may be possible to assume that reducing the amount of categories within the seasonal wetland type could potentially show more difference between the remaining categories. Euliss and Mushet (2011) take this thought one step further, claiming that using wetland classes/type such as temporary, seasonal, and semi-permanent as separators for IPCI metrics degrades the usefulness of this assessment.

Table 5. Average Olsen P and water-extractable P (WEP) concentrations for each wetland type and Index of Plant Community Integrity (IPCI) category for the combined wet meadow and shallow marsh zones.

		Sampling Depth (cm)			
		0-15		15-30	
Wetland	IPCI	Extraction Method		Extraction Method	
Type	Category	Olsen P	WEP	Olsen P	WEP
		------------------------------mg/kg	--------------------------------		
Seasonal	Very Poor§	51.9 (33.1)†a‡	7.73 (8.64)a	33.0 (28.5)a	3.97 (4.27)ac
	Poor	12.4 (5.35)b	3.27 (2.76)a	4.27 (2.29)b	1.34 (1.32)bc
	Fair	11.9 (11.3)b	2.94 (4.77)ab	5.91 (12.4)b	1.87 (3.36)ac
	Good	22.3 (23.1)ab	3.11 (2.99)a	20.2 (28.1)ab	3.93 (5.85)ac
	Very Good	8.75 (8.98)b	0.35 (0.37)b	2.86 (2.18)b	0.24 (0.45)d
Semi-Permanent/Permanent	Poor	10.4 (13.4)a	2.20 (4.36)a	3.29 (3.90)a	0.55 (0.78)a
	Fair	5.27 (3.22)a	0.86 (0.73)a	1.31 (1.36)a	0.35 (0.31)a
	Good	10.7 (13.2)a	0.53 (0.76)a	5.67 (10.1)a	0.62 (0.94)a

†Numbers in parenthesis indicate standard deviation.
‡Different letters within extraction method depth separated by wetland type indicate significant difference at p≤ 0.05 after adaptive Hochberg.
§Number of wetlands for each category can be found in Table 1.

