# WHERE’S THE P IN PRAIRIE POTHOLES? IDENTIFYING PATTERNS OF PHOSPHORUS ACCUMULATION IN CANADIAN PRAIRIE WETLANDS

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## ABSTRACT

Prairie wetlands are in the midst of a disappearing act. The number of Canadian Prairie wetlands has been rapidly declining since the early 1900s largely due to agricultural activities and wetland drainage. The impacts of wetland loss include declining water quality and ecosystem health, in addition to reduced water storage. These negative impacts have spurred an interest in the role that the remaining Prairie wetlands play in nutrient cycling and retention. Research to date has focused on comparing intact wetlands to drained wetlands and assessing differences in nutrient retention, specifically phosphorus (P). Phosphorus is a commonly applied agricultural fertilizer, and an excess or deficit of P can have ecosystem altering effects. Limited research has been done to identify how P concentrations vary in intact Prairie wetlands, and the probable drivers of P concentrations. This gap was addressed by collecting comprehensive data from >140 wetland ponds across the Prairie provinces. These data, along with laboratory-based methods showed that select wetland properties, specifically pondwater alkalinity, pondwater conductivity, sediment clay content (%), and surrounding land-use types (grassland/pasture vs. cropland) are the best predictors for P concentrations in Prairie Pothole Region wetlands. Pondwater alkalinity was the best physicochemical predictor of pondwater P concentrations (total P, dissolved P, and dissolved reactive P) whereas land-use type was the best physiographic predictor of pondwater P concentration, and extractable sediment, and soil P. Sites adjacent to cropland had greater concentrations of P compared to grassland/pasture sites. The differences in P concentrations between land use are likely due to greater fertilizer application in cropland compared to grassland/pasture. This work combines our understanding of P chemistry and the impact of landscape scale processes to identify the key probable drivers in the accumulation of P in Prairie wetlands. This also provides us with a more defined direction for future research, specifically more thoroughly exploring land use influences and ionic composition.

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| | |
|---|---|
| Al | Aluminium |
| C | Carbon |
| Ca | Calcium |
| $CaCl_2$ | Calcium chloride |
| $CaCO_3$ | Calcium carbonate |
| Cl | Chlorine |
| $CaMg(CO_3)_2$ | Dolomite |
| DP | Dissolved phosphorus |
| DRP | Dissolved reactive phosphorus |
| EC | Electrical conductivity |
| Fe | Iron |
| IC | Inorganic carbon |
| K | Potassium |
| Mg | Magnesium |
| N | Nitrogen |
| Na | Sodium |
| OC | Organic carbon |
| OM | Organic matter |
| P | Phosphorus |
| P:A | Pond perimeter to area ratio |
| $PO_4$ | Phosphate (including $HPO_4^{2-}$ and $H_2PO_4^{-}$) |
| PP | Particulate phosphorus |
| PPR | Prairie Pothole Region |
| SC | Specific conductance |
| $SO_4^{2-}$ | Sulfate |
| SOC | Soil organic carbon |
| TC | Total carbon |
| TP | Total phosphorus |

## 1.0: Introduction

### 1.1 Overview

The number and quality of wetlands in the Prairie Pothole Region (PPR) has been declining since the early 1900s (van der Valk 1989). Activities such as agriculture and wetland drainage have caused a disproportionate loss of small wetlands (Evenson et al. 2018), increased average wetland size (Van Meter and Basu 2015), and decreased shoreline to water area ratios (Millar 1971). As a result of wetland drainage and declining local water quality, there is a growing interest in the nutrient retention potential of Prairie wetlands (Cheng and Basu 2017). Research seeking to characterize this wetland behaviour, however, has largely focussed on comparing intact wetlands to drained wetlands, or restored wetlands. For the macronutrients, nitrogen (N) and phosphorus (P), there has been little work done to contrast the varying behaviour of Prairie wetlands with respect to their nutrient pools. The gap in understanding and capacity to identify the probable drivers of P accumulation in Prairie wetlands is a challenge. More research is needed to inform pragmatic wetland management decisions, including strategic wetland conservation and restoration, or approval of wetland drainage. The objective of this research was to **identify the potential physicochemical and physiographic drivers on pondwater P concentrations** across a gradient of wetland conditions, using a survey approach.

This thesis is presented in the traditional thesis format consisting of 5 sections and an appendix. Section 1.0 introduces the research, provides a literature review and identifies objectives. Section 2.0 is materials and methods, outlining the approaches used in the field, analytical techniques employed in the laboratory, and data analysis steps. Section 3.0 presents the results, while 4.0 discusses those results. Lastly, section 5.0 provides conclusions and insights from this study. Appendix A contains supplemental data.

### 1.2 Literature review

#### 1.2.1 Wetlands

Wetlands provide numerous ecosystem services; however, across the world wetlands are being actively destroyed (Mitsch and Gosselink 2000). Occupying 12.1 million $km^2$, or 6% of the world's surface, they are home to rare organisms (Ramsar Convention on Wetlands 2018), mitigate the impacts of flooding (Evenson et al. 2016), and act as biogeochemical hotspots

(Semlitsch and Bodie 1998; Cheng and Basu 2017). As wetlands are removed or degraded by anthropogenic stressors, the amount and quality of ecosystem services they provide declines (Erwin 2009). Upwards of 25% of organisms in these systems have become endangered (Ramsar Convention on Wetlands 2018). Removal or degradation of wetlands impairs water retention abilities, resulting in increased costs from flooding (Pattison-Williams et al. 2018), and alters biogeochemical processes in these systems (Semlitsch and Bodie 1998). We know wetlands are changing, but how they are changing and the implications of such change on our ecosystems are of great concern to those—human and non-human— that interact with wetland systems. More work needs to be done to better understand how wetlands function, to fully understand how they are impacted by ongoing destruction and climate change. Current research indicates that climate change is expected to intensify the degradation of wetland systems and alter biogeochemical processes (Erwin 2009; Niemuth et al. 2010). Thus, these valuable, diverse, and sensitive water bodies are under threat.

### 1.2.2 Prairie pothole region

One region in which wetlands are under immense pressure from anthropogenic stressors is the PPR. One of those anthropogenic stressors is nutrient application, or more specifically P fertilization. Phosphorus is one of the most common agricultural nutrients in the PPR, and is typically in the form of chemical fertilizers or manure (Tilman et al. 2002); however, not all applied P is taken up by crops and an excess of P can remain in the environment (Kalra and Soper 1968). Excess P can move from the point of application into surrounding water bodies, where it contributes to the eutrophication and degradation of water resources. Eutrophication threatens the quality of our drinking water supply and there are high costs associated with management of excess nutrients in order to ensure safe drinking water (Schindler et al. 2012) and healthy ecosystems. This growing concern and financial strain has fuelled a need for solutions for managing excess P. Phosphorus, however, is controlled by an array of physical, chemical and biological processes—processes that are unfolding at varying temporal and spatial scales (Reddy et al. 1999; Orihel et al. 2017). These complex processes give rise to varying concentrations and rates of P accumulation across the PPR.

The PPR covers 750,000 km$^2$ of North America (Figure 1.1), extending through the Canadian Prairies south into the northern plains of the United States (Hayashi et al. 2016). The hummocky to undulating landscape was formed as glaciers retreated over 10,000 years ago

(Christiansen 1979). The glacial retreat left behind glacial till material—rich in calcium carbonates ($CaCO_3$) and clay—in an uneven pattern (Christiansen 1979; Last and Last 2012). This uneven pattern has resulted in a complex of depressions across the landscape (Christiansen 1979). The combination of this variable landscape, a climate with low rates of precipitation, and fine-textured soil with low rates of hydraulic conductivity has limited the development of stream networks (Shook et al. 2013), and instead surface water accumulates in depressions known as Prairie "potholes" or wetlands (Shook et al. 2013). Overall soil, climate, and glacial retreat patterns work together to create the unique PPR landscape where between 16-18% of the area (~135 000 km$^2$) was once wetlands (Dahl 1990).

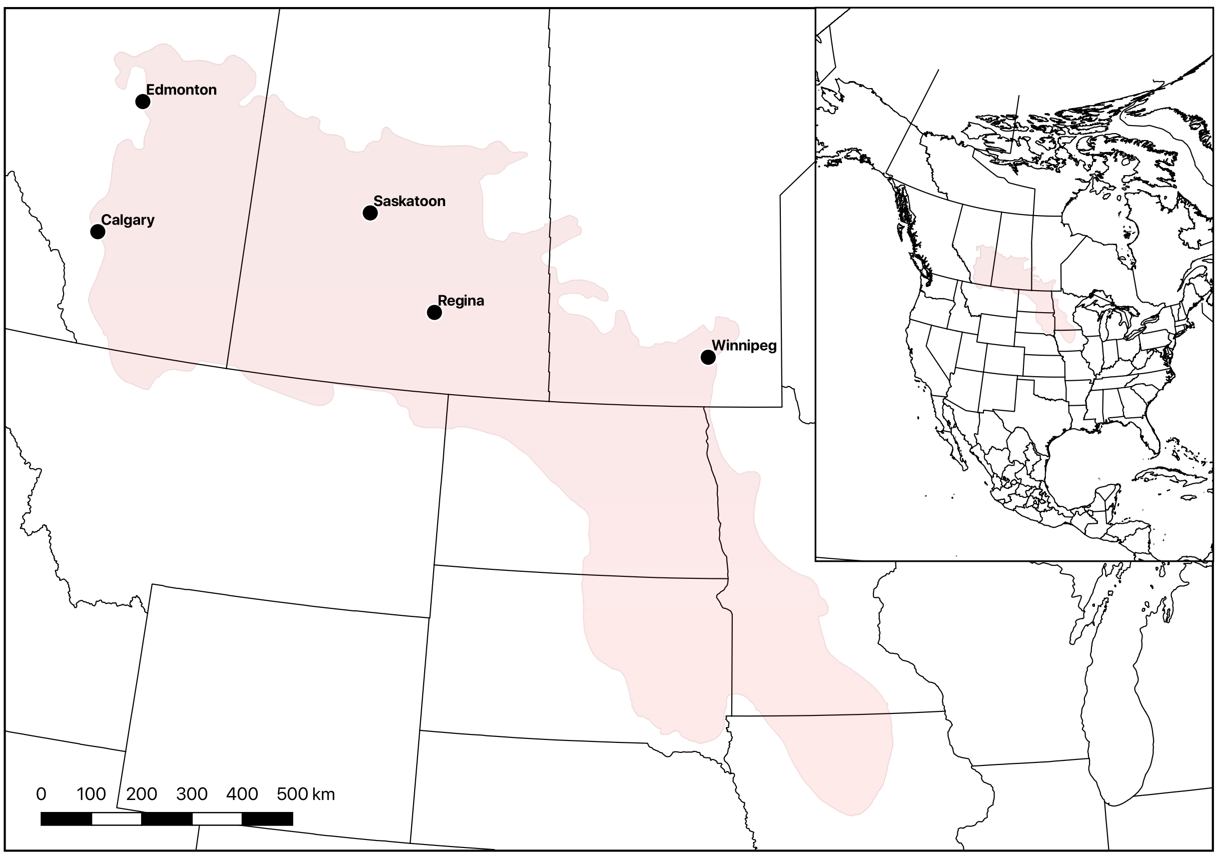


**Figure 1.1** Map of Prairie Pothole region extent (pink shaded area) and largest Canadian cities (black dots) with location of PPR within North America (inset). PPR boundary and political boundary data drawn from North American Political Boundaries from U.S Geological Survey. Information licenced under the Department of Interior Copywrite, Restrictions, and Permissions https://www.doi.gov/copyright. Cities drawn from Statistics Canada, and with information licensed under the Open Government Licence-Canada http://open.canada.ca/en/open-government-licence-canada

The Canadian PPR features a semi-arid to sub-humid climate with alternating multi-year wet and dry cycles. This means that on average the PPR has annual rates of evapotranspiration exceeding the rate of precipitation (Winter 1989; Hayashi et al. 2016). As a consequence of its climate and the hummocky terrain, drainage networks are poorly developed, meaning that in a typical year, much of the precipitation does not manifest as runoff to major streams and rivers. These patterns of water accumulation on the landscape have seen dynamic change over the past 50 years, and the fraction of runoff derived from snowmelt is decreasing, while the fraction of runoff deriving from precipitation increases (Dumanski et al. 2015). With climate change, the patterns and intensity of precipitation are changing and the temperatures in the Prairies are increasing faster than the global average (DeBeer et al. 2016). For example, shifts in seasonal precipitation patterns due to climate change are already happening (DeBeer et al. 2016; Hayashi et al. 2016). These patterns in turn impact farm-scale practices. If a wetland pond dries out earlier in the season because of a decrease in precipitation, it can then be tilled and seeded, increasing the area of productive land (Johnson et al. 2010; Johnston 2013), but a wetland's role as waterfowl habitat and a nutrient sink can be compromised by this activity. Conversely, wetlands that are expanding into cropland because of increases in the amount of precipitation due to shifts in precipitation patterns may be targeted for drainage, to increase long-term usage of the area for crop production (Brown et al. 2017b), once again leading to a loss of habitat and changing wetland nutrient retention potential (Badiou et al. 2018).

#### 1.2.2.1 PPR hydrology

Most surface water movement in the PPR occurs during the snowmelt period. During the winter months, blowing snow accumulates in depressions of the hummocky landscape (Fang and Pomeroy 2009). During spring snowmelt, when evapotranspiration remains low, water flows into these depressions and the storage of surface water in pothole ponds increases (Fang and Pomeroy 2009). While the summer months in the Prairies feature the majority of the precipitation; evapotranspiration rates are high and runoff has historically been limited to intense rainfall events (Hayashi et al. 1998).

Due to potential evapotranspiration exceeding actual evapotranspiration on the landscape, subsurface flow plays an important role in the movement of water between Prairie wetlands. Subsurface flow is generally the main pathway for water movement during dry to normal conditions (Hayashi et al. 2016). Subsurface flow occurs both just below the soil surface and in deeper groundwater pathways. In the case of subsurface pathways, during periods of rising water tables and prolonged soil saturation, effective transmission pathways emerge as soil water storage capacities are exceeded (Brannen et al. 2015). These effective transmission zones connect uplands and ponds and allow for the lateral movement of water and its solutes into ponds via subsurface pathways.

Wetland ponds span a gradient of hydrological connectivity—there are generally three classifications of ponds: recharge, flow-through, and discharge. Typically those in higher landscape positions are connected to deeper groundwater systems with slow transmission pathways (van der Kamp and Hayashi 2009; Figure 1.2). Ponds with water moving downwards "recharging" the groundwater are called recharge wetlands, and these wetlands are typically located in higher landscape positions (Arndt and Richardson 1989). These ponds are mostly freshwater filled by snowmelt and/or fill-and-spill (Arndt and Richardson 1989). Flow-through wetlands are those with changing groundwater recharge and discharge behaviour depending on the water table position (Winter and Rosenberry 1998). Lastly, discharge wetlands, are "filled" by groundwater discharging into the ponds and tend to be lowest in the landscape.

During relatively wet conditions, fill-and-spill processes are important, with wetlands becoming temporarily connected to one another during spill events (van der Kamp and Hayashi 2009). Snowmelt or the rare intense rainfall allow ponds to "fill" with water, and once the wetland is "full" of water, the excess "spills" into wetlands lower in the catchment (van der Kamp and Hayashi 2009). This brief period of surface water connectivity is important for the transport of solutes and nutrients across the landscape (Hayashi et al. 2016), and affirms again the complexity of Prairie wetlands. Even with several different water transmission pathways, these periods of hydrological connectivity are often temporary, and Prairie pothole wetlands continue to be considered geographically isolated (Evenson et al. 2016). Understanding how water accumulates and moves in the PPR is foundational for understanding why each wetland is unique and has a distinctive role to play in the greater PPR landscape.

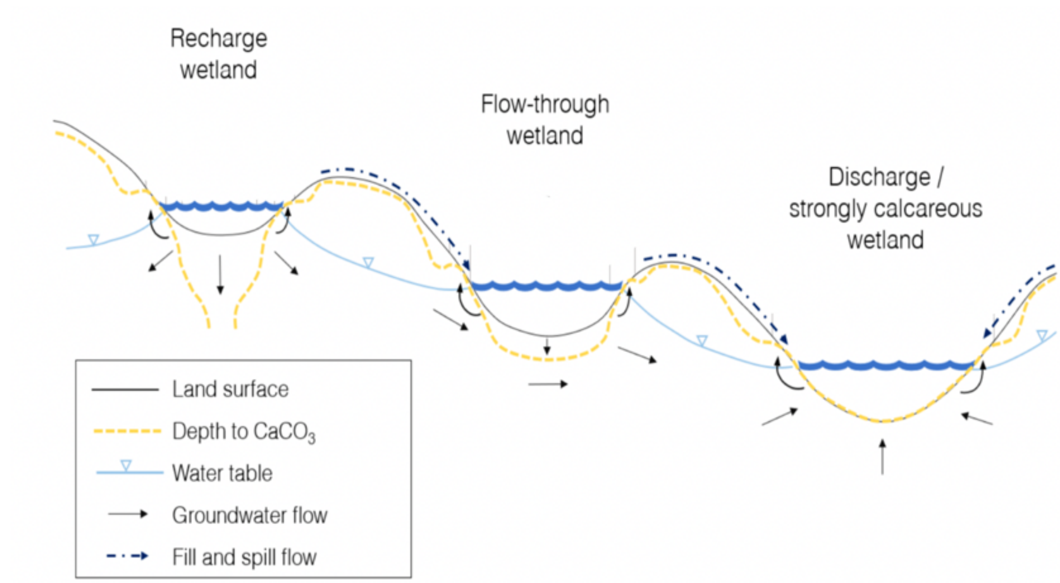


**Figure 1.2** Schematic of depth to $CaCO_3$, groundwater table, and ground water flow directions from Kiss (2018). Based on diagrams from van der Kamp and Hayashi (2009), Pennock et al. (2011) and Pennock et al. (2014).

### 1.2.2.2 PPR soil properties

Another aspect influencing the variability of wetlands across the PPR is soil. Soils are highly spatially variable and influenced by deposition of glacial parent material, precipitation, temperature, topography, and anthropogenic activity (Jenny 1941). Even soil within the same field can look vastly different simply because of topography. Recognizing and understanding the spatial variability of soils is important, as soil properties have an influence on the accumulation and the depletion of nutrients, thus impacting crop growth and adjacent waterbodies.

Specific soil properties need to be examined in order to better understand the importance of soils with regards to possible wetland nutrient probable drivers. Soil properties such as soil texture and carbon (C) content tend to have the greatest influence on the accumulation of nutrients in soils and therefore need to be well understood (Ige et al. 2005; von Wandruszka 2006; Dunne et al. 2010). Across the Prairies, soil textures range from fine silts to heavy clays (Moss and Clayton 1967); however, even within a hummocky clay loam landscape, wetland soils will often be finer than uplands. Soil texture is important because different soil particle sizes have different water-holding and sorption capacities (Zou et al. 2012). Clay particles (<0.002 mm) are much smaller in size than sand (0.05–2 mm) but have significantly more surface area per unit mass, giving clay a greater adsorption capacity because with greater surface area there are more sites available for binding (Zou et al. 2012). While clay also can hold large quantities of water, water movement through a clay soil matrix is slow, which can lead to saturated soil conditions that are unfavourable for some agricultural practices (Bedard-Haughn 2009). Previous work showed that soil texture is an important factor in the short-term control of wetland P dynamics because soils with greater clay content sorb $PO_4$ more tightly then other soil textures (Reddy et al. 2005; Haque et al. 2018b).

Carbon is another important soil property to be considered. Carbon accumulates in two general forms: organic or inorganic C. Organic carbon (OC) represents the forms of C that are in living and non-living organic matter (OM); such as algae and plant material (Reddy and DeLaune 2008a). Inorganic carbon (IC) occurs in rocks such as calcite ($CaCO_3$) and dolomite ($CaMg(CO_3)_2$) (Goh and Mermut 2008).

Organic C accumulations in the PPR are heavily influenced by landscape-scale variability, anthropogenic activity, and climate variability (Bedard-Haughn et al. 2006; Brown et al. 2017b). For example, the concentration of OC in wetland soils and sediments is much greater than the concentration of OC in agricultural soils (Euliss et al. 2006). This is, in part, due to different rates of C accumulation and decomposition between upland agricultural and wetland environments (Euliss et al. 2006). Historically, C concentrations in PPR soils have been in decline, but as a result of the widespread adoption of conservation agriculture and no-till practices in recent decades, C concentrations in some Prairie soils have been on the rise (Awada et al. 2014). Conservation tillage, also known as no-till, leaves more OM on the field to replenish the soil C, typically in the form of crop stubble (Lal et al. 2004). Wetlands, on the other hand, have higher rates OC accumulation owing to the abundance of vegetation in and around wetlands. With an abundance of both above and below ground biomass from vegetation, and a slower rate of decomposition due to their anaerobic conditions, wetlands are one of the greatest land-based C sinks (Euliss et al. 2006). Organic C is also important in driving the accumulation of nutrients like P because decomposing OM can serve as a source for P through decomposition and because OM has many surfaces available for phosphate ($PO_4$) sorption (Reddy et al. 2005). Phosphate (either $HPO_4^{2-}$ or $H_2PO_4^{-}$ at environmentally relevant pH values (Pierzynski et al. 2005; Condron et al. 2005) and referred to here collectively as $PO_4$) is considered to be readily available for plant and microbial uptake. In many environments, crops are encroaching on wetlands to maximize the productive agricultural land, but this practice is reducing or eliminating the OM contributions previously obtained by the presence of vegetation buffers.

Inorganic C has different principal probable drivers than OC. Hydrologic processes control how IC—derived from $CaCO_3$- and $CaMg(CO_3)_2$-rich parent material—is distributed in the PPR environment (Heagle et al. 2007; Last and Last 2012). Accordingly, there is variation in the accumulation of IC across the landscape and between wetlands. For example, in recharge wetlands, water moves downward as groundwater. This downward hydrological movement transfers soluble ions through the soil profile; therefore, IC in the ponds and adjacent soils are typically lower than in groundwater (Figure 1.2 Schematic of depth to CaCO3, groundwater table, and ground water flow directions from Kiss (2018). Based on diagrams from van der Kamp and Hayashi (2009), Pennock et al. (2011) and Pennock et al. (2014).

