

# **Identification of Priority Areas for Wetland Restoration and Conservation: a Hydrogeochemical Study**

by

Md Aminul Haque

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Department of Geological Sciences

University of Manitoba

Winnipeg, Manitoba, Canada

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## **Abstract**

Very few studies have attempted to identify priority criteria for wetland conservation and restoration in the Prairie Pothole Region (PPR), based on specific hydrology- and biogeochemistry-related watershed benefits. To guide the development of such criteria, the objectives of this Ph.D. thesis were to: 1) infer wetland-stream interaction dynamics; 2) study the influence of spatial characteristics on wetland hydrologic function; 3) investigate wetland hydrologic response to individual rainfall-runoff events; and 4) relate wetland soil and hydrological characteristics to wetland biogeochemical function. For the purpose of this thesis, sites (i.e., ten intact, three consolidated and ten fully drained wetlands and a nearby creek) within the Broughton's Creek Watershed (Manitoba, Canada) were monitored over two years for water levels and water-soluble reactive phosphorus (SRP) concentrations. Soil cores were collected to measure soil physiochemical properties such as equilibrium phosphorus concentration. Hydrological behavioural metrics were computed to reflect year-specific, season-specific and event-specific wetland dynamics and were then correlated to wetland physical characteristics to identify landscape controls on wetland hydrologic function. Water column SRP concentrations and soil physiographic properties were also used to infer the time-variable source versus sink behaviour of each of wetland and the factors controlling this behavior. Results indicate a non-stationary behaviour of wetland hydrologic response across different antecedent moisture conditions. No hydrologic response metric showed positive or negative correlations with any of the spatial characteristics for more than 30% of the monitored events. Intact wetlands appeared more likely to provide specific watershed benefits (e.g., water and nutrient retention), compared to consolidated wetlands. Wetland equilibrium phosphorus concentrations ranged between 0 and 2 mg P L<sup>-1</sup>, but wetland phosphorus retention potential could not be related to any landscape

characteristics. Conversely, the area of individual wetlands – together with other landscape characteristics (e.g., storage volume and perimeter) – arose as key surrogate variables to predict some aspects of wetland hydrologic function and were therefore good candidates for developing simple criteria for wetland conservation and restoration. The wetland conservation and restoration scenarios presented at the end of this Ph.D. thesis provide good examples of such criteria and should be confirmed or infirmed in future studies.

## **Acknowledgments**

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## **Dedication**

This thesis is dedicated to three of the most caring persons I have ever seen, my mother Aziza Akter, my father Akhtar Hossain and my lovely wife, Amina Akter

## Table of Contents

<b>Abstract.....</b>	<b>i</b>
<b>Acknowledgments .....</b>	<b>iii</b>
<b>Dedication .....</b>	<b>iv</b>
<b>Table of Contents .....</b>	<b>v</b>
<b>List of Tables .....</b>	<b>x</b>
<b>List of Figures.....</b>	<b>xii</b>
<b>List of Appendices.....</b>	<b>xix</b>
<b>Chapter 1: Background and Rationale .....</b>	<b>1</b>
1.1    Introduction to the Biological, Hydrological and Chemical Importance of Wetlands.....	2
1.2    Wetland Hydrology .....	4
1.3    Wetland Biogeochemistry .....	9
1.4    Wetland Ecology .....	11
1.5    Wetland Conservation and Restoration.....	12
1.6    Research Objectives .....	14
1.7    Study Site Description and Data collection.....	16
1.7.1    Broughton’s Creek Watershed.....	16
1.7.2    Study Sites, Instrumentation and Data Collection .....	18
1.7.3    Laboratory Analyses .....	21
1.8    Ph.D. Thesis Structure.....	21
1.9    Significance of the Ph.D. study .....	23
1.10    Contributions of Authors.....	24
1.11    Connectivity between Wetlands and Streams: Preliminary Analysis .....	25

1.11.1	Introduction.....	25
1.11.2	Study Site and Methods .....	26
1.11.3	Results.....	27
1.11.4	Discussion and Conclusion .....	27
1.12	References .....	30
<b>Chapter 2: Hydrological Dynamics of Prairie Pothole Wetlands: Dominant Processes and Landscape Controls Under Contrasted Conditions .....</b>		<b>39</b>
<b>Abstract.....</b>		<b>40</b>
2.1	Introduction .....	42
2.2	Methods.....	46
2.2.1	Study Site Description and Data Collection .....	46
2.2.2	Data Analysis .....	52
2.3	Results .....	55
2.3.1	Surface Water Storage and Ditch Water-table Dynamics.....	55
2.3.2	Wetland-Stream Interactions .....	60
2.3.3	Landscape Controls on Wetland Hydrological Behavior .....	60
2.4	Discussion .....	66
2.4.1	Dominant Processes Affecting Open-water Wetlands.....	66
2.4.2	Dominant Processes Occurring Downslope of Fully Drained Wetlands.....	70
2.4.3	Strength and Temporal Variability of Wetland – Stream Interaction.....	71
2.4.4	Predictability of Wetlands Dynamics Based on Landscape Characteristics.....	73
2.5	Conclusion.....	76
2.6	Acknowledgments .....	77

2.7	References .....	78
<b>Chapter 3: Event-Based Analysis of Wetland Hydrologic Response in The Prairie Pothole Region..... 88</b>		
	<b>Abstract.....</b>	<b>89</b>
3.1	Introduction .....	91
3.2	Methods .....	96
3.2.1	Study Site Description .....	96
3.2.2	Hydrometric and Climate Data Collection .....	100
3.2.3	Rainfall-Runoff Event Delineation and Response Metrics.....	100
3.2.4	Statistical Analyses .....	104
3.3	Results .....	106
3.3.1	Event Characteristics and Metrics of Wetland Hydrologic Response .....	106
3.3.2	Temporal Variability of Individual Wetland Hydrologic Response.....	110
3.3.3	Spatial and Temporal Variability of Wetland-Stream Interaction.....	113
3.3.4	Temporal Persistence of Spatial Controls on Individual Wetland Hydrologic Response.....	115
3.4	Discussion .....	120
3.4.1	Effective Metrics of Wetland Hydrologic Response .....	120
3.4.2	Temporal Variability in Hydrologic Response .....	122
3.4.3	Characterizing Wetland-Stream Interaction .....	124
3.4.4	Predictability of Spatial Controls on Wetland Hydrologic Response.....	126
3.5	Conclusion.....	128
3.6	Acknowledgments .....	129

3.7	References .....	130
<b>Chapter 4: Hydroclimatic Influences and Physiographic Controls on Phosphorus Dynamics</b>		
<b>in Prairie Pothole Wetlands .....</b>		<b>139</b>
<b>Abstract.....</b>		<b>140</b>
4.1	Introduction .....	142
4.2	Methods .....	146
4.2.1	Study Site Description .....	146
4.2.2	Climate, Hydrometric and Water Quality Data Collection.....	149
4.2.3	Soil Sampling and Physiochemical Analysis.....	151
4.2.4	Graphical and Statistical Analyses.....	153
4.3	Results .....	155
4.3.1	Spatial and Temporal Variability of Wetland Hydrology and Water Quality .....	155
4.3.2	Wetland Soil Physiochemical Properties and P-Sorption Dynamics.....	156
4.3.3	Potential Controls on Wetland P-Sorption Dynamics .....	161
4.4	Discussion .....	163
4.4.1	Linking Hydrology and Water Quality in Pothole Wetlands.....	163
4.4.2	Characterizing the Spatial Variability of Wetland Soil Properties .....	167
4.4.3	Classifying Wetlands According to Their P-Sorption Dynamics .....	170
4.4.4	Identifying Controls on P Dynamics in Pothole Wetlands .....	172
4.5	Conclusion.....	174
4.6	Acknowledgements .....	175
4.7	References .....	177
<b>Chapter 5: Synthesis and Conclusions.....</b>		<b>188</b>

5.1	Summary of Major Findings .....	189
5.2	Candidate Metrics and Criteria for Wetland Conservation and Restoration.....	192
5.3	Implications for Wetland Conservation and Restoration in the Broughton's Creek Watershed.....	196
5.4	Limitations of the Present Study and Recommendations for Future Studies.....	204
5.5	References .....	206
<b>Appendices</b> .....		207
<b>Appendix A:</b> Supplemental Materials Related to Chapter 2.....		208
<b>Appendix B:</b> Supplemental Materials Related to Chapter 4.....		213

## List of Tables

<b>Table 1-1:</b> Summary of controlling factors related to P retention and release. Factors listed in bold and underlined represent most common factors for all wetlands. Factors listed in bold represent unique factors for the respective category. ....	11
<b>Table 1-2:</b> Water samples collected during the 2013 and 2014 open water seasons. ....	21
<b>Table 1-3:</b> Spatial wetland characteristics. I: Intact wetland, C: Consolidated wetland, D: Drained wetland, F: Forest, A: Agriculture. ....	29
<b>Table 2-1:</b> Overview of landscape characteristics associated with the instrumented wetlands. ..	50
<b>Table 2-2:</b> Metrics of open-water wetland behaviour and ditch water-table behaviour .....	54
<b>Table 3-1:</b> Overview of wetland spatial characteristics for open water (i.e., current) and drained (i.e., historic) prairie pothole wetlands. I: intact wetlands; C: consolidated wetlands; D: drained wetlands. ....	99
<b>Table 3-2:</b> Rainfall-runoff events identified from stream hydrographs, and corresponding responses of monitored open water and drained wetlands (PWs). TR = Total event rainfall. ARI = average (event) rainfall intensity. ....	102
<b>Table 3-3:</b> Hydrologic response metrics estimated for each wetland in association with each rainfall-runoff event. ARX = Antecedent rainfall in the “X” days prior to the start of the rainfall event under consideration. ....	103
<b>Table 3-4:</b> Descriptive statistics for the hydrologic response metrics estimated across all instrumented wetlands and all rainfall events. Bold text refers to drained wetlands.....	107
<b>Table 4-1:</b> Landscape characteristics estimated for each of the studied prairie pothole wetlands. Std: standard deviation.....	149
<b>Table 4-2:</b> Physiochemical wetland soil properties inferred from the collected soil cores.....	152

<b>Table 4-3:</b> Behavioral metrics used to compare local hydrological dynamics across the studied prairie pothole wetlands. ....	155
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<b>Table 5-1:</b> Examples of hydrologic response metrics and wetland landscape characteristics that were identified as significantly correlated (95% significance level) at the end of Chapter 2 and 3. For abbreviated wetland characteristics and response metric names, refer to Tables 2-1, 2-2, 3-1 and 3-3. ....	196
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## List of Figures

<b>Figure 1-1:</b> Effects of hydrology on wetland functions and biotic feedbacks (from Mitsch and Gosselink, 2007). .....	6
<b>Figure 1-2:</b> Broughton’s Creek Watershed and sampling site locations around a 5 km study reach. ....	19
<b>Figure 1-3:</b> (a) Autosampler located at the south end of the study reach; (b) Stilling well in an intact wetland; (c) Perched water table well (right) and three nested piezometers in a drainage ditch; (d) Time-lapse camera. ....	20
<b>Figure 1-4:</b> SRP distribution in different wetlands gaining and losing conditions at downstream. Each box has lines at the lower quartile, median, and upper quartile values, while the whiskers extend from each end of the box to show the extent of the rest of the data (minimum and maximum values). Outliers are shown as ‘+’ in red. ....	28
<b>Figure 2-1:</b> (a) Location of the Broughton's creek watershed within Canada. (b) Creek reach and surrounding land under study. (c) Instrumented wetlands in the Broughton's Creek Watershed. I: Intact wetland, C: Consolidated wetland, D: Drained wetland (s) and associated ditch (es). The black cross shows the creek water level measuring location. (d) Example of consolidated wetland, C2; dashed circles indicate historic wetlands that have been drained into C2. ....	47
<b>Figure 2-2:</b> Comparison of meteorological and hydrometric data available for the two study years. (a) and (b) Creek water level and total precipitation for 2013 and 2014, respectively. (c) Measures of antecedent storage for the years 2013 and 2014, relative to long-term means for the study region. Soil moisture data were obtained from AAFC (2017) while temperature and precipitation data were obtained from ENR (2017). For additional details, see Appendix A-1. ....	53

<b>Figure 2-3:</b> Classification of wetland surface water storage dynamics observed in 2013 and 2014. I: Intact wetland; C: Consolidated wetland. For the raw hourly wetland fullness time series, see Appendix A-2.....	56
<b>Figure 2-4:</b> Water-table fluctuations associated with drainage ditches in (a) 2013 and (b) 2014. Dashed lines show the surface elevation of the ditches.....	58
<b>Figure 2-5:</b> (a–f) Year-specific and season-specific variation of behavioural metrics for open-water wetlands and (g–l) ditch water-table dynamics. Each boxplot has lines at the lower quartile, median, and upper quartile values, while the whiskers extend from each end of the box to show the range of the data. Outliers are shown as “+.” For abbreviated metric names, refer to Table 2-2. The numbers “13” and “14” juxtaposed to season names (e.g., Summer13 and Summer14) refer to the years 2013 and 2014, respectively. ....	59
<b>Figure 2-6:</b> Spearman's rank correlation coefficients (rho values) between wetland water level or ditch water-table time series and creek water level time series for different time periods. I: Intact wetland; C: Consolidated wetland; D: Drained wetland (s). The numbers “13” and “14” juxtaposed to season names (e.g., Summer13 and Summer14) refer to the years 2013 and 2014, respectively. “X” signals a period for which no data were collected. Blank cells flag time periods for which the Spearman's rank correlation coefficient was not statistically significant at the 95% level. For correlation coefficient values in tabular format, refer to ESM3. ....	61
<b>Figure 2-7:</b> Spearman's rank correlation coefficients (rho values) between behavioural metrics and wetland characteristics for open-water wetlands for (a) the 2013 and 2014 whole open-water seasons, (b) Spring 2013 and 2014, (c) Summer 2013 and 2014, and (d) Fall 2013 and 2014. For abbreviated wetland characteristics and behavioural metric names, refer to tables 2-1 and 2-2. The numbers “13” and “14” juxtaposed to abbreviated metric names refer to the years 2013 and 2014,	

respectively. Blank cells flag time periods for which the Spearman's rank correlation coefficient between a wetland characteristic and a behavioural metric was not statistically significant at the 95% level ..... 62

**Figure 2-8:** Spearman's rank correlation coefficients (rho values) between behavioural metrics and historic wetland characteristics associated with drainage ditches for (a) the 2013 and 2014 whole open-water seasons, (b) Spring 2013 and 2014, (c) Summer 2013 and 2014, and (d) Fall 2013 and 2014. For abbreviated historic wetland and ditch characteristics and behavioural metric names, refer to tables 2-1 and 2-2. The numbers “13” and “14” juxtaposed to abbreviated metric names refer to the years 2013 and 2014, respectively. Blank cells flag time periods for which the Spearman's rank correlation coefficient between a drained wetland (or ditch) characteristic and a behavioural metric was not statistically significant at the 95% level ..... 64

**Figure 2-9:** Qualitative comparison of hydrological dynamics for open-water wetlands sharing similar landscape characteristics. The range of wetland fullness value on y-axis is 0 to 1.5. Axis tick labels were not included to ensure better readability..... 65

**Figure 3-1:** (a) Location of the Broughton’s creek watershed within Canada. (b) Creek reach and surrounding land under study. (c) Instrumented wetlands in the Broughton’s creek watershed. I: Intact wetland, C: Consolidated wetland, D: Drained wetland(s) and associated ditch(es). The black cross shows the creek water level measuring location. (d) Example of consolidated wetland, C2; dashed circles indicate historic wetlands that have been drained into C2. .... 97

**Figure 3-2:** Examples of creek response (i.e., stream hydrograph) and wetland response (surface and subsurface water level) during individual rainfall-runoff events. .... 104

**Figure 3-3:** Scatter plots of average event rainfall intensity (mm/day) versus total event rainfall (mm). Circles represent individual events. The radius of the circles is proportional to the wetland

fullness or ditch water table rise rate for the respective event; it provides a qualitative way of assessing the correlation – or lack thereof – between rainfall intensity, total rainfall and water level (surface or subsurface) rise rate. .... 108

**Figure 3-4:** Wetland hydrologic response metrics with PCA loadings of more than 0.50 for either the first or the second principal component. Blue circles represent open water wetlands and red circles represent drained wetlands. .... 109

**Figure 3-5:** Spatio-temporal variability of selected wetland hydrologic response metrics. Blue-colored boxes represent open water wetlands (i.e., I1 to C3) while red-colored boxes represent drained wetlands (i.e., D1 to D7). Each box has lines at the lower quartile, median, and upper quartile values, while the whiskers beyond the box show the extent of the rest of the data. Outliers are shown as ‘+’. An outlier is a value that is more than 1.5 times the interquartile range away from the top or the bottom of the box. .... 111

**Figure 3-6:** Spatio-temporal variability of recession dynamics observed across instrumented wetlands. “Partial recession” means that the final wetland water level (or ditch water table level) observed at the end of an event was noticeably higher than the initial value observed at the beginning of an event. “Full recession” rather means that the final wetland water level (or ditch water table level) was similar or lower than the initial value observed at the beginning of an event. .... 112

**Figure 3-7:** Examples of hysteresis patterns observed for selected wetlands and events. TR = Total event rainfall. .... 114

**Figure 3-8:** Temporal distribution of different hysteresis types observed for each instrumented wetland across all 15 rainfall-runoff events under consideration. I = Intact wetland, C = Consolidated wetland, D = Drained wetland. Hysteresis types include: L = Linear (i.e., no

hysteresis), CW = Clockwise, CCW = Counter-Clockwise, ES = Eight-Shaped, and U = Unclear.