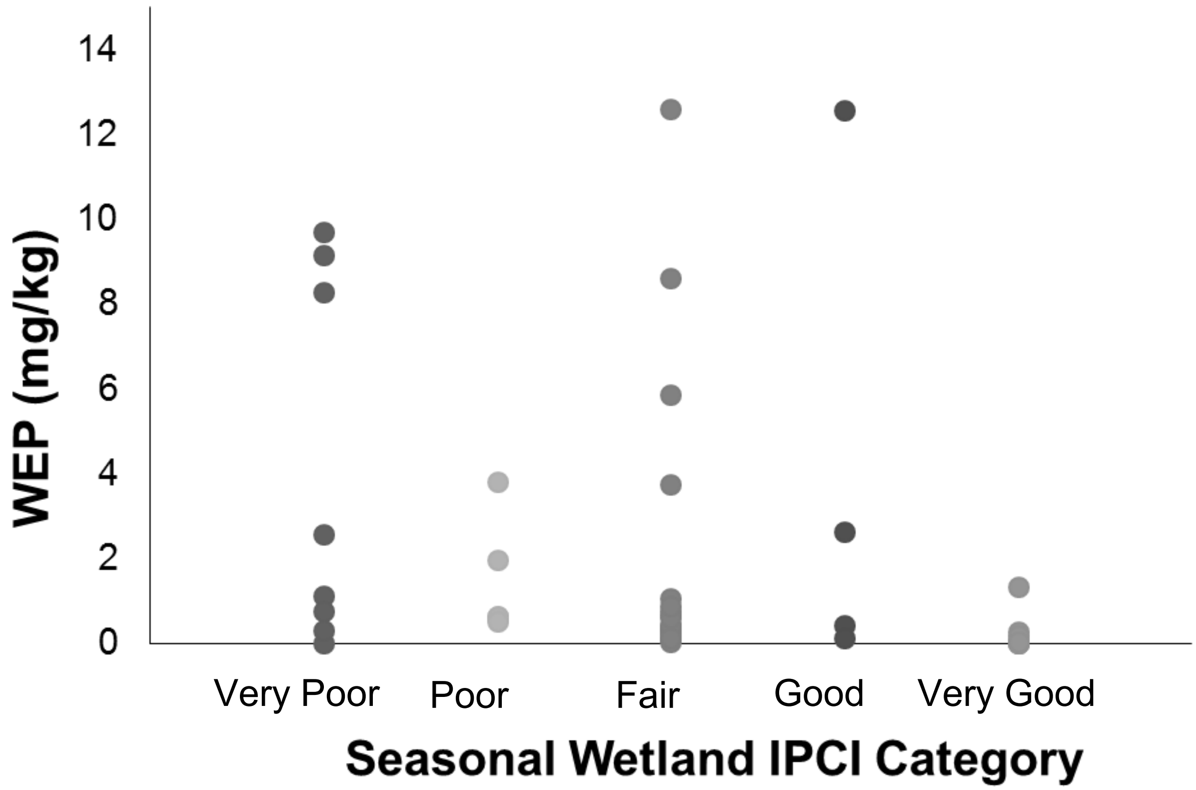

Figure 4. Comparison of water-extractable phosphorus (WEP) 15-30 cm values across seasonal wetland IPCI categories. Mean values for Very Poor, Poor, Fair, Good, and Very Good are 3.97, 1.34, 1.87, 3.93, and 0.24 mg/kg, respectively.

In the semi-permanent/permanent wetlands, EC was significantly greater in the Poor than the Fair category at a depth of 0-15 cm and the pH was significantly greater in the Fair rather than the Good category at a depth of 15-30 cm (Table 6). Difference in the seasonal wetlands was only observed at a depth of 15-30 cm where the pH of the Very Good category was significantly greater than the Very Poor.

Table 6. Average soil electrical conductivity (EC) and pH for each wetland type and Index of Plant Community Integrity (IPCI) category for the combined wet meadow and shallow marsh landscape positions.

		Sampling Depth (cm)			
		0-15		15-30	
Wetland Type	IPCI Category	EC	pH	EC	pH
		---dS/m---		---dS/m---	
Seasonal	Very Poor	1.0 (1.2)†a‡	6.9 (0.6)a	0.99 (0.99)a	7.0 (0.6)b
	Poor	0.54 (0.31)a	7.7 (0.4)a	0.45 (0.26)a	7.7 (0.2)ab
	Fair	0.91 (0.69)a	7.5 (0.8)a	1.0 (0.8)a	7.7 (0.8)ab
	Good	0.4 (0.21)a	6.8 (1.3)a	0.26 (0.15)a	6.9 (1.2)ab
	Very Good	1.4 (1.3)a	7.6 (0.3)a	0.96 (0.94)a	7.7 (0.3)a
Semi-Permanent/ Permanent	Poor	1.6 (1.6)a	7.8 (0.3)a	1.6 (1.6)a	7.9 (0.6)ab
	Fair	0.8 (0.26)b	7.8 (0.2)a	0.9 (0.6)a	8.0 (0.2)a
	Good	1.2 (0.8)ab	7.6 (0.3)a	0.97 (0.77)a	7.8 (0.3)b

†Numbers in parenthesis indicate standard deviation.
‡Different letters within extraction method depth separated by wetland type indicate significant difference at p≤ 0.05 after adaptive Hochberg.

NDRAM

Across NDRAM categories at a depth of 0-15 cm, average Olsen P and WEP values ranged from 8.73 to 22.8 and 2.24 to 3.18 mg/kg, respectively (Table 7). At a depth of 15-30 cm, average Olsen P and WEP values ranged from 2.13 to 7.96 and 0.61 to 2.03 mg/kg, respectively. The average Olsen P value at a depth of 0-15 cm for the Fair High category (8.73 mg/kg) was significantly less than the Fair Low and Poor categories (17 and 22.8 mg/kg, respectively). However, at the same depth for WEP, the Fair Low and Poor categories (3.18 and 2.92 mg/kg, respectively) were significantly greater than the Good category (2.24 mg/kg). At a depth of 15-30 cm, Olsen P for Fair High was significantly less than all other categories and no differences were observed within WEP.

Table 7. Average Olsen P and water-extractable P (WEP) concentrations for each North Dakota Rapid Assessment Model (NDRAM) category for the combined wet meadow and shallow marsh landscape positions.

	Sampling Depth (cm)			
	0-15		15-30	
NDRAM	Extraction Method		Extraction Method	
Category	Olsen P	WEP	Olsen P	WEP
	---------mg/kg---------			
Poor	22.8 (22.5)†a‡	2.92 (4.16)a	11.2 (15.5)a	1.30 (2.13)a
Fair Low	17.0 (22.6)a	3.18 (4.93)a	10.5 (19.8)a	2.03 (3.42)a
Fair High	8.73 (11.8)b	2.59 (5.12)ab	2.13 (2.69)b	0.61 (1.16)a
Good	12.0 (16.0)ab	2.24 (4.95)b	7.96 (13.4)a	1.09 (2.42)a

†Numbers in parenthesis indicate standard deviation.
‡Different letters within extraction method depth indicate significant difference at p≤ 0.05 after adaptive Hochberg.