; Arndt and Richardson 1989). This IC-rich groundwater then supplies water to discharge wetlands, resulting in pondwater and soils that are rich in IC and secondary carbonates (Arndt and Richardson 1989). As the groundwater moves upwards, $CaCO_3$ and $CaMg(CO_3)$ precipitation occurs in the soil, resulting in secondary carbonate deposition (Arndt and Richardson 1989; Bedard-Haughn and Pennock 2002). Pondwater conductivity changes are also influenced by $CaCO_3$, and $CaCO_3$ precipitation. Precipitation generally begins at 1000 µs $cm^{-1}$ and provides a useful threshold for the divisions between freshwater ponds and saline ponds (Pennock et al. 2014). Understanding the role and patterns in IC and OC accumulation is key to unlocking the potential of nutrient processes and accumulation in PPR wetlands.

The impact that C—both OC and IC—and soil texture have on stimulating nutrient cycling and enhancing nutrient adsorption makes them important to consider in the context of nutrient processes and PPR wetlands (McGill and Cole 1981; Reddy et al. 1999). Soil texture and C are just two components contributing to the immense variability of the PPR; recognizing and then understanding this variability are two steps of many towards understanding patterns of P accumulation and retention in PPR wetlands.

### 1.2.3 Phosphorus biogeochemistry

Phosphorus is critical for the health of our ecosystems, but a delicate balance must be maintained. A depletion of P means that plants fail to grow, starving out the fauna that feed on them and impacting the food web in a multitude of ways; but an influx of P leads to largely unrestricted bacteria growth at rates beyond ecosystem equilibrium thresholds and potentially leading to more death in the food web. Like all ecosystems, the PPR thrives in a specific range of nutrient accumulation and depletion, but we have begun to see symptoms that suggest we are at the extreme end of that range.

There are numerous dynamics that drive the transformation and accumulation of P, all of which can be influenced by pedogenesis, pH, redox potential, soil texture, climate, land use, and vegetation (Pierzynski et al. 2005; Condron et al. 2005). The forms of P can be most generally divided into organic P (P bonded to C) (Stewart and Tiessen 1987) and inorganic P. The proportion of each form of P in an environment is variable; organic P can make up anywhere between 0% and 100% of the total P in a given environment (McKelvie 2005). There are many subgroups within these organic and inorganic P forms, but only a single form of inorganic P ($PO_4$). There is mounting interest in the biogeochemical process that replenish $PO_4$ (Cordell et al. 2009; Richardson and Simpson 2011) as other forms of P cannot be directly used by plants (Weihrauch and Opp 2018). For example, mycorrhiza and bacteria can change the soil environment by releasing phosphatases and/or organic acids, which can release $PO_4$ to the soil solution for uptake by plants (Weihrauch and Opp 2018).

The potential for transformation of other forms of P to $PO_4$ in the soil solution makes knowing the size of the P pool—in all its forms—and the controls that facilitate the transformation of P incredibly useful when making decisions about P management in the environment. Some of the processes that facilitate this key transformation of P include immobilisation and mineralization; sorption and desorption; and precipitation and solubilization. Immobilization is the processes of $PO_4$ being taken up by organisms, and either transformed into organic P compounds or stored in cells as $PO_4$ or polyphosphates (chains of $PO_4$) (Condron et al. 2005). Specifically, $PO_4$ is absorbed into the cells of OM such as microbes or vegetation (Condron et al. 2005). Mineralization is the opposite of this processes, it is the release of $PO_4$ by decomposition of organic matter from microbes or vegetation, or the release of $PO_4$ by hydrolyzation of organic P compounds by P-specific enzymes (phosphatases) (Condron et al. 2005). Sorption and desorption are processes that mediate the retention of $PO_4$ and some organic P compounds such as DNA or phytate in the environment (Reddy et al. 2005). Adsorption is the process that affects how both organic and inorganic P compounds from the soil solution or pondwater accumulate on the surface of clay or minerals such as iron (Fe) or aluminum (Al) (oxy)hydroxides. This is a more temporary and rather quick processes that can be easily reversed compared to other P transformation processes. Desorption is the opposite of adsorption. It is the processes of adsorbed $PO_4$ and organic P compounds being released into solution from the surface of clay or minerals. Lastly, precipitation and solubilization. Precipitation is the processes by which ions such as $Al^{3+/2+}$, $Fe^{3+/2+}$ and $Ca^{2+}$ react with $PO_4$ and some organic P compounds in the soil to form less-soluble P complexes (Reddy et al. 2005). This process tends of be slower and more difficult to reverse than sorption (Reddy et al. 2005). Solubilization is the reverse of precipitation: the release of precipitated P compounds back into the soil solution.

Understanding each of these processes is important as both inorganic and organic P compounds are involved. However, before diving deeper into the specifics of how these abiotic and biotic processes specifically facilitate the transformation of P it is important to note that while the role of organic P merits a place in the discussion regarding the accumulation of P in PPR wetlands, the detailed lab analysis required to distinguish the accumulation of both organic P and inorganic P compounds is beyond the scope of this project.

#### 1.2.3.1 Abiotic controls of P retention

Phosphorus retention in wetlands is mediated by several abiotic controls. Some of these major abiotic controls include reactions with Fe, Al, $Ca^{2+}$, sulfate ($SO_4^{2-}$), potassium ($K^+$), magnesium ($Mg^{2+}$), chlorine ($Cl^-$), and sodium ($Na^+$). The controls also include changes in pH, redox potential, OC, and clay content. This section will begin by outlining the three main elements known to play an important role in regulating P retention, as most inorganic P complexes fall within one of two groups: Ca-containing complexes, or Fe- and/or Al-containing complexes (Reddy et al. 2005). Understanding the variability and unique conditions of these complexes is important as the ionic composition of wetland pondwater is highly variable across the PPR; in some regions $Na^+$, $K^+$, and $SO_4^{2-}$ are dominant ions, while in others $Mg^{2+}$ and $Cl^-$ dominate (LaBaugh 1989). As the following section will explore how the stability of these complexes is governed by pH and redox conditions (Reddy et al. 2005), the impact of changing pH and redox on the complexes will be discussed within each element section. It is also vital to mention that processes that facilitate the transformation of P are not exclusively mediated by the aforementioned complexes; OC and clay content also play an important role (see below).

Iron-$PO_4$ complexes are the most abundant forms of inorganic P found in acid, freshwater, and brackish environments (Reddy and DeLaune 2008b). Iron-$PO_4$ is susceptible to changing redox conditions; specifically, ferric Fe (III) in $FePO_4$ is reduced to ferrous Fe (II) under anaerobic conditions (Reddy et al. 2005). In the PPR wetland environment further complexity is added as $SO_4^{2-}$ is common in the pondwater (Jensen et al. 2009). In anaerobic environments, the reduction of $SO_4^{2-}$ by microbial activity means that ferrous sulfides may be forming before Fe-$PO_4$ compounds, and therefore the buffering capacity of Fe to retain $PO_4$ compounds is diminished (Lamers et al. 1998; Hoffmann et al. 2009). Not only does the reduction of $Fe^{3+}$ increase the solubility of $PO_4$ compounds in the soil solution, but under the low oxygen conditions $Fe^{3+}$ is used as an alternate electron receptor (Reddy et al. 2000, 2005) by microbes found in the soil environment. This complex relationship can ultimately result in increased concentrations of inorganic $PO_4$ available for organisms in the wetland environment (Jensen et al. 2009)

Calcium-phosphate complexes, on the other hand, are the most abundant form of inorganic P in alkaline and saline environments (Reddy and DeLaune 2008b). Calcium-phosphates are found in many forms including Ca-phosphate, dicalcium phosphate, beta-tricalcium phosphate, octacalcium phosphate, and hydroxyapatite. For the most part these complexes are not redox sensitive, remaining unavailable under anaerobic conditions. This, however, is not always the case as the solubility of some Ca-phosphates such as tricalcium phosphate is sensitive to changes in pH associated with redox conditions. Overall, the insolubility of Ca-$PO_4$ compounds can decrease the overall bioavailability of P in ecosystems. In the PPR, $Ca^{2+}$ and other ions such as $SO_4^{2-}$ and $Mg^{2+}$ are common as they are derived from the weathered till parent material common across region (Goldhaber et al. 2014). These ions are then transported into the wetland through groundwater (Euliss et al. 2014). Calcium carbonate has also recently been identified as an important control on $PO_4$ sorption in PPR wetlands soils, as research showed that wetlands both rich and depleted in $CaCO_3$ had the same amount of total P, but available P was six times greater in $CaCO_3$-depleted wetlands than $CaCO_3$-rich ones (Brown et al. 2017a). Soils that are rich in OM can also have up to 72% of the TP present as Ca or Mg-bound $PO_4$ (Reddy and DeLaune 2008b). When the release of $PO_4$ from Fe-$PO_4$ compounds occurs in alkaline ponds—common in the PPR—the excess $PO_4$ released can react with $Ca^{2+}$. Calcium-$PO_4$ compounds form and precipitate from the water column more easily than Al and Fe compounds. It is important to mention that $K^+$, $Mg^{2+}$, $Cl^-$, and $Na^+$ also have some degree of regional variation and influence P accumulation (Last and Last 2012).

As mentioned in the previous section, pH is important for controlling the dominant $PO_4$ compounds (Reddy and DeLaune 2008b). Changing pH impacts the ion repulsion of $PO_4$

compounds or results in the dissolution of precipitation $PO_4$ compounds (Penn and Camberato 2019). Acid release during decomposition of OM and by plants and microbes in the soil can decrease pH, resulting in solubilization of Ca-$PO_4$.

Higher clay content and OM usually results in a larger number of binding surfaces for $PO_4$ sorption (Reddy and DeLaune 2008b). Organic matter complexed with Fe and Al is also responsible for additional $PO_4$ sorption (Reddy et al. 1999). Clay is also usually high in Fe and Al oxides, which furthers its capacity as to bind $PO_4$ (Reddy and DeLaune 2008b). For example, due to the high surface area of clay particles there may be a greater abundance of $PO_4$ compounds in clay-rich sediment than compared to sand or silt. Research has also reported that in soils with high IC and P concentrations, $PO_4$ in solution will precipitate, forming insoluble P complexes and that are not readily available for plant uptake (Stewart and Tiessen 1987). Oxidation of OM can also facilitate the conversion of organic P compounds to $PO_4$ (Reddy and DeLaune 2008b). However as previously mentioned, binding capacity of Fe and Al (oxy)hydroxides is pH-dependent.

All things being equal, the behaviour of $PO_4$ compounds is also in part due to the concentration of P compounds in the pond or porewater or soil, as dissolved $PO_4$ compounds will strive to find an equilibrium between the sediment and water (Reddy et al. 2005). If the pondwater has a greater concentration of P than soil, then the dissolved $PO_4$ will be sorbed or precipitated on and into the soil until an equilibrium between the water and soil is found.

#### 1.2.3.2 Biotic processes in P dynamics

As alluded to in the abiotic processes section, P cycling is also influenced by a number of processes. Biotic processes include the following: assimilation and immobilization of P by vegetation, plankton, and microorganisms, decomposition of OM, and mineralization of organic P. While for the most part these biotic processes impact the organic P fraction, the biotic processes that mediate the transformation of organic P to $PO_4$ is incredibly important as organic P can represents up to 100% of the total P in some environments making the processes that mediate transformation to $PO_4$ especially influential (Reddy et al. 2013). These biotic process are also inherently linked to the abiotic P cycling processes.

Phosphate can be taken up and stored by plants, with the amount and kind of vegetation playing an important role. Immobilization of $PO_4$ occurs during this process as $PO_4$ is converted into organic P by microbes and other organism to be integrated into living cells, or is stored in cells as $PO_4$ or polyphosphates. For example, macrophytes can absorb $PO_4$ directly from the water column (Reddy et al. 1999). Processes like bioturbation by aquatic organisms and plant growth will also impact the rate of P deposition from or resuspension to the water column.

The decomposition of organic matter can result in $PO_4$ being released back into the environment, unless plants are removed from the site, as occurs through harvest (Reddy et al. 1999). The release of $PO_4$ is primarily mediated by hydrologic enzymes that mineralize organic P compounds. The mineralization of organic P is an important biotic process influencing P dynamics. The hydrologic enzymes break the bonds between $PO_4$ and OM, which results in $PO_4$ being released back into the environment. It is important to note that the relationship and presence of microbes and enzyme processes which facilitate the release of $PO_4$ drops significantly when in $PO_4$-anaerobic or water saturated environments (Condron et al. 2005). Thus, biotic processes become an important consideration in both the short-term and long-term retention of P in wetlands. Other biotic processes mediate the abiotic chemical processes, such as the release of organic acids by microbes and plants, to release sorbed $PO_4$ compounds.

### 1.2.4 Physiographic groupings

This section will explore the physiographic groupings that have emerged as tools for better understanding patterns and probable drivers of nutrient accumulation—or more specifically P—in the PPR landscape. Specifically this section will outline the role of the soil climate zones, perimeter to surface water area ratios (P:A), pond permanence classes, pond salinity classes, land use, and watershed classes.

Prairie soils can be generally grouped into four soil climate zones (Brown, Dark Brown, Black and Grey). Named after the color of their soils, which reflects the soil OM levels, each of the soil zones have unique climate conditions and vegetation (Fuller 2010). Soil zones are a relatively well recognized grouping within the agronomic sphere and can be used to assign some level of land value, as well as to determine suitability for select crops (Campbell et al. 2002).

Wetland P:A is a physiographic grouping to consider, as recent work suggests that ponds with larger P:A act as biogeochemical hotspots (Cheng and Basu 2017). This metric may be more useful for PPR ponds than area or perimeter individually because area and perimeter were generally dynamic due to changes in water depth, whereas P:A remains generally consistent (Cheng and Basu 2017; Johnston and McIntyre 2019). Secondly, these results highlight that P:A data can be effectively used to identify patterns and probable drivers in wetland nutrient accumulation given a large enough dataset (Cheng and Basu 2017; Johnston and McIntyre 2019).). In recent literature P:A has also been used to understand how PPR wetlands and wetland distributions have changed over time (Van Meter and Basu 2015). Use of P:A ratio for investigations of ponds in the Canadian portion of the PPR has not been pursued to date.

Another way to understand wetland behaviour is through pond permanence. Stewart and Kantrud (1971) proposed a wetland classification with five main classes: ephemeral, temporary, seasonal, semi-permanent, and permanent ponds. Ephemeral ponds typically have very brief periods of surface water accumulation, occurring mostly in the spring. Temporary ponds maintain surface water for a little longer than ephemeral ponds, and have wet-meadow vegetation, with sedges and grasses. Seasonal ponds are those that typically dry out early in the summer; these ponds can disappear completely during periods of drought and can range from freshwater to moderately brackish. Semi-permanent ponds are those that dry out later in the summer and are characterized by deep marsh vegetation. Lastly, permanent ones are those that have open water all year. However, pond permanence can be a difficult physiographic grouping to identify as many anthropogenic activities—such as drainage or tilling of ephemeral to semi-permanent ponds, thereby removing characteristic vegetation—or natural variability may present challenges for classifying pond permanence. Nonetheless pond permanence class is one of the longest standing classification systems in the PPR, suggesting it could be a useful grouping for understanding potential nutrient drivers, as the aspects identified to classify ponds such as vegetation play and the oxidation of sediments that occurs with changing pond wetness also have an impact on nutrient accumulation (Reddy and DeLaune 2008b).

Pondwater specific conductance (SC) varies strongly across pothole ponds and was another classification grouping proposed by Stewart and Kantrud (1971) as a proxy for salinity. Specific conductance is used to classify ponds as either saline (>1000 µs cm$^{-1}$) or freshwater (<1000 µs cm$^{-1}$) (Arndt and Richardson 1989; Pennock et al. 2014). In the PPR, pondwater conductivity changes are initially controlled by $CaCO_3$, and $CaCO_3$ precipitation generally begins at 1000 µs cm$^{-1}$, therefore providing a useful threshold for the divisions between freshwater ponds and saline ponds (Pennock et al. 2014). This is of particular relevance from a nutrient retention perspective, as $CaCO_3$ is abundant in the PPR and influences the binding of specific nutrients (von Wandruszka 2006; Last and Last 2012). The interaction that $CaCO_3$ has with nutrients and the metric that the saline vs freshwater pond SC grouping provides may be a key tool in understanding the probable drivers of P accumulation in PPR wetlands.

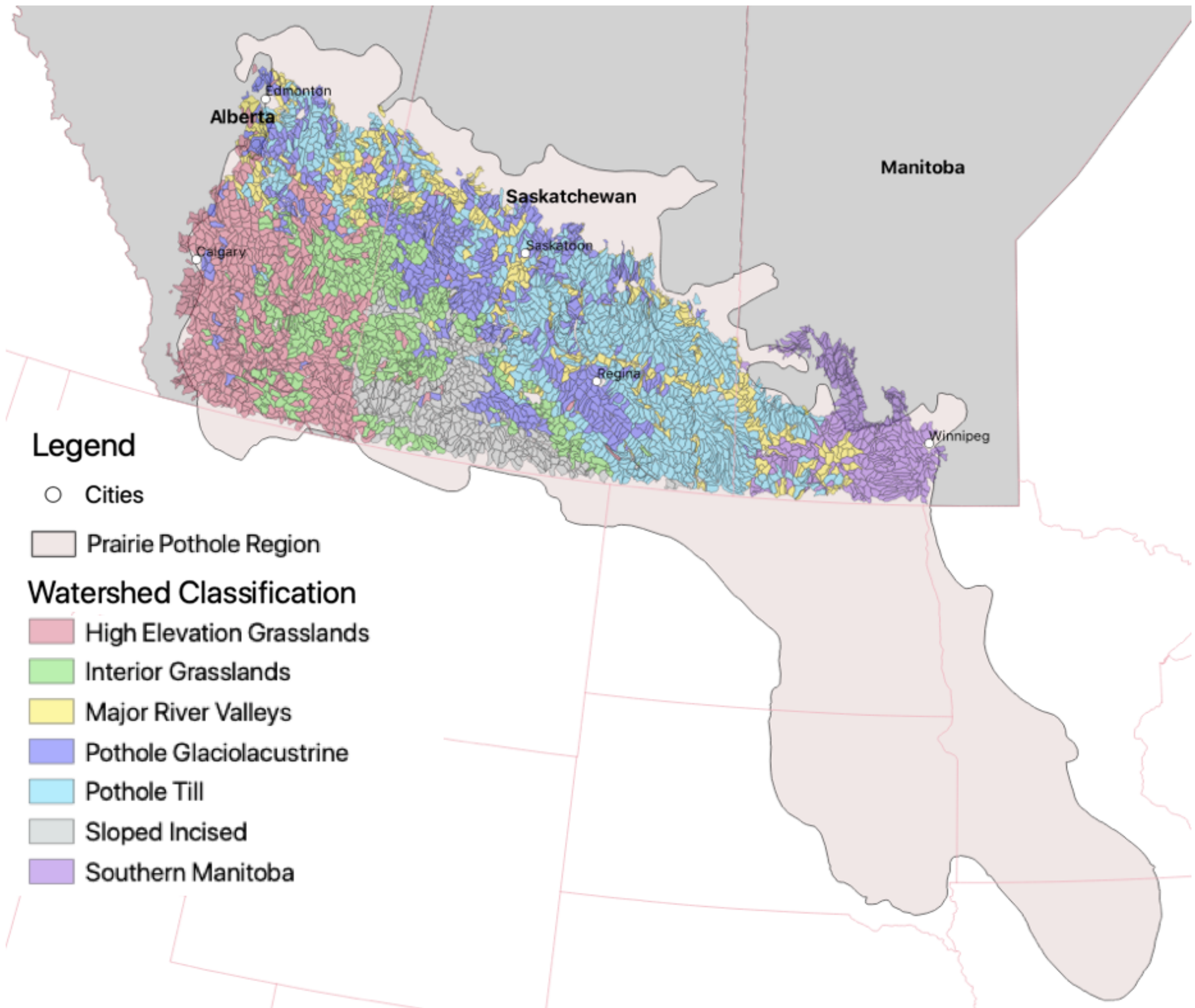


**Figure 1.3** Map of prairie watershed classifications (Wolfe at al. 2019), Prairie Pothole region extent (pink shaded area) and prairie provinces (dark grey area). See text above for details on select prairie watershed classifications. Watershed classification drawn from Wolfe at al. 2019, used with permission. PPR boundary and political boundary data drawn from North American Political Boundaries from U.S Geological Survey. Information licenced under the Department of Interior Copywrite, Restrictions, and Permissions https://www.doi.gov/copyright. Cities drawn from Statistics Canada, and with information licensed under the Open Government Licence-Canada http://open.canada.ca/en/open-government-licence-canada.

Watershed classes are another way of investigating physiographic factors in an integrative way. Small watersheds across the PPR have been classified based on land use, soil zone, climate, wetland density, topography (Wolfe et al. 2019). The seven watershed classes are:

Southern Manitoba, Pothole Till, Pothole Glaciolacustrine, Major River Valleys, Interior Grasslands, High Elevation Grasslands, and Sloped Incised (**Error! Reference source not found.**). For example, the Pothole Till watershed class has the greatest density of wetlands, highest non-contributing area, and lowest unmanaged grassland area (Wolfe et al. 2019). In contrast, the Southern Manitoba watershed classes have the greatest area under agricultural land use and lowest non-contributing area (Wolfe et al. 2019). Comparing these different watersheds highlights the variability in landscape properties that these watershed groupings are capable of capturing, rather than reporting each unique property independently. This may equate to functional differences among watershed classes, which could manifest in pothole pond nutrient patterns.

As we group ponds based on similar pond characteristics, watershed properties or adjacent land uses, we can begin to tease out the importance of the specific variabilities and find probable drivers in the patterns of PPR wetland nutrient accumulations. Understanding and unifying the key probable drivers of nutrient retention in pothole ponds is a vital step towards continued management of the PPR landscape.