..... 115

**Figure 3-9:** Temporal persistence of spatial controls on selected wetland hydrologic response metrics for open water wetlands across all events. Each colored segment of the pie chart represents a percentage of monitored rainfall events for which the correlation coefficient between the column variable (wetland spatial characteristic) and the row variable (wetland response metric) was statistically significant (i.e.,  $p\text{-value} < 0.05$ ). Light gray: positive correlation; black: negative correlation. See Tables 3-1 and 3-3 for the descriptions of wetland spatial characteristics and wetland response metrics, respectively..... 118

**Figure 3-10:** Temporal persistence of spatial controls on selected wetland hydrologic response metrics for drained wetlands across all events. Each colored segment of the pie chart represents a percentage of monitored rainfall events for which the correlation coefficient between the column variable (wetland spatial characteristic) and the row variable (wetland response metric) was statistically significant (i.e.,  $p\text{-value} < 0.05$ ). Light gray: positive correlation; black: negative correlation. See Tables 3-1 and 3-3 for the description of wetland spatial characteristics and wetland response metrics, respectively..... 119

**Figure 4-1:** Prairie pothole wetlands and creek reach under study within the Broughton's Creek Watershed. I = Intact wetland, C = Consolidated wetland. .... 148

**Figure 4-2:** Variability of SRP concentrations in the water column of the studied wetlands. Each box has lines at the lower quartile, median, and upper quartile values, while the whiskers beyond the box show the extent of the rest of the data. Outliers are shown as red plus (“+”) signs..... 156

**Figure 4-3:** Timeseries of wetland fullness (daily data) and SRP concentrations (biweekly data) in each of the studied wetlands across the study period. Biweekly data points are linked to produce a

SRP timeseries for easier visualization and comparison with the wetland fullness timeseries. The blue vertical dashed line separates 2013 data (left) from 2014 data (right). The red horizontal dashed lines represent the EPC value for each wetland. rho is the Spearman's rank correlation coefficient between wetland fullness and SRP concentrations for each wetland. '\*' signals correlation coefficients that were statistically significant at the 95% level (i.e., p-value < 0.05). Missing data points on the timeseries signal the malfunction of a water level logger (e.g., wetland I9) or dry conditions..... 157

**Figure 4-4:** Texture triangle showing the variability in wetland soil particle size among the studied wetlands. .... 158

**Figure 4-5:** Wetland-specific and depth-specific (a) organic matter; and (b) and (c) total, organic and inorganic carbon content estimated from the collected soil cores. OM: organic matter. OC: organic carbon. IC: inorganic carbon..... 159

**Figure 4-6:** (a) Temporal variability of P-source (blue cells) versus P-sink (red cells) behaviour for each of the studied wetlands. White cells represent dry conditions or no data. (b) and (c): Temporal variation of daily potential evapotranspiration (PET) and antecedent rainfall (AR) across all sampling dates. ARX: cumulative rainfall for the "X" days prior to each sampling date. The blue vertical dashed line separates 2013 data (left) from 2014 data (right)..... 162

**Figure 4-7:** Differences in soil physiochemical properties between wetlands that persistently acted as P sinks (i.e., S group) and wetlands that switched between P-sink and P-source behaviour (i.e., SS group) during the 2013 and 2014 open water seasons. Each box has lines at the lower quartile, median, and upper quartile values, while the whiskers beyond the box show the extent of the rest of the data. Outliers are shown as red plus signs. KWP: p-value associated with the Kruskal-Wallis test. .... 164

<b>Figure 4-8:</b> Differences in wetland hydrologic response characteristics between wetlands that persistently acted as P sinks (i.e., S group) and wetlands that switched between P-sink and P-source behaviour (i.e., SS group) during the 2013 and 2014 open water seasons. Each box has lines at the lower quartile, median, and upper quartile values, while the whiskers beyond the box show the extent of the rest of the data. Outliers are shown as red plus signs. KWP: p-value associated with the Kruskal-Wallis test.....	165
<b>Figure 4-9:</b> Differences in landscape characteristics between wetlands that persistently acted as P sinks (i.e., S group) and wetlands that switched between P-sink and P-source behaviour (i.e., SS group) during the 2013 and 2014 open water seasons. Each box has lines at the lower quartile, median, and upper quartile values, while the whiskers beyond the box show the extent of the rest of the data. Outliers are shown as red plus signs. KWP: p-value associated with the Kruskal-Wallis test. ....	166
<b>Figure 4-1:</b> Spatial distribution of %WetHFull metric values for open water wetlands (i.e., intact and consolidated) in the northwest corner of the BCW.....	198
<b>Figure 4-2:</b> Outcomes of a hypothetical wetland conservation scenario, according to which intact wetlands that have %WetHFull metric values of less than 50% would be selected in priority. Results are shown only for the northwest corner of the BCW. ....	201
<b>Figure 4-3:</b> Outcomes of a hypothetical wetland restoration scenario, according to which consolidated wetlands that have %WetHFull metric values of more than 50% would be selected in priority. Results are shown only for the northwest corner of the BCW. ....	202
<b>Figure 4-4:</b> Outcomes of a hypothetical wetland restoration scenario, according to which drained wetlands that have %WetHFull metric values of less than 50% would be selected in priority. Results are shown only for the northwest corner of the BCW. ....	203

## **List of Appendices**

<b>Appendix A-1:</b> Long-term historical data for (a) pre-season moisture deficit (computed as precipitation minus potential evapotranspiration, and (b) current season precipitation. In (a), pre-season values are computed between April 1 <sup>st</sup> of the year preceding the one indicated on the x-axis, and March 31 <sup>st</sup> of the year indicated on the x-axis. In (b), “Current season” refers to the open water season of the year indicated on the x-axis and ranges from April to October. To obtain the timeseries in panels (a) and (b), temperature and precipitation data were obtained from ENR (2017) (see manuscript for full bibliographic reference).....	210
<b>Appendix A-2:</b> Hourly wetland fullness timeseries for (a) 2013, and (b) 2014. ....	211
<b>Appendix A-3:</b> Spearman’s rank correlation coefficient values between creek water level and wetland fullness or ditch water table timeseries. The numbers “13” and “14” juxtaposed to season names (e.g., Summer13, Summer14) refer to the years 2013 and 2014, respectively. “X” signals a period for which no data were collected. Empty table cells indicate time periods for which the Spearman’s rank correlation coefficient was not statistically significant at the 95% level. ....	212
<b>Appendix B-1:</b> Long-term historical data for pre-season moisture deficit computed as precipitation minus potential evapotranspiration. Pre-season values are computed between April 1 <sup>st</sup> of the year preceding the one indicated on the x-axis, and March 31 <sup>st</sup> of the year indicated on the x-axis. To obtain the timeseries temperature and precipitation data were obtained from ENR (2017) .....	214

## **Chapter 1: Background and Rationale**

## **1.1 Introduction to the Biological, Hydrological and Chemical Importance of Wetlands**

Wetlands are the only type of ecosystems that have been deemed worthy of conservation through an international convention (Environment Canada, 2013). It has been estimated that Canada has about 24% of the world's wetlands and they cover about 14% of the country's total landmass (Warner & Rubec, 1997). In the last 100 years, 50% of the world's wetlands have been lost while in Canada, this loss exceeds 70% in some areas (Touzi, 2011). In the province of Manitoba alone, where 43% of the landmass is wetlands (Halsey et al., 1997), the current wetland loss rate is estimated to be 8.5 square miles per year (Yang et al., 2010). In Manitoba as well as in the rest of the Prairie Pothole Region (PPR), wetlands have been lost or degraded mainly because of the conversion of wetland basins to agricultural land. Usually, this involves the partial or total drainage of wetland basins or the consolidation of multiple wetland basins into one, and then using the drained wetland basins for cropping (Pattison et al., 2011).

Wetland loss, especially in the Prairie Pothole Region, has been positively correlated with a significant increase in downstream drainage areas, peak flow during floods (Wang et al., 2010), nutrient export (Brunet et al., 2012), sediment loading (Preston et al., 2013), and greenhouse gas emissions (Holloway, et al., 2011). Wetlands play an important role in maintaining good water quality in rivers and streams by filtering excess amounts of nutrients from surface runoff. They act as the kidneys of a watershed and can absorb significant amounts of available nitrogen and phosphorus, thus acting as a sink for these nutrients (Mitsch & Gosselink, 2007). Wetlands also act as a major sink for greenhouse gasses like CO<sub>2</sub> and they have the highest carbon density among terrestrial ecosystems (Kayranli, 2010). They are active sites of groundwater recharge and this, combined with their natural filtering properties, makes them critical to the provision of safe drinking water (U.S. EPA, 2006; Environment Canada, 2013). Due to their water purification

properties, wetlands have also been used for wastewater treatment: the establishment of constructed wetlands that mimic the processes governing natural wetlands has been proven a cost-effective way for wastewater and storm water treatment (U.S. EPA, 2001). Wetlands are also very helpful in attenuating floods: one acre of wetland can hold up to 1-1.5 million gallons of floodwater (U.S. EPA, 2001).

This Ph.D. thesis focuses on information needed to support the conservation or restoration of wetland function because when wetlands are degraded, many of the aforementioned benefits are no longer available. There have been numerous efforts to restore wetlands to mitigate the negative impact of their loss (Mitsch, 2005) and in most cases, wetland restoration efforts are driven by the policy of “no net loss”, which promotes the restoration of the same area of wetland that has been degraded or lost (Mitsch, 2005). Brooks et al. (2005) argued that this “no net loss” approach did not work, as it does not consider the functions of wetlands when deciding to restore them solely based on their surface area. Although research has shown that wetlands can be created and have a positive impact in places where there were no historical wetlands (Amon et al., 2005), it has been argued that, for watershed-scale benefits, suitable wetland restoration sites should be selected based on specific criteria (White & Fennessy, 2005). The main hypothesis of the current Ph.D. thesis is that these criteria should be driven by the hydrological and biogeochemical functions of wetlands. The following paragraphs therefore provide a succinct review of wetland hydrological and biogeochemical functions before highlighting the research gaps associated with wetland conservation and restoration and the unique contribution of this Ph.D. research.

## 1.2 Wetland Hydrology

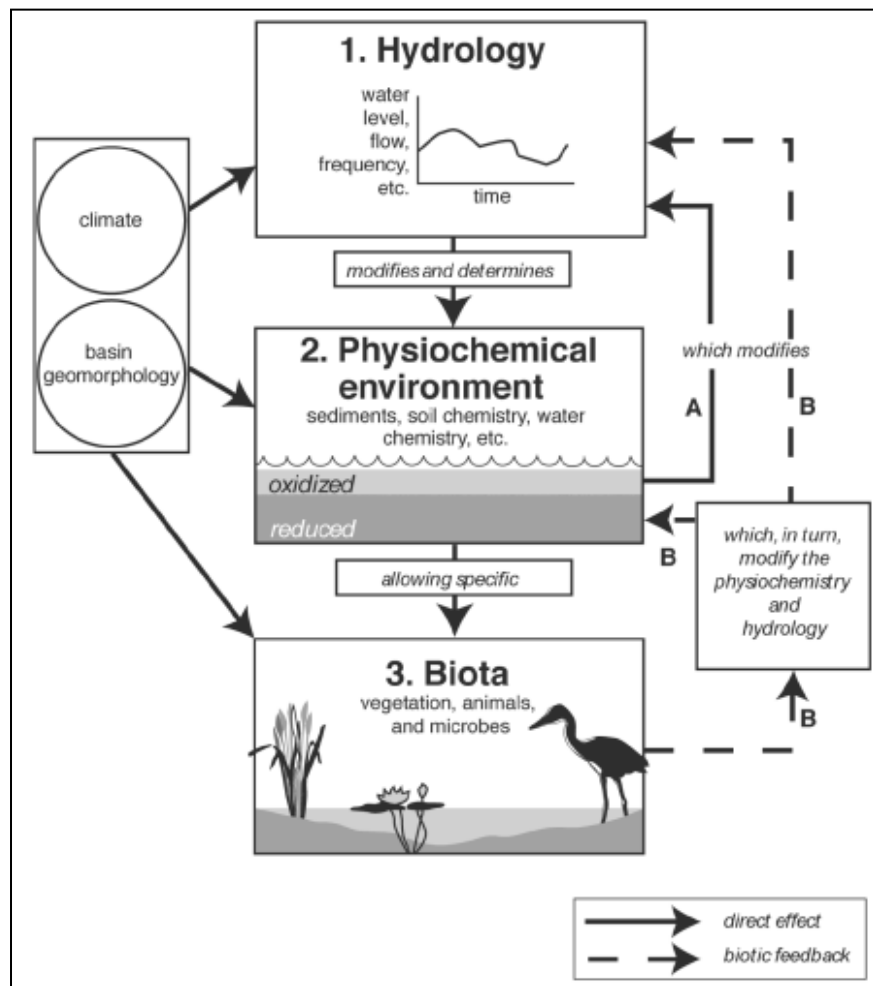
*“Hydrology is probably the single most important determinant of the establishment and maintenance of specific types of wetlands and wetland processes”* (Mitsch & Gosselink, 2007, p. 108). Indeed, wetlands are unique hydrologic units (Winter et al., 2003) whose positions largely depend on climate and basin geomorphology (Mitsch & Gosselink, 2007). Wetland hydrology (or the hydrologic state of a wetland) can be described using a few variables such as water level, hydropattern, water residence time, and groundwater and/or surface water exchanges (U.S. EPA, 2008; Mitsch & Gosselink, 2007). The water level simply refers to the elevation of the wetland water surface relative to the wetland bottom, and it can serve as an indicator of the dissolved oxygen state of the soil-water system. The hydroperiod describes the seasonal pattern of wetland water level (i.e., the frequency and duration of wetland soil saturation), and it depends on the balance between water inflows to and outflows from the wetland, basin geomorphology, and subsurface conditions. The hydropattern is a relatively new term that expands the concept of hydroperiod by incorporating timing and areal extent of saturation into it (U.S. EPA, 2008). Knowing the timing and areal extent of saturation together with the hydroperiod is useful to understand large wetland complexes that contain different wetland features (U.S. EPA, 2008; Mitsch & Gosselink, 2007). Water residence time refers to the travel time of water through a wetland: it is the ratio of the volume of water within the wetland to the rate of flow through the wetland, and it is often related to the hydropattern. A wetland can receive surface flows (inflows) of several types (e.g. overland flow, streamflow etc.) and can feed surface flows (outflows) through continuous, channelized connections with streams or via intermittent connections (e.g., over spilling). The interaction of wetlands with groundwater is often described as one of their most important attributes, though it is not true for all wetland types. In general, based on their interaction

with groundwater, wetlands can be of three types: (i) discharge wetlands that receive groundwater, (ii) recharge wetlands that recharge groundwater, and (iii) wetlands that have very little interaction with groundwater. Moreover, recharge wetlands can act as discharge wetlands and vice versa due to changes in hydrologic and basin climatic conditions (U.S. EPA, 2008; Mitsch & Gosselink, 2007).

The hydrology of wetlands influences abiotic factors such as, but not limited to, soil anaerobiosis, nutrient availability, and salinity (in coastal wetlands). Combined with water level fluctuations, these factors determine the plant and animal species that develop in a wetland. Wetlands biotic components also depend on wetland hydrology and other physiochemical features. Hydrology has a direct influence on physical and chemical wetland properties such as oxygen availability, pH, toxicity, and sediment and nutrient transport (Mitsch & Gosselink, 2007). Thus, hydrology controls species composition and richness, primary productivity, organic accumulation, and nutrient cycling in wetlands. Wetlands with a high flow-through of water and nutrients or pulsing hydroperiods have high productivity. Also, sequences of dry and wet conditions are more favorable than constant standing water for decomposition (Mitsch & Gosselink, 2007).

Figure 1-1 illustrates the effects of hydrology on wetland functions. Wetland functions such as maintaining downstream water quality and storing excess water during heavy precipitation are closely related to the concept of wetland-stream connectivity. Here the term ‘connectivity’ refers to the degree to which wetlands are connected to downstream water bodies and the degree of their interactions through different transport mechanisms (U.S. EPA, 2015). Since all parts of a watershed are connected to each other to some degree, the connectivity continuum can be described through the frequency, timing, magnitude, duration, and rate of change of physical and chemical fluxes between wetlands and streams (U.S. EPA, 2015). The degree of connectivity

varies with the placement of wetlands within a watershed. A recent literature review by the U.S. Environmental Protection Agency on wetland and stream connectivity has broadly classified wetlands into two categories, namely floodplain wetlands and non-floodplain wetlands (U.S. EPA, 2015); they have much common functionality but differ in their hydrologic behavior, specifically their connectivity to streams. Most non-floodplain wetlands – such as Prairie potholes – are considered as geographically isolated wetlands (GIWs) or upland-embedded wetlands (Mushet et al., 2015; Neff & Rosenberry, 2018; Tiner 2003) due to their lack of channelized connection with nearby streams or water bodies.



**Figure1-1:** Effects of hydrology on wetland functions and biotic feedbacks (from Mitsch and Gosselink, 2007).

It has been argued that “isolated” is a relative term and has many dimensions that can be expressed in both hydrologic and/or biotic terms. Thus, it is very difficult to conclude whether a wetland is fully isolated or fully connected; it is rather suggested to describe a wetland in the context of an isolation-connectivity continuum (Leibowitz, 2003). The isolation-connectivity continuum can be explained through combining the time and space continuum. On one hand, space isolation could be due to the distance between wetlands. On the other hand, time isolation can be expressed as the frequency of connection between wetlands (Leibowitz, 2003). Studies from Prairie regions, Carolina Bays, and coastal plains of Florida showed that wetland-stream connectivity can mainly occur through wetland water spillage, temporal or discontinuous surface runoff, near-surface water movement, and groundwater movement (Brunner et al., 2009; Pyzoha et al., 2008; Rains et al., 2006; Wilcox et al., 2011; Winter & LaBaugh, 2003). Depending on the geological, regional, and topographic settings, wetland-stream connectivity can exist thanks to one or more of these processes and thus, it should be viewed as a probability event (e.g., low versus high degree of connectivity) rather than defining it as a binary property (i.e., connected versus disconnected) (Leibowitz & Vining, 2003). Therefore, for important policy evaluations, a process-based understanding of wetland connectivity and isolation is needed to measure the extent to which a so-called GIW is connected to a stream, and how that connectivity varies across different temporal and spatial scales as well as across different geographic regions (Golden et al., 2014).

GIWs can influence overall watershed behavior in many ways. Some GIWs act as transient storage of water in wet periods and recharge groundwater in dry periods (Tiner, 2003). Prairie potholes can function as discharge, recharge, or flow-through systems, and reversals in the direction of flows occur and contribute to local and regional groundwater flows (Tiner, 2003; Winter & LaBaugh 2003; Rosenberry & Winter 1997). Even small isolated depressions

(< 1000 m<sup>2</sup>) in Prairie regions play significant roles in storing snowmelt water, increasing evapotranspiration and promoting groundwater infiltration; they can be connected to the overall hydrologic system or reduce water flow to streams by trapping water, thus influencing the hydrologic cycle on a watershed scale (Hayashi et al., 2003).

The ability of GIWs to reduce water flow to streams, thereby attenuating peak flows, is a key wetland function worth preserving or restoring (McAllister et al. 2000). Although sufficient evidence is present in the literature about the role of the GIWs in governing overall watershed behavior, studies addressing the hydrological and chemical function of GIWs across different timescales are still lacking in regions where these wetlands have a significant presence (e.g., the PPR). For example, research is needed to understand the fundamental hydrologic processes in GIWs; more specifically, to know the frequency, duration and amount of flow from GIWs to water bodies (U.S. EPA, 2015). Understanding these processes for GIWs in regions like the PPR is closely related to the concept of non-contributing areas (McCauley et al., 2014), i.e., areas that do not contribute runoff to the outlet of a watershed under normal conditions. Non-contributing areas can increase or decrease depending on the temporal variation of hydrologic connectivity between GIWs and streams (Shaw, 2009), but also change due to human alterations to the landscape. While intact potholes can prevent runoff water from reaching waterways and thus act as non-contributing areas, drained or altered wetlands do not have the same ability to disconnect basins from waterways. Continuous monitoring of wetland hydrology (e.g., water level, hydroperiod, etc.) will help us understand the fundamental hydrological processes dominant in GIWs in the PPR. **It is, however, unclear whether the changes in hydrologic wetland function (i.e., local wetland water storage, wetland-stream interaction) can be predicted based on the degree of wetland alteration, and that knowledge gap represents a new research avenue to explore.**

### **1.3 Wetland Biogeochemistry**

Influenced by wetland hydrology, wetland biogeochemistry involves the transformation and transport of elements via physical, chemical and biological processes that are unique to wetland environments and determine overall wetland productivity (Figure 1-1). Water quality functions of GIWs largely depend on hydrologic connectivity since it can change the pH, specific conductance and salinity of water in GIWs (Leibowitz & Vining, 2003). Usually, GIWs act as sinks for nutrients and other chemical constituents through sediment retention, phosphorus adsorption, and denitrification; thus, GIWs act as long time storage vessels for a wide variety of chemicals, and the loss of GIWs can cause downstream water degradation (Whigham & Jordan 2003). Prairie pothole wetlands usually receive nutrients like phosphorus (P) through dry deposition, shallow subsurface flow, surface runoff, and occasional flooding (Brunet & Westbrook, 2012), and similar to other GIWs, they can provide long-term retention of P (Hauer, et al., 2002). Accumulation of silt/clay-sized particles means that the wetland is receiving relatively more P as silt/clay particles are suitable for P transport (Preston et al., 2013). Loss of P from this type of wetlands occurs through groundwater seepage and subsurface flow of dissolved P, and during spilling. These processes are controlled by the hydrologic condition of wetlands (Hauer, et al., 2002).

In general, wetlands store P in different forms (mainly particulate) by different processes (mainly by sedimentation). However, at the same time wetlands can mobilize stored P via different processes such as dissolution, desorption and decomposition of organic matter. The balance between storage and mobilization determines a wetland's role as a source or a sink. If P storage processes are dominant, then the wetland will act as a sink and vice versa (White et al., 2000). In a laboratory experiment, P release from re-wetted soil is limited within few months (Aldous et al.,

2007) whereas in an actual field scenario, P can be released from re-wetted sediments for several years depending of Al- and Fe-bounded P in soil (Kinsman-Costello et al., 2014). A significant amount of P released due to re-wetting can be recycled within the wetland by transforming the mobilized reactive soluble P into less reactive forms (Kinsman-Costello et al., 2014). Table 1-1 shows a list of wetland properties/parameters that are mentioned in the literature as controlling factors for P release or retention mechanisms. Similar parameters are grouped together, e.g., sediment properties include sediment redox potential, size fraction, organic matter, Fe and Al content. There are some common factors like redox potential, sediment characteristics, and nutrient loading that were reported both for retention and release mechanisms. On the other hand, a few factors are specific to the retention or release of P in wetlands. **One research gap is that those biogeochemical factors have not been extensively studied in all wetland settings, and very few studies have been done on P sorption/desorption mechanisms in Prairie pothole wetlands.** As pothole wetlands are GIWs and therefore hydrologically different from the floodplain wetlands that are the focus of most past studies, it is unclear whether P sorption/desorption processes are also different between these systems. An improved understanding of wetland hydrologic and biogeochemical functions and the interaction between them is therefore needed for Prairie potholes. It has been hypothesized that the examination of wetland landscape characteristics, soil properties and water quality will give us valuable insights into their role as nutrient sinks or sources (Dunne et al., 2006; Holloway et al, 2011; White et al., 2000).