Across NDRAM categories at a depth of 0-15 cm, average EC and pH values ranged from 0.86 to 1.6 dS/m and 7.3 to 7.8, respectively (Table 8). Likewise, at a depth of 15-30 cm, average EC and pH values ranged from 0.86 to 1.7 dS/m and 7.4 to 8.0. Across all categories and depths, there was no significant difference observed among EC values. However, in both depths, the average pH values were significantly greatest in the Fair High (7.8, 0-15 cm; 8.0, 15-30 cm) compared to other categories.

As with the IPCI assessment, the NDRAM P values were unsuccessful in providing a gradient of significant difference between all assessment categories. Rokosch et al., (2009) conducted a study on forested wetlands to determine if soil indicators, including P, could be used with the Ohio Rapid Assessment Method (ORAM) to estimate soil quality. Although Rokosch et al.'s, (2009) study analyzed soil samples from only six wetlands, both studies included wetlands with various assessment scores. They found that the soil characteristics analyzed (total soil P, Nitrogen (N), enzyme activity, etc.) were unsuccessful in separating the sampled wetlands along a gradient with the ORAM scores, although some indicators (excluding total soil P and pH) did correlate with the scores. Even though analysis of other soil parameters and correlation are beyond the scope of this study, this similar result suggests that soil P, despite availability, is not a useful soil parameter for use in current wetland assessment procedures at this time. Further refinement of assessments and longer monitoring periods may help determine possible relationships between P availability and changing conditions in a wetland for inclusion in future assessments.

The IPCI and NDRAM Olsen P values appear to be generally close to the expected Olsen P average value for the state. The North Dakota state average P value from North Dakota State University (NDSU)-tested fields between 1992 to 2001 was 14 mg/kg using a sodium bicarbonate extraction solution (Cihacek et al., 2009). The Olsen P extraction method for the IPCI Seasonal Very Poor category (both depths) and the NDRAM Poor and Fair Low categories (0-15 cm) are all above 14 mg/kg. Based on the IPCI and NDRAM results, this may indicate that higher than state average P values may indicate higher amounts of wetland disturbance. However, the IPCI Seasonal Good category for both depths was higher than 14 mg/kg and the Semi-Permanent/Permanent Poor category was below 14 mg/kg. Even though the state P average was determined from fields instead of wetlands, IPCI and NDRAM category average values would be expected to be fairly similar. This may indicate that wetland categories based on current characteristics may not reliably reflect soil parameters.

Variability is also an issue with the IPCI and NDRAM P values even though more wetlands are included per category. For example, the Seasonal IPCI Good category consisted of two wetlands. The average Olsen P value at a depth of 0-15 cm for one wetland was 3.30 and 3.55 mg/kg (wet meadow and shallow marsh zone, respectively) while the other wetland's values were 32.0 and 50.4 mg/kg (wet meadow and shallow marsh zone, respectively). In contrast, the average lowest and highest Olsen P value pairs of the Seasonal Fair category (which consisted of 10 wetlands) at the same depth was 2.60 and 4.96 mg/kg for the lowest pair (wet meadow and shallow marsh zone, respectively) and 38.1 and 38.2 mg/kg for the highest pair (wet meadow and shallow marsh, respectively). Variability in soil P values between reference, converted cropland, and excavated wetlands was also problematic in Paper I. In this, a set of previously determined reference wetlands showed higher P values than comparable converted cropland and excavated (disturbed) wetlands. However, the problem of variability of P concentrations in wetlands is not a new one. For example, Rokosch et al., (2009) originally collected eight soil samples randomly from a one meter square plot from each wetland. The sample size from one wetland had to be reduced to six for analysis due to frequent outliers. Rokosch et al., (2009) also had to use a statistical analysis (multi-variate based using principle components analysis) to account for variability within soil data to determine the absence of a total P gradient with ORAM scores. Different site conditions (as discussed in Paper I) as well as other changes which may not be currently detected with wetland assessments could also account for high variability. This would be further encouraged by differences in wetland type and hydrology. If soil parameters are to be included in wetland assessments, the amount of variability would have to be taken into account.