### 1.2.5 PPR wetlands and phosphorus retention

Based on the previous sections, it is clear that PPR wetlands, their physiographic groupings, and P cycling processes share something in common—they're highly complex. Improved wetland management has been proposed as a partial solution to the problem of growing financial strain on those water treatment plants managing P-related issues (Marton et al. 2015). Wetlands can manage excess nutrients by acting as a buffer between upland and aquatic systems (Kleinman et al. 2015). However, the limited research so far has focused on the differences in P retention between intact and drained wetlands, with intact wetlands being more effective nutrient sinks than drained ones (Badiou et al. 2018; Haque et al. 2018b). Research has also compared ponds with different soil types (calcareous and non-calcareous) and while they have the same amounts of total P, soil test P (Kelowna extraction method) was greater in non-calcareous wetland soils (Brown et al. 2017a). While these results on PPR wetland P retention are exciting it is a limited body of research, that has really only focused on specific regions of the PPR (Broughton's Creek Watershed, Manitoba) or conducted with a limited number of sites ($n$ = 2). None of the PPR wetland research so far has accounted for the varying rates of P fertilizer application occurring in the landscape adjacent to the wetlands, focus has been centered on the wetlands themselves rather than surrounding landscape conditions. This is a difficult as fertilizer and P application rates vary both spatially and temporally. Research has yet to explore what P accumulation looks like across a range of intact wetlands, across the entire PPR.

Prairie wetlands can act as invaluable P sinks, but in the face of regional stressors, there is a need to identify which wetland-specific properties or physiographic properties make them effective P retention and transformation areas, as this can factor into efforts to mitigate wetland drainage, target restoration, and improve wetland management. In order to understand how we can best leverage PPR wetlands as nutrient storage units, we need to better understand the biological, physical and chemical processes that control transformation and accumulation of P in water, soil, and sediments. By investigating a number of different ponds across a number of different conditions we stand to learn a lot about what PPR wetlands have to offer as P sinks, and which specific properties—either physical or chemical—make some wetlands more effective at retaining P than others.

### 1.2.6 Conclusion and objectives

Given the complexity of the PPR landscape, and the potential dynamics within individual ponds, this research will seek to investigate potential drivers of pothole pond P considering both physicochemical and physiographic factors. The central objective of this research is to identify the physicochemical and physiographic drivers on pondwater P across a gradient of wetland conditions. Pulling from the literature review above, the physicochemical factors of interest include pondwater pH, pondwater conductivity, sediment clay content, sediment OC, sediment IC, soil clay content, soil OC, and soil IC. Similarly, watershed class, P:A, pond permanence, soil zone and land use are the physiographic features that will be explored as potential drivers for P in PPR wetlands. This research will answer the following questions:

1. Which physicochemical characteristics (e.g. water chemistry, sediment and soil properties) are drivers of wetland pond surface water P characteristics?
2. Which physiographic characteristics (e.g. P:A, pond permanence, watershed class, and land-use) are drivers of P concentrations in the surface water?

## 2.0: Materials and Methods

### 2.1 Study area

The Canadian portion of the PPR is approximately 520 000 $km^2$ in size and the dominant land use is agriculture. Air temperatures are generally regulated by latitude effect, rather than topography or large water bodies. Winters are cold and harsh (–9 to –18ºC in January), while summers are short and warm (26 to 14ºC in July) (Environment and Climate Change Canada 2020). The mean annual precipitation of the region varies, with annual precipitation between 300 to 550 mm generally decreasing from north and east to southwest (Millett et al. 2009).

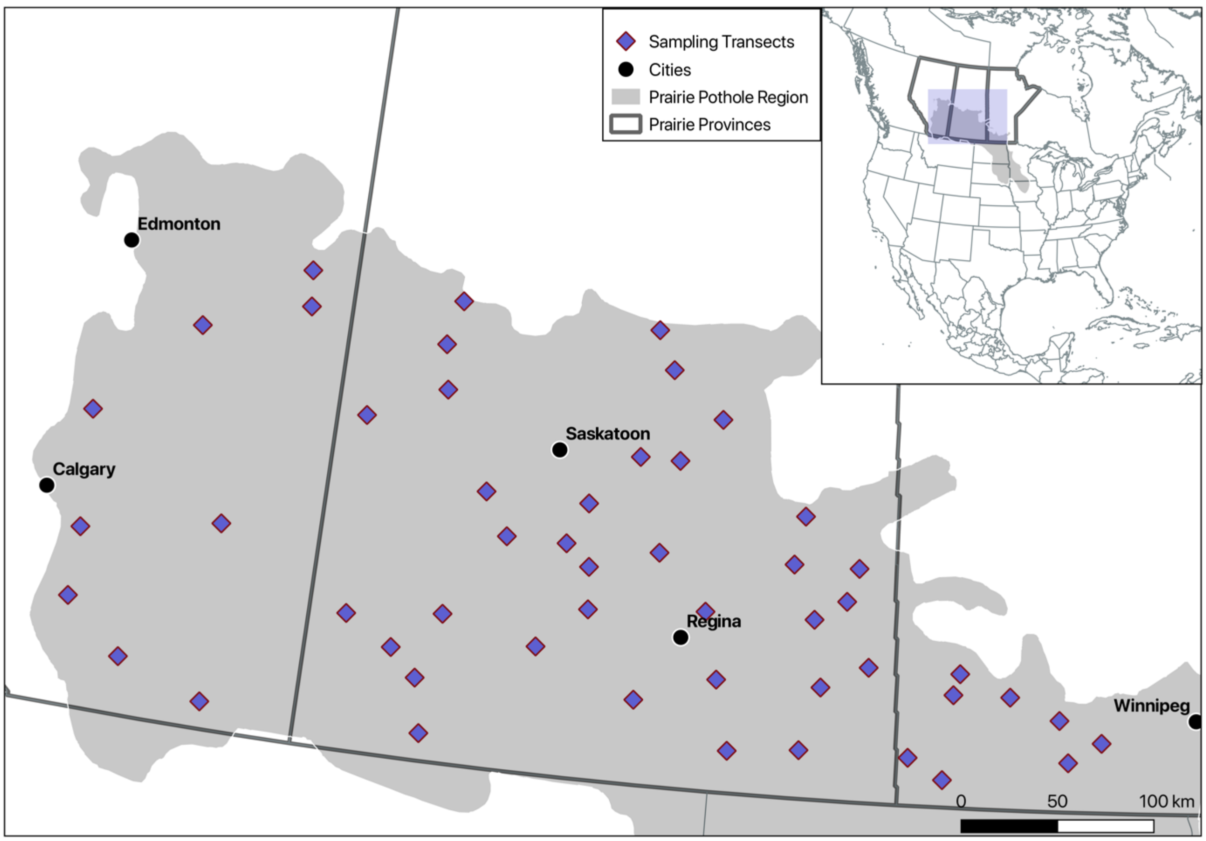


**Figure 2.1** Study area in western Canada (inset) with extent of Prairie Pothole Region (grey area), locations of sampling transects (purple diamonds), and major cities shown. Drawn with PPR boundary and political boundary data from North American Political Boundaries from U.S Geological Survey. Information licenced under the Department of Interior Copywrite, Restrictions, and Permissions https://www.doi.gov/copyright. Cities drawn from Statistics Canada, and with information licensed under the Open Government Licence-Canada http://open.canada.ca/en/open-government-licence-canada.

A total of 150 wetland ponds were sampled from 51 unique transects that are distributed across the PPR (Figure 2.) over an 11-day period (26 April–6 May 2019) during or shortly following snowmelt. In the majority of cases, three wetland ponds were sampled along each transect, with the ponds being within ~100 m of the nearest road. Wetlands were located within road access to allow for easiest and most efficient pond access given the limited sampling period. Most of the roads used to access the wetlands were gravel grid roads with little traffic. With few exceptions, the transects are located in areas of annual cropland.

**2.2 Field sampling**

The wetland sites were sampled for pondwater chemistry, sediments and pond-adjacent soil properties. Site observations were used to describe wetland buffer vegetation composition and coverage (%), estimated wetland basin fill (%), connection to other wetlands (Y/N), macroinvertebrates present (Y/N), seeded field (Y/N; Figure A.1) and pond permanence (seasonal, semi-permanent, or permanent). All other physiographic pond properties (soil zone, watershed class, P:A, and land-use type) were characterized using existing datasets, as described below (Section 2.4).

Water samples were collected from all 150 wetland ponds, and water temperature (°C), specific conductance which is also known as electrical conductivity (µS $cm^{-1}$), dissolved oxygen (mg $L^{-1}$), and pH values were measured in the field using a multiparameter handheld probe (YSI 600 XLM, Yellowstone Scientific Instruments). Pondwater samples were collected in areas with water depth >0.5 m (wherever possible) and care was taken to avoid disturbing the wetland sediments. Bulk pondwater samples were collected in acid-washed and triple-rinsed HDPE bottles from 10–20 cm below the pond surface. Samples were stored in coolers on ice (4ºC) during transport, and samples were processed daily by dividing water samples into four subsamples according to intended lab analyses: 1) raw unfiltered for pH, SC, and alkalinity values, 2) acidified ($H_2SO_4$) for total phosphorus (TP), 3) filtered (0.45 µm) and acidified ($H_2SO_4$) for dissolved phosphorus (DP), and 4) a subset of samples ($n$ = 59) that were returned to the laboratory within ~24 h of collection, which were syringe filtered (0.45 µm) and analysed for dissolved-molybdate-reactive P (DRP) immediately.

At 140 sites, sediment samples were collected in triplicate from the uppermost 10 cm using a polycarbonate tube. Samples were collected where the pondwater depth was <0.5 m. Prior to collection, sediment tubes were rinsed three times using pondwater. Efforts were made to collect sediment from areas free of vegetation wherever possible. Samples were composited in a plastic bag and stored cool (4ºC) for transport to the laboratory. At select sites fewer than three samples were collected, including 10 sites where no samples were collected, due to site conditions (e.g. compacted sediment).

At 148 sites, soil samples were collected from outside the pondwater perimeter but within the wetland catchment, on the mid-slope adjacent to the pond. Soil sampling location was determined using a decision tree to maintain consistency in sampling location across wetlands and among different surveyors (Figure A.2), and to avoid areas that may be seasonally inundated. Three soil samples (0–15 cm) were collected at 1-m intervals along a transect perpendicular to the shoreline using a Dutch Auger, and composited. GPS coordinates of the soil sampling location were recorded. Samples were not collected at two sites due to surveyor error. Composite soil samples were stored in plastic bags and kept cool for transport to the laboratory.

**2.3 Physicochemical water analyses**

The subset of water samples ($n$ = 59) that were returned to the laboratory within ~24 h of collection were analysed for dissolved molybdate-reactive P (DRP; EPA 365.1) using the SmartChem™ 170 discrete analyzer at the University of Saskatchewan (WESTCO Scientific Instruments, Inc. Brookfield, CT). Water samples for TP and DP analysis were digested ($(NH_4)_2S_2O_8$) in the lab using an autoclave, frozen, and stored in the dark. Samples were analyzed for TP and DP colorimetrically (ammonium-molybdate ascorbic-acid, method WP3D) using the SmartChem™ 170 discrete analyzer at the University of Saskatchewan. All P analyses were conducted in duplicate, with a detection limit of 1 μg $L^{-1}$. Particulate P (PP) concentrations were calculated as the difference between TP and DP. All remaining pondwater samples were stored at 4ºC in the dark prior to analysis of pH, SC, and alkalinity. Specific conductance (SC) (μS $cm^{-1}$) and pH values were analyzed using a multiparameter probe (6561 pH sensor, YSI 600 XLM, Yellowstone Scientific Instruments). Alkalinity analysis (mg $L^{-1}$ $CaCO_3$) was done on all the pondwater samples using the SmartChem™ 170 discrete analyzer (method ALK-001-A).

**2.4 Physicochemical soil and sediment analyses**

Soil and sediment samples were inventoried and stored at 4ºC in the dark immediately upon return from the field. The number of soil and sediments analyzed was different for individual analyses, because the collection of samples for some sites was not possible or because insufficient sample material was available. Prior to air-drying the soils and sediments, porewater P was analyzed in all sediment samples for which extraction yielded a sufficient volume of water ($n$ = 130). Samples were manually homogenized, and a 40-g subsample was placed in a 50-mL Falcon tube for centrifugation at 3800 RPM for 15 min. Pore-water supernatant was decanted, filtered (0.45 µm) and acidified ($H_2SO_4$) prior to analysis for DP (method WP3D; SmartChem 170 autoanalyzer). The soil ($n$ = 148) and sediment ($n$ = 140) samples were air-dried at room temperature, manually homogenized and then ground and passed through a 2-mm sieve prior to analysis for pH, SC values and $CaCl_2$-extractable P. Samples for particle size analysis were combusted at 400°C for 10 h in a muffle furnace. Finally, samples for TC and OC analyses were ground finely to a fine powder with a ball mill.

All samples were analyzed for pH and C. Soil pH and SC values were measured at a soil:deionized water mass ratio of 0.5 (1:2) (Hendershot et al. 2008; Miller and Curtin 2008). Sediment pH and SC was measured at a soil:deionized water mass ratio of 1:5, due to the high amount of OM (Hendershot et al. 2008; Miller and Curtin 2008). The $CaCl_2$-extractable P concentrations were determined using 0.01 M $CaCl_2$ extracts for all sediment and soil samples, followed by colorimetric analysis using a discrete analyzer at the University of Saskatchewan (Self-Davis et al. 2009). Particle size for all of the sediment samples ($n$ = 135) and a subset of the soil samples ($n$ = 40) was analyzed in triplicate via laser ablation (Horiba Particle Size Analyser LA-950 V2) in the Ecosystems Research Group Lab at Trent University to determine fractions of clay, silt, sand, and geometric mean particle size (Geomean). Prior to laser ablation, samples were soaked overnight in Calgon (sodium hexametaphosphate) dispersing agent (Levasseur et al. 2020). Total C was determined in all sediment and soil samples by loss-on-ignition; samples were ignited at 1350°C in a Ni lined ceramic boat, using a LECO-C632 C analyzer in the Soils Teaching Lab at the University of Saskatchewan (Skemstad and Baldock 2008). Organic C percentage was determined using the same procedure with the additional pre-treatment of samples with sulfuric acid ($H_2SO_4$) until reactions stopped (Skemstad and Baldock 2008). Inorganic C percentages were calculated as the difference between TC and OC.

### 2.5 Physiographic properties

Soil zone, watershed class, pond perimeter, P:A, and land-use type for each wetland were identified by examining publicly-available datasets in QGIS (http://www.qgis.org; version 3.12). Prairie soil zones were delineated at a resolution of 1:1,000,000 in accordance with the Soil

Landscapes of Canada V3.1 (AAFC 2013). Watershed classifications for the region were described by Wolfe et al. (2019). Sites that were not associated with a watershed class (Figure A.3) (e.g. those outside the Prairie Ecozone; *n* = 16) were characterized as being outside the classification. Watersheds were delineated according to the HydroSHEDs database (Lehner and Grill 2013). Soil zone and watershed classifications for a given wetland were confirmed by identifying the polygon that each wetland fell within (QGIS tool: zonal statistics). Pond area and pond perimeter were derived using two different datasets: CanVec and the Canadian Wetland Inventory. In CanVec, the hydrographic features dataset was used. This included watercourses, waterbodies, water wells and a number of other hydrological features at 1:50,000 resolution (NRC 2016). Wetland sampling sites were plotted using a 50 m buffer (tool: buffer) to identify corresponding CanVec hydrographic polygons. Of the 150 wetlands, only 76 had a CanVec hydrographic polygons that overlapped with the site. Manual verification was performed to ensure that each CanVec feature identified was intersecting with the appropriate wetland polygon. The Canadian Wetland Inventory (CWI) was used to supplement the number of identified wetlands for calculating wetland P:A, an additional 30 wetlands were identified. However, only parts of the PPR have been completed so far (DUC, 2021). Between the two datasets, a total of 106 wetlands had perimeter and area data available for calculating P:A. Where sampled wetlands had both CanVec and CWI physiographic data available (area, perimeter, and P:A; *n* = 25), data were averaged. Therefore, the pond physical properties data are combination of CanVec, CWI, and CanVec-CWI averages. For each of the wetlands, area and perimeter of the polygon were calculated (QGIS tool: calculate geometry) and used to determine P:A, with higher P:A values indicating more complex pond shapes.

Land use was determined according to the Annual Crop Inventory (ACI) for 2019 (AAFC 2019; 30 m resolution) in two ways. To determine land use adjacent to each wetland in each year, a 500-m buffer around each soil sampling site was used (tool: buffer), because smaller buffers (e.g. 100 m) were deemed too limited for use, as “wetland” was the most frequently occurring land-use type. Land use within the 500-m buffer for each year was characterized as the most frequently occurring land-use type (QGIS tool: zonal statistics). To get a better understanding of what has been happening to the landscape around the wetlands over a period of time, the crop rotation type (simple or diverse) from 5 years prior to sampling (2014–2019) was determined using these data. Simple crop rotation type was assigned to cropland sites that had both oilseed and cereal crops over the 5-year observation period. Diverse crop rotation type was assigned cropland sites that had pulse crops and either oilseed and cereal crops or as a combination of pulse crops, oilseed, and cereal crops growing over the 5-year observation period. The key aspect of the diverse crop rotation assignment is that pulse crops had to be cultivated on the cropland at least once over the 5-year observation period.

### 2.6 Data analysis and statistics

All statistical analyses were performed using R: A Language and Environment for Statistical Computing (R Core Team 2020, version 4.0.0). Across the analysis, an alpha value of 0.05 was used as the threshold for statistical significance and false discovery rate (FDR) correction was used to provide each test with a 95% confidence interval.

#### 2.6.1 Data analysis

Prior to statistical analysis, two of the datasets were grouped into classes for analysis: pond salinity groups and land-use classifications. Pond groupings according to SC were done two ways. The first was a two-class grouping, as either freshwater (<1000 µS $cm^{-1}$) or saline pond (>1000 µS $cm^{-1}$). This grouping was used because the precipitation of $CaCO_3$ occurs around 1000 µS $cm^{-1}$, making it an important threshold for exploring pondwater geochemistry (Arndt and Richardson 1989). The second proposed pond SC values classification had three classes: saline, brackish and freshwater. In this instance, ponds were grouped according to freshwater (<500 µS $cm^{-1}$), slightly brackish ponds (500–2000 µS $cm^{-1}$), and saline (>2000 µS $cm^{-1}$) ponds, according to Stewart and Kantrud (1971). This classification has been used to understand the hydrological processes and movements of pondwater across the PPR (Nachshon et al. 2013).

Land-use classification was done according to the 2019 ACI land-use classification (Table A.1).**Table A.*1*** Breakdown of assigned land use groups, taken from Annual Crop Inventory crop classification (used in analysis). Land use was classified into grassland/pasture and cropland. Grassland and pasture were grouped owing to potential uncertainty in these land-use classifications and the low number of grassland ($n$ = 14) and pasture ($n$ = 3) adjacent sites. Cropland data were further divided according to crop type (pulses, oilseed, and cereal), and used for additional analysis.

### 2.6.2 Statistical analysis

Before performing any statistical analysis, data were determined to be non-normal through histograms, quantile-quantile plots (R Core Team (2021) package: 'stats'; function [qqnorm]) and the Shapiro-Wilk test (R Core Team (2021) package: 'stats'; function [shapiro.test]). We also checked for interactions between physicochemical and physiographic conditions (site latitude, site longitude, pond permanence, salinity, soil zones, watershed classes, and land-use type) using Pearsons Chi-Squared test (R Core Team (2021) package: 'stats'; function [chisq.test]). When covariates were confounded (e.g. soil zones and cropland type) one of the physiographic conditions (cropland type) was removed from the analysis that follows below as it required additional analysis outside the scope of this project.

Correlations were used to test for relationships among pondwater P (TP, DP, DRP, and PP concentrations) and pond chemistry and physical properties (area, perimeter, and P:A). Correlation was tested using Spearman rank correlation (R Core Team (2021) package: 'stats'; function: [cor.test]). For correlations conducted with pond area, perimeter and P:A, one outlier was excluded (area: 5.9 $km^2$; perimeter: 20 km) as it had an area nearly six times greater than the next largest pond (area: 0.2 $km^2$; perimeter: 3.8 km, and this was not consistent with observations from the field. A different subset of samples ($n$ = 44) were used for correlations with DRP. For correlations and analysis conducted with DRP, five data points were excluded because concentrations for DRP were greater than TP; these data were also excluded from other tests involving DRP.

Kruskal-Wallace and Mann-Whitney tests were performed to identify differences in P pools (TP, DP, DRP, PP, porewater DP, soil and sediment $CaCl_2$-extractable P) across different physicochemical and physiographic (pond permanence, salinity, soil zones, watershed classes, and land-use type) conditions. Nonparametric analysis was used as efforts to transform the data were unsuccessful for many of the variables and also eliminated important outliers. For all pond groupings with three or more classes (permanence, soil zones, and watershed class), the Kruskal-Wallace test (R Core Team (2021) package: 'stats'; function: [kruskal.test]) was used. A post-hoc Wilcoxon test (R Core Team (2021) package: 'stats'; function: [pairwise.wilcox.test]) was subsequently carried out with the $p$-values FDR-corrected for a 95% confidence interval. When comparing pond salinity classes and land use with only two groups, a Mann-Whitney test (R Core Team (2021) package: 'stats'; function: [wilcox.test]) was used.

Lastly a generalized least squares regression (GLS) (Pinheiro et al. (2021) package: ‘nlme’; function: [gls]) was done to identify predictors of pondwater, sediment porewater, sediment and soil $CaCl_2$ extractable P. Generalized least squares regression was used as it is suitable for datasets that have evidence of heteroskedasticity, and allows for autocorrelation; both these conditions are present in these data. Other regression models would not have allowed us to accurately or confidently account for the natural variability that is shown within these data (Zuur et al. 2009).

Multiple covariates of P were considered for the following observed P pools: TP, DP, DRP, Porewater DP, $CaCl_2$-extractable sediment P, and $CaCl_2$-extractable soil P. Each of the modelled P properties had a different number of available observations (due to sampling challenges described above). Correlation analysis informed which covariates to consider as candidates for the models, while latitude and longitude were also explored as potential covariates. The number of covariates was eight for pondwater P (TP, DP, and DRP), nine for sediment porewater P and $CaCl_2$-extractable sediment P models, and eight covariates were also considered in the $CaCl_2$-extractable soil P models.

Covariates in each of models were tested for collinearity by ensuring that all variance inflation factors (vif) were less than 5 (Fox and Weisberg (2019) package: ‘car’; function [vif]). Alternate predictors were also used in the models because several predictors described similar factors. For example, salinity grouping and pondwater conductivity describe the same pond properties. When two or more of these predictors were included, the vif was exceeded and there were issues with collinearity. Therefore, only one (alternate) predictor variable was used in the model at a time, the model was run repeatedly with changing predictor variables. In the sediment P samples there were three sets of alternate predictors: C properties (soil zone, TC, OC, and IC), salinity (water conductivity and freshwater/saline pond grouping), and land-use properties (land use). Model analysis outlined above was also repeated with the P:A data due to differences in the sample sizes when P:A data were included (with P:A data: $n = 106$, without P:A data: $n = 149$; Table A.6).