**Table 1-1:** Summary of controlling factors related to P retention and release. Factors listed in bold and underlined represent most common factors for all wetlands. Factors listed in bold represent unique factors for the respective category.

Intact Wetlands		Restored Wetlands	
Factors controlling the retention of P	Factors controlling the release of P	Factors controlling the retention of P	Factors controlling the release of P
<ul style="list-style-type: none"> <li>• <b><u>Sediment properties</u></b></li> <li>• Nutrient load</li> <li>• Hydraulic residence time</li> <li>• <b>Channelization</b></li> <li>• <b>P concentration gradient</b></li> </ul>	<ul style="list-style-type: none"> <li>• <b><u>Sediment properties</u></b></li> <li>• Nutrient load</li> <li>• Sediment pore water P</li> <li>• Precipitation event</li> </ul>	<ul style="list-style-type: none"> <li>• <b><u>Sediment properties</u></b></li> <li>• Nutrient load</li> <li>• Hydraulic residence time</li> <li>• <b>Hydraulic diffusion</b></li> <li>• Location</li> <li>• <b>Organic amendments</b></li> <li>• <b>Sulfate reduction</b></li> <li>• <b>Wetland size to drainage area ratio</b></li> </ul>	<ul style="list-style-type: none"> <li>• <b><u>Sediment properties</u></b></li> <li>• Fe/P ratio</li> <li>• <b>Historic P content</b></li> <li>• <b>Water level fluctuation</b></li> </ul>

#### 1.4 Wetland Ecology

As described in Figure 1-1, wetland hydrology and climatic variability are the key drivers of ecology in wetland environments. Studies have shown strong linkages between ecological response and water regime and noted the influence of groundwater and surface water interactions on wetland ecology (Casanova & Brock, 2000; Jolly et al., 2008). Vegetation is a key component of wetland ecology, and a wetland's hydropattern can influence the establishment and community shift of specific plant populations, which eventually have an impact on wetland biogeochemistry (Casanova & Brock, 2000; Gerritsen, 1989). For example, higher than usual soil P concentrations can have negative impact on indigenous plant communities, which act as short-term P storage. Loss of these indigenous plant communities can increase the water column P concentration due to the release of up to 75% of total plant associate P (Richardson, 1985; White 2004). Wetland plant

species such as helophytes can absorb P from water by incorporating it into their growing parts. These species usually release the adsorbed P to surface water through leaching and decomposition after dieback (Hoffmann et al., 2009). Concentrations of soluble reactive phosphorus can influence the uptake and retention of P by wetland vegetation. Richardson and Marshall (1986) have noted that a low level of P load is favorable to retain P in the litter-microorganism compartments through chemical sorption, while a higher P load can decrease the P removal efficiency up to 22% for the same species. They also noted that plant uptake of P can be negligible if there are seasonal variations of P concentration and the majority of P input occurs outside of the growing season. It is evident that wetland vegetation plays an important role in defining wetland P sink versus source behaviours. However, over a long timescale, plant and microbial uptake of P usually represents a short-term storage, as the stored P is eventually released back to the soil and water through decomposition. This is especially true in low-abundance P environments that restrict the uptake ability of plants (Richardson & Marshall, 1986; Verhoeven et al., 1990). Formation of peat under suitable conditions only is considered a sustainable long-term P storage (Rothe et al., 2016; Rydin & Jeglum, 2006). On a similar note, wetland vegetation plays an important role in defining wetland hydrologic behaviour during individual rainfall-runoff events and at seasonal scale; however, the influence of vegetation on wetland hydrology at the annual scale is minimal (Mitsch & Gosselink, 2007).

## **1.5 Wetland Conservation and Restoration**

The above paragraphs highlight the fact that degradation or loss of wetlands alters the overall hydrology and biogeochemistry of watersheds. In the last 100 years, more wetlands have been lost compared to the number of intact wetlands at present and as such, the conservation of

the remaining wetlands is not enough and restoring lost wetlands is important. Moreover, the annual rate of wetland loss is more than what we can restore (Bartzen et al., 2010). Wetland restoration has been successfully done in different parts of the world and in Canada in particular (Begley et al., 2012; Kaza & Bendor, 2013; Kennedy and Mayer, 2002; Oneal et al., 2008; Pattison et al., 2011; Stevens et al., 2003). Multiple studies have shown that restored wetlands can increase sediment and nutrient retention (Kennedy & Mayer, 2002), increase groundwater discharge and reduce flood intensity (Kennedy & Mayer, 2002). Restoration of historically isolated wetlands can provide additional phosphorus storage and could significantly influence nutrient cycling on a watershed scale (Dunne et al., 2006). The conservation and restoration of wetlands have many dimensions (hydrological, biogeochemical, ecological) but in the current thesis, conservation refers to maintaining wetlands in their existing hydrologic and biogeochemical state (function) while restoration refers to the re-establishment of lost/degraded wetlands to their historical hydrologic and biogeochemical state.

Though wetland restoration has been proven successful, it requires considerable financial support (Prato & Hey, 2006) and involves dealing with socioeconomic issues as most of the altered wetlands are on private properties. This brings to the forefront the need to prioritize areas for wetland conservation and restoration to ensure that critical wetland function (hydrological, biogeochemical) is preserved or re-established. Again, while the identification of critical areas for wetland conservation or restoration has many dimensions, including ecological ones, this Ph.D. thesis is only concerned with the hydrobiogeochemical aspects (i.e., wetland restoration for flood protection and water quality mitigation) that have received a tremendous amount of attention in the province of Manitoba. Here the term “priority areas” refers to those wetlands that should be preserved or restored in priority because their drainage will result (or has resulted) in the most

significant adverse impacts on water quality, watershed connectivity and flooding. **There is a lack of studies focusing on the development and use of hydrogeochemical criteria to identify priority areas for wetland restoration.** Such scientific criteria to prioritize restoration efforts in time are especially needed because due to landscape heterogeneity and various hydro-meteorological conditions, wetland functions and “efficiencies” are not spatially or temporally uniform. Since it is not possible to restore all pre-existing wetlands, and most wetland protection needs to occur in agriculturally dominated areas (where wetlands are privately owned), it is believed that future efforts should take two different forms: a) convincing landowners to preserve existing wetlands, and b) targeting priority areas for wetland restoration. Both aspects of this dual strategy will be addressed in the synthesis sections of the present thesis (i.e., section 6.2 and 6.3). It has been hypothesized that the observation of wetlands hydrologic and biogeochemical properties will be helpful to identify the drivers that govern the ability of said wetlands to perform critical wetland function (i.e., water quality improvement, flood protection) (Hayashi et al., 2016; Shaw, 2009; Shook et al., 2013). This approach, however, does not explicitly consider wetland ecological functions and may therefore lead to the preservation or restoration of hydrologic and biogeochemical benefits at the detriment of ecological benefits. Therefore, policy decisions (i.e., focusing on certain wetland functions) will play a vital role when choosing wetland conservation and restoration criteria.

## **1.6 Research Objectives**

As discussed above, due to different socioeconomic issues, the restoration of all degraded/lost wetlands back to historical levels is rarely possible. This shows the need for criteria to prioritize wetlands for conservation or restoration in order to achieve specific watershed benefits

(e.g., flood retention, nutrient control). Before the development of such criteria can be attempted, a clearer understanding of the effects of wetlands on watershed processes (e.g., connectivity between wetlands and streams) and the identification of specific wetland properties that govern wetland functions is needed. This will allow the future development of statistical or physics-based models for evaluating wetland conservation and restoration scenarios.

The overall aim of this Ph.D. work is therefore, to develop scientific criteria to identify priority areas for wetland conservation and restoration and, if possible, to use those criteria as a basis for extrapolating the outcomes of different wetland conservation and restoration scenarios. This Ph.D. thesis is guided by five main research objectives as outlined below:

**Research objective #1:** Infer wetland-stream interaction by comparing hydrologic and chemical signatures of streams and wetlands.

**Research objective #2:** Study the influence of spatial characteristics on wetland hydrologic function, specifically the ability of wetlands to store water and decrease peak flows in nearby waterways over annual to seasonal timescales.

**Research objective #3:** Examine wetland hydrologic response and wetland-stream interaction (together with their controls) during individual rainfall-runoff events.

**Research objective #4:** Relate wetland soil and hydrological characteristics to wetland biogeochemical function, specifically the ability of a wetland to act as a nutrient sink.

**Research objectives #5:** Combine the new process knowledge acquired through thesis objectives #1 to #4, so as to identify criteria to select priority areas for wetland conservation and restoration and explore simple wetland conservation and restoration scenarios.

## **1.7 Study Site Description and Data collection**

### **1.7.1 Broughton's Creek Watershed**

The Broughton's Creek Watershed (BCW) is located in southwestern Manitoba and is part of the Little Saskatchewan River Conservation District. Below are some of the study area characteristics that have been summarized from the Little Saskatchewan River Conservation District reports (2002, 2005) and a research report on the Broughton's Creek Watershed submitted to Ducks Unlimited Canada (DUC) by Yang et al. (2008).

The total area of the watershed is about 260 km<sup>2</sup>. The Broughton's Creek, originated from the Little Saskatchewan River, flows southeast into Lake Wahtopanah located about 4.8 km upstream of the town of Rivers, Manitoba. The Little Saskatchewan River flows into the Assiniboine River which, in turn, flows into the Red River and finally into Lake Winnipeg. Physiographically, the Broughton's Creek Watershed is part of the Prairie Pothole Region: it represents typical Prairie Pothole Region landscapes, which have been created on top of the surficial deposits left by the retreat of the Assiniboine glacial lobe from 20,000 to 12,000 B.C. Due to its geological settings, the area is broadly described as a hummocky till plain that comprises numerous potholes, sloughs, and lakes. The watershed has been further subdivided into two sub-regions based on topographic relief: although both sub-divisions share similar topography, the northeast sub-region has a higher average topographic relief (more than 3 m) and elevations higher than 550 m above sea level. On the other hand, the other sub-region that comprises the southern and southwest part of the watershed has low average topographic relief (less than 3 m) and elevations lower than 550 m above sea level. Within 40 km, the main stem of the Broughton's Creek drops about 140 m. The numerous undrained depressions in the Broughton's Creek Watershed range from potholes to small lakes. In addition, sloughs are also present: they are

relatively larger but shallower compared to potholes, usually elongated in the northwest to southeast direction.

Newdale association soils (i.e., mainly Orthic Black Chernozems) are found throughout the watershed and they are usually well drained. Other minor soils like Dorset, Dorkan, Eroded SI and Jaymar associations are also found in the areas adjacent to streams. Land use in the BCW consists of agriculture (72%), rangeland (11%), wetland (10%), forest (4%) and others (3%). Historical data shows the continuous loss or degradation of wetlands in the BCW: it has been estimated that between 1968 and 2005, about 6,000 wetland basins have been lost or degraded due to drainage for agricultural uses. This is about 70% of the total number of wetlands historically present in the watershed.

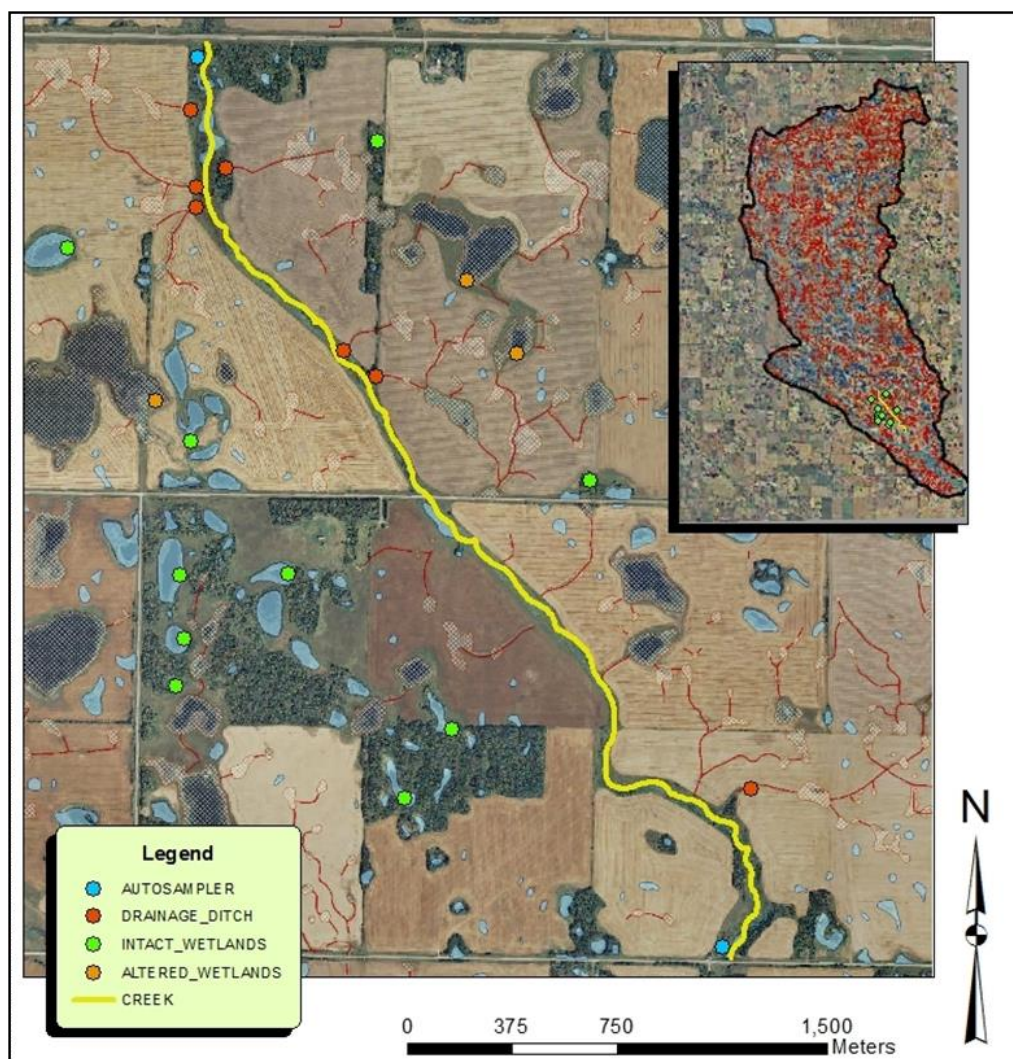
An application of the Soil Water Assessment Tool (SWAT) in the watershed has estimated that wetland loss has resulted in a 31% increase in contributing area, 37% increase in peak flow during heavy rainfall events, 62% increase in downstream water flow, 32% increase in phosphorus load, 57% increase in nitrogen load, and 85% increase in sediment load (Yang et al., 2010). These predictions highlight the necessity for wetland conservation and restoration. However, there is a significant lack of scientific knowledge about spatially and temporally varying hydrological processes, and that knowledge is needed to prioritize wetlands (for conservation or restoration) based on their importance or influence on overall wetland dynamics. Therefore, the reasons behind the selection of the Broughton's Creek Watershed for the study are mostly related to knowledge gaps and data availability. Knowledge gaps include but are not limited to: (i) a lack of understanding of sediment, nutrient and water movements in hummocky, pothole-dominated environments, and (ii) a lack of understanding of the influence of degraded wetlands and drainage ditches on stream-wetland interaction (i.e., connectivity). Background data for the present study

include: (i) a comprehensive wetland inventory dating back to 1968; (ii) high precision digital elevation (i.e., LIDAR) data; (iii) some insights on wetland dynamics obtained through research projects (mostly, model-based) conducted at that site in the last decade.

### **1.7.2 Study Sites, Instrumentation and Data Collection**

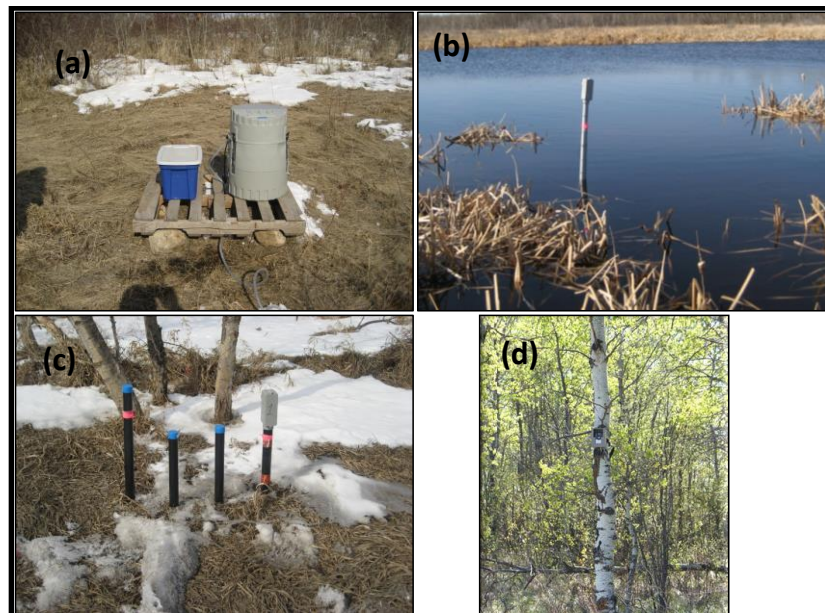
For the purpose of this study, a 5 km study reach was selected in the Broughton's Creek Watershed for continuous monitoring (Figure 1-2). The two extremities of the study reach, namely the north-end (upstream) and the south-end (downstream), were equipped with electronically controlled, battery-powered automated water-samplers (Figures 1-2 & 1-3a). Each autosampler uses a peristaltic pump and a tube to collect water samples from the creek at regular intervals (at least once every two days), thus allowing the collection of water samples between field visits. Samples were retrieved every two weeks during field visits and a new set of bottles was installed in each of the autosamplers to allow uninterrupted sampling. The batteries of the autosamplers were also recharged and replaced in a timely manner. The positioning of the autosamplers at the north and south ends of the study reach was to allow the characterization of upstream-downstream differences in water quality. A capacitance water level logger was also installed at the downstream end of the creek reach to record creek water level fluctuations at a 15-minute frequency. Ten (10) intact and undisturbed wetlands, three (3) consolidated wetlands, and seven (7) drainage ditches adjacent to the study reach were selected (see Figure 1-2 for the exact locations). The ditches were selected based on current and historical maps showing that their role, in the landscape, is to move runoff away from past wetlands (that have since been drained) towards the creek reach under study. Stilling wells (i.e., above ground wells) were deployed in the intact and consolidated wetlands to monitor fluctuations in surface water level (Figure 1-3b), while perched water table wells (i.e.,

below ground) were drilled below the drainage ditches (Figure 1-3c) to a depth of 1 m to monitor shallow groundwater fluctuations. All stilling wells and perched water table wells were equipped with capacitance water level loggers so that fluctuations in surface water and shallow groundwater could be recorded every 15 minutes. To investigate the chemical composition of wetland surface water, grab samples were taken in all 13 intact and consolidated wetlands during each site visit (once every two weeks).



**Figure 1-2:** Broughton's Creek Watershed and sampling site locations around a 5 km study reach.

Table 1-2 below summarizes the number of samples collected in 2013 and 2014. Lastly, a time-lapse camera was setup to capture overall site conditions (snow cover, water stage level, etc.) at one of the targeted intact wetlands (Figure 1-3d); pictures were captured every 15 minutes. In 2015, soil cores were collected from the wetlands identified in Figure 1-2 and the clay content and other physiochemical characteristics of the wetland soils were determined. Landscape characteristics (such as Area, Volume, Perimeter, Contributing Area, and Total Drainage Area) were estimated for the targeted wetlands based on the current and historic wetland inventory dataset. Those datasets were provided by Ducks Unlimited Canada and assembled on the basis of Canada Wetland Inventory specifications (Ducks Unlimited Canada, 2016) as well as field reconnaissance information.



**Figure 1-3:** (a) Autosampler located at the south end of the study reach; (b) Stilling well in an intact wetland; (c) Perched water table well (right) and three nested piezometers in a drainage ditch; (d) Time-lapse camera.

**Table 1-2:** Water samples collected during the 2013 and 2014 open water seasons.

Sampling locations	Number of samples	
	2013	2014
Creek reach – Upstream, north end	59	133
Creek reach – Downstream, south end	70	132
Intact and consolidated wetlands (surface water)	123	146

### **1.7.3 Laboratory Analyses**

All samples listed in Table 1-2 were tested for pH, electrical conductivity (EC), total dissolved solids (TDS) and salinity in the field using a handheld water-quality pocket tester (Eutech Instruments Multi-Parameter PCSTestr<sup>TM</sup> 35). Samples were then put on ice in a cooler and transported back to the University of Manitoba, Fort Garry Campus for further analysis. The external housing of the automated water samplers deployed at both ends of the study creek reach is designed to regulate changes in sample temperature throughout the scheduled sampling period; therefore, even the samples left in the autosampler for several days before retrieval were kept in cool conditions. Back to the University of Manitoba and in a laboratory setting, samples were filtered through a 0.45 µm membrane and soluble reactive phosphorus (SRP) concentrations were estimated using a United States Environmental Protection Agency (USEPA)-compliant colorimeter (LaMotte SMART3<sup>TM</sup> Colorimeter) and the associated reagents. In 2015, soil cores were collected, and the clay content of the soils was determined by the laser diffraction method with a Malvern Mastersizer 2000 particle size analyzer.