Table 8. Average soil electrical conductivity (EC) and pH for each North Dakota Rapid Assessment Model (NDRAM) category for the combined wet meadow and shallow marsh landscape positions.

	Sampling Depth (cm)			
	0-15		15-30	
NDRAM Category	EC	pH	EC	pH
	---dS/m---		---dS/m---	
Poor	0.86 (0.9)†a‡	7.3 (0.7)b	0.86 (0.82)a	7.4 (0.6)b
Fair Low	1.5 (2.8)a	7.5 (0.7)b	1.3 (1.9)a	7.7 (0.6)b
Fair High	1.6 (1.4)a	7.8 (0.3)a	1.7 (1.4)a	8.0 (0.4)a
Good	1.4 (1.4)a	7.4 (0.8)b	1.2 (1.6)a	7.6 (0.9)b

†Numbers in parenthesis indicate standard deviation.
‡Different letters within extraction method depth indicate significant difference at $p \leq 0.05$ after adaptive Hochberg.

Landscape Position

Across landscape positions, at a depth of 0-15 cm, average Olsen P and WEP values ranged from 5.47 to 16.1 and 1.54 to 2.34 mg/kg, respectively (Table 9) and at 15-30 cm, ranged from 1.83 to 9.46 and 0.64 to 1.60 mg/kg, respectively. Average Olsen P values in both depths show the upland position as having significantly lower P than the other two positions. However, no significant differences were observed in WEP averages for either depth across landscape positions. This may be due to the lower amount of detectable P associated with the method used to determine WEP in this study. Although there is not a significant gradient continuous with elevation, the amount of available P increases toward the center of the wetland. Cheesman et al., (2010) also encountered an increase in total P within the top 10 cm of soil from the upland pasture (117 mg/kg) to the shallow marsh (171 mg/kg) and the deep marsh zone (371 mg/kg) from four wetlands surrounded by pastures in Florida.

Table 9. Average Olsen P and water-extractable P (WEP) concentrations for each landscape position.

Landscape Position	Sampling Depth (cm) 0-15 Extraction Method Olsen P	0-15 WEP	15-30 Extraction Method Olsen P	15-30 WEP
	---------------mg/kg			---------------
Upland	5.47 (6.18)†b‡	1.54 (2.52)a	1.83 (2.26)b	0.64 (1.12)a
Wet Meadow	11.1 (14.2)a	2.22 (4.13)a	5.24 (11.2)a	0.88 (1.85)a
Shallow Marsh	16.1 (20.7)a	2.34 (4.36)a	9.46 (16.5)a	1.60 (2.94)a

†Numbers in parenthesis indicate standard deviation.
‡Different letters within extraction method depth indicate significant difference at $p \leq 0.05$ after adaptive Hochberg.

Electrical conductivity follows the same pattern as Olsen P values with the upland landscape position showing significantly less EC than the other two positions for both depths (Table 10). Between both depths, EC ranged from 0.65 to 1.7 dS/m. These values are higher than those reported by Freeland et al., (1999), which ranged from an estimated 0.3 to 0.8 dS/m for the wet meadow and 0.25 to 1.0 dS/m for the shallow marsh zones in four North Dakota PPR wetlands. Average pH values showed that only the upland was only significantly less than the wet meadow position at a depth of 0-15 cm while there were no differences at 15-30 cm. Excluding the wetland surrounded by cultivation in Freeland et al., (1999), similar pH values were estimated for the wet meadow (7.4 to 7.7) and shallow marsh zones (7.1 to 7.5).

Table 10. Average soil electrical conductivity (EC) and pH for each landscape position.

	Sampling Depth (cm)			
	0-15		15-30	
Landscape Position	EC	pH	EC	pH
	---dS/m---		---dS/m---	
Upland	0.65 (0.55)†b‡	7.4 (0.5)b	0.68 (0.77)b	7.7 (0.4)a
Wet Meadow	1.7 (0.64)a	7.7 (0.6)a	1.6 (2.0)a	7.8 (0.7)a
Shallow Marsh	1.1 (0.85)a	7.4 (0.6)ab	1.1 (0.96)a	7.6 (0.6)a

†Numbers in parenthesis indicate standard deviation.
‡Different letters within extraction method depth indicate significant difference at $p \leq 0.05$ after adaptive Hochberg.