An information-theoretic approach (Akaike Information Criterion corrected for small sample size; $AIC_c$) was used to select the best model for each of the P pools with the exception of PP (Mazerolle (2021) package: ‘AICcmodavg’; function [gls]). The model with the lowest AICc is deemed best (Burnham and Anderson 2004). Models with $AIC_c$ values $\leq 2$ and $<4$ are considered well-supported and plausible (Burnham and Anderson 2004). Model selection was done using maximum likelihood estimation (Pinheiro et al. (2021) package: ‘nlme’; function [gls], method = "ML"), but parameter estimates were calculated using restricted maximum likelihood estimation (Pinheiro et al. (2021) package: ‘nlme’; function [gls], method = "REML") Restricted maximum likelihood estimations (REML) are used to provide accurate parameter estimates that are unbiased compared to maximum likelihood (ML) estimates (Zuur et al. 2009). The $\beta \pm SE$ of the best models are reported unless otherwise stated.

## 3.0: Results

### 3.1 General summary of properties

Pondwater properties ranged considerably across the sites. Specific conductance varied from 90–6770 µS $cm^{-1}$ (Table 3.1). The pH values of ponds were generally slightly basic, ranging from 7.3 to 8.8. The average pondwater alkalinity ($CaCO_3$) was 229 mg $L^{-1}$ and ranged considerably (28–952 mg $L^{-1}$). Pondwater temperatures also varied (1.6–6.7 °C) during the sampling days.

Generally, $CaCl_2$-extractable P, Clay (%), Geomean (µm), SC, and pH were greater in soils than in sediments (Table 3.1). The soil SC values also had a considerably wider range (66–7050 µS $cm^{-1}$) than pond sediments (54–4430 µS $cm^{-1}$). Only mean TC and OC were greater in the sediments than the soils (Table 3.1).

The sampled ponds were classified in several physiographic (pond permanence, soil zones, watershed classes, and land-use type,) and physicochemical (SC) groupings (Table 3.2). The wide variability in the pondwater, soil, and sediment properties across the study sites was also explored for the different physiographic and physicochemical groups. The sections below explore some of the variability, first looking at the range in pondwater, soil, and sediment properties between the physiographic and physicochemical groups and secondly focussing on P.

**Table 3.1** Mean wetland pondwater, sediment and soil physicochemical characteristics. Values shown are mean with standard deviations shown in parentheses.

| Sample From | Variable | n | | Value |
|---|---|---|---|---|
| Pondwater | Temperature | 145 | °C | 8.4 (3.3) |
| | pH | 149 | | 8.1 (0.59) |
| | Specific conductance | 149 | $\mu S\ cm^{-1}$ | 1088 (1053) |
| | Alkalinity ($CaCO_3$) | 149 | $mg\ L^{-1}$ | 229 (144) |
| Pond Physical Properties | Pond perimeter (P) | 105 | m | 713 (614) |
| | Pond area (A) | 105 | $m^2$ | 272 (365) |
| | P:A | 105 | | 0.047 (0.029) |
| Sediment | Specific conductance | 127 | $\mu S\ cm^{-1}$ | 675 (677) |
| | pH | 130 | | 6.7 (1.1) |
| | $CaCl_2$-extractable P | 130 | $mg\ P\ kg^{-1}$ | 3.0 (0.51) |
| | Total Carbon | 138 | % | 8.3 (5.5) |
| | Organic Carbon | 133 | % | 8.9 (6.1) |
| | Inorganic Carbon | 132 | % | 0.2 (0.4) |
| | Clay | 135 | % | 3.7 (1.3) |
| | Geomean | 135 | μm | 30 (14) |
| Soil | Specific conductance | 144 | $\mu S\ cm^{-1}$ | 1674 (1832) |
| | pH | 146 | | 7.1 (0.6) |
| | $CaCl_2$-extractable P | 145 | $mg\ P\ kg^{-1}$ | 3.4 (1.13) |
| | Total Carbon | 144 | % | 3.7 (1.8) |
| | Organic Carbon | 145 | % | 3.6 (1.8) |
| | Inorganic Carbon | 142 | % | 0.2 (0.5) |
| | Clay | 40 | % | 5.6 (3.1) |
| | Geomean | 40 | μm | 31 (15) |

**Table 3.2** Summary of the physicochemical (pondwater SC values groups) and physiographic pondwater groupings (permanence, soil zones, watershed classes, and land use) assigned to the sampled wetlands ($n$ = 150).

| Pondwater SC* | n | Permanence | n | Soil Zones | n | Watershed Class | n | Land use** | n |
|---|---|---|---|---|---|---|---|---|---|
| Freshwater (<1000 µS $cm^{-1}$) | 91 | Seasonal | 26 | Black | 68 | High Elevation Grasslands | 13 | Cropland | 133 |
| Saline (>1000 µS $cm^{-1}$) | 58 | Semi-Permanent | 41 | Dark Brown | 47 | Pothole Glaciolacustrine | 21 | Grassland/pasture | 17 |
| | | Permanent | 83 | Brown | 23 | Major River Valleys | 7 | | |
| | | | | Grey | 12 | Interior Grassland | 20 | | |
| | | | | | | Pothole Till | 67 | | |
| | | | | | | Sloped Incised | 2 | | |
| | | | | | | Southern Manitoba | 4 | | |
| | | | | | | Outside Watershed Classes | 16 | | |

*Pondwater properties unavailable for one pond, therefore $n$ = 149.

** Land use in the year of sampling (2019). Detailed breakdown of agricultural land-use type classification available in Table A.1.

### 3.1.1 Pondwater properties summary

Of the pondwater properties analysed (alkalinity, SC, and pondwater pH), only alkalinity and pondwater pH had significant differences among the physiographic and physicochemical groups (Table A.2). Pondwater pH (Kruskal-Wallis test $p = 0.004$) was significantly lower in Black soil zones than all of the other soil zones (Black < Brown, Wilcoxon test $p = 0.047$; Black < Dark Brown, Wilcoxon test $p = 0.019$; Black < Grey, Wilcoxon test $p = 0.043$). Pondwater pH (Kruskal-Wallis test $p = 0.001$) was significantly greater in permanent ponds than in other pond types (permanent > seasonal, Wilcoxon test $p = 0.003$; permanent > semi-permanent, Wilcoxon test $p = 0.001$). Lastly, there were differences in pondwater pH between the watershed classes (Kruskal-Wallis test $p = 0.001$). Pondwater pH of ponds outside of the watershed classification was significantly greater than the pH of ponds in the Major River Valleys (Wilcoxon test $p = 0.016$) and Pothole Till (Wilcoxon test $p = 0.011$) watershed classes. The pH of the ponds located in the Interior Grasslands was also significantly greater than the pH of ponds in the Major River Valleys (Wilcoxon test $p = 0.037$). Pondwater alkalinity (Mann-Whitney U test $p = 0.001$) was significant greater in the saline ponds than in the freshwater ponds. Pondwater alkalinity (Kruskal-Wallis test $p = 0.032$) was also different between the different crop types. The alkalinity of ponds adjacent to oilseed crops was greater than the alkalinity of ponds located adjacent to cereal crops (Wilcoxon test $p = 0.038$). Mean pondwater SC values were similar among the physiographic and physicochemical groupings (Table A.2).

### 3.1.2 Sediment properties summary

Of all the sediment properties analysed (OC, IC, SC, and pH) only OC varied significantly among physiographic groups (Table A.3). Sediment OC in the soil zones had significant differences (Kruskal-Wallis test $p = 0.024$). The OC in the Black soil zone were greater than the Dark Brown soil zone (Wilcoxon test $p = 0.014$). Among the watershed classes (Kruskal-Wallis test $p = 0.002$), the Pothole Glaciolacustrine watershed OC was greater than either the High Elevation grassland, (Wilcoxon test $p = 0.038$) or the Pothole Till watershed (Wilcoxon test $p = 0.038$), whereas the Pothole Till watershed had a greater OC than outside the watershed classification (Wilcoxon test $p = 0.027$). Mean IC, sediment SC and sediment pH values did not differ among the physiographic and physicochemical groupings (Table A.3).

### 3.1.3 Soil properties summary

Many significant differences were shown in the soil properties (OC, IC, Soil SC, and Soil pH). Soil OC in the soil zones (Kruskal-Wallis test $p = 0.005$) and watershed classes (Kruskal-Wallis test $p = 0.001$) were significantly different (Table A.4). Soil OC in the Black and Grey soil zones were both significantly greater than the Dark Brown (Black: Wilcoxon test $p = 0.001$; Grey: Wilcoxon test $p = 0.049$) and Brown (Black: Wilcoxon test $p = 0.001$; Grey: Wilcoxon test $p = 0.028$) soil zones. Among the watershed classes, OC of Pothole Till soils were significantly greater than OC of the Interior Grassland (Wilcoxon test $p = 0.048$). Only the watershed classes (Kruskal-Wallis test $p = 0.005$) had significant IC differences (Table A.4). The soil IC analysis by watershed class revealed that the Pothole till watershed had a significantly greater IC concentration than the Interior Grassland watershed (Wilcoxon test $p = 0.032$). When looking at soil SC and soil pH, values differed significantly between land-use type (SC: Mann-Whitney U test $p = 0.005$; pH: Mann-Whitney U test $p = 0.004$). For both soil SC and soil pH, cropland values (SC: 1824 $\mu S\ cm^{-1}$; pH: 7.1) were greater than those for grassland/pasture (SC: 473 $\mu S\ cm^{-1}$; pH: 6.5).

### 3.1.4 Summary of physiographic properties among physiographic and physicochemical groupings

When investigating interactions among physiographic conditions, the crop rotation type strongly interacted with the soil zone data or rather crop rotation types differed by soil zone (Pearsons Chi-Squared test $p = 0.001$). Given this interaction, crop rotation could not be considered as independent and was removed from further analyses. When investigating the differences in P:A across the different physiographic groupings, the only significant difference in P:A was shown among soil zones (Kruskal-Wallis test $p = 0.002$) and watershed class (Kruskal-Wallis test $p = 0.001$) groups (Table A.5). The P:A was significantly greater in Black soil zones than all of the other soil zones (Black > Brown, Wilcoxon test $p = 0.012$; Black > Dark Brown, Wilcoxon test $p = 0.012$; Black >Grey, Wilcoxon test $p = 0.012$). Among watershed classes, Pothole Till P:A was significantly greater than the P:A of the ponds Outside the Watershed classes (Wilcoxon test $p = 0.021$). All other P:A data were comparable (Table A.5).

### 3.2 The relationships of phosphorus pools with pondwater, soil and sediment properties

The different P metrics varied across sites (Table 3.3). Total pondwater P, DP, and DRP were significantly and positively correlated with pondwater alkalinity and specific conductance values (Table 3.4) but these relationships were weak. Likewise, $CaCl_2$-extractable sediment P was positively correlated with TP, DP, and Porewater DP concentrations (Table 3.4). With porewater DP, sediment P, and soil P concentrations, each was correlated with at least one type of C (TC, OC, and IC; Table 3.5). Clay was negatively correlated with both sediment and soil $CaCl_2$-extractable P concentrations (Table 3.5).

**Table 3.3** Pondwater total phosphorus (TP), dissolved phosphorus (DP), dissolved reactive phosphorus (DRP), particulate phosphorus (PP), and porewater dissolved phosphorus, $CaCl_2$-extractable sediment and $CaCl_2$-extractable soil phosphorus pools for sampled wetlands. Values shown are mean with standard deviations shown in parentheses.

| Parameter | Units | n | Mean |
|---|---|---|---|
| Pondwater TP | mg $L^{-1}$ | 150 | 0.67 (0.61) |
| Pondwater DP | mg $L^{-1}$ | 150 | 0.56 (0.63) |
| Pondwater DRP | mg $L^{-1}$ | 54 | 0.49 (0.64) |
| Pondwater PP | mg $L^{-1}$ | 150 | 0.11 (0.23) |
| Porewater DP | mg $L^{-1}$ | 130 | 0.39 (0.90) |
| $CaCl_2$ -extractable sediment P | mg P $kg^{-1}$ | 130 | 3.01 (0.51) |
| $CaCl_2$-extractable soil P | mg P $kg^{-1}$ | 145 | 3.44 (1.13) |

**Table 3.4** Spearman rank correlation coefficients among concentrations of total phosphorus (TP), dissolved phosphorus (DP), dissolved reactive phosphorus (DRP), particulate phosphorus (PP), and porewater dissolved phosphorus in sampled pond surface water with specific conductance (SC), pH, and alkalinity values and concentrations of $CaCl_2$-extractable sediment and soil P.

| Variable | Sample Type | Pondwater TP | Pondwater DP | Pondwater DRP | Pondwater PP | Porewater DP |
|---|---|---|---|---|---|---|
| SC (µS $cm^{-1}$) | Pondwater | **0.16**$^{*}$ | **0.23**$^{+}$ | **0.41**$^{+}$ | –0.06 | **0.19**$^{*}$ |
| pH | | 0.05 | 0.05 | –0.08 | 0.06 | 0.16 |
| Alkalinity | | **0.24**$^{+}$ | **0.28**$^{+}$ | **0.40**$^{+}$ | 0.02 | **0.20**$^{*}$ |
| $CaCl_2$-extractable sediment P | P Pools | **0.20**$^{*}$ | **0.18**$^{*}$ | 0.15 | –0.01 | **0.33**$^{+}$ |
| $CaCl_2$-extractable soil P | | 0.08 | 0.04 | –0.13 | 0.13 | 0.04 |

Correlations are denoted by $^{*}$($p \leq 0.05$) or a $^{+}$ ($p \leq 0.01$) (with *p*-values FDR corrected).

**Table 3.5** Spearman rank correlation coefficients among concentrations of porewater DP, $CaCl_2$ extractable sediment, and soil phosphorus in sampled wetland sediment and soil with total carbon (TC), organic carbon (OC), inorganic carbon (IC) concentrations and clay content (%).

| Variable | Sample Type | Porewater DP | $CaCl_2$-extractable sediment P | $CaCl_2$-extractable soil P |
|---|---|---|---|---|
| SC (µS $cm^{-1}$) | Sediment | 0.11 | 0.01 | N/A |
| pH | Sediment | 0.01 | **–0.26**$^{+}$ | N/A |
| TC (%) | Sediment | **0.18**$^{*}$ | **0.45**$^{+}$ | N/A |
| OC (%) | Sediment | **0.13**$^{*}$ | **0.44**$^{+}$ | N/A |
| IC (%) | Sediment | 0.01 | **–0.24**$^{+}$ | N/A |
| Clay (%) | Sediment | **–0.11**$^{*}$ | **–0.21**$^{*}$ | N/A |
| SC (µS $cm^{-1}$) | Soil | 0.17 | N/A | 0.06 |
| pH values | Soil | 0.04 | N/A | **–0.19**$^{*}$ |
| TC (%) | Soil | –0.06 | N/A | 0.07 |
| OC (%) | Soil | –0.08 | N/A | **0.17**$^{*}$ |
| IC (%) | Soil | –0.09 | N/A | –0.16 |
| Clay (%) | Soil | –0.28 | N/A | **–0.32**$^{*}$ |

Correlations are denoted by $^{*}$($p \leq 0.05$) or a $^{+}$ ($p \leq 0.01$) (with *p*-values FDR corrected).

### 3.2.1 Pond physiographic groupings and properties

Between the physiographic groups (seasonal, semi-permanent, and permanent and saline vs. freshwater), only DRP was significantly different, with higher concentrations in saline ponds (Mann-Whitney U test $p = 0.002$). None of the mean concentrations of the other P pools varied between the seasonal, semi-permanent, and permanent ponds (Figure A.5) or the saline and freshwater ponds (Figure A.6). Of the ponds where pond area, perimeter and P:A were available only $CaCl_2$-extractable sediment P was positively correlated with P:A physiographic properties (Table 3.6).

**Table 3.6** Spearman rank correlation coefficients among concentrations of total phosphorus (TP), dissolved phosphorus (DP), dissolved reactive phosphorus (DRP), particulate P (PP), porewater DP, $CaCl_2$-extractable sediment, and soil P in pond P:A

| Variables | Pondwater TP | Pondwater DP | Pondwater DRP | Pondwater PP | Porewater DP | $CaCl_2$-extractable sediment P | $CaCl_2$-extractable soil P |
|---|---|---|---|---|---|---|---|
| | | | mg $L^{-1}$ | | | mg P $kg^{-1}$ | mg P $kg^{-1}$ |
| Area ($m^2$) | –0.07 | –0.08 | –0.13 | 0.04 | –0.04 | **–0.28**$^{+}$ | 0.23 |
| Perimeter (m) | –0.08 | –0.10 | –0.13 | 0.06 | –0.11 | **–0.30**$^{+}$ | 0.02 |
| P:A | 0.02 | 0.04 | 0.10 | –0.03 | –0.08 | **0.22**$^{*}$ | –0.01 |

Correlations are denoted by $^{*}$ ($p \leq 0.05$) or a $^{+}$ ($p \leq 0.01$) (with *p*-values FDR corrected), with major outlier removed.

### 3.2.2 Soil zones

Among soil zones, only porewater DP pools were significantly different (Figure 2; porewater DP: Kruskal-Wallis test $p = 0.047$). Concentrations of total P, DP, PP, and sediment and soil $CaCl_2$-extractable P (not shown) were not different among the soil zones. Dissolved reactive P could not be compared because there were no DRP samples collected at sites in the Brown soil zone.

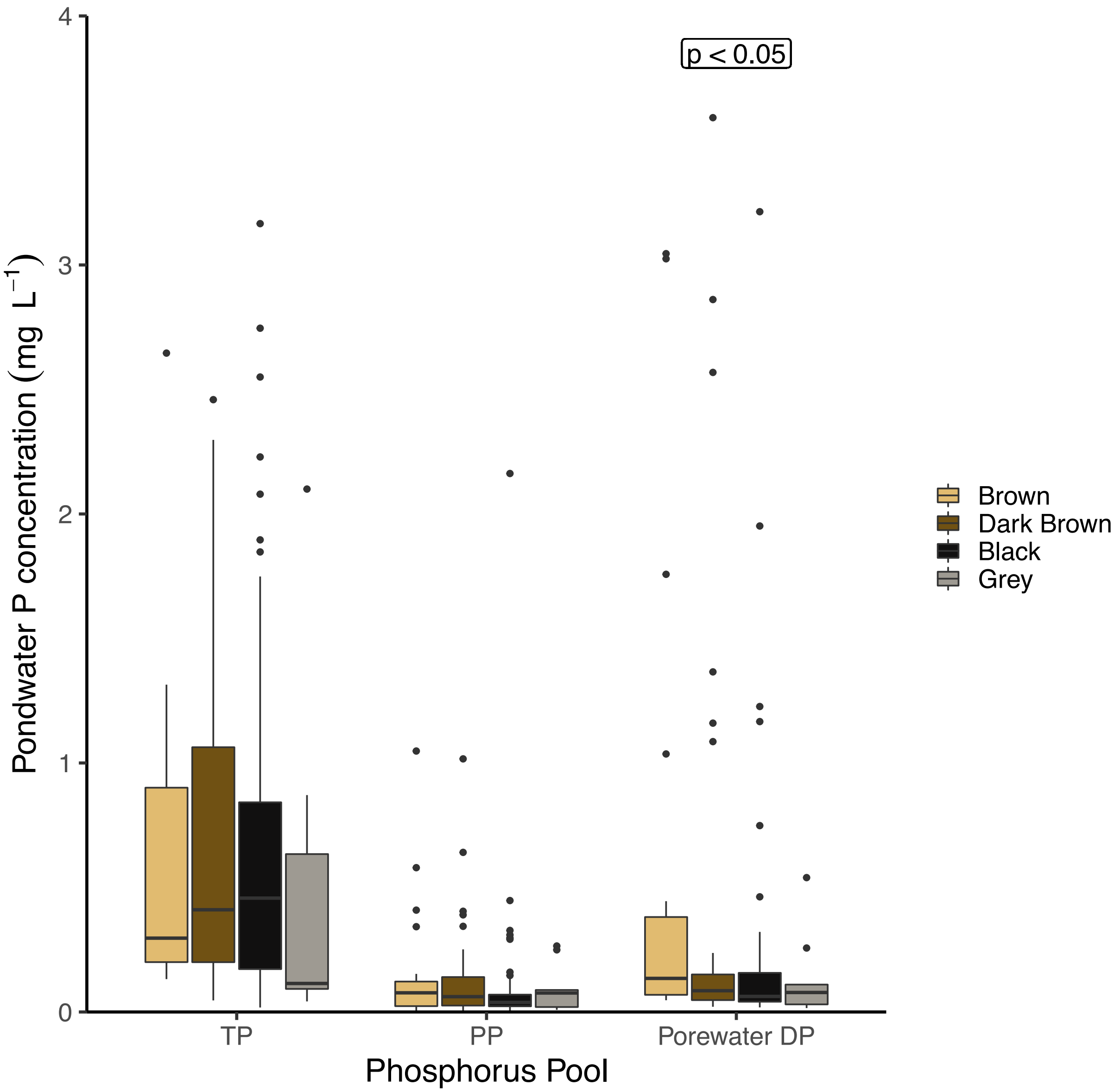


**Figure 2.1** Boxplot of concentrations of P in various P pools (Pondwater TP, PP, and porewater DP between the soil zones (Brown, Dark Brown, Black and Grey; p-values reported from a Kruskal-Wallace test, and FDR corrected). The boxplot displays data distribution (median, two hinges [$25^{th}$ and $75^{th}$ percentiles], whiskers [max and min range], and outlying points).

### 3.2.3 Watershed classes

Particulate P was the only P pool that had significant differences among watershed classes (Figure 3.2; Kruskal-Wallis test $p = 0.005$). Mean PP in ponds of Pothole Glaciolacustrine watershed (PP: 0.21 mg $L^{-1}$) was twice that of ponds in Pothole Till watershed (PP: 0.10 mg $L^{-1}$; Wilcoxon test $p = 0.003$). Concentrations of the other P pools (TP, DP, sediment porewater DP, $CaCl_2$-extractable soil, and sediment P) were not different among watershed classes.