## **1.8 Ph.D. Thesis Structure**

This Ph.D. thesis is written in the Grouped Manuscript Style (Sandwich Thesis), following the guidelines of the Department of Geological Sciences and the Faculty of Graduate Studies at the University of Manitoba. The first chapter of this thesis (i.e., the current chapter) contains an

introduction to the research topics, background information for each of the research themes, research objectives, the study site description, a summary of data collection methods, and the general outline of the thesis. Chapters 2, 3 and 4 are the major data chapters of this thesis and have been structured as three separate manuscripts. Each data chapter addresses a research objective outlined in section 1.5. In summary:

- Section 1.11 addresses research objective #1 and is an extended abstract (i.e., short research paper) published in *Elements*, the newsletter of the Canadian Geophysical Union (Summer 2015, Volume 33, Number 2, page 7). This special, short publication was an invited one after I won the Campbell Scientific Award for best poster at the 2015 annual meeting of the Canadian Geophysical Union.
- Chapter 2 addresses research objective #2 and has already been published as: Haque, A., Ali, G., and Badiou, P. (2018). 'Hydrological dynamics of prairie pothole wetlands: dominant processes and landscape controls under contrasted conditions'. *Hydrological Processes*, 32: 2405–2422, doi: 10.1002/hyp.13173.
- Chapter 3 addresses research objective #3 and has been submitted to *Journal of Hydrology*, where it is currently under review.
- Chapter 4 addresses research objective #4 and has already been published as: Haque, A., Ali, G., Macrae, M., Badiou, P., and Lobb, D. (2018). 'Hydroclimatic influences and physiographic controls on phosphorus dynamics in prairie pothole wetlands'. *Science of the Total Environment*, 645: 1410-1424, doi: 10.1016/j.scitotenv.2018.07.170.

The three data chapters have been reformatted from the published or submitted versions for inclusion in this thesis. Chapter 5 addresses the fifth research objective while providing a synthesis

of the previous three data chapters. In addition to showing how chapter-specific findings are linked to one another, Chapter 5 also includes a section that addresses the implications of the thesis research results for wetland conservation and restoration, and it provides a list of study limitations as well as recommendations for future research.

## **1.9 Significance of the Ph.D. study**

This Ph.D. thesis aims to define hydrogeochemical criteria that could later be used to identify priority areas for wetland conservation and restoration. The chosen approach for establishing the criteria (listed in the concluding chapter, i.e., Chapter 5) was to examine the impact of wetlands (intact and altered) on the overall hydrological and geochemical functioning of a watershed (see specific objectives above). Because the focus of this Ph.D. research concerned the hydrogeochemical benefits of wetlands, the dynamics between shallow groundwater and wetland water in different undrained, partially drained and fully drained wetlands were examined and linked to wetland spatial characteristics (related to wetland geometric properties as well as landscape topographic and soil properties).

One novel aspect of this Ph.D. research is the use of high-frequency (15 minutes to biweekly) data to characterize the hydrological and biogeochemical function of prairie potholes along a continuum of alteration (i.e., intact, consolidated and fully drained wetlands). Such data richness provided a unique opportunity to look at the spatial but also temporal variability of wetland function over a two-year period, and such collection and analysis of high-frequency wetland-related data is unprecedented. Characterization of wetlands based on the hydrological and biogeochemical criteria for GIWs in the PPR will also be a unique and valuable contribution of my Ph.D. research to the field. These characterizations are strikingly different from wetland

classifications that usually rely on ecological characteristics or water permanence. Characterizing wetlands based on several aspects of hydrological and biogeochemical wetland behavior may show that wetlands which belong to the same ecological class can still be very different from a hydrobiogeochemical standpoint, thereby enhancing our understanding of the fundamental hydrologic and biogeochemical processes that control GIWs in the PPR. The hydrological and biogeochemical criteria developed in this thesis could be used not only in future research, but also in a policy-making sense to support the designation of priority areas for wetland conservation or restoration, establish a hierarchy of mitigation strategies, and strengthen exiting wetland-focused programs. Indeed, as mentioned earlier in this thesis, restoration and conservation of wetlands should be prioritized based on their functions, and the surface area of a given wetland may not be the best – or sole – proxy or surrogate measure of wetland function. One significant contribution of this Ph.D. research is to clearly identify instances in which wetland area may be a good indicator of wetland function, as opposed to other instances where other surrogate measures of wetland function should be considered for adequate wetland management and science-based policy making.

## **1.10 Contributions of Authors**

All the chapters were written by me while my advisor provided guidance and suggested revisions. Co-authors of the submitted and published manuscripts provided revisions and suggestions ahead of the manuscript being sent to journals for evaluation. I have done the analysis, discussion and synthesis of data for all the chapters, with guidance from my advisor. All the field set up, fieldwork, data collection and laboratory analyses were done with the help of my “lab

mates”. Ducks Unlimited Canada provided spatial data and wetland characteristics data for Chapters 2, 3, 4 and 5.

## **1.11 Connectivity between Wetlands and Streams: Preliminary Analysis**

### **1.11.1 Introduction**

The loss or alteration of Prairie Pothole wetlands, which are usually considered as geographically isolated, has modified the frequency of water and pollutant exchanges between land and streams and thus affected regional water quality (Wang et al., 2010). Indeed, while intact wetlands act as nutrient sinks by effectively trapping runoff and associated pollutants, lost or altered wetlands are prone to release nutrients to nearby streams. Although the general impacts of wetlands loss or alteration at the regional water quality dynamics are well understood (Wang et al., 2010), little is known about the local wetland properties that drive those dynamics. In general, the role of wetlands in maintaining downstream water quality by storing excess nutrient (e.g., phosphorus) is closely related to wetland hydrology and the concept of wetland-stream connectivity (U.S. EPA, 2015): both are influenced by climate and basin geomorphology (Mitsch and Gosselink, 2007) and are known to vary based on the specific location of wetlands within a watershed (U.S. EPA, 2015). One possible way to address the influence of local wetland properties on stream water quality is to examine the synchronicity (or lack thereof) between nutrient dynamics in wetlands and adjacent streams. The main research objective of this study was therefore to infer wetland- stream connectivity via the comparison of water level and phosphorus concentrations in a stream reach and in a range of potholes located in the lateral contributing area to the stream reach.

### 1.11.2 Study Site and Methods

Ten intact and undisturbed pothole wetlands (hereafter simply referred to as “wetlands”), three consolidated wetlands and seven ditches (historically used to drain wetlands) were selected adjacent to a 5 km study reach in the Broughton’s Creek Watershed (BCW, southwestern Manitoba, Canada). Physiographically, the BCW is typical of the Prairie Pothole Region and intact wetlands are assumed not to contribute water and nutrients to the creek during dry to normal conditions. Stilling wells were installed at the downstream end of the creek reach as well as in intact and consolidated wetlands, while 1m-deep perched water table wells were drilled below the drainage ditches: capacitance-based loggers were used to record surface and subsurface water level fluctuations at a 15-minute frequency in May to September 2013 and 2014. During the same period, water samples were collected at least every two days at the two extremities of the study reach, and every two weeks from all intact and consolidated wetlands and from multi-depth piezometers installed below the drainage ditches.

All water samples were analyzed for soluble reactive phosphorous (SRP) concentrations. To assess the temporal variability of SRP in the study reach, sampling dates were categorized into gaining conditions (downstream SRP > upstream SRP) and losing conditions (downstream SRP < upstream SRP), which potentially reflect opposite scenarios of wetland- stream connectivity and phosphorus export. To test the hypothesis that gaining and losing conditions are driven by the conditions prevailing in the lateral contributing area to the reach, Kruskal-Wallis tests were performed to assess if SRP concentrations in wetlands and ditches were statistically different between “gaining dates” and “losing dates”. The Spearman’s rank correlation coefficient between stream water level and wetland water level (or ditch water table depth) was also used as a surrogate measure for wetland- stream hydrologic connectivity.

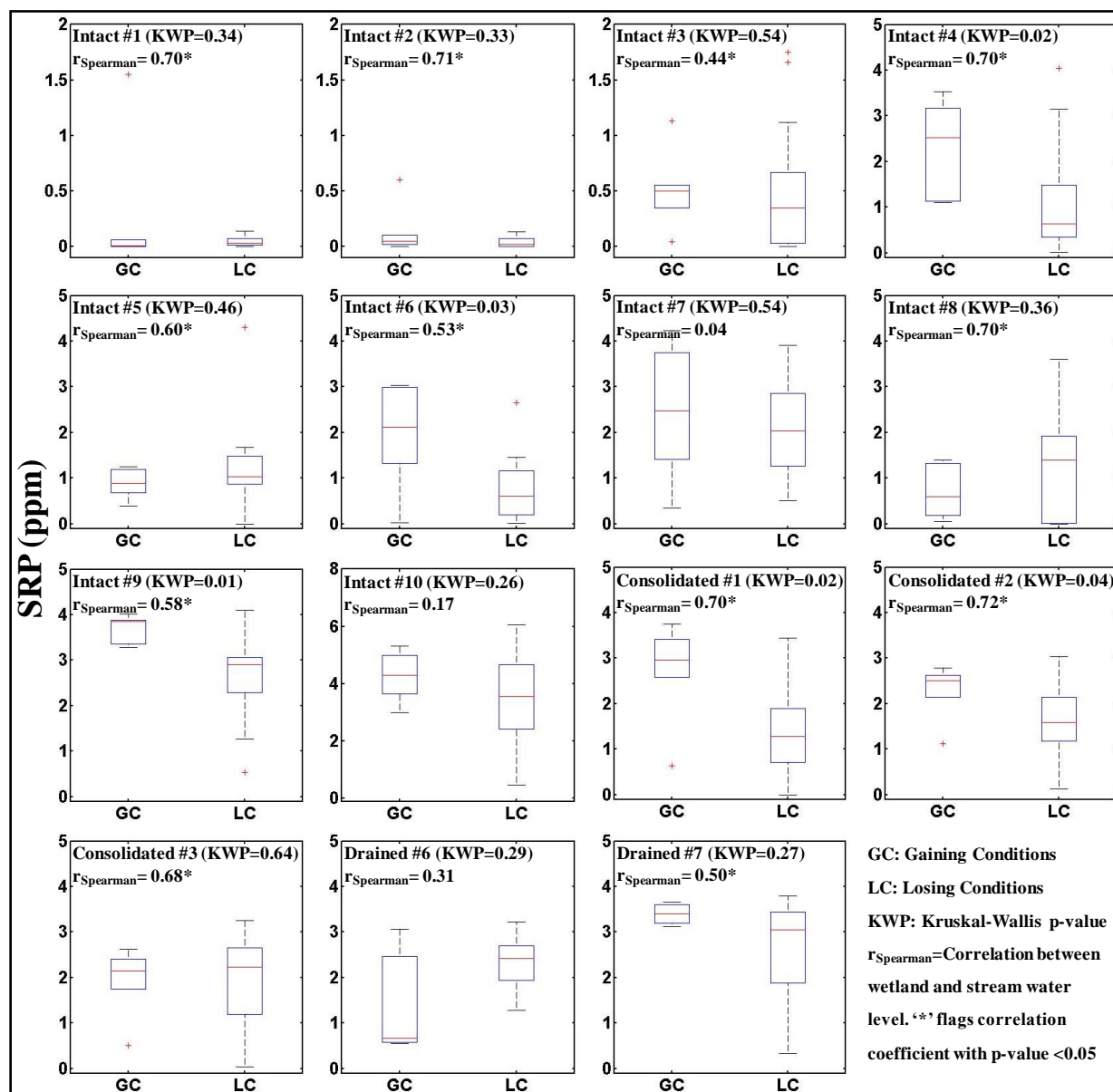
### **1.11.3 Results**

Almost all wetlands showed statistically significant correlations between wetland water level and stream water level (Figure 1-4). Based on the 2013 and 2014 dataset, losing conditions prevailed for 76% of the time. The median SRP concentration at the creek downstream end was 1.21 ppm, with minimum and maximum values of 0.01 ppm and 10.85 ppm, respectively. At the upstream end, median, minimum and maximum values were 1.86 ppm, 0.01 ppm, and 3.75 ppm. As shown in Figure 1-4, intact wetlands #1, 2, 3 and 5 had generally low SRP concentrations (<1.5 ppm) compared to other wetlands. Kruskal-Wallis tests revealed that only three intact (#4, 6, 9) and two consolidated (#1, 2) wetlands showed a statistically significant difference in SRP concentrations between gaining and losing dates, with higher values associated with gaining dates (Figure 1-4). Most of the wetlands are located within agricultural areas (Table 1-3).

### **1.11.4 Discussion and Conclusion**

The studied intact and consolidated wetlands are located at least 0.5 km away from the stream (Table 1-3) and have no visible surface connections between them and the stream. Spillage events were not observed; therefore, it is likely that wetland-stream hydrologic connectivity, when it exists, is due to shallow or deep subsurface flow. The use of a correlation coefficient between stream and wetland water level as an indication of wetland- stream connectivity could however be challenged as it might not necessarily reflect causality between wetland and stream dynamics but rather highlight the fact that similar drivers are behind stream and wetland dynamics. Many wetlands did not show any difference in SRP concentrations between gaining and losing conditions, which is probably an indication that they are not connected to the stream or do not respond to the same climatic drivers as the stream. Wetlands that showed a significant difference in SRP concentrations between gaining and losing conditions, and a significant correlation

between stream and wetland water level, probably an indication that they are connected to the stream.



**Figure 1-4:** SRP distribution in different wetlands gaining and losing conditions at downstream.

Each box has lines at the lower quartile, median, and upper quartile values, while the whiskers extend from each end of the box to show the extent of the rest of the data (minimum and maximum values). Outliers are shown as '+' in red.

**Table 1-3:** Spatial wetland characteristics. I: Intact wetland, C: Consolidated wetland, D: Drained wetland, F: Forest, A: Agriculture.

	I1	I2	I3	I4	I5	I6	I7	I8	I9	I10	C1	C2	C3	D6	D7
Dominant land cover in upslope area	F	F	F	A	F	A	A	A	A	F	A	A	A	A	A
Distance from stream (km)	0.90	1.02	0.60	1.00	0.75	0.50	0.47	0.56	0.58	0.60	0.55	0.65	0.65	0.22	0.22

These wetlands are located within agricultural fields and do not have any forest in their upslope area (Table 1-3), a factor that may play a role in their subsurface flow-driven connectivity with the stream. Consolidated wetlands did not appear to behave in a significantly different manner than intact wetlands, making it unclear whether changes in wetland-stream connectivity can be predicted based on the degree of wetland alteration.

In conclusion, the preliminary data analyses performed here suggest possible exchanges of water and SRP between so-called geographically isolated wetlands and the nearby stream in the BCW. However, those analyses do not provide specific information on the extent to which individual wetlands contribute (or not) to those exchanges, information about the hydrological processes that may be the root cause for such exchanges. Therefore, studies of individual wetland hydrological behaviour and phosphorus dynamics are needed to confirm or infirm the role of subsurface flow and other hydrological processes in wetland- stream connectivity, and to identify specific wetland characteristics that control it. Such studies are reported in Chapters 2, 3 and 4 of this thesis.

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**Chapter 2: Hydrological Dynamics of Prairie Pothole Wetlands: Dominant Processes and  
Landscape Controls Under Contrasted Conditions**

## **Abstract**

Numerous studies have examined the impact of prairie pothole wetlands on overall watershed dynamics. However, very few have looked at individual wetland dynamics across a continuum of alteration status using subdaily hydrometric data. Here, the importance of surface and subsurface water storage dynamics in the Prairie Pothole Region was documented by (1) characterizing surface fill–spill dynamics in intact and consolidated wetlands; (2) quantifying water-table fluctuations and the occurrence of overland flow downslope of fully drained wetlands; (3) assessing the relation (or lack thereof) between intact, consolidated or drained wetland hydrological behaviour, and stream dynamics; and (4) relating wetland hydrological behaviour to landscape characteristics. Focus was on southwestern Manitoba, Canada, where ten intact, three consolidated, seven fully drained wetlands, and a nearby creek were monitored over two years with differing antecedent storage conditions. Hourly hydrological time series were used to compute behavioural metrics reflective of year-specific and season-specific wetland dynamics. Behavioural metrics were then correlated to wetland physical characteristics to identify landscape controls on wetland hydrology. Predictably, more frequent spillage or overland flow was observed when antecedent storage was high. Consolidated wetlands had a high degree of water permanence and a greater frequency of fill–spill events than intact wetlands. Shallow and highly responsive water tables were present downslope of fully drained wetlands. Potential wetland–stream connectivity was also inferred via time-series analysis, while some landscape characteristics (e.g., wetland surface, catchment area, and storage volume) strongly correlated with wetland behavioural metrics. The nonstationarity of dominant processes was, however, evident through the lack of consistent correlations across seasons. This, therefore, highlights the importance of combining

multiyear high-frequency hydrometric data and detailed landscape analyses in wetland hydrology studies.

**Keywords:** landscape controls, Prairie Pothole Region, storage dynamics, wetland drainage, wetland hydrology, wetland–stream connectivity.

## 2.1 Introduction

Depending on their geographic and topographic position, wetlands can be critical in buffering streamflow, storing floodwater, and reducing peak flow after heavy precipitation (Acreman & Holden, 2013; Bullock & Acreman, 2003; McLaughlin et al., 2014). Therefore, storage dynamics are a critical component of local wetland hydrology, as well as a driver of wetland–stream connectivity. Here and throughout the rest of this paper, the term “dynamics” is used as a general sense to refer to any variation in the magnitude or intensity of hydrological processes. As for “connectivity,” it refers to the degree to which wetlands are interacting with downstream water bodies, and it can be quantified in terms of frequency, intensity, magnitude, and duration of different water transport mechanisms (EPA, 2015; Leibowitz, 2003; Rains et al., 2006). Non-floodplain wetlands are often called geographically isolated wetlands not only because they are surrounded by uplands but also because they lack surface, channelized connections to nearby streams (Cohen et al., 2016; Leibowitz, 2003, 2015; Tiner, 2003). It has, however, been argued that the term “isolated” is a misnomer (Leibowitz, 2015; Mushet et al., 2015) as geographically isolated wetlands can still affect the integrity of watersheds, either by transferring material to streams via pathways other than surface runoff or by preventing material from reaching streams through retention mechanisms (Marton et al., 2015; McLaughlin et al., 2014; McLaughlin & Cohen, 2013; Pringle, 2003; Rains et al., 2016). The fact that a single wetland might, at times, orchestrate enhanced material transfer and, at other times, be a catalyst for material retention has prompted researchers to evaluate wetland–stream interaction not in a binary fashion (i.e., isolated or connected) but rather within an isolation-connectivity continuum framework (Leibowitz, 2003). Studies focusing on prairie regions, Carolina bays, and the coastal plains of Florida have shown that wetland–stream connectivity can occur through wetland water spillage, temporary or

discontinuous surface runoff, shallow subsurface water movement, and deep groundwater movement (Brunner et al., 2009; Cohen et al., 2016; EPA, 2015; McDonough et al., 2015; McLaughlin et al., 2014; Pyzoha et al., 2008; Rains et al., 2006; Wilcox et al., 2011; Winter & LaBaugh, 2003) and should be viewed as a probability event (e.g., low versus high degree of connectivity) (Cohen et al., 2016; Leibowitz & Vining, 2003; Mushet et al., 2015). A strong, process-based understanding of wetland–stream interaction is therefore needed to quantify the extent to which non-floodplain wetlands, such as prairie potholes, generate flow and transmit it to streams (EPA, 2015; Golden et al., 2014; Leibowitz, 2015).