Wet Meadow NWCA, Olsen P

Significant difference was observed between the NWCA pits and tri-sample site when all wetlands were combined (p=0.0023) and when wetlands were divided into IPCI seasonal and semi-permanent/permanent (p=0.0241 and p=0.0462, respectively) (Figure 5). The difference in soil sample depth is one factor that may contribute to this difference. Wet meadow tri-samples were consistently taken from a depth of 0-15 cm, while the range for a comparable depth for NWCA samples varied since sample divisions were based on horizon layers. As a result, data from NWCA samples which fell within the 0-15 cm depth was used for comparison. Differences in extraction methods could also be a factor, since extractions were performed on wet tri-samples versus air-dry NWCA samples. There is limited information available comparing wet and dry Olsen samples, likely due to the traditional and ease of use associated with dry samples. However, both the NWCA Olsen P and tri-sample P values showed no difference among any categories within the IPCI and NDRAM assessment methods.

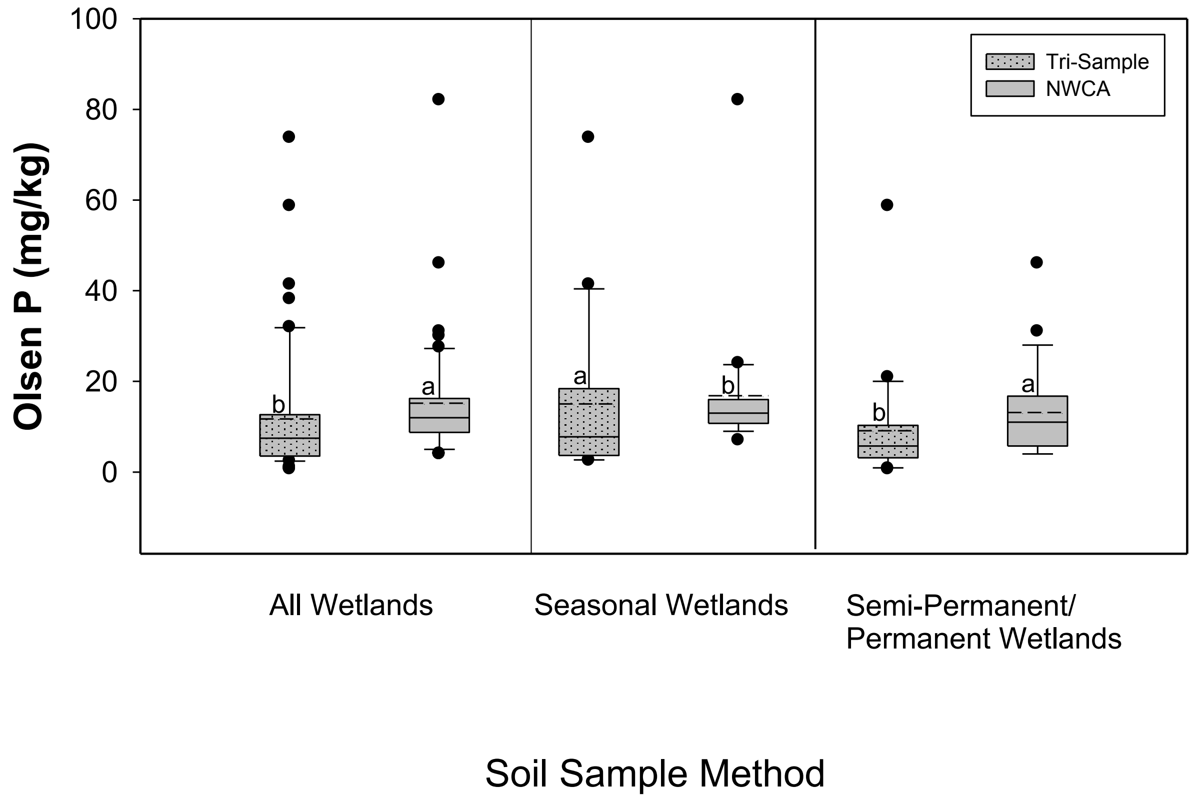

Figure 5.Comparison of P extracted using data from the NWCA's one pit and the average of 3 pits (tri-sample) in the wet meadow land position for seasonal, semi-permanent/permanent, and all wetlands. The dotted line represents mean while the solid line represents the median. From bottom to top, the whiskers and box represent the 10th, 25th, 75th, and 90th percentiles. Different letters within soil sample method wetland type indicate significant difference at $p \leq 0.05$ after adaptive Hochberg.