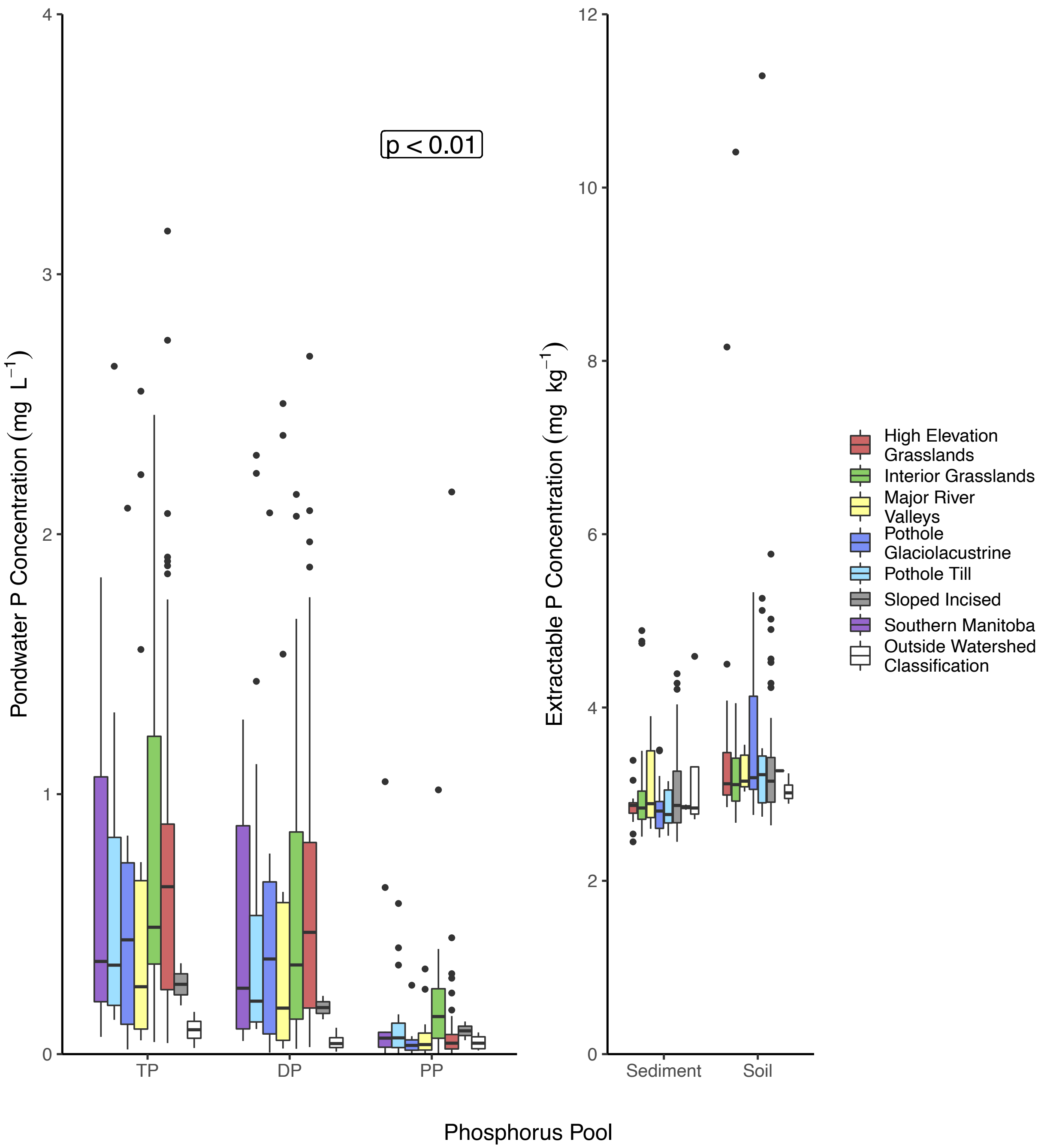


**Figure 3.2** Boxplot of comparison of concentrations of P in various P pools (Pondwater TP, DP, PP, sediment and soil $CaCl_2$-extractable P) across the watershed classes (High Elevation Grasslands, Interior Grasslands, Major River Valleys, Pothole Glaciolacustrine, Pothole Till, Sloped Incised, Southern Manitoba, and Outside Watershed Classification) (*p*-values reported from a Kruskal-Wallace test, and FDR corrected). Boxplots displays data distribution (median, hinges [25$^{th}$ and 75$^{th}$ percentiles], whiskers [max and min range], and outlying points).

### 3.2.4 Land use

Land-use type appears to have some influence on pondwater P concentrations (Figure 3). When comparing P pools (Figure 3), some of the P concentrations were greater in the cropland (oilseed, pulses, and cereal), than the grassland/pasture sites. The TP in ponds adjacent to cropland (0.71 mg $L^{-1}$) was more than three times greater than in grassland/pasture-adjacent ponds (0.32 mg $L^{-1}$; Mann-Whitney U test $p = 0.006$). Dissolved P (0.60 mg $L^{-1}$) in cropland sites was nearly three times greater than in grassland/pasture-adjacent sites (0.23 mg $L^{-1}$; Mann-Whitney U test $p = 0.007$). Porewater DP was not different between cropland and grassland/pasture-adjacent sites (cropland: 0.39 mg $L^{-1}$, grassland/pasture: 0.43 mg $L^{-1}$). Dissolved reactive P could not be compared because there were no DRP samples collected at grassland/pasture-adjacent sites.

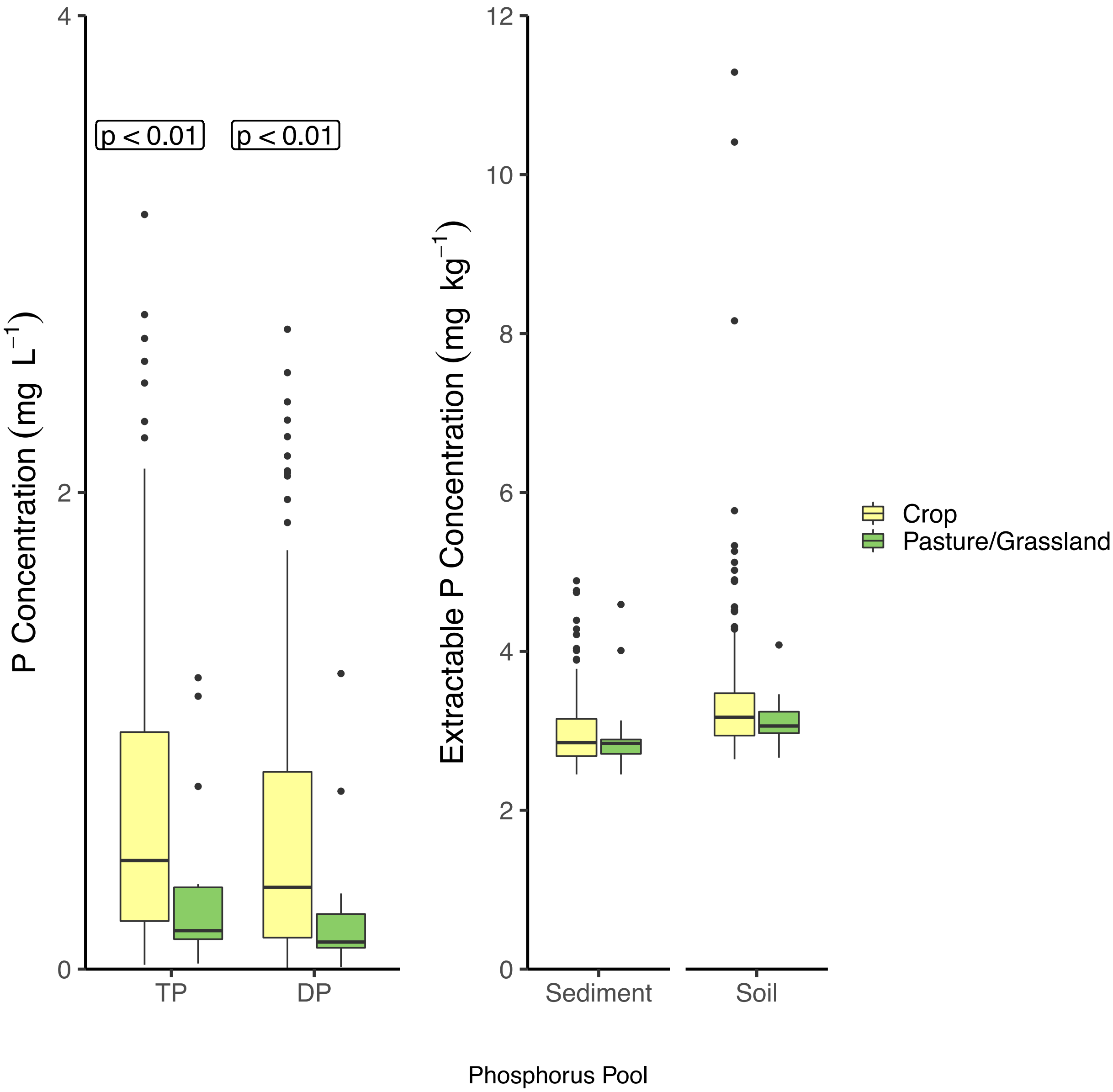


**Figure 3.3** Boxplot comparison between two land-use types; cropland ($n$ = 131) and grassland/pasture ($n$ = 17) for concentrations of P in various P pools (TP, DP, sediment and soil $CaCl_2$-extractable P; p-values reported from Mann-Whitney test, and FDR corrected). Boxplots displays data distribution (median, hinges [25$^{th}$ and 75$^{th}$ percentiles], whiskers [max and min range], and outlying points).

### 3.3 Modelling of possible phosphorus drivers in PPR wetlands

Having identified differences in P pools across various physiographic and physicochemical features, we used a model to synthesize this information (six predictive models total). Probable drivers differed among the models, but land use was the most frequently occurring predictor in the models (Table 3.7). Probable drivers of pondwater P consistently included alkalinity and land use (cropland compared to grassland/pasture; Table 3.7). Probable drivers of the sediment porewater DP were a combination of physical sediment properties and pondwater chemistry (Table 3.7). Model analysis was also done using a different dataset that included P:A data; however, none of the top models included P:A data and that model is therefore not shown (Table A.6).

**Table 3.7** Ranking of models and parameter estimates explaining variation of pondwater, sediment, and soil P concentrations of sampled wetlands (β(SE): beta of parameter estimates with standard error, $AIC_c$: Akaike's information criterion corrected for small sample size, *Δ*AIC: the difference between the best model and the other models in the dataset, K: the number of model parameters)

| P pools | | | β(SE) | $AIC_c$ | ΔAIC | K |
|---|---|---|---|---|---|---|
| Pondwater TP | Parameter | Alkalinity | 0.000939 (0.000411) | – | – | – |
| | Parameter | Land use–Grassland & Pasture | –0.304 (0.116) | – | – | – |
| | Model | Alkalinity, Land use | – | 281.6 | 9.2 | **5** |
| | | Intercept & model structure only (statistical null) | – | 290.8 | – | **3** |
| Pondwater DP** | Parameter | Alkalinity | 0.00104 (0.000387) | – | – | – |
| | Parameter | Land use–Grassland & Pasture | –0.282 (0.0845) | – | – | – |
| | Model | Alkalinity, Land use | – | 252.6 | 17.1 | **5** |
| | | Intercept & model structure only (statistical null) | – | 269.7 | – | **3** |
| Pondwater DRP | Parameter | Alkalinity | 0.00194 (0.000615) | – | – | – |
| | Model | Alkalinity | – | 83.2 | 4.6 | **4** |
| | | Intercept & model structure only (statistical null) | – | 87.7 | – | **3** |
| Porewater DP | Parameter | Sediment pH | –0.0342 (0.0187) | – | – | – |
| | Parameter | Clay (%) | –0.0248 (0.0159) | – | – | – |
| | Model | Sediment pH, Clay (%) | – | 95.5 | 2.7 | **5** |
| | | Intercept & model structure only (statistical null) | – | 98.2 | – | **3** |
| $CaCl_2$-Extractable sediment P | Parameter | Organic Carbon (%) | 0.0383 (0.00894) | – | – | – |
| | Parameter | Clay (%) | 0.0432 (0.0228) | – | – | – |
| | Model | Organic Carbon (%), Clay (%) | – | 75.4 | 12.3 | **5** |
| | | Intercept & model structure only (statistical null) | – | 87.7 | – | **3** |
| $CaCl_2$-Extractable soil P | Parameter | Land use–Grassland & Pasture | –0.449 (0.144) | – | – | – |
| | Model | Land use–Grassland & Pasture | – | 346.9 | 5.6 | **4** |
| | | Intercept & model structure only (statistical null) | – | 352.5 | – | **3** |

**Plot of actual vs predicted model available in Appendix A-Figure A.4

## 4.0: Discussion

Among the potential drivers of wetland P explored, pondwater alkalinity and land use were the most important. Pondwater alkalinity was positively correlated with TP concentrations in wetland pondwater. Alkalinity also commonly emerged as a predictor of pondwater P concentrations in the statistical models (Table 3.4, Table 3.5, Table 3.7). Pondwater TP and DP concentrations were greater in wetlands adjacent to cropland than wetlands adjacent to grasslands or pasture (Figure 3, Table 3.7). As the overall objective of this research was to identify the drivers of P concentrations across a gradient of wetland conditions, the following section will explore the role of select physicochemical (pondwater SC and alkalinity, sediment IC, OC, pH, and clay, soil IC, OC, pH, and clay), and physiographic (P:A, permanence, watershed class, and land-use) characteristics in greater detail. This section discusses the implications of their role—or lack thereof—in wetland P accumulation, and the importance of the results in the context of wetland removal, restoration, and retention across the Prairies.

### 4.1 Important physicochemical and physiographic properties

#### 4.1.1 Salinity/Specific conductance

Like so much previous wetland research (Stewart and Kantrud 1971; van der Kamp and Hayashi 2009; Nachshon et al. 2014), SC was identified as a useful metric for grouping wetland function, providing important insight into PPR wetland variability and P retention potential. The results re-affirm this usefulness as pondwater SC positively correlated with several of the pondwater P pools (TP, DP, and Porewater DP). Pondwater DRP concentrations were also significantly greater in saline wetlands than in freshwater wetlands (Table 3.4, Figure A.6).

Total P and DP concentrations also increase with increasing pondwater conductivity. The reason for increases in TP with increasing salinity is largely due to the fact that salinity is an aggregate measure of the concentration of ions (LaBaugh 1989; Last and Last 2012). In general, a greater concentration of ions results in a greater number of sites for P binding (Reddy et al. 2005). Rather than available P being taken up by plants or other microorganisms, it is complexed with ions—specifically $SO_4^{2-}$, $Fe^{3+/2+}$, $Al^{3+/2+}$, $Mg^{2+}$, $K^{+}$ or $Ca^{2+}$— that make up salinity measurements (Reddy et al. 1999).

Nachshon et al. (2013) showed that understanding the ions which make up salinity measurements in the PPR is important. The ionic composition of wetland pondwater is highly variable; in some regions $Na^+$, $K^+$, and $SO_4^{2-}$ are the major ions while in others $Mg^{2+}$ and $Cl^-$ dominate (LaBaugh 1989). The salts in PPR wetlands are generally in the form of $SO_4^{2-}$ (Nachshon et al. 2013). The identification of $SO_4^{2-}$ as the dominant anion in PPR ponds is important, as water bodies with high concentrations of $SO_4^{2-}$ have greater rates of sediment P release, which contributes to greater P concentrations in the pondwater (Jensen et al. 2009). In anaerobic environments, the reduction of $SO_4^{2-}$ by microbial activity means that ferrous sulfides may be forming before $Fe\text{-}PO_4$ compounds, and therefore the buffering capacity of Fe to retain $PO_4$ is diminished (Lamers et al. 1998; Hoffmann et al. 2009). This can ultimately result in greater inorganic P or available P in the wetland environment (Jensen et al. 2009). Other ions like $Na^+$, $Mg^{2+}$, and $Cl^-$ have some degree of regional variance (Last and Last 2012). The inextricable tie of salinity with P accumulation though ion complexes makes it vital to explore the patterns that drive distribution of salinity in the PPR landscape (Arndt and Richardson 1989). From this insight is it easy to note that these would have been strengthened if analysis on the ionic composition the pondwater—specifically looking at the concentration of $SO_4^{2-}$, $Fe^{3+/2+}$, $Al^{3+/2+}$, $Mg^{2+}$, $K^+$ or $Ca^{2+}$—was performed, though much can still be learned from the relationships identified in this research (LaBaugh 1989). Illustrating salinity as an important driver in pondwater P concentrations is invaluable information in the conversation regarding P and the P retention potential of varying PPR wetlands. The application of these results in the context of wetland P retention potential will be explored in further detail in section 5.0.

### 4.1.2 Alkalinity and inorganic carbon

Pondwater alkalinity and IC were two of the most important physicochemical probable drivers of P concentrations. As pondwater alkalinity increased so did concentration of P in the pondwater (Table 3.4); comparatively, as sediment IC increased, the $CaCl_2$-extractable P in the sediments declined (Table 3.5). Alkalinity was also the most frequently occurring physicochemical parameter, and in some cases the only parameter identified, when modelling pondwater P concentrations (Table 3.7). It is however important to consider that the P-alkalinity relationship was rather poorly parametrized, as the SE was regularly high.

These results are largely due to the relationships between $CaCO_3$ and P (Richardson et al. 1994; Reddy et al. 2005; Müller et al. 2016; Orihel et al. 2017). An abundance of $CaCO_3$—

derived from the limestone parent material of the PPR region—has directly contributed to the pondwater alkalinity and sediment and soil IC observed in the PPR region (Arndt and Richardson 1993; Goldhaber et al. 2014 LaBaugh et al. 2018). Alkalinity and IC are linked with $CaCO_3$ because in the pondwater, $CaCO_3$ is not specifically identified but contributes to the measured alkalinity, whereas in the soil and sediments $CaCO_3$ is an identified aggregate during IC analysis. Although pondwater alkalinity and IC did not correlate with one another (Table A.), pondwater alkalinity and IC concentrations are partly governed by similar processes and will be discussed consecutively in this section.

The interaction between $CaCO_3$ and $PO_4$ occurs where Ca—derived from $CaCO_3$—binds with $PO_4$ though adsorption (Reddy et al. 2005). As a result $PO_4$ is often sorbed to $CaCO_3$ or co-precipitated with $Ca^{2+}$, depending on the anaerobic conditions and especially in alkaline environments (Reddy et al. 2005). However, the presence of $Ca^{2+}$ alone does not mean an immediate abundance of calcium-phosphates. Key factors affecting P accumulation with $Ca^{2+}$ in water and availability include pH, vegetation such as periphyton, humic material, and temperature (Orihel et al. 2017). Depending on the pH of the environment, different ions—such as $Fe^{3+/2+}$, $Al^{3+/2+}$ or $Ca^{+}$—react with $PO_4$ compounds (Penn and Camberato 2019). In alkaline environments $PO_4$ compounds can co-precipitate with $Ca^{2+}$ or $Mg^{2+}$, while in acidic environments Fe and Al are more dominant (Reddy et al. 2005). During decreases in pondwater pH, $PO_4$ compounds may co-precipitated with $Ca^{2+}$ (Reddy et al. 2005). In some cases, periphyton has been the source of changing pH (Dodds 2003). Previous work from St. Denis National Wildlife Area found that Fe, manganese, and Al pondwater concentrations all positively correlated with pondwater TP (Witham unpublished data). Data on the concentration of $Ca^{2+}$ and pondwater pH was not available but this still serves to highlight the role of ions in P retention. In environments where there is an abundance of humic material, in addition to the abundance of P originating from the OM, phosphates can also be bound with humic materials by bridging through ions such as $Fe^{3+/2+}$ and $Ca^{2+}$ into more stable and less available forms of P (Alvarez et al. 2004). Similarly, higher temperatures can stimulate $PO_4$ precipitation and adsorption (Reddy et al. 2005). Overall, there are many factors affecting how $CaCO_3$ interacts with P compounds in the pondwater, but overall $Ca^{2+}$ needs to be present in the pondwater in order for these reactions to occur in the first place.

Soil and sediment IC might actually have even greater role or equally great role to play in the accumulation of P in the PPR landscape as pondwater chemistry. Brown et al. (2017a) showed that concentrations of available P (Kelowna extraction method) were six times greater in Ca-depleted wetland soils than Ca-rich wetland soils. Our results present the same findings, but rather than using a binary Ca-depleted/Ca-rich approach, we found that sediment $CaCl_2$-extractable P was negatively correlated with IC (Table 3.5). However, even with this relationship, IC and P concentrations are still subject to changing soil moisture, vegetation, and pH which in turn drive additional P dynamics. For example, Penn and Camberato (2019) recently reviewed the body of literature exploring the effects of pH on P uptake in soils and reaffirmed that P availability in the solution is greatest when pH is near neutral. The effect of changing IC on P accumulation reaffirms that P is dynamic and developing a greater understanding in the differences in P accumulation across the PPR landscape would be a huge asset. Overall, understanding the differences in P accumulation across a gradient of wetland conditions is particularly important as currently very little is known about the properties potentially driving P accumulation in PPR wetlands.

### 4.1.3 Organic carbon

Organic carbon in the sediment and soil was positively correlated with porewater DP, sediment $CaCl_2$-extractable P, and soil $CaCl_2$-extractable P (Table 3.5). Sediment OC was also one of two parameters—the other being clay content—in the model for sediment $CaCl_2$-extractable P (Table 3.7). **Error! Reference source not found.**In light of these results and the fact that recent PPR research (Lane and Autrey 2016; Badiou et al. 2018; Haque et al. 2018a) identified the role of OC and OM in wetland P dynamics—with greater concentrations of P in greater OC environments—this section will focus exclusively on the influence of OC on P pools.

Organic C is linked with P because of two key roles, first it provides a substrate for binding of P complexes and secondly OM contains P (Weihrauch and Opp 2018). Humic material plays an important role, as it provides a surface for ions to bind, with the ions eventually becoming sorption sites for P (McGill and Cole 1981; Reddy et al. 1999; Alvarez et al. 2004; Weihrauch and Opp 2018). While the inherent presence of OM means that immobilization is occurring as $PO_4$ is being taken up by plants uptake $PO_4$, the decomposition of OM and mineralization of organic P facilitate the release of $PO_4$ back into the environment. The role of OC in P accumulation is clearly supported by the current findings, which identified that $CaCl_2$-

extractable P concentrations were greater in environments with high OC (**Error! Reference source not found.**Table 3.5). While such observations are consistent with the literature, the role of OC in facilitating P-complexes and the rates of immobilization and mineralization vary depending on the environment; redox conditions, freeze-thaw, and temperature fluctuation can change the role that OC has in P accumulation or depletion (Lane and Autrey 2016; Badiou et al. 2018; Weihrauch and Opp 2018).