While directly measuring wetland–stream connectivity is difficult, it is possible to infer its probability of occurrence by focusing on individual wetland dynamics as well as correlations between wetland dynamics and nearby stream dynamics. In prairie wetland complexes, fill–spill dynamics control wetland-to-wetland and wetland-to-stream connectivity and determine the actual area that contributes runoff to a given outlet (Hayashi et al., 2016; Shaw et al., 2012). While the magnitude and frequency of runoff events are important to consider when assessing the likelihood of wetland spillage, pre-existing surface storage and basin memory effects strongly dictate the timing of spillage (Hayashi et al., 2016; Shaw et al., 2012; Shook et al., 2015). Indeed, wetland-water surface area and surface storage are a function of water depth (Hayashi et al., 2016), and pre-existing storage determines the effectiveness with which a given wetland basin can retain or detain excess floodwater (Huang et al., 2011; Huang et al., 2013). That effectiveness is, however, not linear since storage changes temporally as a function of prior water inputs and outputs, also known as storage memory or memory effects (O’Kane & Flynn, 2007; Shook & Pomeroy, 2011). Memory effects have been shown to manifest in different ways, such as the presence of autocorrelated storage time series in response to uncorrelated precipitation time series or the

existence of a time delay (i.e., hysteresis) between precipitation inputs and storage changes, both of which can result in nonlinear wetting and drying cycles affecting wetland basins (Shook et al., 2015; Shook & Pomeroy, 2011). To a lesser extent, subsurface flow processes may also have an impact on prairie wetland hydrology. During wet periods, for instance, when water tables are close to the surface, shallow subsurface flow can occur (Brannen et al., 2015) due to the formation of an effective transmission zone (van der Kamp & Hayashi, 2009). Hydrologic connectivity via shallow subsurface flow can, therefore, play a major role in maintaining wetland basin storage and surface connections between individual wetlands or wetland complexes and streams (Brannen et al., 2015). However, Hayashi et al. (2016) found that deep groundwater-driven hydrologic connectivity is rare in prairie wetland complexes and is hypothesized to take place only during extremely wet periods.

The published literature extensively discusses the role of intact, non-floodplain wetlands in determining overall watershed hydrologic behaviour (Bullock & Acreman, 2003; EPA, 2015; Gleason et al., 2007; Haan & Johnson, 1968; Hayashi et al., 2016; Hayashi et al., 2003; Huang et al., 2013; Huang et al., 2011; Hubbard & Linder, 1986; Johnston, Detenbeck, & Niemi, 1990; Lindsay et al., 2004; McAllister et al., 2000; McLaughlin et al., 2014; Tiner, 2003; Wang et al., 2010; Yang et al., 2010). Even small, intact isolated depressions (<1,000 m<sup>2</sup>) in prairie regions can play a major role in storing water, increasing evapotranspiration, and promoting groundwater infiltration, thus influencing the hydrologic cycle on a watershed scale (Hayashi et al., 2003). However, few studies have relied on high-frequency field data rather than a modelling framework to further our understanding of the impacts, individual or cumulative, of natural landscape characteristics and wetland drainage at local scales. Over the last 200 years, more than 50% of the World's wetlands have been lost (Davidson, 2014), including 31% of natural wetlands between

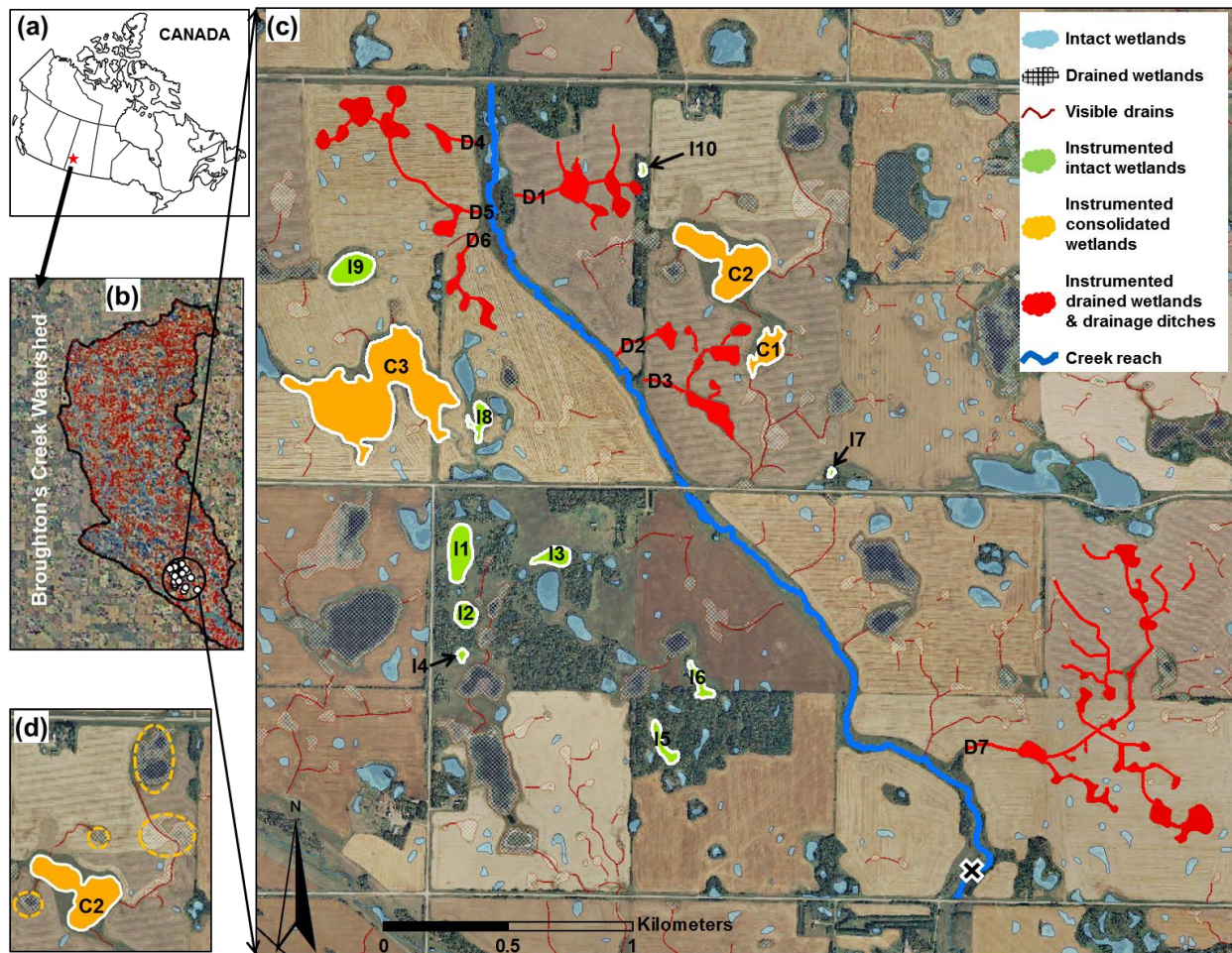
1970 and 2008 (Dixon et al., 2016). Wetland drainage has been extensive in the Prairie Pothole Region (PPR) of North America (Van Meter & Basu, 2015), where wetland loss has been attributed to rapid agricultural expansion (Johnston, 2013). Wetlands in the PPR can vary in size from 100 m<sup>2</sup> to 30,000 m<sup>2</sup> (Zhang et al., 2009), and historically, there has been a tendency to drain small wetland features as they inconvenience the operation of farm machinery and are the easiest to drain (Van Meter & Basu, 2015). In some instances, wetland consolidation is preferred to wetland drainage and infilling: Consolidation occurs when two or more small wetlands are drained (consolidated) into a single large one in an attempt to retain some of the wetland hydrologic functions in the landscape (McCauley et al., 2015). Research gaps remain regarding the changes in water storage and discharge dynamics that may exist across a gradient of wetland alteration (e.g., intact, consolidated, and fully drained wetlands) or regarding the landscape and climatic drivers that influence the storage and release dynamics that regulate wetland–stream connectivity. The hydrologic function of intact and consolidated wetlands is mainly defined by their relative ability to “fill and spill” (Shaw et al., 2012), hence their assumed – although debated – usefulness in reducing flood intensity and storing water during periods of high flow (Ehsanzadeh, van der Kamp, & Spence, 2016; Hayashi et al., 2016; McCauley et al., 2015). In contrast, fully drained wetlands no longer have an open-water component, but they are still likely to affect regional hydrology, either through overland flow occurring downstream of them or through shallow subsurface flow when the ground is not frozen. Dynamics of surface water storage for open-water wetlands and shallow subsurface water storage for drained wetlands and their associated drainage ditches need to be characterized over a range of timescales, not only to improve our process understanding but also to inform wetland conservation and restoration policies. The overall objective of the current paper was, therefore, to quantify the spatiotemporal variability of pothole

wetland water storage and infer the hydrological processes that may be the root cause of such variability. By relying on high-frequency hydrometric data as well as a detailed landscape analysis in the PPR, four specific research objectives were addressed: (1) characterize fill–spill dynamics in intact and consolidated wetlands; (2) quantify water-table fluctuations and the occurrence of overland flow downslope of fully drained wetlands; (3) assess the relation (or lack thereof) between intact, consolidated, or drained wetland hydrological behaviour and stream dynamics; and (4) relate wetland hydrological behaviour to landscape characteristics. All analyses were carried out using data collected in a dry year and a wet year to contrast dynamics prevailing under different antecedent storage conditions.

## **2.2 Methods**

### **2.2.1 Study Site Description and Data Collection**

The Broughton's Creek Watershed (BCW) is located in southwestern Manitoba, Canada (Figure 2-1a), and drains an area of about 252 km<sup>2</sup> (Figure 2-1b). The Broughton's Creek, which is a third-order stream and a tributary of the Little Saskatchewan River, flows into the Assiniboine River which, in turn, flows into the Red River and finally into Lake Winnipeg. Physiographically, the BCW represents a typical PPR landscape, as it was created on top of the surficial deposits left by the retreat of the Assiniboine glacial lobe from 20,000 to 12,000 B.C. (Wang et al., 2010; Yang et al., 2010). The glaciated landscape is peppered with water-holding depressions, including sloughs, pothole wetlands, and lakes that sit in hummocky moraines or near-level outwash plains common in the region. The maximum elevation along the watershed boundary exceeds 600 m above sea level (a.s.l.), while the maximum relief is 135 m (Svenson & McGinn, 2003).



**Figure 2-1:** (a) Location of the Broughton's creek watershed within Canada. (b) Creek reach and surrounding land under study. (c) Instrumented wetlands in the Broughton's Creek Watershed. I: Intact wetland, C: Consolidated wetland, D: Drained wetland (s) and associated ditch (es). The black cross shows the creek water level measuring location. (d) Example of consolidated wetland, C2; dashed circles indicate historic wetlands that have been drained into C2.

Due to the geological setting of the BCW, the numerous sloughs, pothole wetlands, and lakes present in the till plain and moraines are oriented in the northwest-southeast direction. Under normal conditions (described as the 1-in-2-year flood in the region), potholes are thought not to contribute to streamflow via overland flow (Stichling & Blackwell, 1957) and rather act as closed

basins, which are isolated from the hydrographic network (Godwin & Martin, 1975; Martin, 2001; PFRA-Hydrology-Division, 1983). Conversely, under wet conditions, the topology of wetlands and streams allows connectivity to occur, either via spilled water that forms temporary surface connections towards downgradient streams or via shallow lateral subsurface flow. Newdale association soils (i.e., mainly Orthic Black Chernozems, with solum depths ranging from 25 to 98 cm) are found throughout the watershed: They are usually well drained at shallow depths (~45 cm or less) due to thick and humus-rich A and B horizons. Other minor soils like Dorset, Drokan, Eroded Slope Complex, and Jaymar associations are also found in areas adjacent to streams (MAFRI, 2008; Wang et al., 2010). Newdale, Dorset, and Jaymar associations are similar to the Udic Boroll subgroups in the US soil taxonomic classification (Soil classification working group, 1998). Land use in the BCW consists of agriculture (71.4%), rangeland (10.8%), wetland (9.8%), forest (4%), and others (4%). Historical data shows the continuous alteration of wetlands in the watershed, with 6,000 wetland basins lost or degraded between 1968 and 2005 (Wang et al., 2010).

For targeted monitoring of wetland dynamics, a 5-km study reach (as well as surrounding land on each side of it) was selected in the BCW (Figure 2-1c). A capacitance-based water level logger (Odyssey™, Dataflow Systems) was installed at the downstream end (south end) of the creek reach to record hourly creek water-level fluctuations. Ten intact and hydrologically undisturbed wetlands, three consolidated wetlands, and seven drainage ditches located downslope of fully drained wetlands were selected adjacent to the study reach (see Figure 2-1c for the exact locations). In the BCW, historical maps reveal that intact wetlands have not undergone major morphological changes in the past 60 years, while wetland consolidation was achieved by draining multiple pre-existing wetlands into a single larger basin (see example in Figure 2-1d). None of the selected intact and consolidated sites are cropped, but all of them have well-developed hydric soils.

The selected study sites also have ponded water and aquatic vegetation in them every year, which fits the definition of prairie pothole wetlands clarified by van der Kamp, Hayashi, Bedard-Haughn, and Pennock (2016). The monitored drainage ditches were selected on the basis of current and historical maps that confirm their role in the diversion of runoff away from historical wetlands (that have since been drained) towards the creek. It is worth noting that those drainage ditches are now barely incised swales, as they have been subjected to infilling since their creation; there is usually no water in them except during very wet conditions. Landscape characteristics (Table 2-1) were estimated for the targeted wetlands based on the current and historic wetland inventory dataset. Those datasets were provided by Ducks Unlimited Canada and assembled on the basis of Canada Wetland Inventory specifications (Ducks Unlimited Canada, 2016) as well as field reconnaissance information. Volume estimation methods varied depending on wetland condition. For open-water wetlands with emergent communities and open water, Ducks Unlimited Canada used the v-a-h (i.e., volume-area-depth) method as described in Pomeroy et al. (2010). The v-a-h method was chosen for its ability to account for capacity below the standing water at the time of LiDAR acquisition. For completely drained wetlands, the volume was estimated from LiDAR data by measuring the storage contained below the plane of the historical wetland extent using the Surface Volume Tool in ESRI ArcGIS. The perimeter of the instrumented open-water wetlands (i.e., intact and consolidated) varies from  $1.0 \times 10^2$  to  $2.8 \times 10^3$  m, while their area and storage volume range from  $7.4 \times 10^2$  to  $1.5 \times 10^5$  m<sup>2</sup> and  $7.4 \times 10^1$  to  $4.4 \times 10^4$  m<sup>3</sup>, respectively. Open-water wetlands are located at least  $4.7 \times 10^2$  m (Euclidean distance) away from the study creek and lack surface channelized connections to it. The historically drained wetlands associated with

**Table 2-1:** Overview of landscape characteristics associated with the instrumented wetlands.

Name	Abbreviation	Description (units)	Mean value (Range)
<b>Open water wetlands</b>			
Area	Area	Surface area of wetland (m <sup>2</sup> )	$2.2 \times 10^4$ ( $7.4 \times 10^2$ - $1.5 \times 10^5$ )
Perimeter	Peri	Edge length of wetland (m)	$6.3 \times 10^2$ ( $1.0 \times 10^2$ - $2.8 \times 10^3$ )
Storage volume	Vol	Storage volume of wetland (m <sup>3</sup> )	$7.4 \times 10^3$ ( $7.4 \times 10^1$ - $4.4 \times 10^4$ )
Bottom elevation	Elev	Elevation of wetland bottom above sea level (m)	$5.3 \times 10^2$ ( $5.2 \times 10^2$ - $5.3 \times 10^2$ )
Spill level	SL	Height of water (relative to stilling well bottom) at which wetland will spill (m)	$8.8 \times 10^{-1}$ ( $6.1 \times 10^{-1}$ - $1.2 \times 10^0$ )
Catchment area	CatArea	Natural basin area (or drainage area) of wetland (m <sup>2</sup> )	$7.5 \times 10^4$ ( $6.0 \times 10^3$ - $3.1 \times 10^5$ )
Catchment to wetland ratio	Cat2Area	Ratio of wetland catchment area to wetland surface area (-)	$8.0 \times 10^0$ ( $1.8 \times 10^0$ - $4.2 \times 10^1$ )
Catchment to volume ratio	Cat2Vol	Ratio of wetland catchment area to wetland storage volume (m <sup>-1</sup> )	$6.9 \times 10^1$ ( $6.2 \times 10^0$ - $5.6 \times 10^2$ )
Catchment perimeter	CatPeri	Edge length of wetland catchment (m)	$1.6 \times 10^3$ ( $4.1 \times 10^1$ - $4.8 \times 10^3$ )
Shortest distance to creek	Distance	Euclidean distance between wetland center and nearby creek (m)	$6.8 \times 10^2$ ( $4.7 \times 10^2$ - $1.0 \times 10^3$ )
Incremental contributing area	ConArea	Increase in wetland catchment area due to human alteration, i.e., drainage (m <sup>2</sup> )	$1.0 \times 10^4$ ( $0.0 \times 10^0$ - $1.2 \times 10^5$ )
Incremental contributing perimeter	ConPeri	Edge length of wetland incremental contributing area (m)	$2.4 \times 10^2$ ( $0.0 \times 10^0$ - $2.4 \times 10^3$ )
Total drainage area	TotDrainArea	Sum of wetland catchment area and wetland incremental contributing area (m <sup>2</sup> )	$8.5 \times 10^4$ ( $6.0 \times 10^3$ - $3.8 \times 10^5$ )
Total drainage area to catchment area ratio	TotDrain2CatArea	Ratio of wetland total drainage area to wetland catchment area (-)	$4.0 \times 10^{-2}$ ( $0.0 \times 10^0$ - $4.5 \times 10^{-1}$ )
<b>Drained wetlands</b>			
Area	AreaD	Surface area of historic wetland (m <sup>2</sup> )	$6.8 \times 10^3$ ( $7.8 \times 10^2$ - $1.3 \times 10^4$ )
Perimeter	PeriD	Edge length of historic wetland (m)	$3.8 \times 10^2$ ( $1.0 \times 10^2$ - $6.6 \times 10^2$ )
Storage volume	VolD	Storage volume of historic wetland (m <sup>3</sup> )	$2.8 \times 10^3$ ( $1.5 \times 10^2$ - $7.0 \times 10^3$ )
Surface elevation	ElevD	Elevation of ditch bottom above sea level (m)	$5.3 \times 10^2$ ( $5.2 \times 10^2$ - $5.3 \times 10^2$ )
Catchment area	CatAreaD	Natural basin area (or drainage area) of historic wetland (m <sup>2</sup> )	$4.2 \times 10^4$ ( $9.8 \times 10^3$ - $9.0 \times 10^4$ )
Catchment to wetland ratio	Cat2AreaD	Ratio of historic wetland catchment area to wetland surface area (-)	$7.2 \times 10^0$ ( $3.6 \times 10^0$ - $1.3 \times 10^1$ )
Catchment to volume ratio	Cat2VolD	Ratio of historic wetland catchment area to wetland storage volume (m <sup>-1</sup> )	$2.4 \times 10^1$ ( $6.8 \times 10^0$ - $6.7 \times 10^1$ )
Catchment perimeter	CatPeriD	Edge length of historic wetland catchment (m)	$1.3 \times 10^3$ ( $6.6 \times 10^2$ - $2.1 \times 10^3$ )
Ditch length	DLength	Length of drainage ditch routing water from historic wetland to nearby creek (m)	$1.7 \times 10^2$ ( $9.3 \times 10^1$ - $2.2 \times 10^2$ )
Incremental contributing area	ConAreaD	Increase in historic wetland catchment area due to human alteration (m <sup>2</sup> )	$1.6 \times 10^4$ ( $0.0 \times 10^0$ - $6.2 \times 10^4$ )
Incremental contributing perimeter	ConPeriD	Edge length of historic wetland incremental contributing area (m)	$5.3 \times 10^2$ ( $0.0 \times 10^0$ - $1.8 \times 10^3$ )
Total drainage area	TotDrainAreaD	Sum of historic wetland catchment area and wetland incremental contributing area (m <sup>2</sup> )	$5.8 \times 10^4$ ( $9.8 \times 10^3$ - $1.2 \times 10^5$ )
Total drainage area to catchment area ratio	TotDrain2CatAreaD	Ratio of historic wetland total drainage area to wetland catchment area (-)	$3.1 \times 10^{-1}$ ( $0.0 \times 10^0$ - $1.3 \times 10^0$ )