CONCLUSION

The absence of a gradient established by extracting plant-available P at either depth based on corresponding condition categories associated with the IPCI and NDRAM suggests that P may not be a reliable metric for these assessments. High variability in P availability, as also observed in Paper I, has likely influenced these results. The different number of wetlands in each category may also have been a contributor to this outcome. Plant-available P would be a useful soil metric for the IPCI and NDRAM assessments if significant difference between each category was present which would help accurately reflect a wetland’s condition.

While soil EC and pH may not be significantly useful individually as metrics in wetland assessments, they can still provide background information which may be linked to other soil and plant metrics. Significant gradients were not observed in P, EC, or pH with the IPCI and NDRAM assessments, although significant differences existed between some condition categories within the assessments. This may indicate that if condition scale, condition categories, or division of wetland types within the assessments is re-assessed or condensed into fewer groups, a gradient between conditions may be prevalent. A significant gradient in plant-available P among landscape positions was absent and P availability generally increased toward the center of the wetland. The Olsen extraction method indicated significantly less P from 0-30 cm in the upland position compared to the wet meadow and shallow marsh zones. This suggests that the shallow marsh zone may act as a sink for P in the surrounding landscape. If the removal of P and possibly other soil nutrients is a priority for wetland remediation, these results may suggest focusing on controlling the accumulation of these nutrients in the shallow marsh zone.

Sampling methods for plant-available P should be carefully evaluated before use in prairie pothole wetlands. Significant difference was present between the NWCA samples and tri-samples even though they represented the same wetlands. As mentioned earlier, the shallow marsh zone appears to be the wetland zone with the most accumulation of P in the wetlands used for this study. When only the wet meadow zone P availability was assessed for the NWCA and tri-sample sites, no difference was present in any category with either assessment. However, with the inclusion of the shallow marsh zone for the tri-sample sites, some categories within the IPCI and NDRAM assessments were significantly different. This shows that differences in sampling methods and extraction methods can play a significant role in accurately representing a wetland’s condition. A consistent sampling method would be helpful in future wetland assessments to determine if a soil metric should be included in an assessment.

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GENERAL CONCLUSIONS

Soil metrics such as P, EC and pH are not suggested for use in wetland condition assessments such as the IPCI and the NDRAM since no gradient in P between Very Poor/Poor to Very Good/Good was apparent. However, information provided by these soil characteristics could still be useful to support or help explain abnormalities which may be encountered at specific sites. In support of previous research, the Olsen P values were generally higher than WEP values in both papers. This relationship between both extractions could be used in future wetland studies involving P availability to help eliminate incorrect values since Olsen P values should typically be higher. If fewer wetland types or condition categories are used in assessments, a gradient in P levels may be more prominent since there will be larger groups of wetlands across fewer divisions. If the IPCI and NDRAM assessments are changed in the future, P availability should be studied to determine if a gradient in P levels related to wetland condition is present as a result of changes to the assessments.

Overall, variability exhibited in P availability in wetland soils makes it difficult to use P as a reliable soil metric in wetland assessments and to determine the effectiveness of remediation projects such as sediment removal. Issues associated with variability were somewhat lessened with the use of proper statistical analysis. However, the disproportionate number of wetlands distributed between wetland assessment condition categories may have contributed to additional complications in analysis. Other factors which may have contributed to the variability of one or both studies include hydrology differences between wetlands, depth of excavation, depth of sampling and recovery time for the plant community.

No significant differences in condition categories were present when the wet meadow zone was compared between the Olsen P and WEP procedures. However, with the inclusion of the shallow marsh zone, the replicated samples showed significant differences in some condition categories. A recommendation from this study would be that research being done within each region uses the same sample preparation and extraction procedures so that cross-study data are comparable.

A significant gradient in P was absent between landscape positions although the Olsen P extraction showed significantly less P in the upland compared to the wet meadow and shallow marsh zones. The amount of P generally increased toward the middle (shallow marsh zone) of the wetlands indicating that overland flow concentrates P in or near the wetland boundary or that plants in these regions are able to extract the P from the soil and after senescence concentrate P in the upper soil regions.

Although sediment removal has been used to remediate or improve wetland function, this method may not fully remove P from the shallow marsh zone. Ramifications of P retention include decreased water quality and possible promotion competitive species such as the hybrid cattail (*Typha x glauca*) over native plants. Although the results from Paper II indicate that P may not be completely removed, removal of sediment increased the depth of the water column and thereby decreasing the ability of *Typha x glauca* to become established.