As mentioned, the abundance and availability of $PO_4$ varies depending on soil and sediment properties; pH, the amount of OM, rate of OM decomposition, the microbial community and presence or absence of specific plants can change the kind and strength of P-complexes (Weihrauch and Opp 2018). For example in acidic environments, Fe and Al complex with negatively charged humic substances; therefore, $PO_4$ retention on OM is dominated by Fe and Al-P complexes (Weihrauch and Opp 2018). As previously mentioned, in alkaline environments P retention is regulated by humic substances, as these provide a bridge for $PO_4$ to bind more easily with $Ca^{2+}$ ions (Weihrauch and Opp 2018). Humic material also inhibits Ca-$PO_4$ transformation and can decrease the availability of P in the soil or sediment (Alvarez et al. 2004). Plants also facilitate the release of $PO_4$ from OC by secreting phosphatase, or developing symbiotic relationships with fungi to encourage them to excrete phosphatase and organic acids to change the soil pH to facilitate the release of P (Filippelli 2014). It is important to note that the same interactions between OC and P have also been explored in agricultural studies which showed that phosphates released in response to demand for P (2000). Wetland phosphatase activity has also been identified as a possible early warning indicator for wetland eutrophication in some regions (Newman et al. 2003). Research exploring the influence of phosphatases and other enzymes in the PPR wetland is lacking; however recent research has shown significant differences in alkaline phosphates and other extracellular enzymes between different groundwater table levels in the PPR suggesting that variable pond wetness may pay a role (Shahariar et al. 2021).

However, while much of the literature on this topic (Lane and Autrey 2016; Badiou et al. 2018; Haque et al. 2018a) was conducted in much greater detail in terms of sampling frequency, it is regionally specific (Broughton's Creek Watershed, Manitoba & Florida, USA), and has a limited wetland selection size ($n$ = 12–55). Therefore, the presented results affirm that OC plays an important role in the patterns of P accumulation, and not just in a select area of the PPR. The results can also be used to inform future modeling of P retention potential in soils across the PPR, not just southern Manitoba.

### 4.1.4 Texture

Soil and sediment clay content were negatively correlated with $CaCl_2$-extractable soil and sediment P (Table 3.5). Sediment clay content was also negatively correlated with porewater DP and identified as a key parameter in the porewater DP and $CaCl_2$-extractable sediment P models (Table 3.5, Table 3.7). This is because with greater clay content there are more sites available for P binding compared to sand or silt (Mozaffari and Sims 1994). Our results indicated that with increasing clay content, porewater DP, and $CaCl_2$-extractable P from sediment and soil declined, suggesting that more of the P has been bound by clay. Haque et al. (2018a) and Ige et al. (2005) also showed that clay content was an important driver for understanding P pools in both agriculture and wetland soils of the PPR.

### 4.1.5 Pond physical characteristics

Differences in P concentrations between P:A, pond permanence, and watershed classes were explored as these characteristics have been identified or strongly speculated as important drivers to consider when exploring the biogeochemical behaviour of small water bodies (Cheng and Basu 2017; Haque et al. 2018b; Wolfe et al. 2019). These results however showed that the previously identified properties were not the best predictors for P concentrations in these data (Table 3.7). Only sediment $CaCl_2$-extractable P concentrations positively correlated with P:A (Table 3.7). When comparing the watershed classes, only particulate P was different between the classes (Figure 3.2) and there were no significant differences in TP, DP, or PP across pond permanencies (Figure A.5). There are a number of possible reasons that P:A, pond permanence, and watershed class were not identified as statistically significant drivers; these will be discussed in detail below.

#### 4.1.5.1 Perimeter to area ratio

Understanding the size of local waterbodies is important because smaller waterbodies have been shown to have greater nutrient loading rates per unit area than larger water bodies (Cheng and Basu 2017). Likewise, P:A can be expected to be a useful metric for understanding the role of pond physical properties in nutrient retention (Millar 1971; van der Kamp and Hayashi 2009), because of the potential for higher loading per unit area where P:A is high. While these results did not show that smaller water bodies had higher nutrient loading rates than larger water bodies, we did find that sediment $CaCl_2$-extractable P was positively correlated with P:A (Table 3.7).

There are several possible reasons why $CaCl_2$-extractable P was positively correlated with P:A, while other forms of P in the pondwater were not. It is possible that ponds with greater P:A retain more P due to their greater potential P loading capacity. Wetlands with greater P:A have the potential to receive more P from the surrounding landscape relative to the area of the pond in comparison to low P:A sites. Given that the relationship was significant for sediment extractable P, but not for surface water, it is possible that P loading is partitioned into the sediment, rather than remaining in the water column. Consistent sediment flooding promotes long-term OM accumulation, which can control long-term P storage (Dunne et al. 2007). It is possible that inorganic P received in these systems is taken up by algae, and transferred to the sediments as organic P, some of which may ultimately become bound to the sediment as extractable P. Accumulation of P in the sediment in this manner may increase the $CaCl_2$-extractable sediment P. Additional research identifying the total sediment P concentrations would be beneficial in this situation to see if increasing $CaCl_2$-extractable P is related to increasing total P in the sediment. Sediment P pools are an important consideration, as internal P loading is important in small Prairie lakes (Orihel et al. 2017), and could also play a role in seasonal P behaviour in wetland ponds. It might also be expected that water level drawdown is higher in higher P:A ponds due to lateral transfer of water during dry periods, and this may promote transfer of P to the wetland sediments. This is consistent with the sampling being done partway through a multi-year dry period across much of the study region.

Although the influence of P:A was not observed for measurements of P other than sediment $CaCl_2$-extractable P, previous research suggests that wetlands with high P:A should continue to be a priority in wetland restoration and retention efforts. Wetlands with greater P:A are associated with greater wetland-upland connectivity and their more complex perimeter has important implications for waterfowl habitat and vegetation (Van Meter and Basu 2015). These results are useful in highlighting the continued need for a dynamic repository of wetland pond physical properties in the Canadian PPR, specifically to confirm the role of wetlands as biogeochemical hotspots on a greater scale.

### 4.1.5.2 Permanence

Research suggests that, given the hydrological processes that contribute to the development of different PPR wetland pond types, patterns of P accumulation would appear between different pond permanence classes (LaBaugh et al. 2018). Our results, however, showed there were no differences in the concentrations of P among these classes (permanent, semi-permanent, seasonal, Figure A.5)

Patterns of P accumulation were expected to vary by pond permanence due to the fact that previous research observed marked differences in pondwater chemistry between wetlands of different recharge or permanencies in the PPR (Goldhaber et al. 2014). For example, Goldhaber et al. (2014) found that even wetlands in close proximity (~ 200 m) had variable ionic compositions. In upland recharge wetlands, they found that bicarbonate and calcium bicarbonate dominated, while $SO_4^{2-}$ and magnesium sulfate were dominant in the discharge wetlands Goldhaber et al. 2014). While our results did not look at the ionic composition of pondwater, results showed that although permanent ponds had the greatest mean pondwater conductivity and alkalinity, both were highly variable (Table A.2). Therefore, it is possible that the inherent variability of wetland hydrogeochemistry is having a greater influence on the patterns of P accumulation and P availability than pond permanencies.

### 4.1.6 Watershed classes

The P patterns across watershed classes were explored because these classes were derived using a suite of landscape information. Among P forms, only PP was significantly different among the watershed classes (Figure 3.2); however, similar P concentrations were observed for similar watershed classes (Table A.2). While natural variability within wetlands for each of the watershed classes remains high, this observation of similar mean P concentrations shown within the similar watershed classes may give us insight into the potential role of watershed-scale landscape factors. For example, the Pothole Till and Pothole Glaciolacustrine watershed classes are both characterized as having the highest amount of non-effective contributing area, due to the areas having the greatest density of wetlands of all the watershed classes. This could contribute to P accumulation rather than export to drainage networks. Likewise the Pothole classes have the highest cover of cropland (Wolfe et al. 2019) while the grassland watersheds (High Elevation Grassland and Interior Grasslands) have the largest fractions of unmanaged grasslands (Wolfe et al. 2019). Given what is known about the influence of agricultural practices on the accumulation of P in runoff (Cade-Menun et al. 2013), it seems reasonable that Pothole wetlands could have higher P compared to the grassland classes. While there is no significant difference between TP concentrations in the Pothole classes and the Grassland classes TP concentrations are higher in the Pothole classes (Table A.2).

The Southern Manitoba watershed class had a low mean TP concentration compared to the other watershed classes though it has a similar amount of area under agriculture compared to the other watershed classes (Table A.2). The reason for the difference in P accumulation may be because the Southern Manitoba watershed has the fewest number of samples ($n$ = 4), an alternate explanation for the results could be due to the small non-effective area and greater annual precipitation. The Southern Manitoba watershed class was the only watershed class with no mean moisture deficit (Wolfe et al. 2019). This lack of moisture deficit or rather greater abundance of precipitation may be contributing to a dilution in the P concentration; however it is possible that greater moisture means greater rates of runoff in the region and more P is consistently transported away from the system.

Overall, the patterns of P accumulation seen in the watershed classification still provide useful insight into the importance of understanding and accounting for all the landscape properties, from soil moisture deficit to land-use percentage in the PPR region. The P models also affirm the importance of accounting for a variety of properties in the PPR, as they too have many variables contributing to the models (Table 3.7). However, the results are largely useful for providing a direction for future research. Specifically, they highlight that focus on the impact of land use on the movement and accumulation of P in PPR wetlands would provide the best insight into how landscape characteristics impact patterns of P accumulation in PPR wetlands. The impact of land use will be briefly explored in the following section.

### 4.2 Patterns in land-use influencing pondwater phosphorus

Land-use type was the best and most frequently occurring physiographic parameter in the wetland pondwater, sediment, and soil P models, and in some cases, it was the only parameter in the model (Table 3.7). There were also significant differences in P concentrations between wetlands adjacent to different land-use types (Figure 3). The ponds adjacent to cropland held three times more TP and DP than the ponds adjacent to grasslands or pasture (Figure 3). The difference in vegetation between the two land-use types is likely not responsible for the observed differences in P accumulation but this result is nonetheless consistent with current knowledge of agricultural practices.

The variability in P concentrations between land-use type is likely attributed to many different potential drivers. These may include differences in fertilizer application rates, the kind of fertilizer applied, tillage practices, timing of fertilizer application, previous land uses, presence or absence of crop stubble, livestock access to wetland ponds for water, time of harvest, and time of seeding. While the research did not explore the probable drivers mentioned above specifically, previous research suggest that many hold a strong degree of influence. For example, fertilizers and manure are commonly applied on cropland and some pasturelands (Schindler et al. 2012; Yates et al. 2012), and while the hope is that all of the applied nutrients are taken-up by crops or retained in the soil and available for plants in future, this is not always the case. In the PPR and many other areas of the world, nutrient-rich runoff is a major contributor to the eutrophication of large water bodies (Schindler et al. 2012; Rattan et al. 2017). Not only is nutrient runoff concerning, but depending on the time of year that nutrients were applied or the form (fertilizer, manure etc.), they may be more susceptible to transport by runoff (Cade-Menun et al. 2013). Previous work in the PPR showed that an intact grassland wetland site that did not receive any recorded nutrient application (manure, fertilizer, etc.) had the lowest concentrations of TP and DP in the runoff compared to runoff from other sites (Badiou et al. 2018). Phosphorus concentrations in wetland sediments have also been greater in the sediments of wetlands adjacent to agricultural land compared to wetlands adjacent to native prairie (Preston et al. 2013).

The timing and methods of fertilizer application, the type of fertilizer applied, and the presence of cattle on pasture could also influence P accumulation. The merits of synthetic or organic fertilizer use and the timing of fertilizer application are long-discussed topics, as the mobilization of fertilizers can change depending on the time of application and the subsequent weather events (Kleinman et al. 2011). For example, upon initial additional of fertilizer, P is more labile and can potentially move though the landscape during a well-timed storm event; however, over time the P becomes more stable and less likely to move, making the timing of fertilizer application and subsequent weather conditions potentially important factors (Audette et al. 2016). The kind of fertilizer is also important as animal manure (organic fertilizer) has been shown to increase P in both pasture and crop soils but have been particularly beneficial in increasing organic P (Dodd and Sharpley 2015). The kind of animal manure applied is also important as differences in soil test P (kind of test) have been identified between soils with swine and cattle manure application (Qian et al. 2004).

It is important to acknowledge that while pasture and grassland sites were grouped in this study, the presence of cattle on pasture landscapes can impact P mobility. For example, the presence of cattle on the landscape during a rainfall event can increase the amount of P in the runoff (Vadas et al. 2015). Research in the PPR has not yet compared P in PPR wetlands and adjacent soils between different animal grazing types; however, pastured cattle—specifically dairy cattle—can contribute between 55 and 70% of non-point source P to adjacent streams (James et al. 2007), and timing and location of livestock relative to water bodies is beginning to receive consideration as a beneficial management practice. Additional research suggests that grazing itself may not be the issue, with the fraction of native and non-native vegetation located in the pasture having a greater impact on wetland water properties (Dunne et al. 2010). This is, however, if the cattle on the landscape have left the wetland vegetations relatively intact and capable of continuing to thrive in the environment. Thus, while the literature suggests the potential for differences in P patterns between pasture and grassland adjacent wetlands, significant differences between cropland and grassland/pasture adjacent sites were nonetheless observed in this study, suggesting that any role of livestock at the small number of pasture sites did not act to obscure what emerged as notable differences in P concentrations between cropland and grassland/pasture land uses. The effect of livestock on P accumulation appears to depend on a range of factors, and targeted research is necessary to understand the impact of cattle and other grazing animals on PPR wetland P accumulation, and how wetlands adjacent to grassland and pasture may be different.

Beyond fertilizer and the presence or absence of cattle, there is a strong possibility for redistribution of P through agricultural practices, like landscape grading, tilling, or harvesting. These practices will differ strongly between cropland and grassland or pasture, and can mobilize organic P or soil P via the creation of dust. While these airborne pathways of P transport have not been quantified in this region, dust associated with harvest is visible and can be expected to transport some P to wetland ponds.

While we haven’t measured the factors mentioned above in our study sites, based on previous research and the reported differences between cropland and grassland/pasture wetlands P concentrations, differences in management practices between cropland and grassland/pasture sites are likely an important driver of wetland P concentrations. The land-use results also invoke another important insight: they present an opportunity to reconsider the patterns of cropland and grassland/pasture wetland P accumulation through the lens of one of PPR wetlands' greatest threats—wetland drainage. If these results are considered through the lens of wetland drainage and the utility of wetlands for P retention, then cropland wetlands have a very unique role to play. Cropland-adjacent wetlands play a greater role in P retention simply because they are expected to receive higher P loads from runoff. Therefore, with accompanying P loads, the P retention will be greater but also the potential P losses with cropland-adjacent wetlands will be greater. If drained, cropland-adjacent wetlands may contribute three times more TP to downstream environments than grassland/pasture-adjacent wetlands. Sediment and soil P would be susceptible to loss through 1) re-wetting and subsequent drainage or 2) plant uptake (Venterink et al. 2002).

Ideally the patterns explored and outlined above will be used to inspire and prompt future research exploring the impact of land use on P accumulation in PPR wetlands. Overall, research has brought even more evidence to the table that PPR wetlands are great nutrient sinks but the results suggest that intact cropland-adjacent wetlands may have an even greater P retention potential while they may also have higher P loading than intact grassland/pasture adjacent wetlands.

## 5.0 Conclusions

### 5.1 Implications for wetland management

This work is some of the first to explore patterns of P accumulation across PPR wetlands. We highlight that land use and specific physicochemical wetland properties (pondwater alkalinity, pondwater conductivity, and sediment OC) are important to consider when exploring the role of wetlands in the PPR landscape and P retention. Identifying the role of land use and physicochemical properties in P accumulation gives us the chance to better understand where on the landscape we can decrease the risk of excess P and leverage the P retention potential of wetlands. As the results highlighted, cropland-adjacent wetlands likely have greater P retention potential simply because the additions of P into the wetlands are expected to be greater. High pondwater alkalinity concentrations were also associated with greater pondwater P concentrations. Given this, our results suggest that wetlands located adjacent to cropland with high pondwater alkalinity should be protected from drainage. As these wetland systems may be performing an even greater role in the PPR landscape than originally anticipated due to the additional P loading from cropland, both protection from drainage and action to reduce nutrient loading to these systems could be beneficial for downstream water quality. Otherwise, it is possible that given enough time and consistent nutrient application in cropland that these adjacent wetlands will potentially be transformed from P sinks to P sources. The potential of these wetland systems to become sources of P should be considered, as legacy P can have damaging impacts on adjacent waterbodies but also may be used to facilitate the management of sustainable P stocks (Rowe et al. 2016; Menezes-Blackburn et al. 2018).

The physiographic and physicochemical properties outlined impact wetland P concentrations in different ways, therefore management needs to use different lenses and contexts when picking management strategies. However, before any decisions are made it is important to remember the context. For example, if a priority for a management strategy is to provide habitat for waterfowl, then wetlands with high P:A would be highly sought after, compared to those with low P:A. Other wetland strategies may be trying to prioritize wetlands with high runoff-retention potential, therefore those in optimal locations for water storage could be prioritized. These examples serve to underline that discussion about wetland P retention and drainage are best suited for management plans that are in the context of nutrient retention. Each of these wetland types have a unique role to play and the wetland management practices suitable for one scenario—like P retention—will not be suitable for all.

Understanding the unique role that different wetland types have to play in P retention will greatly improve our decision-making ability. However, more analysis is needed to understand just how greatly P retention and accumulation vary between cropland-adjacent wetland sites. Within the context of physicochemical properties, identifying and recognizing the inherent ability of wetlands to bind P because of physicochemical properties is a considerable advantage and can be used to identify wetlands that may be more suitable for drainage over others due to a lacking role in P retention. These results are useful in highlighting that biogeochemistry of PPR ponds is an often-overlooked part of wetland nutrient retention. If some wetlands are better at retaining nutrients based on existing hydrological or physicochemical properties, then we need to leverage such information during management decisions. Overall, both the physiographic and physicochemical properties highlighted as key potential drivers in PPR wetland P patterns need to be used when it comes to wetland decision making. The results have hopefully highlighted the need to account for all of the unique PPR wetland dynamics when planning for wetland management.

### 5.2 Future research directions

As this project is the first of its kind—exploring patterns of P accumulation across a gradient of wetlands in the PPR—we identified a number of future research directions and insights that will be useful for those who explore PPR wetland P dynamics in the future. Ongoing PPR wetland research is needed to continue supporting PPR wetland management but, based on the results above, research could focus on one or all of the three following areas: understanding patterns in P accumulation across a greater range of PPR wetland ionic compositions, more research on the impact of land-use characteristics on P accumulation and lastly, research needs to expand beyond operationally-defined forms of P.

First, this section will explore the possible insight gained from exploring ionic composition in PPR wetland pondwater. Insight would be gained by knowing the ionic composition of PPR wetland pondwater, as different ions play different roles with changing pondwater properties, which can impact P retention or release (Reddy et al. 2000). While this research was useful in re-affirming the relationship between salinity, alkalinity, and P concentrations, additional work exploring P concentrations across all wetland salinity and variable ionic compositions would be beneficial, specifically identifying the dominant ions in the pondwater environment. Euliss et al. (2014) also previously remarked that identifying the composition of the pondwater solution would be useful in understanding the relationship of individual wetlands to groundwater. Additional exploration into the presence and impact of the regional variance of pondwater ions may some provide insight into the potential drivers of P accumulation in PPR wetlands.

Secondly, better understanding and controlling for landscape-scale variability would be a huge asset to any future widescale PPR wetland research. Based on the results presented above, a detailed inventory on the landscape properties—more specifically P:A, land use, and P application rates—on PPR wetland P accumulation is needed. The P:A data used in the analysis were derived from multiple wetland inventories with varying extents and spatial resolutions, which may limit the strength of the analysis. This highlights that there is no wetland inventory—like those in the United States—that capture Canadian PPR wetland physiographic features using consistent methods. This data gaps makes it much harder to conduct widespread analysis of the role of P:A and similar features in the Canadian PPR.

The previous section also speculated that several land-use properties need to be considered in order to accurately identify the impact of land use on P accumulation in PPR wetlands. For example, the results showed that soil zone and crop type were related to one another; therefore, future projects will need to take crop type into consideration during sampling design. Accounting for the differences in P concentrations between the soil zones, and the inherent differences in crop type between them gives the opportunity to better identify differences in P concentrations between crop and other land-use types. Additional soil-related information such as the rate of fertilizer application and historical land use or crop type would be useful to get a complete picture of the soil environment. Data collection over multiple years is also needed to better understand long-term vs short-term P accumulation patterns between different land uses. Details regarding that process that facilitate the accumulation and depletion of P in PPR wetland ecosystems, also need to be considered in future research. Specifically, this may include the presence or absence of cattle or livestock, land management practices including the kind of equipment used, and historical land use. Research on P export from agricultural fields in streams in the PPR has established that high P export is associated with fertilizer application, livestock density, and sewage (Rattan et al. 2017); similar research is also needed specifically in a PPR wetland setting. In general, research that controls for differences in soil properties between soil zones and land uses while collecting detailed information on the pre-existing and current land-use conditions and P application rates would vastly improve understanding of P accumulation across different landscapes.