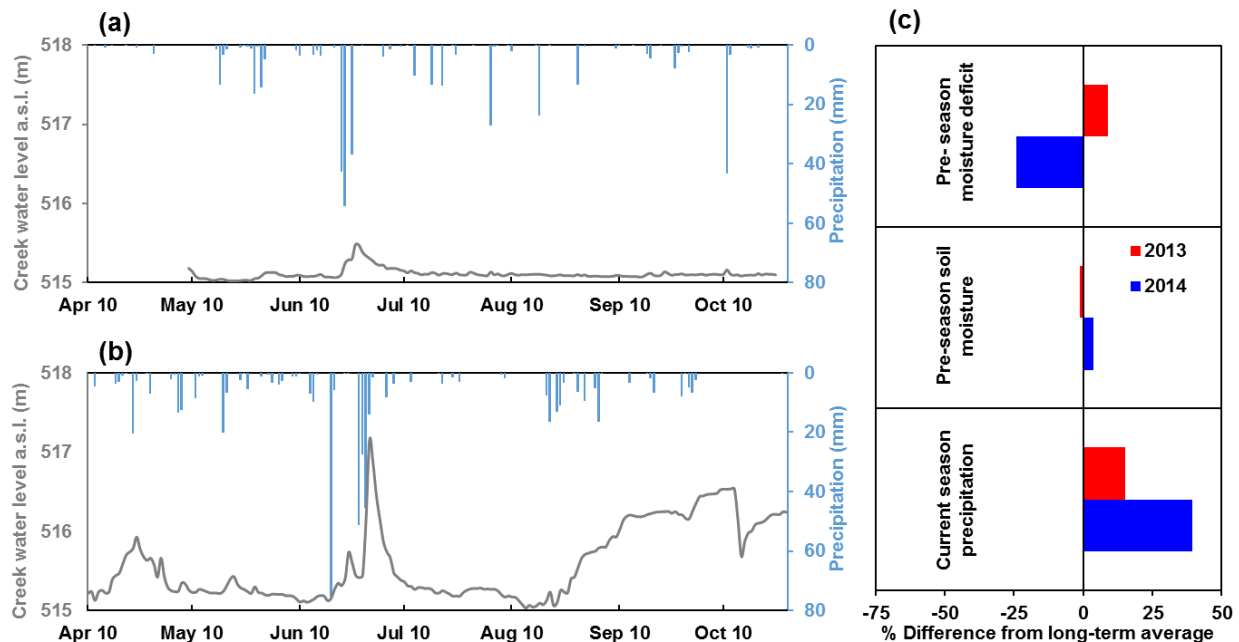
the instrumented ditches had areas and storage volumes ranging from  $7.8 \times 10^2$  to  $1.3 \times 10^4$  m<sup>2</sup> and  $1.5 \times 10^2$  to  $7.0 \times 10^3$  m<sup>3</sup>, respectively. It is worth noting that some of the landscape characteristics included in Table 2-1, notably the incremental contributing area (ConArea) and the total drainage area (TotDrainArea), were meant to capture changes associated with consolidated and drained (i.e., impacted) wetlands and therefore had zero values for intact wetlands and nonzero values for impacted wetlands. Assuming that wetlands X and Y have been consolidated into wetland Z, the incremental ConArea to Z corresponds to the catchment areas of wetland X and Y: While the latter wetlands no longer exist in the landscape, their catchments now “shed” water to wetland Z. TotDrainArea for wetland Z is, therefore, the sum of its original catchment area and its incremental ConArea. Stilling wells (i.e., above-ground wells) were deployed in the intact and consolidated wetlands to monitor fluctuations in surface water level, while water-table wells were drilled below the drainage ditches to a depth of 1 m to monitor shallow groundwater fluctuations. All stilling wells and water-table wells were equipped with capacitance-based water-level loggers (Odyssey™, Dataflow Systems) so that fluctuations in surface water and shallow groundwater could be recorded hourly. All measurements were made during the 2013 and 2014 open-water seasons (i.e., April to October).

Climate data are available from an Environment Canada weather station (Brandon A, climate ID: 5010481, World Meteorological Organization ID: 71140) located about 30 km southwest from the study site; previous studies have considered this weather station representative of weather in the BCW (Wang et al., 2010; Yang et al., 2010). Mean air temperatures were similar across the two open-water seasons included in the period of study (~13.5 °C in April–October 2013 and 2014). Total precipitation for the open-water season was 402.8 mm in 2013 compared with 523.4 mm in 2014 (Figure 2-2a and 2-2b; Appendix A-1), with both values above the long-

term mean (1960–2014) for the region. Antecedent basin storage, as approximated by pre-season moisture deficits and actual soil moisture measured in late April, was different between the two study years. The pre-season moisture deficit, computed as precipitation minus potential evapotranspiration and estimated for the April–March period preceding each open-water season, was, respectively, larger and smaller than the long-term mean (1960–2014) in 2013 and 2014 (Figure 2-2c). Similarly, soil moisture measured at the beginning of the open-water season was smaller than the long-term mean in 2013 and larger than the long-term mean in 2014 (Figure 2-2c; AAFC, 2017). Hence, based on measures of pre-existing storage, 2013 was considered to be a dry year, while 2014 was a wet year.

### **2.2.2 Data Analysis**

The originally recorded surface and subsurface water-level time series were transformed before analysis. We used hourly wetland fullness for open-water wetlands, hourly water-table height a.s.l. for drainage ditches, and hourly water level a.s.l. for the creek. For open-water wetlands, wetland fullness was computed by dividing the wetland water depth by the mean wetland depth (or spillage level) in a manner similar to Euliss Jr. et al. (2014): This was done to normalize surface water-level data and allow comparisons to be made among wetlands. Wetland fullness is therefore not a binary indicator but rather a continuous measure to assess whether a wetland is dry (fullness  $<0.05$ ), less than half-full ( $0.05 \leq \text{fullness} \leq 0.5$ ), more than half-full ( $0.5 \geq \text{fullness} < 1$ ), at the point of spillage (fullness = 1), or actively spilling with ponding water around it (fullness  $\geq 1$ ). For the drainage ditches, each water-table measurement was converted to elevation a.s.l. Ditch water-table height values greater than the ditch surface elevation a.s.l. were interpreted as surface water ponding resulting in saturation- excess overland flow downslope of fully drained wetlands.



**Figure 2-2:** Comparison of meteorological and hydrometric data available for the two study years. (a) and (b) Creek water level and total precipitation for 2013 and 2014, respectively. (c) Measures of antecedent storage for the years 2013 and 2014, relative to long-term means for the study region. Soil moisture data were obtained from AAFC (2017) while temperature and precipitation data were obtained from ENR (2017). For additional details, see Appendix A-1.

The temporal variability in surface and subsurface water level captured by the high-frequency time series was summarized using a set of behavioural metrics (Table 2-2). For open-water wetlands, surface storage dynamics were assessed by calculating metrics such as the percent time a wetland was dry (%WetDry) or spilling (%WetSpillage). For ditches, subsurface storage was assessed by calculating the percent time a ditch was dry (%WT100cm) or had water ponding (%WTground). To evaluate the importance of short-term storage change, metrics such as the percent time surface and subsurface water levels were rising (%WetRis/WTRis), falling

(%WetFal/WTFal), or unchanged (%WetNC/WTNC) were evaluated. All behavioural metrics were calculated for the years 2013 and 2014 separately and for different seasons (i.e., spring, summer, and fall) in each year to compare dynamics under contrasted conditions.

**Table 2-2:** Metrics of open-water wetland behaviour and ditch water-table behaviour.

Properties	Metrics (Abbreviated names)
<b>Open water wetland/ditch water table behaviour</b>	
<b>Storage regime</b>	Percentage of time (over a year or season) during which a wetland was dry ( <b>%WetDry</b> ) or the water table below a ditch was deeper than 100 cm ( <b>%WT100cm</b> )
	Percentage of time (over a year or season) during which wetland spillage occurred ( <b>%WetSpillage</b> ) or the water table was above ground, exfiltrating in the ditch ( <b>%WTground</b> )
	Percentage of time (over a year or season) during which a wetland was half-full ( <b>%WetHFull</b> ) or the water table below a ditch was within 50 cm of the ground surface ( <b>%WT50cm</b> )
	Temporal variability of wetland fullness ( <b>WetStd</b> ) or ditch water table depth ( <b>WTStd</b> ), expressed as the yearly or seasonal standard deviation based on hourly data
<b>Short-term storage change</b>	Percentage of time (over a year or season) during which the wetland fullness was rising ( <b>%WetRis</b> ) or the ditch water table was rising ( <b>%WTRis</b> )
	Percentage of time (over a year or season) during which the wetland fullness was falling ( <b>%WetFal</b> ) or the ditch water table was falling ( <b>%WTFal</b> )
	Percentage of time (over a year or season) during which the wetland fullness did not change ( <b>%WetNC</b> ) or the ditch water table did not change ( <b>%WTNC</b> )

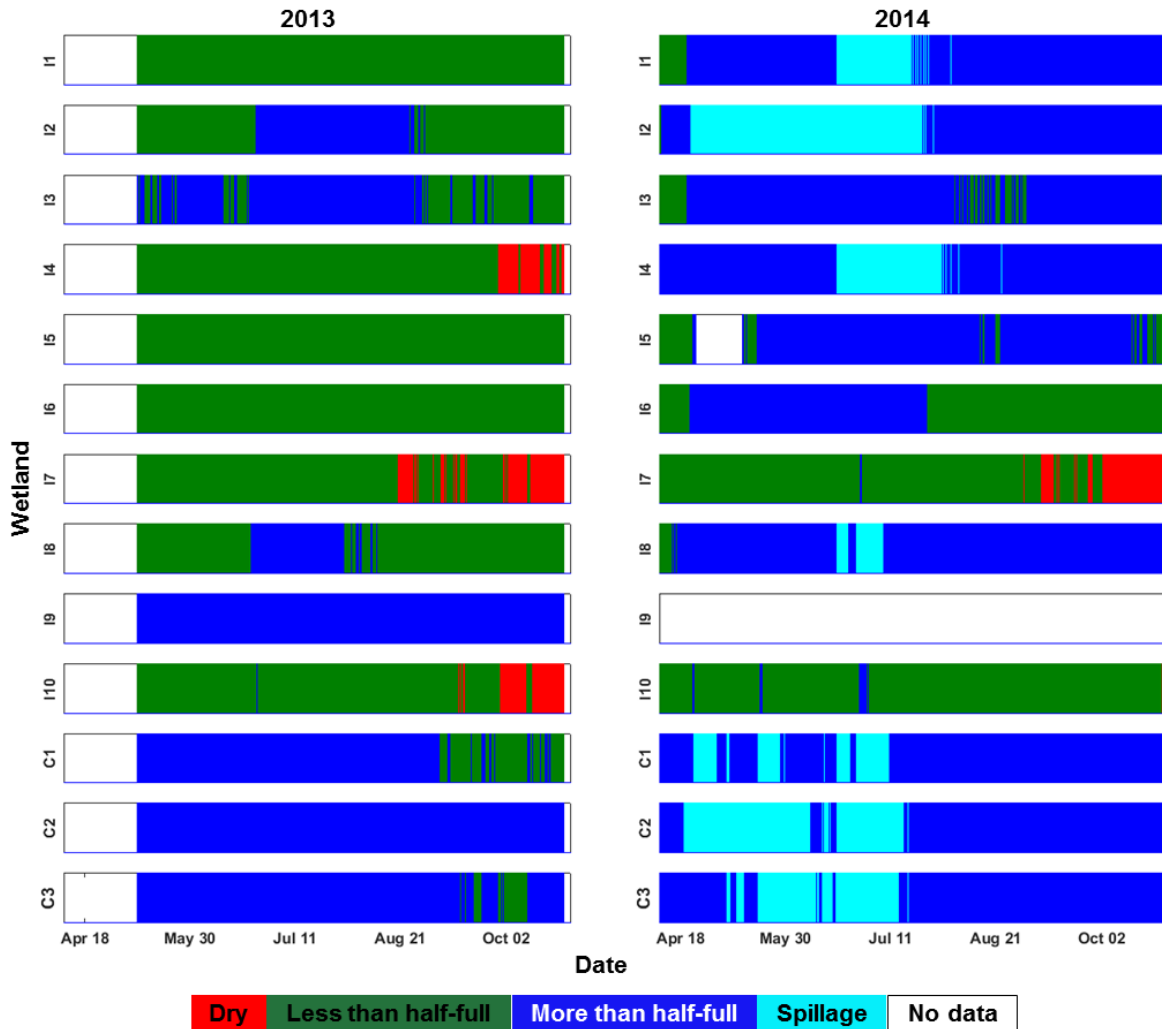
To assess hydrological dynamics in open-water wetlands, wetland fullness time series and behavioural metrics related to surface storage and storage change were compared (research objective 1). To evaluate the occurrence of overland flow downslope of fully drained wetlands, water-table fluctuations were examined relative to the surface elevation of drainage ditches (research objective 2). The relation (or lack thereof) between wetland hydrological behaviour and creek dynamics was inferred via correlation analysis and used as an implicit indicator of potential wetland–stream (dis) connectivity (research objective 3). Here, we refer to “potential” wetland–stream interaction or connectivity because the presence of a correlation may – but does not necessarily – reflect the presence of an actual water transfer from wetland to stream. Spearman's

rank correlation coefficients were calculated and evaluated at the 95% significance level between the creek water-level time series and either wetland fullness or ditch water-table height time series. Here, the Spearman's rank correlation coefficient was used – rather than the more common Pearson correlation coefficient – because it does not assume a normal distribution of the data and is suitable to assess both linear and nonlinear relations between variables (Sokal & Rohlf, 2012). Lastly, to relate wetland hydrological behaviour to landscape controls (research objective 4), Spearman's rank correlation coefficients were also calculated and evaluated at the 95% significance level between individual wetland characteristics (Table 2-1) and the behavioural metrics associated with open-water wetlands and fully drained wetlands (Table 2-2). The MATLAB Statistics and Machine Learning Toolbox, Release 2017b, was used for all statistical analyses. Lastly, for selected open-water wetlands with similar landscape characteristics, wetland fullness values were plotted against creek water levels to produce hysteresis curves and evaluate spatial controls on wetland–stream interactions. In our experimental setup, all studied wetlands are part of the “lateral” contributing area to the same 5-km-long creek reach, and the water travel time between the upstream end and the downstream end of the reach is relatively short. Hence, we assumed that any differences in wetland fullness-stream stage hysteresis would be mostly due to differences in wetland behaviour and not to channel routing effects.

## **2.3 Results**

### **2.3.1 Surface Water Storage and Ditch Water-table Dynamics**

The high-frequency time series of creek water level (Figure 2-2a,b) and wetland fullness (Appendix A-2; Figure 2-3) reflected the differences in antecedent storage and open-water season precipitation between both study years: Most of the open-water wetlands were more than half-full



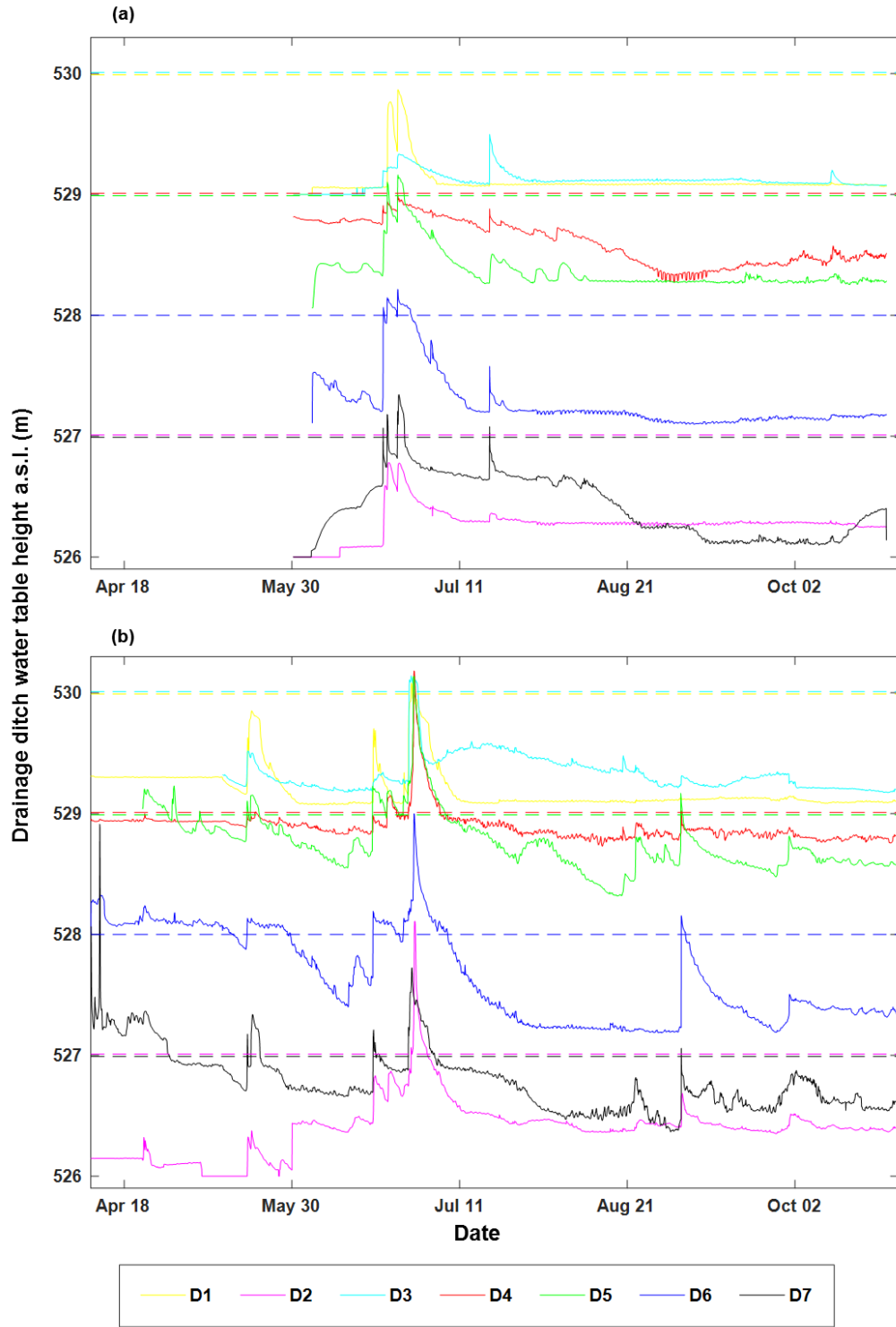
**Figure 2-3:** Classification of wetland surface water storage dynamics observed in 2013 and 2014.

I: Intact wetland; C: Consolidated wetland. For the raw hourly wetland fullness time series, see Appendix A-2.

with occasional spillage during the wettest year, that is, 2014 (Figure 2-3). Not every wetland was equally responsive to wetter conditions. For example, half of the intact wetlands did not spill in 2014, while spillage occurred in all three consolidated wetlands (Figure 2-3). Wetland I4 appeared

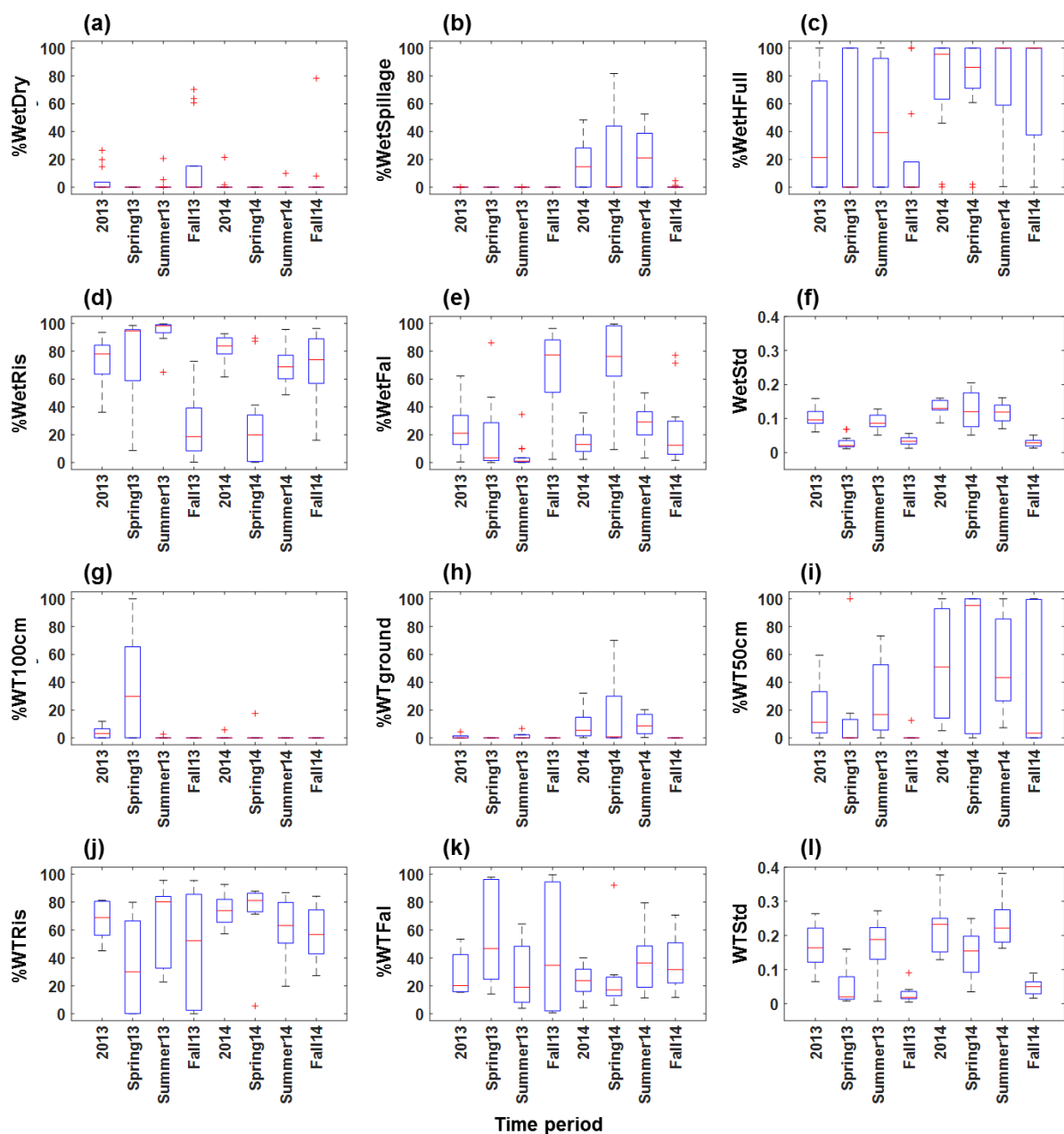
highly responsive to wetter conditions, as it went from mostly dry in 2013 to periods of spillage in 2014. Conversely, wetland I7 ran dry in both years. None of the consolidated wetlands was ever dry during the study period (Figure 2-3). Moreover, the event responses of consolidated wetlands were different than those of intact wetlands, with multiple peaks in wetland fullness observed in consolidated wetlands after rainfall, as opposed to single-peak responses for intact wetlands (Appendix A-2).

Differences in ditch water-table dynamics were also evident between both study years, as the water table was often recorded above the surface of the drainage ditches in 2014 (Figure 2-4), thus leading to ponding and overland flow into the adjacent creek. Fewer instances of such ponding and runoff were observed for the drainage ditches in 2013 (Figure 2-4). Ditch water-table response was well synchronized with rainfall events for both years. However, similar to open-water wetlands, not all ditches were equally responsive to wetter conditions. Compared with others, D5, D6, and D7 produced overland flow in both years, albeit more frequently in 2014, while others did not produce overland flow in 2013 and only did so once in 2014 (Figure 2-4). Due to logistical difficulties at the onset of the study, the beginning of the spring freshet was not captured by any of the 2013 time series but was included in the 2014 dataset. The major differences in the dynamics observed between both years can, however, not be explained solely by the fact that the 2013 spring freshet was not fully captured. For open-water wetlands, behavioural metrics such as %WetDry did not vary a lot, as opposed to %WetSpillage, %WetHFull, WetStd, %WetRis, and %WetFal that showed considerable variation across years and seasons (Figure 2-5). As for the water table-related metrics associated with fully drained wetlands and drainage ditches, %WT50cm, WTStd, %WTRis, and %WTFal were the most variable, temporally (Figure 2-5).



**Figure 2-4:** Water-table fluctuations associated with drainage ditches in (a) 2013 and (b) 2014.

Dashed lines show the surface elevation of the ditches.



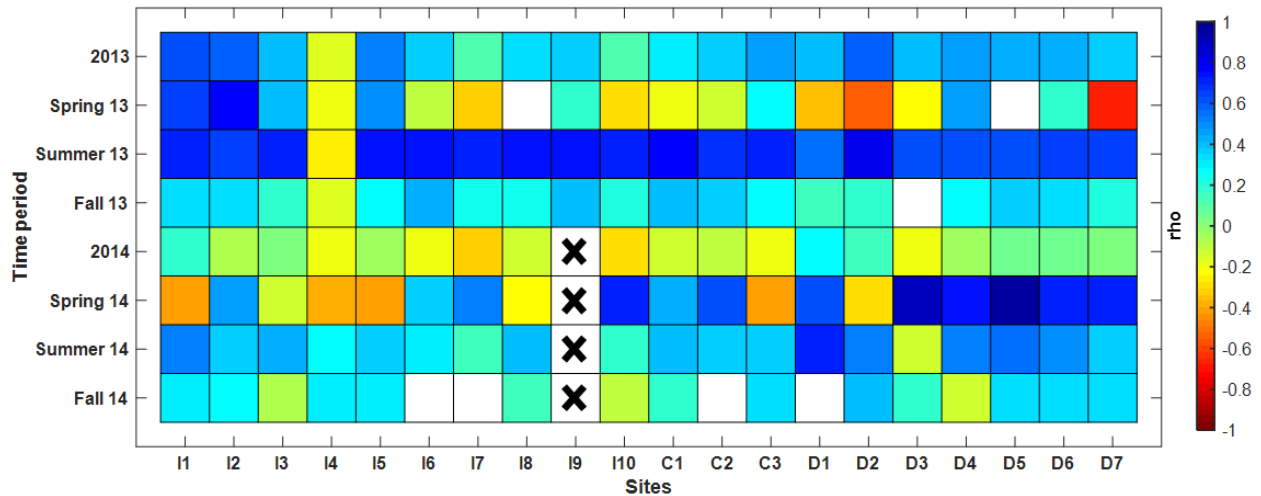
**Figure 2-5:** (a–f) Year-specific and season-specific variation of behavioural metrics for open-water wetlands and (g–l) ditch water-table dynamics. Each boxplot has lines at the lower quartile, median, and upper quartile values, while the whiskers extend from each end of the box to show the range of the data. Outliers are shown as “+.” For abbreviated metric names, refer to Table 2-2. The numbers “13” and “14” juxtaposed to season names (e.g., Summer13 and Summer14) refer to the years 2013 and 2014, respectively.

### **2.3.2 Wetland-Stream Interactions**

For the most part, correlation analyses between creek water level and wetland fullness or ditch water-table time series revealed statistically significant, positive correlations in 2013. Conversely, statistically significant, negative correlations occurred in 2014 (Figure 2-6; Appendix A-3). Across both years, summer seasons were associated with positive correlation coefficients between creek dynamics and wetland dynamics for all sites but two, namely, I4 in summer 2013 and D3 in summer 2014. In spring, however, opposite correlation results were observed in the two study years: The negative correlations observed in Spring 2013 were positive in Spring 2014 and vice versa (Figure 2-6). Some open-water wetlands (e.g., I1, I2, I3, I5, and C3) as well as water-table time series below two ditches (i.e., D4 and D6) showed consistent positive correlation with creek water level throughout 2013, regardless of season. However, only water-table dynamics recorded below D5, D6, and D7 showed consistent positive correlation with creek dynamics throughout 2014. Wetland fullness in I4 showed consistent negative correlation with creek water-level dynamics during all considered time periods except summer and fall 2014 (Figure 2-6).

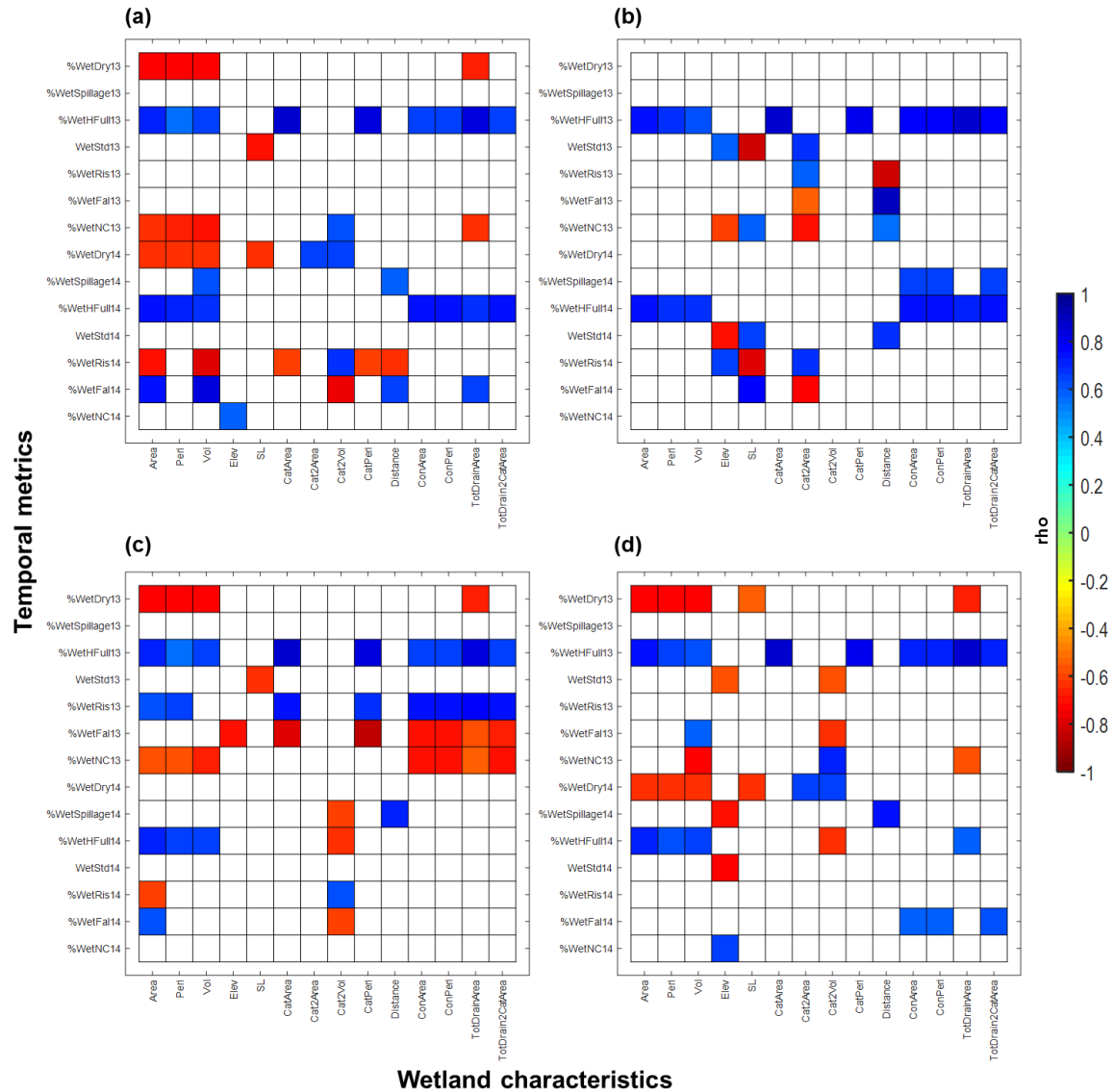
### **2.3.3 Landscape Controls on Wetland Hydrological Behavior**

Correlations between open-water wetland characteristics and metrics of wetland hydrological behaviour showed considerable variability across specific years and seasons (Figure 2-7a to 2-7d). Out of all the metrics that characterize surface water storage regime, %WetHFull had the largest number of positive correlations with wetland characteristics, namely, Area, Peri, Vol, ConArea, ConPeri, TotDrainArea, and TotDrain2CatArea, regardless of the time period considered. However, differences across years were present, as %WetHFull was positively correlated with all the wetland characteristics except Elev, Spill Level (SL), Cat2Area, Cat2Vol, and Distance for all time periods in 2013 but not in 2014 (Figure 2-7a to 2-7d).



**Figure 2-6:** Spearman's rank correlation coefficients (rho values) between wetland water level or ditch water-table time series and creek water level time series for different time periods. I: Intact wetland; C: Consolidated wetland; D: Drained wetland (s). The numbers “13” and “14” juxtaposed to season names (e.g., Summer13 and Summer14) refer to the years 2013 and 2014, respectively. “X” signals a period for which no data were collected. Blank cells flag time periods for which the Spearman's rank correlation coefficient was not statistically significant at the 95% level. For correlation coefficient values in tabular format, refer to ESM3.

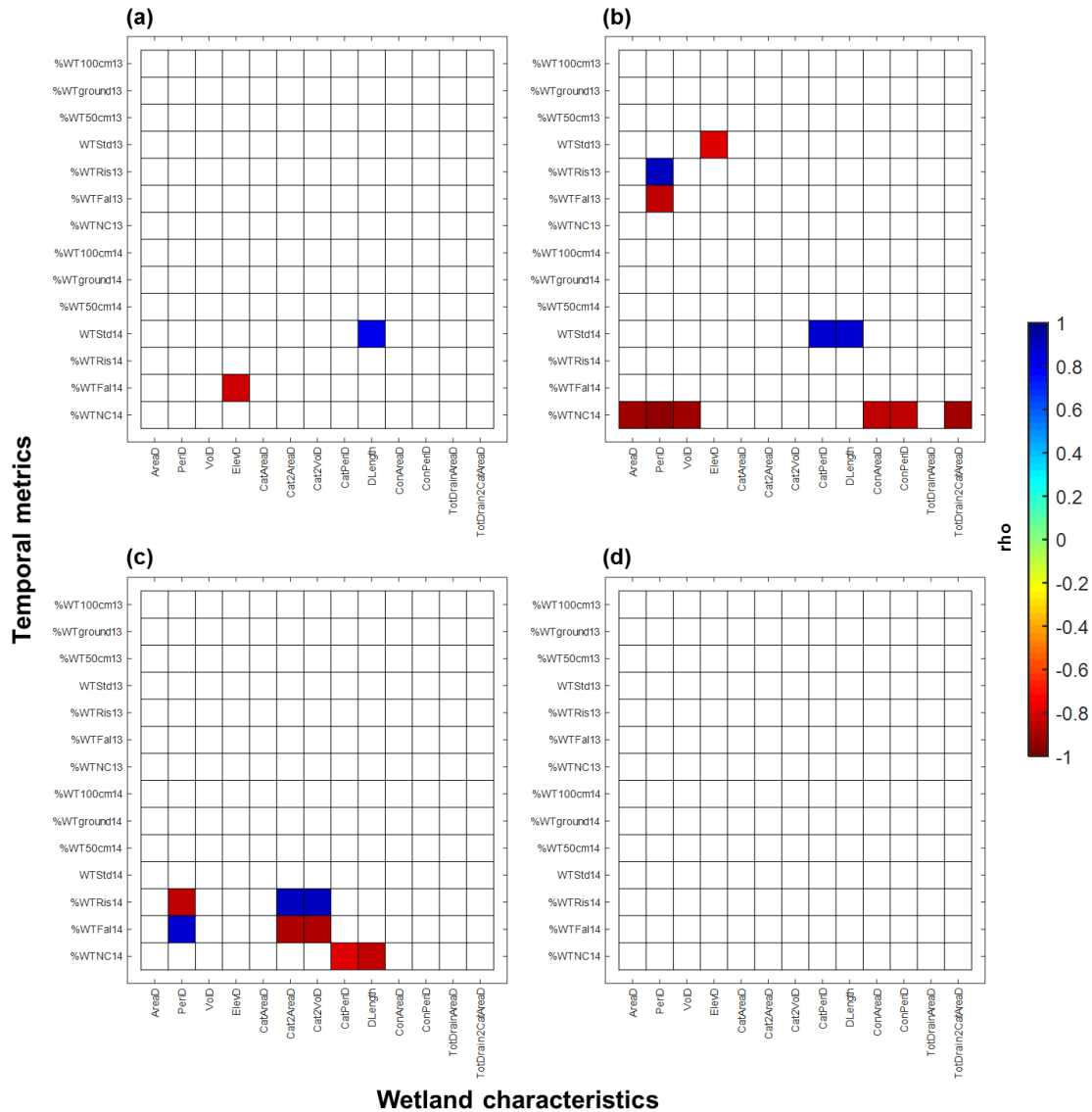
The %WetDry metric was negatively correlated with Area, Vol, and Peri for most of the time periods considered and negatively correlated with SL in fall of both years (Figure 2-7a,c,d). Landscape characteristics did not consistently show statistically significant correlation with %WetSpillage or WetStd (Figure 2-7a to 2-7d). For instance, the %WetSpillage metric was positively correlated with Distance in most time periods in 2014 but not in 2013. As for the WetStd metric, it was negatively correlated with SL for most of the time periods in 2013 and with Elev in the fall season of both years (Figure 2-7a to 2-7d). Regarding short-term storage change metrics,



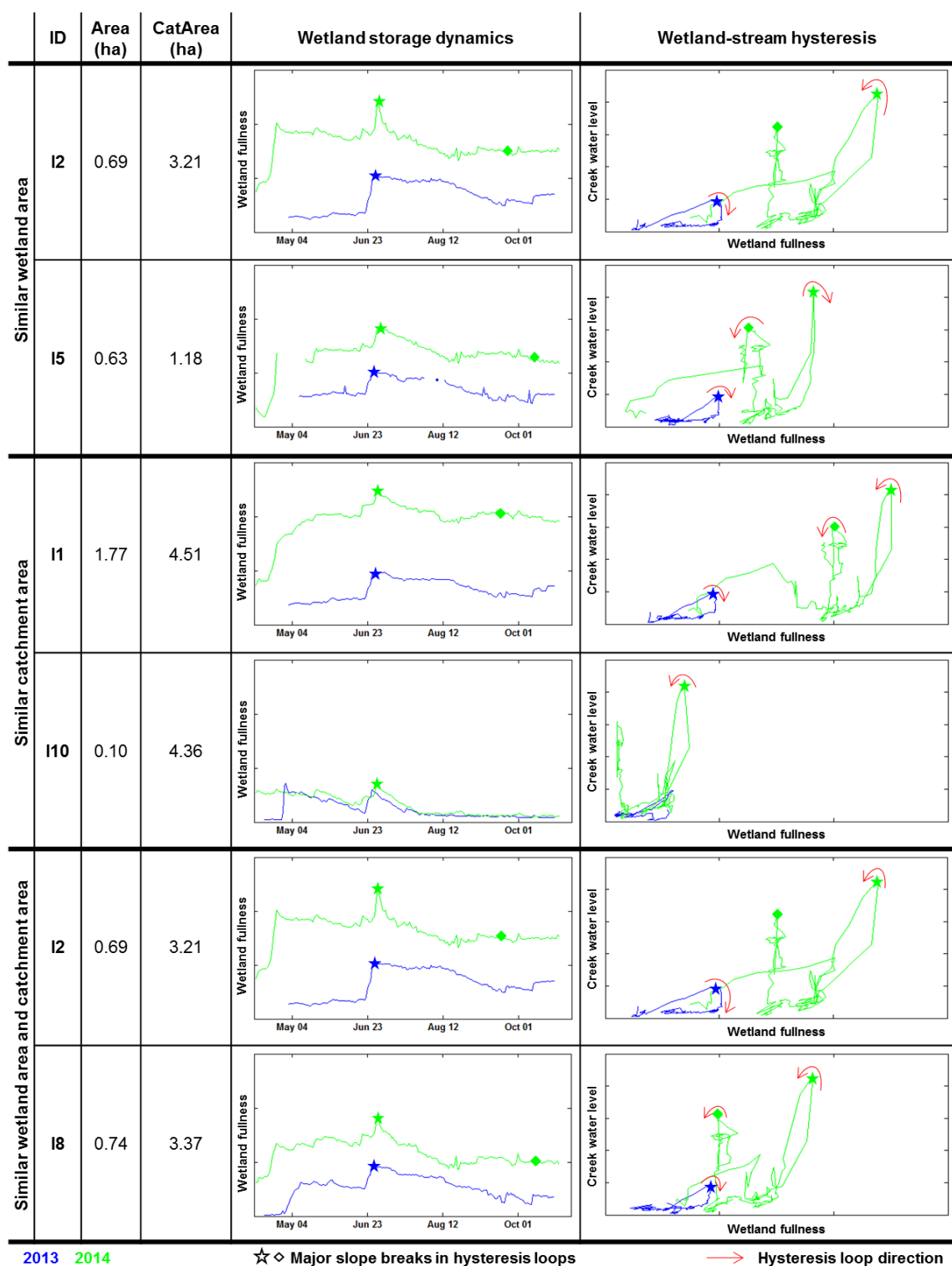
**Figure 2-7:** Spearman's rank correlation coefficients (rho values) between behavioural metrics and wetland characteristics for open-water wetlands for (a) the 2013 and 2014 whole open-water seasons, (b) Spring 2013 and 2014, (c) Summer 2013 and 2014, and (d) Fall 2013 and 2014. For abbreviated wetland characteristics and behavioural metric names, refer to tables 2-1 and 2-2. The numbers “13” and “14” juxtaposed to abbreviated metric names refer to the years 2013 and 2014, respectively. Blank cells flag time periods for which the Spearman's rank correlation coefficient between a wetland characteristic and a behavioural metric was not statistically significant at the 95% level.