Lastly, this research—and other PPR wetland research to date—only explored operationally-defined P pools. A huge gap in understanding remains as no research has investigated differences in organic P forms or accumulation in PPR wetlands to date. Understanding patterns and forms of organic P accumulation is imperative, as organic P may account for 90% of all P found in PPR wetlands (Reddy and DeLaune 2008b). The growing capacity to chemically identify P forms, rather than using operationally defined techniques, should be leveraged in PPR wetlands. The emergence of techniques such as X-Ray adsorption near edge structure (XANES) and $^{31}$P nuclear magnetic resonance ($^{31}$P-NMR) has allowed researchers to chemically identify the forms of organic and inorganic P (Cade-Menun and Liu 2014; Sato et al. 2005). Research using $^{31}$P-NMR has indicated that the types and dominance of P forms change between different kinds of manure applications, landscape positions, and in aquatic systems (Dou et al. 2009; Menezes-Blackburn et al. 2018). Seeing as all forms of P are potentially bioavailable given enough time and the right conditions, understanding and identifying the specific controls and drivers on organic P transformation is invaluable. By identifying how the organic P concentrations and forms vary across the PPR, research could begin to identify the potential controls that mediate the transformation of P in wetlands and explore the implications of transformation. Overall, PPR wetlands are dynamic and future research needs to account for this dynamic nature. The hope is that the insights presented will give future PPR wetland researchers a roadmap, highlighting potential potholes and areas of concern while exploring the longstanding questions about P dynamics in PPR wetlands.

Overall, this project strived to identify potential physiographic and physicochemical drivers on pondwater, sediment, and soil P concentrations across a gradient of wetland conditions and was successful in identifying a number of controls that had degrees of influence depending on the P pool identified. The fact that the physiographic and physicochemical drivers identified were found across such a variety of wetland conditions is a testament to the influence of these drivers have on the accumulation of P in PPR wetlands. These results also serve as call for better understanding of how both physiographic and physicochemical drivers and controls work together to impact nutrient biogeochemical processes in wetlands.

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## Appendix: Supplemental Information

**Site ID:** ____________

Date (yyyy-mm-dd): ____________

Time: ____________

Initials (team): ____________

Lat ____________

Long ____________

Crop type: ____________
(if known: stubble, canola, cereal, pulses, grassland)

Seeded field: ____________
(Y/N)

Macroinvertebrates ____________
(Y/N)

Bubbles present: ____________
(High/Low/None)

Sediment smell: ____________
(like rotten egg when collecting water?: Y/N)

Sediment cover: ____________
(Bare/vegetated (incl. rooted macrophytes))

Sediment composition: ____________
(soft/normal/hard)

Connected: ____________
(to other wetlands Y/N)

**Dominant Buffer Veg. (indicate % where >25)**

| | | | |
|---|---|---|---|
| trees | | shrubs | |
| rushes | | reeds | |
| sedges | | grasses | |
| cattails | | other | |

Buffer Coverage (circumference): ____________
0, 1 = 1-25%; 2 = 26-50%; 3 = 51-75%; 4 = >76%

Permanence ____________
(seasonal, semi-permanent, permanent)

Basin Fill: ____________
1 = 1-25%; 2 = 26-50%; 3 = 51-75%; 4 = >76%

WQ/MET

| | | | |
|---|---|---|---|
| Water temp | | | |
| Barometric | | pH (optional) | |
| Mean wind | | Cond (optional) | |

Site observations:

| Checklist | | | | Exetainer IDs: | |
|---|---|---|---|---|---|
| Site photo(s) | | Sediment | | R1 | |
| Bulk water (1 or 2) | | Soil lat | | R2 | |
| Pesticides (2) | | Soil long | | ATM | |
| Isotopes | | Soil distance | | | |

**Figure A.1** Copy of qualitative data collection form used in wetland survey (26 April–6 May 2019)

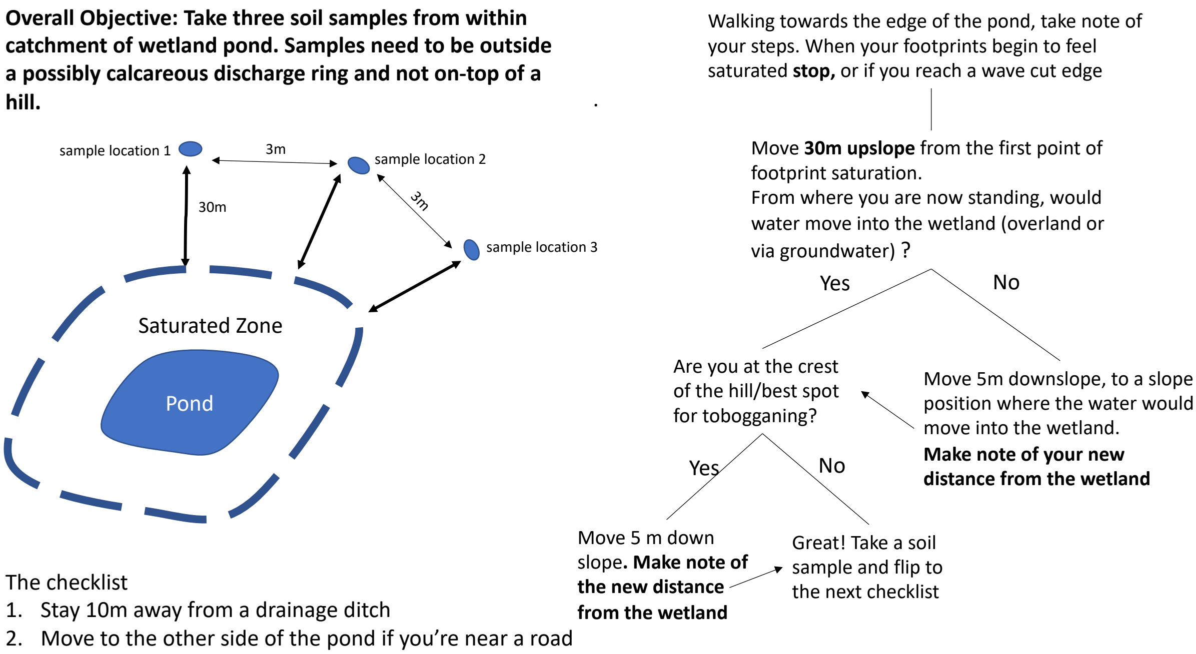


**Figure A.2** Copy of soil sampling decision tree used in wetland survey to determine the collection of a soil sampled within the wetland catchment (26 April–6 May 2019)

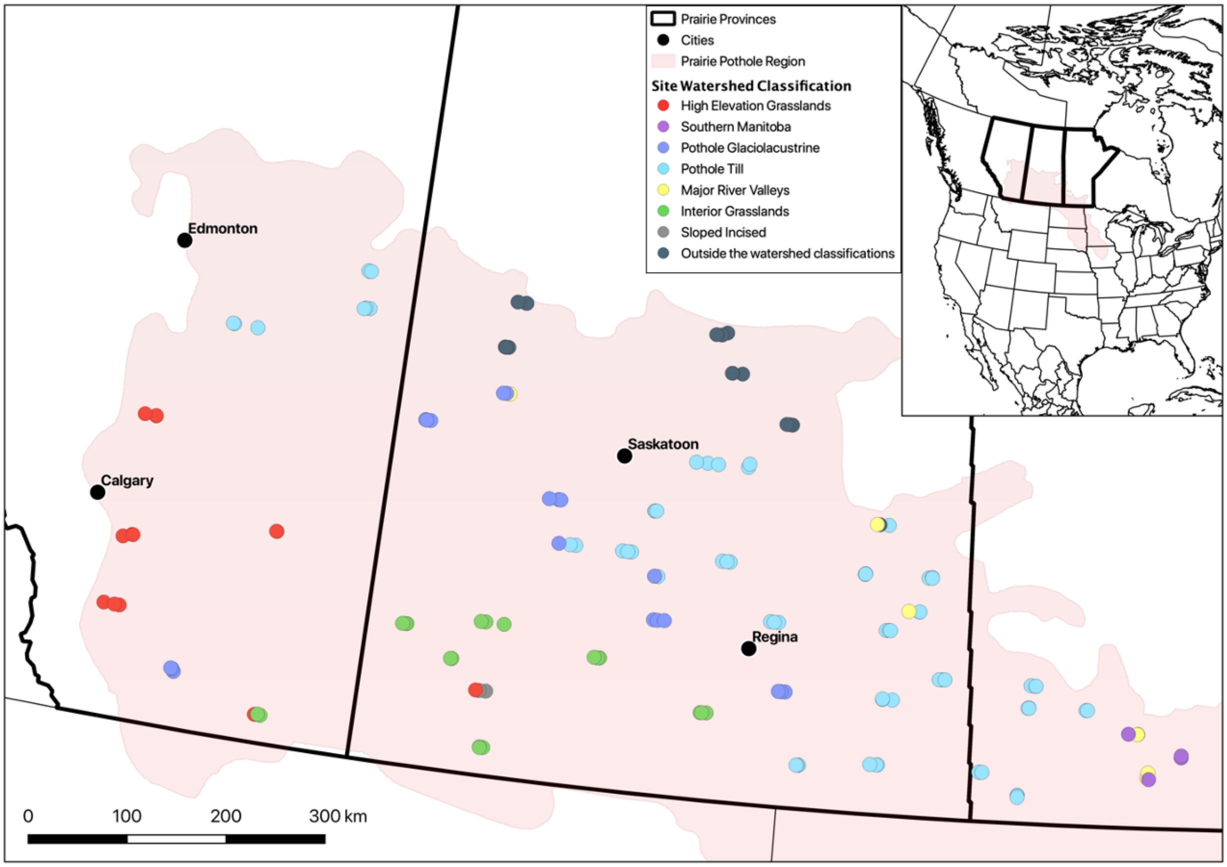


**Figure A.3** Study area in western Canada (inset) with extent of Prairie Pothole Region (pink), locations of sampling sites with associated watershed class and major cities shown. Watershed identification drawn from Wolfe at al. 2019, used with permission. PPR boundary and political boundary data drawn from North American Political Boundaries from U.S Geological Survey. Information licenced under the Department of Interior Copywrite, Restrictions, and Permissions https://www.doi.gov/copyright. Cities drawn from Statistics Canada, and with information licensed under the Open Government Licence-Canada http://open.canada.ca/en/open-government-licence-canada.

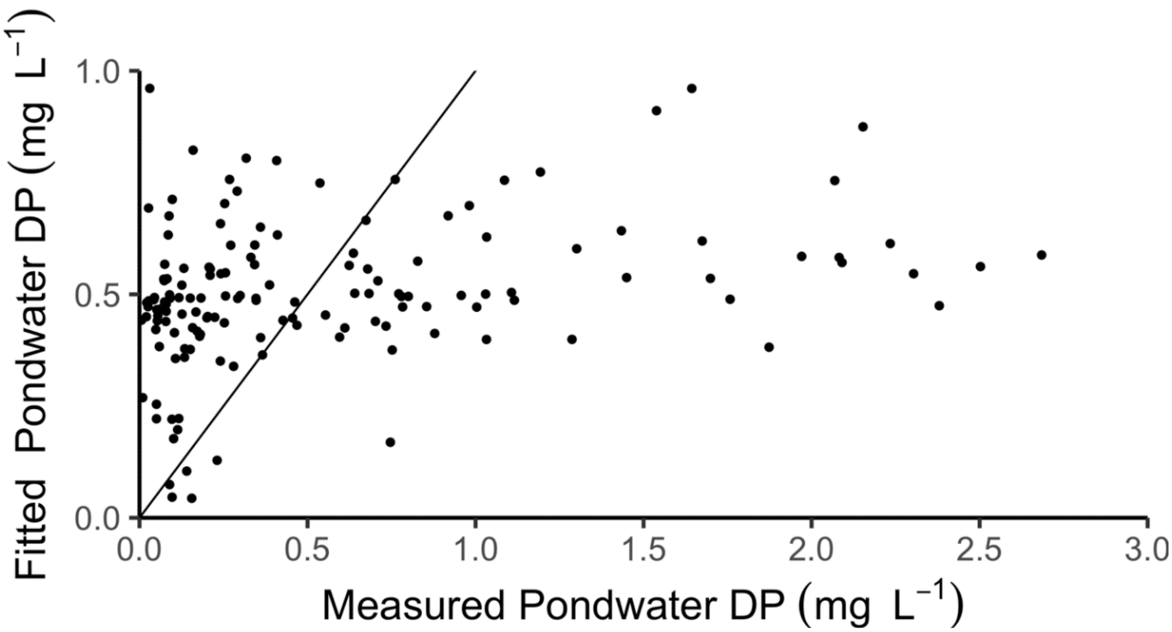


**Figure A.4** Measured and fitted pondwater DP (from GLM model [Pondwater DP ~ Alkalinity + Land use]) concentrations from sampled PPR wetland

**Table A.1** Breakdown of assigned land use groups, taken from Annual Crop Inventory crop classification (used in analysis).

| Land use group assigned for analysis | Oilseed | Cereal | Pulses | Grassland/Pasture | Cropland |
|---|---|---|---|---|---|
| | Flax | Barley | Lentils | Grassland | Flax |
| | Mustard | Wheat | Soybeans | Pasture / Forages | Mustard |
| | Canola | Oats | Peas | | Canola |
| | | Rye | | | Barley |
| | | | | | Wheat |
| | | | | | Oats |
| | | | | | Rye |
| | | | | | Lentils |
| | | | | | Soybeans |
| | | | | | Peas |

**Table A.2** Mean pondwater chemistry properties (SC, alkalinity, pH, and TP), between physicochemical (pondwater SC values groups) and physiographic pondwater groupings (permanence, soil zones, watershed classes, land use, and agricultural land use) assigned to the sampled wetlands (*n* = 150). Values shown are averages with standard deviations shown in parentheses

| Physiographic Group | | n | SC ($\mu S\ cm^{-1}$) | Alkalinity ($CaCO_3$) | pH | Total Phosphorus (TP) |
|---|---|---|---|---|---|---|
| Pondwater SC | Freshwater (<1000 $\mu S\ cm^{-1}$) | 91 | 455 (224)[a] | 168 (76)[a] | 7.8 (0.5)[a] | 0.619 (0.642) |
| | Saline (>1000 $\mu S\ cm^{-1}$) | 58 | 2071 (102)[a] | 323 (172)[a] | 8.5 (0.4)[a] | 0.744 (0.694) |
| Permanence | Seasonal | 26 | 839 (846) | 187 (95) | 7.8 (0.6)[a] | 0.568 (0.487) |
| | Semi-Permanent | 41 | 765 (484) | 195 (91) | 7.9 (0.5)[a] | 0.679 (0.713) |
| | Permanent | 83 | 1312 (1255) | 256 (170) | 8.3 (0.6)[a] | 0.689 (0.687) |
| Soil Zones | Black | 68 | 814 (666) | 203 (117) | 7.9 (0.6)[a] | 0.681 (0.696) |
| | Dark Brown | 47 | 1393 (1178) | 280 (179) | 8.2 (0.7)[a] | 0.731 (0.660) |
| | Brown | 23 | 1473 (1571) | 194 (98) | 8.3 (0.7)[a] | 0.601 (0.603) |
| | Grey | 12 | 576 (354) | 226 (159) | 8.3 (0.4)[a] | 0.445 (0.596) |
| Watershed Class | High Elevation Grasslands | 13 | 1345 (1188) | 248 (228) | 8.3 (0.5) | 0.628 (0.572) |
| | Interior Grasslands | 20 | 1550 (1627) | 196 (78) | 8.2 (0.8)[a] | 0.608 (0.627) |
| | Major River Valleys | 7 | 448.0 (232) | 174 (67) | 7.7 (0.4)[ab] | 0.608 (0.724) |
| | Pothole Glaciolacustrine | 21 | 988 (853) | 280 (213) | 8.2 (0.7) | 0.807 (0.736) |
| | Pothole Till | 67 | 1071 (967) | 228 (126) | 8.0 (0.5)[b] | 0.710 (0.652) |
| | Sloped Incised | 2 | 1707 (2237) | 103 (61) | 8.1 (0.2) | 0.269 (0.115) |
| | Southern Manitoba | 4 | 525 (402) | 316 (217) | 7.4 (0.6) | 0.0934 (0.0591) |
| | Outside Watershed Classes | 16 | 734 (617) | 202 (119) | 8.5 (0.3)[b] | 0.612 (0.794) |
| Land use | Cropland | 133 | 1037 (918) | 226 (120) | 8.1 (0.6) | 0.709 (0.679)[a] |
| | Grassland/Pasture | 16 | 1508 (1830) | 250 (268) | 7.9 (0.7) | 0.323 (0.366)[a] |
| Agricultural Land use | Cereal | 62 | 957 (928) | 201 (111)[a] | 8.1 (0.6) | 0.628 (0.635)[a] |
| | Oilseed | 59 | 1132 (947) | 252 (128)[a] | 8.2 (0.5) | 0.759 (0.715)[a] |
| | Grassland/Pasture | 16 | 1508 (1830) | 250 (268) | 7.9 (0.7) | 0.323 (0.366)[ab] |
| | Pulses | 12 | 984 (697) | 231 (111) | 8.1 (0.6) | 0.878 (0.721)[b] |

Means with same letter in same column are significantly different according to Kruskal-Wallace and Mann-Whitney tests ($P>0.05$; with p-values FDR corrected).

**Table A.3** Mean pond sediment properties (TC, OC, IC, SC, and pH), between physicochemical (pondwater SC values groups) and physiographic pondwater groupings (permanence, soil zones, watershed classes, land use, and agricultural land use) assigned to the sampled wetlands (n = 150). Values shown are averages with standard deviations shown in parentheses.

| Physiographic Group | | n | TC (%) | OC (%) | IC (%) | SC (µS cm$^{-1}$) | pH |
|---|---|---|---|---|---|---|---|
| Pondwater SC | Freshwater (<1000 µS cm$^{-1}$) | 91 | 8.5 (6.4) | 9.2 (7.0) | 0.1 (0.5) | 378 (408)[a] | 6.4 (1.2)[a] |
| | Saline (>1000 µS cm$^{-1}$) | 58 | 8.0 (3.8) | 8.5 (4.3) | 0.2 (0.4) | 1104 (757)[a] | 7.2 (0.7)[a] |
| Permanence | Seasonal | 26 | 8.2 (4.8) | 8.8 (5.2) | 0.1 (0.3) | 483 (392) | 6.5 (0.4) |
| | Semi-Permanent | 41 | 9.8 (6.7) | 10.5 (7.7) | 0.1 (0.5) | 531 (461) | 6.7 (1.1) |
| | Permanent | 83 | 7.5 (4.8) | 8.2 (5.2) | 0.1 (0.3) | 801 (802) | 6.7 (1.1) |
| Soil Zones | Black | 68 | 9.6 (5.5)[a] | 10.1 (6.3)[a] | 0.1 (0.3) | 631 (517) | 6.5(1.2) |
| | Dark Brown | 47 | 5.9 (3.4)[a] | 6.6 (3.8)[a] | 0.1 (0.2) | 769 (707) | 7.0 (0.9) |
| | Brown | 23 | 8.5 (3.9)[a] | 8.9 (4.2)[a] | 0.2 (0.5) | 719 (1080) | 6.2 (1.2) |
| | Grey | 12 | 10.6 (10.1) | 11.0 (10.9) | 0.2 (0.6) | 423 (483) | 6.9 (1.1) |
| Watershed Class | High Elevation Grasslands | 13 | 7.6 (2.8) | 8.4 (1.9) | 0.2 (0.5) | 787 (649) | 6.7 (1.5) |
| | Interior Grasslands | 20 | 8.5 (4.4) | 9.3 (4.8) | 0.2 (0.6) | 806 (1162) | 6.9 (1.2) |
| | Major River Valleys | 7 | 11.1 (7.9) | 11.5 (9.3) | 0.4 (0.8) | 403 (440) | 7.3 (0.5) |
| | Pothole Glaciolacustrine | 21 | 5.2 (2.5)[a] | 5.7 (2.6)[a] | 0.1 (0.3) | 661 (696) | 6.6 (1.4) |
| | Pothole Till | 67 | 9.2 (6.1)[a] | 9.8 (6.8)[a] | 0.1 (0.3) | 694 (618) | 6.6 (1.1) |
| | Sloped Incised | 2 | 3.9 (3.3) | 4.0 (3.3) | N/A | 101 (44) | 5.8 (1.1) |
| | Southern Manitoba | 4 | 11.1 (4.8) | 11.5 (5.7) | 0.1 (0.2) | 482 (355) | 7.3 (0.4) |
| | Outside Watershed Classes | 16 | 5.6 (5.6)[a] | 5.8 (5.7)[a] | 0.2 (0.3) | 371 (360) | 6.3 (1.3) |
| Land use | Cropland | 133 | 8.2 (5.6) | 8.8(6.6) | 0.2 (0.4) | 650 (586) | 6.7 (1.1) |
| | Grassland/Pasture | 16 | 8.7 (4.2) | 9.6(4.5) | 0.1 (0.3) | 845 (1142) | 6.5 (1.4) |
| Agricultural Land use | Cereal | 62 | 8.4 (6.0) | 9.3 (6.7) | 0.1 (0.2) | 540 (574) | 6.6 (1.2) |
| | Oilseed | 59 | 8.2 (5.6) | 8.5 (6.2) | 0.2 (0.5) | 745 (569) | 6.8 (1.1) |
| | Grassland/Pasture | 16 | 8.7 (4.2) | 9.6 (4.5) | 0.1 (0.3) | 845 (1142) | 6.5 (1.4) |
| | Pulses | 12 | 7.7 (2.8) | 8.0 (2.9) | 0.1 (0.1) | 760 (698) | 6.7 (1.1) |

Means with same letter in same column are significantly different according to Kruskal-Wallace and Mann-Whitney tests ($P>0.05$; with p-values FDR corrected).

**Table A.4** Mean pond soil properties (TC, OC, IC, SC, and pH), between physicochemical (pondwater SC values groups) and physiographic pondwater groupings (permanence, soil zones, watershed classes, land use, and agricultural land use) assigned to the sampled wetlands (n = 150). Values shown are averages with standard deviations shown in parentheses.

| Physiographic Group | | n | TC (%) | OC (%) | IC (%) | SC (µS cm$^{-1}$) | pH |
|---|---|---|---|---|---|---|---|
| Pondwater SC | Freshwater (<1000 µS cm$^{-1}$) | 91 | 3.7 (1.9) | 3.7 (1.9) | 0.2 (0.5) | 1425 (1742)$^{a}$ | 6.9 (0.7) |
| | Saline (>1000 µS cm$^{-1}$) | 58 | 3.7 (1.7) | 3.5 (1.7) | 0.3 (0.4) | 2068 (1917)$^{a}$ | 7.3 (0.5) |
| Permanence | Seasonal | 26 | 3.4 (1.5) | 3.3 (1.5) | 0.3(0.8) | 1609 (1832) | 7.0 (0.6) |
| | Semi-Permanent | 41 | 3.3 (1.4) | 3.4 (1.2) | 0.2 (0.4) | 1942 (2016) | 6.9 (0.7) |
| | Permanent | 83 | 3.9 (2.1) | 3.8 (2.2) | 0.2 (0.3) | 1520 (1730) | 7.1 (0.6) |
| Soil Zones | Black | 68 | 4.3 (2.0)$^{a}$ | 4.2 (2.0)$^{a}$ | 0.3 (0.6) | 1924 (1970) | 6.9 (0.8) |
| | Dark Brown | 47 | 3.0 (1.2)$^{ab}$ | 2.9 (1.2)$^{ab}$ | 0.2 (0.3) | 1711 (1806) | 7.3 (0.3) |
| | Brown | 23 | 2.7 (1.3)$^{ab}$ | 2.7 (1.2)$^{ab}$ | 0.1 (0.2) | 1279 (1633) | 7.0 (0.7) |
| | Grey | 12 | 4.8 (2.5)$^{b}$ | 4.6 (2.2)$^{b}$ | 0.3 (0.3) | 495 (276) | 7.0 (0.7) |
| Watershed Class | High Elevation Grasslands | 13 | 3.9 (2.5) | 3.9 (2.4) | 0.2 (0.4) | 1705 (2006) | 6.6 (0.9) |
| | Interior Grasslands | 20 | 2.7 (1.2)$^{a}$ | 2.7 (1.2)$^{a}$ | 0.2 (0.3) | 1855 (1901) | 7.1 (0.6) |
| | Major River Valleys | 7 | 3.1 (1.3) | 3.2 (1.3) | 0.1 (0.1) | 352 (198) | 6.6 (1.0) |
| | Pothole Glaciolacustrine | 21 | 3.0 (1.5)$^{a}$ | 3.1 (1.5) | 0.1 (0.1)$^{a}$ | 1505 (1698) | 7.2 (0.4) |
| | Pothole Till | 67 | 3.8 (1.4)$^{a}$ | 3.6 (1.3)$^{a}$ | 0.3 (0.6)$^{a}$ | 1874 (1919) | 7.1 (0.6) |
| | Sloped Incised | 2 | 1.7 (0.3) | 1.6 (0.3) | 0.1 (0.1) | 321 (84.9) | 7.8 (0.2) |
| | Southern Manitoba | 4 | 6.4 (5.2) | 8.2 (6.4) | 0.1(0.1) | 393 (233) | 7.1 (0.7) |
| | Outside Watershed Classes | 16 | 4.0 (2.0) | 4.1 (1.8) | 0.1(0.3) | 2171 (2286) | 7.1 (0.6) |
| Land use | Cropland | 133 | 5.6 (1.6) | 3.5 (1.5) | 0.2 (0.5 | 1824 (1884)$^{a}$ | 7.1(0.6) |
| | Grassland/Pasture | 16 | 4.3 (3.3) | 4.5 (3.7) | 0.1 (0.2) | 473 (491)$^{a}$ | 6.6 (0.8) |
| Agricultural Land use | Cereal | 62 | 3.6 (1.8) | 3.4 (1.6) | 0.3 (0.4) | 1676 (1692) | 7.2 (0.5)$^{a}$ |
| | Oilseed | 59 | 3.6 (1.4) | 3.5 (1.3) | 0.3 (0.6) | 1938 (2037) | 7.1 (0.6)$^{a}$ |
| | Grassland/Pasture | 16 | 4.3 (3.3) | 4.5 (3.4) | 0.1 (0.2) | 473 (491) | 6.6 (0.8)$^{ab}$ |
| | Pulses | 12 | 3.5 (1.3) | 3.8 (1.6) | 0.1 (0.1) | 1995 (2104) | 7.0 (0.6)$^{b}$ |

Means with same letter in same column are significantly different according to Kruskal-Wallace and Mann-Whitney tests ($P>0.05$; with p-values FDR corrected).

**Table A.5** Mean pond physiographic properties (pond perimeter, pond area, and P:A) between physicochemical (pondwater SC values groups) and physiographic pondwater groupings (permanence, soil zones, watershed classes, land use, and agricultural land use) assigned to the sampled wetlands (n = 105). Values shown are averages with standard deviations shown in parentheses

| Physiographic Group | | n | Pond perimeter (P) | Pond area (A) | P:A |
|---|---|---|---|---|---|
| Pondwater SC | Freshwater (<1000 $\mu S\ cm^{-1}$) | 43 | 705 (655) | 273 (406) | 0.049 (0.025) |
| | Saline (>1000 $\mu S\ cm^{-1}$) | 62 | 726 (558) | 271(302) | 0.046 (0.035) |
| Permanence | Seasonal | 16 | 502 (431) | 172 (252) | 0.054 (0.030) |
| | Semi-Permanent | 28 | 699 (642) | 219 (324) | 0.049 (0.021) |
| | Permanent | 61 | 776 (637) | 323 (412) | 0.045 (0.033) |
| Soil Zones | Black | 47 | 612 (672)[a] | 223(415)[a] | 0.056 (0.27)[a] |
| | Dark Brown | 31 | 892 (663)[a] | 349 (352)[a] | 0.045(0.039)[a] |
| | Brown | 19 | 618 (284) | 208(170) | 0.038 (0.012)[a] |
| | Grey | 8 | 848 (537) | 416 (411) | 0.032 (0.015)[a] |
| Watershed Class | High Elevation Grasslands | 8 | 797 (552) | 384 (439) | 0.035 (0.021) |
| | Interior Grasslands | 15 | 614 (303) | 212 (185) | 0.038 (0.014) |
| | Major River Valleys | 4 | 264 (92.0) | 48.7 (32.8) | 0.061 (0.015) |
| | Pothole Glaciolacustrine | 16 | 1143 (777) | 429 (347) | 0.043 (0.029) |
| | Pothole Till | 51 | 58 (607) | 176 (288) | 0.057 (0.034)[a] |
| | Sloped Incised | 2 | 586 (246) | 154 (39.3) | 0.037 (0.006) |
| | Southern Manitoba | 1 | 794 (N/A) | 237 (N/A) | 0.033 (N/A) |
| | Outside the Watershed Classes | 7 | 1109 (592) | 791 (662) | 0.024 (0.013)[a] |
| Land use | Cropland | 94 | 728 (643) | 284 (383) | 0.048 (0.031) |
| | Grassland/Pasture | 12 | 2203 (5600) | 175 (120) | 0.042 (0.016) |
| Agricultural Land use | Cereal | 47 | 755 (588) | 256 (333) | 0.046 (0.026) |
| | Oilseed | 36 | 756 (772) | 217 (478) | 0.052 (0.039) |
| | Grassland/Pasture | 12 | 587 (236) | 175 (120) | 0.042 (0.016) |
| | Pulses | 11 | 541 (362) | 163 (174) | 0.045 (0.017) |

Means with same letter in same column are significantly different according to Kruskal-Wallace and Mann-Whitney tests ($P>0.05$; with p-values FDR corrected).

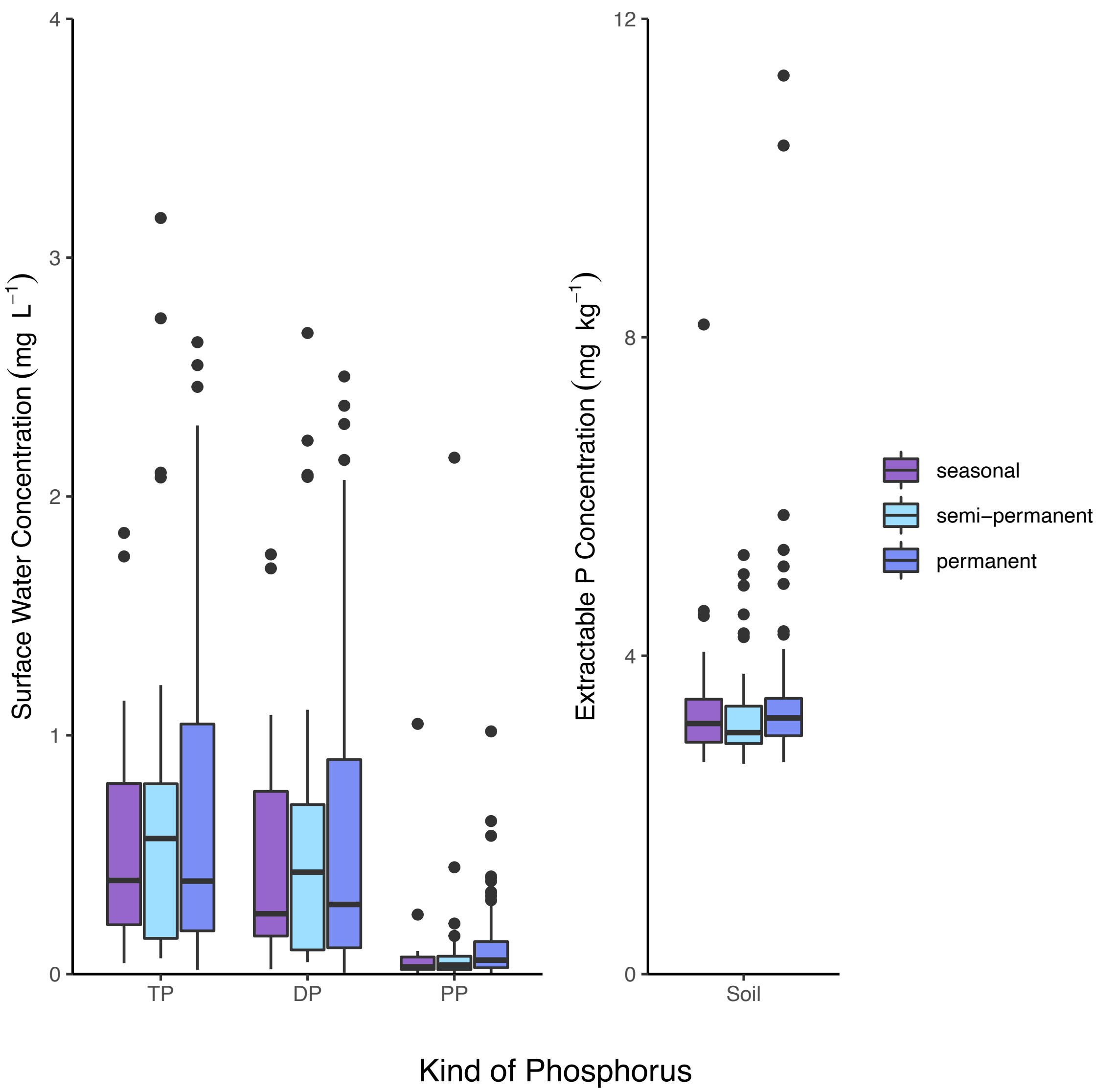


**Figure A.5** Boxplot comparison between pond permanence (seasonal, semi-permanent, and permanent) and concentrations of P in various P pools (TP, DP, PP, and soil $CaCl_2$-extractable P). The boxplot displays data distribution (median, hinges [$25^{th}$ and $75^{th}$ percentiles], whiskers [max and min range], and outlying points).

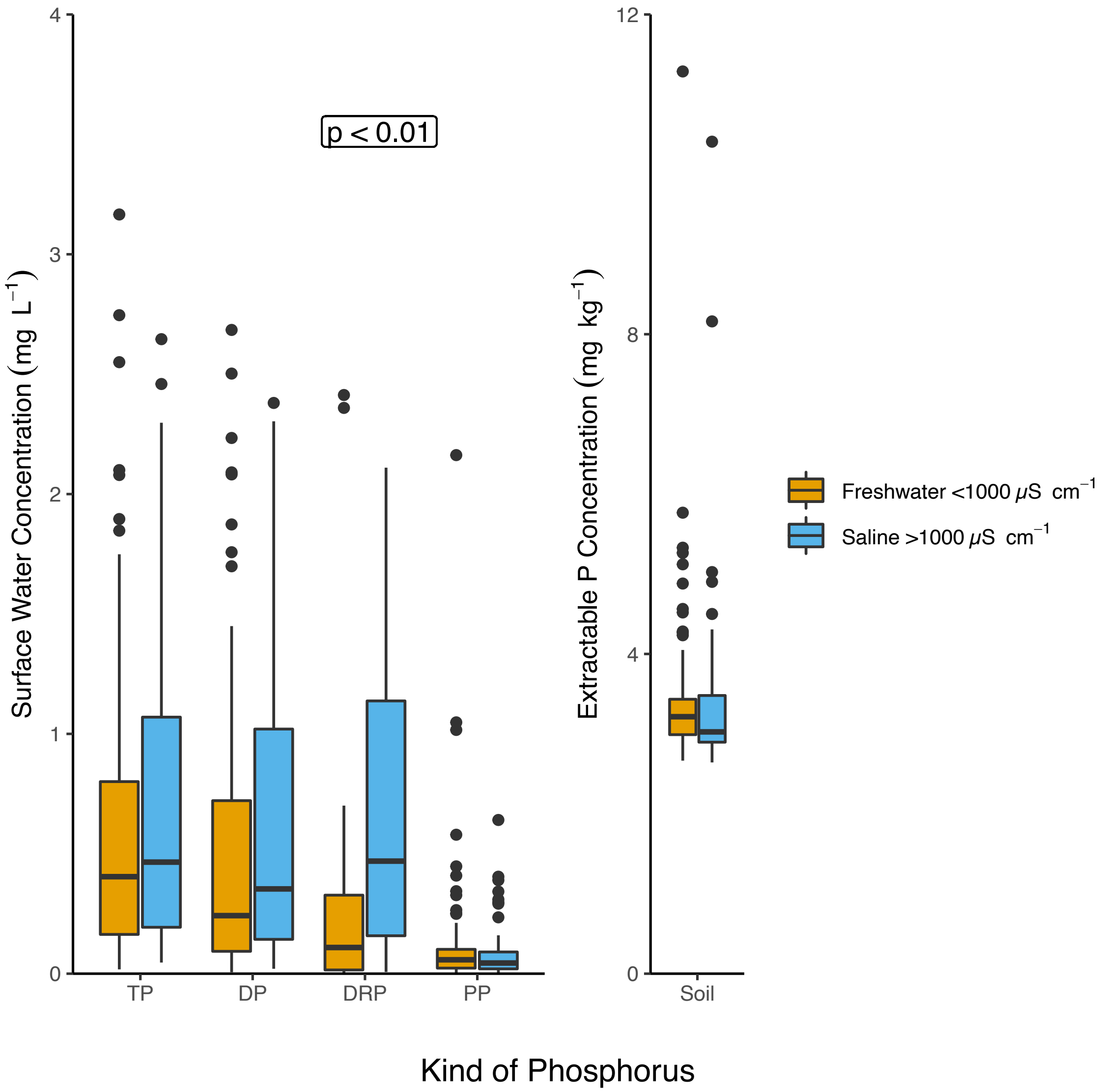


**Figure A.6** Boxplot comparison between pondwater conductivity classes (saline and freshwater) and concentrations of P in various P pools (pondwater TP, DP, DRP, PP, and soil $CaCl_2$-extractable P; *p*-values reported from Mann-Whitney test, and FDR corrected). The boxplot displays data distribution (median, hinges [25$^{th}$ and 75$^{th}$ percentiles], whiskers [max and min range], and outlying points)

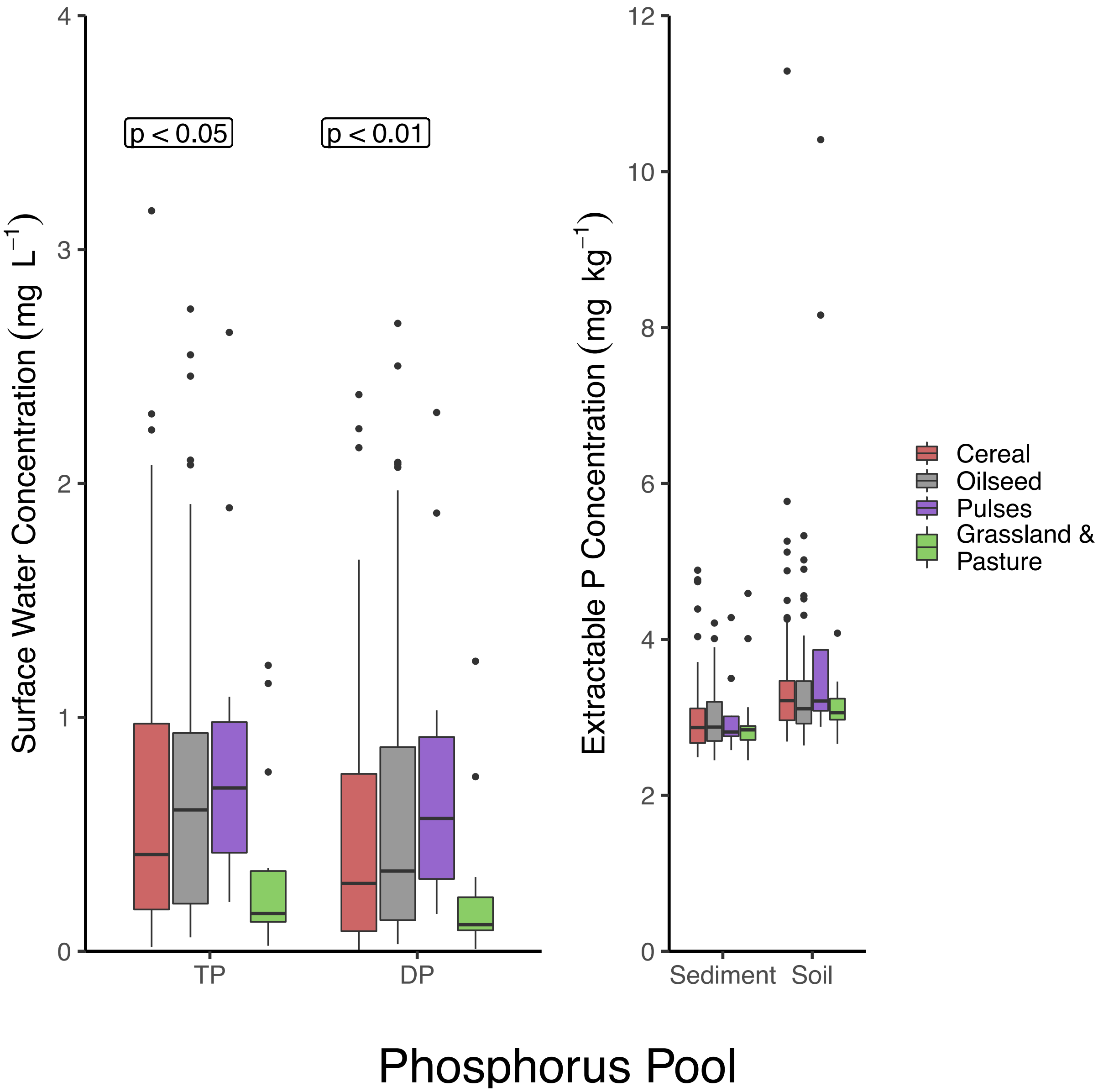


**Figure A.7** Boxplot comparison between land uses (oilseed, cereal, pulses, grassland and pasture) for concentrations of P in various P pools (TP, DP, sediment, and soil $CaCl_2$-extractable P; *p*-values reported from a Kruskal-Wallace test, and Wilcoxon corrected). The boxplot displays data distribution (median, hinges [25$^{th}$ and 75$^{th}$ percentiles], whiskers [max and min range], and outlying points).

**Table A.6** Ranking of models and parameter estimates explaining variation of pondwater, sediment, and soil P concentrations of sampled wetlands, including pond physiographic properties (pond perimeter, pond area, and P:A) (β(SE): beta of parameter estimates with standard error, AICc: Akaike's information criterion corrected for small sample size, *Δ*AIC: the difference between the best model and the other models in the dataset, K: the number of model parameters)

| P pools | | | β(SE) | AICc | ΔAIC | K |
|---|---|---|---|---|---|---|
| Pondwater TP | Variable | Land use–Pasture/Grassland | –0.364 (0.140) | – | – | – |
| | Variable | Pondwater pH | –0.161 (0.106) | – | – | – |
| | Model | Land use, Pondwater pH | – | 212.1 | 3.5 | **5** |
| | | Intercept & model structure only (statistical null) | – | 215.6 | 0 | **0** |
| Pondwater DP | Variable | Land use | –0.378 (0.117) | – | – | – |
| | Model | Land use | – | 193.3 | 7.8 | **4** |
| | | Intercept & model structure only (statistical null) | – | 201.1 | 0 | **0** |
| Pondwater DRP | Variable | Saline Group | 0.531 (0.154) | – | – | – |
| | Model | Saline Group | – | 72.5 | 8.3 | **4** |
| | | Intercept & model structure only (statistical null) | – | 80.8 | 0 | **0** |
| Porewater DP | Variable | Clay (%) | –0.0463 (0.0113) | – | – | – |
| | Variable | Sediment pH | –0.0347 (0.0125) | – | – | – |
| | Variable | Alkalinity | 0.000363 (0.000104) | – | – | – |
| | Model | Total Carbon (%), Sediment pH, Alkalinity | – | –24.7 | –15 | **5** |
| | | Intercept & model structure only (statistical null) | – | –9.7 | 0 | **0** |
| $CaCl_2$-Extractable sediment P | Variable | Total Carbon (%) | 0.0394 (0.0111) | – | – | – |
| | Model | Total Carbon (%) | – | 60.6 | 8.7 | **4** |
| | | Intercept & model structure only (statistical null) | – | 69.3 | 0 | **0** |
| $CaCl_2$-Extractable soil P | Variable | Pond Permanence–Seasonal | –0.490 (0.196) | – | – | – |
| | Variable | Pond Permanence–Semi-Permanent | –0.337 (0.201) | – | – | – |
| | Variable | Soil pH | –0.337 (0.182) | – | – | – |
| | Variable | Land use–Pasture/Grassland | –0.419 (0.164) | – | – | – |
| | Model | Pond Permanence, soil pH, Land use | – | 236.3 | 5.1 | **7** |
| | | Intercept & model structure only (statistical null) | – | 241.4 | 0 | **0** |

**Table A.7** Spearman rank correlation coefficients among concentrations of pondwater alkalinity with IC sediment and soil

| | Sediment IC | Soil IC |
|---|---|---|
| Alkalinity | 0.05 | 0.09 |

Correlations are denoted by $^{*}$($p \leq 0.05$) or a $^{+}$ ($p \leq 0.01$) (with *p*-values FDR corrected)