%WetRis was negatively correlated with Area, Vol, and Distance and positively correlated with Cat2Vol in 2014, but not in 2013 (Figure 2-7a). The %WetRis metric was also positively correlated with Cat2Area in the spring season of both years (Figure 2-7b). The %WetFal metric showed statistically significant correlation with the same wetland characteristics as the %WetRis metric and for the same time periods, albeit they were of opposite direction (i.e., negative correlation where it was positive for %WetRis and vice versa; Figure 2-7a,b). As for drained wetlands and their ditches, there was almost no statistically significant correlation between water-table behavioural metrics and wetland characteristics (Figure 2-8a to 2-8d).

Lastly, qualitative comparisons of hydrological behaviour between wetlands sharing similar landscape characteristics were also done. For instance, when two open-water wetlands with similar Area were compared (i.e., I2 and I5, see Figure 2-9), the one with the highest value of CatArea (i.e., I2) appeared to be more responsive to rainfall events: It demonstrated larger wetland fullness and flashier dynamics with shorter duration rising and falling limbs than the wetland with a smaller value of CatArea (i.e., I5) in 2014 (Figure 2-9). Wetlands with similar CatArea but different Area also exhibited differences in their responsiveness to wetter conditions. On one hand, the small wetland I10 showed nearly identical dynamics in 2013 and 2014, as revealed by its very similar 2013 and 2014 wetland fullness time series in Figure 2-9. Conversely, the larger wetland, I1, showed noticeably larger wetland fullness in 2014 than 2013 (Figures 2-3 and 2-9). Moreover, I1 had larger wetland fullness values than I10 in both years. Two wetlands with similar CatArea and Area values (i.e., I2 and I8) displayed very similar responses to rainfall events, notably similar rising and falling limb durations. Both wetlands also had larger wetland fullness values in 2014 than in 2013 (Figures 2-3 and 2-9). Regardless of their physical characteristics, all wetlands showed partial recession following spring melt in 2014, meaning that the post-melt wetland



**Figure 2-8:** Spearman's rank correlation coefficients (rho values) between behavioural metrics and historic wetland characteristics associated with drainage ditches for (a) the 2013 and 2014 whole open-water seasons, (b) Spring 2013 and 2014, (c) Summer 2013 and 2014, and (d) Fall 2013 and 2014. For abbreviated historic wetland and ditch characteristics and behavioural metric names, refer to tables 2-1 and 2-2. The numbers “13” and “14” juxtaposed to abbreviated metric names refer to the years 2013 and 2014, respectively. Blank cells flag time periods for which the Spearman's rank correlation coefficient between a drained wetland (or ditch) characteristic and a behavioural metric was not statistically significant at the 95% level.



**Figure 2-9:** Qualitative comparison of hydrological dynamics for open-water wetlands sharing similar landscape characteristics. The range of wetland fullness value on y-axis is 0 to 1.5. Axis tick labels were not included to ensure better readability.

fullness did not recede back to the pre-melt wetland fullness level (Figure 2-9). Summer recession dynamics were, however, different depending on antecedent storage conditions. Indeed, fortuitously the data collected in 2013 and 2014 captured major rainstorm events around approximately the same time in both years (end of June, see Figure 2-2a and 2-2b), allowing a more straightforward comparison between the 2 years. In 2014, when antecedent storage was high, almost all wetlands showed a fast and complete recession after rainfall events, meaning that shortly after the event-driven peak was achieved, wetland fullness receded to a level that was similar to or smaller than pre-event wetland fullness. Conversely, slower, partial recessions were observed in 2013 (Figure 2-9). The only exception to that rule was wetland I10, which demonstrated complete and fast recessions after peak wetland fullness during June rainstorms in both years. Hysteresis dynamics between creek water level and wetland fullness were also different depending on antecedent storage and showed both major and minor loops (Figure 2-9). In summer and fall 2014, two major loops were observed in succession for each site and all but one of those loops were counter-clockwise. However, in 2013, a single major clockwise loop was generally observed in summer at each site (Figure 2-9).

## **2.4 Discussion**

### **2.4.1 Dominant Processes Affecting Open-water Wetlands**

The wetland fullness time series and behavioural metrics computed to characterize open-water wetlands showed considerable variation, not only in time (from one season or year to another) but also in space (from one wetland to another; Figures 2-3 and 2-5). That variation is likely due to the different hydrological processes operating at different times. For instance, the %WetDry metric did not vary a lot, temporally, for individual wetlands but it did vary across

wetlands (Figure 2-5a): This could indicate that processes responsible for wetland dryness (e.g., evapotranspiration and/or groundwater recharge) varied spatially rather temporally, due to some wetlands being recharge zones for shallow groundwater for most of the time whereas others were not (Hayashi et al., 2016). While no groundwater data were presented here in association with intact wetlands, recent analyses of electrical conductivity data in the BCW revealed spatial variability but relative temporal stability in the major groundwater discharge and recharge areas (Ali, Haque, Basu, Badiou, & Wilson, 2017). Hayashi et al. (2016) also mentioned the impact of wetland pond size on wetland dryness: Shallow groundwater outflow per unit area of wetland is typically larger in small wetlands than in large wetlands, due to evapotranspiration in the moist wetland margin. Euliss and Mushet (1996) pointed out that large permanent wetlands are sustained by groundwater whereas small ephemeral wetlands mainly receive water from spring melt. However, based on a water balance approach, Hayashi et al. (2016) disagreed with the assumption of major groundwater input to prairie potholes and argued that wetland pond permanence depends on lateral water input and wetland catchment size. Based on the detailed wetland inventory available for the BCW, the ratio of catchment to wetland surface area was greater for smaller wetlands than larger ones (i.e., per unit of wetland surface area, small wetlands have greater contributing areas than large wetlands), which is aligned with Shook et al. (2013).

In the current paper, some behavioural metrics such as %WetHFull were correlated with CatArea, which would support the process conceptualization put forward by Hayashi et al. (2016). However, our qualitative comparison exercise did not show similar peak wetland fullness and recession behaviour for two wetlands with similar CatArea, thus suggesting that CatArea alone may not be a strong predictor of overall wetland surface water storage dynamics (Figure 2-9). With regards to the role of wetland alteration status on hydrological dynamics, the larger wetland

fullness values generally observed for consolidated wetlands suggest that they are receiving water from different storage sources (e.g., shallow subsurface flow, fill–spill) throughout the year (Figure 2-3). This result is aligned with McCauley et al. (2015) who concluded that due to the drainage of multiple wetlands into one, consolidated wetlands have higher water levels and a larger surface area, in addition to being more permanent than intact wetlands. Indeed, wetland drainage and consolidation can alter surface water storage and groundwater recharge (or discharge) dynamics (van der Kamp & Hayashi, 2009), not only due to the infilling of topographic depressions but also due to the increase of water flow through connecting ditches (McCauley et al., 2015; van Der Kamp & Hayashi, 1998). Additionally, wetland consolidation and drainage can greatly reduce groundwater infiltration and evapotranspiration that would otherwise be occurring in unaltered wetlands (Anteau, 2012; Spaling & Smit, 1995; van Der Kamp & Hayashi, 1998).

Throughout the study period, all consolidated wetlands were more than half-full and never went dry, while some intact wetlands did go dry (Figure 2-3). This could be an indication that consolidated wetlands are acting as discharge zones whereas most of the intact wetlands are acting as depression-focused shallow groundwater recharge zones (Berthold et al., 2004; Hayashi et al., 2003; Hayashi et al., 2016). One could argue that the distinct behaviour of consolidated wetlands – with persistently high wetland fullness – is not due to their alteration status but rather to their relatively large size. This hypothesis is, however, less plausible since some relatively large intact wetlands (e.g., I1 and I2) were associated with high fullness values only in the wetter year of study (2014). Consolidated wetlands were consistently fuller than intact wetlands despite having smaller Cat2Area and Cat2Vol values than intact wetlands. In fact, some of the intact wetlands with large Cat2Area values were dry most of the time (e.g., I7 and I10) or had similar wetland fullness values and recession rates in both years (e.g., I10), despite the wetter antecedent conditions prevailing in

2014 (Figure 2-9). Close inspection of the wetland fullness time series also revealed that while intact wetlands generally had single-peak responses to rainfall events, consolidated wetlands consistently showed multipeak responses (Appendix A-2), thus supporting the hypothesis of consolidated wetland dynamics being driven by asynchronous surface and subsurface water inputs. Another explanation for the different intact versus consolidated wetland dynamics might be due to emergent vegetation, which can greatly affect evapotranspiration rates and hence wetland fullness. Detailed vegetation surveys were, however, not available to further explore that hypothesis. The variation of the %WetSpillage metric between both study years, with more spillage events in 2014 than in 2013 (Figure 2-5b), indicated that fill–spill dynamics depend on antecedent storage conditions and precipitation inputs that are large enough to exceed the available surface storage. Wiltermuth (2014) found that wetland consolidation increases the size of the remaining wetlands and causes their water level to rise towards their spillage points, thus making them more permanent and more prone to chronic flooding. Consolidated wetlands are therefore thought to be less effective in storing floodwater than intact wetlands that are less permanent, rarely reach their spillage point and often dry out. This chronic flooding condition was observed in the current study as all of the consolidated wetlands were above or close to the spillage point for a good portion of the study period. In contrast, for intact wetlands, spillage happened occasionally and only for a few sites in 2014 (Figure 2-3). Variations of the %WetHFull, WetStd, and %WetRis metrics across study years and seasons suggest that the factors governing short-term storage change – such as antecedent moisture conditions and precipitation intensity, frequency, and duration – are highly variable in time (Figure 2-5c to 2-5f).

#### **2.4.2 Dominant Processes Occurring Downslope of Fully Drained Wetlands**

The high or near-surface water tables often observed in association with the ditches located downslope of fully drained wetlands are likely evidence of a strong influence of rainfall inputs and local infiltration on subsurface flow dynamics (Figure 2-4). Water-table responses were also, typically, flashy following rainfall events (Figure 2-4). McCauley et al. (2015) found that complete wetland drainage could increase the flooding probability of a terminal depression located within a wetland complex. However, Hayashi et al. (2016) suggested that wetland drainage may not have any major impact on river discharge unless said drainage targets terminal wetlands (i.e., deep basins located at the end of fill–spill networks). For the purpose of the current study, the drainage ditches selected for monitoring all provided a direct connection between historic wetlands or wetland complexes (even though they did not necessarily fit the definition of “terminal” depressions when they were present on the landscape) and the creek (Figure 2-1c). In this context, we assumed that wetland drainage can strongly contribute to creek dynamics thanks to the structural connectivity provided by the ditches. Furthermore, the role of the ditches as transient shallow subsurface flow pathways can be inferred from the water-table fluctuations (Figure 2-4). For most of the study period, water tables were rarely or never below 100 cm (Figure 2-5g) and often within 50 cm of the ground surface (Figure 2-5i), suggesting the continuous presence of favourable conditions (i.e., hydraulic gradients directed towards the creek) for shallow subsurface flow to occur. Within 50 cm or less of the ground surface, water tables are in soil horizons of higher transmissivity, thus giving rise to large volumes of lateral subsurface flow consistent with the concept of transmissivity feedback (Bishop, Seibert, Nyberg, & Rodhe, 2011) or that of effective transmission zone (Brannen et al., 2015; van der Kamp & Hayashi, 2009). Moreover, during heavy rainfall events, water tables often rose to the near-surface, eventually producing

saturation-excess overland flow contributing to the creek. “Drained” (D) monitoring sites that were associated with historic wetland complexes and relatively long ditches (e.g., D5, D6, and D7, see Figure 2-1c) tended to produce more frequent overland flow (Figure 2-4) than others. These observations partly contradict the assumption of Hayashi et al. (2016), according to which an increase of subsurface storage capacity and transpiration by crops after wetland drainage compensates for the loss of surface storage and evaporation loss from a drained wetland. Both process conceptualizations might be true, and their observation may be strongly dependent on local topographic or geological setting in the PPR. In the current study, the evaluation of the impacts of wetland drainage relied on wetland-specific hydrologic time series as well as the interaction between wetland-specific and creek hydrologic time series to provide multiple lines of evidence and allow process inference (see below).

### **2.4.3 Strength and Temporal Variability of Wetland – Stream Interaction**

Moderately high (absolute) values of the Spearman's rank correlation coefficients between wetland fullness or ditch water-table time series and creek water-level time series could indicate strong wetland–stream (creek) interaction (Figure 2-6). While positive correlation coefficients could be an indication of potential connectivity between wetlands and the creek, negative correlations could also indicate potential connectivity but with asynchronous water-level peaks reflecting important time delays. The direction of correlation (i.e., positive or negative) is, therefore, likely influenced by antecedent wetness conditions and wetland surface storage. For example, wetland fullness at site I4 showed negative correlation with creek water level throughout the study period except in summer and fall 2014. In these two seasons, site I4 experienced very different local dynamics (i.e., less than half-full in summer 2013 versus spillage in summer 2014; dry in fall 2013 versus more than half-full in fall 2014, see Figures 2-3 and 2-6). Similarly, changes

in the direction of correlations between creek water level and wetland fullness or ditch water table, from overly negative in spring 2013 to mostly positive in spring 2014, can also be attributed to the larger antecedent storage and soil moisture at the beginning of the open-water season in 2014. Physical connections between the creek and the ditches were obvious and routinely observed during field visits (Figure 2-1c). However, the temporal variability in correlation strength between ditch water table and creek water-level time series hints at the nonstationary interaction between drained wetlands and the creek (Figure 2-6). The occurrence of overland flow events in all drainage ditches and the correlation between water table and creek water-level time series suggest the possibility of both surface and subsurface flow pulses reaching the creek through the ditches located downslope of drained wetlands (Figures 2-4 and 2-6). The saturation-excess overland flow events observed, more frequently in 2014 than 2013, also suggest the presence of stronger surface water–groundwater interactions in 2014 because of larger antecedent subsurface storage, that is, larger soil moisture levels in 2014 than 2013 (Figure 2-2). These observations support the hypothesis that wetland drainage can convert a non-contributing area into a contributing area and thus increase hydrologic connectivity (Anteau, 2012; Wiltermuth, 2014). It is worth noting that the use of a correlation coefficient to assess potential wetland–stream connectivity could be challenged, as it might not necessarily reflect causality between wetland and stream dynamics but rather highlight the fact that similar drivers are behind stream and wetland dynamics. Regardless, the variation of the computed Spearman's rank correlation coefficients across years and seasons provided insights into temporally variable hydrological processes – and the factors controlling those processes – and how that could ultimately affect the establishment of wetland–stream connectivity.

#### **2.4.4 Predictability of Wetlands Dynamics Based on Landscape Characteristics**

There were some consistent positive correlations across years and seasons between several wetland characteristics (e.g., Area, Peri, Vol, ConArea, and ConPeri) and the %WetHFull metric, implying that landscape factors exert an important and temporally persistent control on wetland storage dynamics (Figure 2-7a to 2-7d). It should, however, be noted that some wetland geometry properties (e.g., Area, Peri, and Vol) are inherently intercorrelated due to the way they were computed; for instance, Minke, Westbrook, and van der Kamp (2010) used a simplified v-a-h method to estimate water storage in prairie wetlands and identify a strong relation between wetland area and wetland perimeter. The strong interrelations that exist among some landscape characteristics and among some of the wetland behavioural metrics estimated in this paper may have led to the large number of statistically significant correlations observed (Figure 2-7a to 2-7d). This, however, means that when attempting to predict wetland storage dynamics, several choices of landscape characteristics may be available and different scientists and practitioners may be able to only select the ones that are easier for them to quantify. Negative correlations between the %WetDry metric and Area, Vol, and Peri (Figure 2-7a,c,d) support the assumption that larger wetlands rarely dry out (Anteau, 2012; Hayashi et al., 2016; Wiltermuth, 2014). However, the lack of consistent, statistically significant correlations across years between other behavioural metrics (i.e., %WetSpillage and WetStd), and individual wetland characteristics (Figure 2-7a to 2-7d) made it obvious that the relationships between physical landscape characteristics and storage dynamics are nonstationary.

All wetland characteristics, except Distance, never showed any statistically significant correlation with the %WetSpillage metric (Figure 2-7a to 2-7d): This suggests that the occurrence of wetland spillage might be less controlled by static landscape characteristics and instead

controlled by temporally dynamic precipitation inputs, general antecedent storage conditions, and the storage memory of individual wetlands. Indeed, in the current study, memory effects manifested themselves through different recession dynamics in 2013 versus 2014, namely, slow and partial recessions in the presence of low antecedent storage in 2013 and rather fast (i.e., steep sloped) and full recessions in the presence of larger antecedent storage in 2014 (Figure 2-9; Appendix A-2). Slow and partial recessions were identified through large wetland fullness values persisting long after major events: That temporal persistence indicates strong autocorrelation in wetland water level despite temporally random rain events, which is evidence of memory effects as described by Shook et al. (2015). The fact that Cat2Area was positively correlated with %WetRis and negatively correlated with %WetFal in spring of both years (Figure 2-7b) suggests that wetlands with catchments that are disproportionately large in comparison with their surface area are more responsive to water inputs in spring. Therefore, wetlands with larger Cat2Area tend to have greater wetland fullness rising rates and faster recession rates than wetlands with smaller Cat2Area. This hypothesis is also supported by the qualitative comparison of wetland hydrologic responses for sites sharing similar physical characteristics (e.g., I2, I5, and I8; see Figure 2-9). However, wetland I10 with its large Cat2Area value did not support that hypothesis; this may be due to a very short storage memory, as was suggested by Shook et al. (2015) while examining other prairie pothole wetlands in Alberta, Saskatchewan, and Manitoba.

For the major rainfall event in June 2013, clockwise hysteretic relations were observed across sites, indicating that creek water level peaked before wetland fullness. This would imply that wetland– stream connectivity, if present in 2013, was not established via fill– spill and overland flow. On the contrary, in 2014, the majority of hysteretic relations were counter-clockwise (Figure 2-9), implying that wetlands reached a critical peak storage value before the

creek reached its peak water level, a scenario compatible with the idea of wetland– stream connectivity via overland flow. Those opposite hysteretic dynamics in 2013 versus 2014 are undoubtedly the root cause of the opposite results obtained earlier via correlation analyses (Figure 2-6). These observations are also aligned with other studies that examined hysteresis loops and reported the influence of storage memory on wetland hydrologic response (Shook et al., 2015; Shook & Pomeroy, 2011). In the current study, in both years, all major and minor hysteresis loops returned to their original starting points: This agrees with the return-point memory concept associated with the Preisach hysteresis model (Flynn, McNamara, O’Kane, & Pokrovskii, 2006) that was relied upon in previously published prairie pothole wetland studies (e.g., Shook et al., 2015; Shook & Pomeroy, 2011). In 2014, the two major hysteresis loops were parallel and mostly congruent, which is also in agreement with the Preisach hysteresis model. The reverse dynamics observed across the majority of wetlands in 2013 versus 2014 stress the importance of considering overall antecedent storage conditions, which may override any influence of local landscape properties on potential and actual wetland-stream connectivity (Figures 2-6 and 2-9). Overall, the presence of hysteresis between wetland fullness and stream stage might signal a delay between wetland storage release and stream stage fluctuations and hence be associated with the travel time of water between wetland and stream. It should be noted, however, that while we assumed channel routing to have a negligible effect on our hysteresis relations, there remains a slight possibility that this assumption might be erroneous. Lastly, almost no statistically significant correlation between drained wetland characteristics and behavioural metrics could be found (Figure 2-8): This suggests that the dynamics of drained wetlands are modified in such a way that they are no longer influenced by “natural” landscape physical characteristics in a manner that is statistically predictable.

## 2.5 Conclusion

This study aimed to enhance our hydrological understanding of intact and human-altered prairie pothole wetlands by examining their storage dynamics across space and time and identifying landscape controls on these dynamics. The focus was on high-frequency time series of wetland fullness, water-table fluctuations, and creek water level collected under contrasted conditions. Antecedent storage and memory effects were found to be critical determinants of wetland behaviour. Consolidated wetlands were more susceptible to spillage than intact wetlands. Water tables downslope of drained wetlands were very responsive to precipitation and their response quasi-synchronous with that of the creek, suggesting the potential for enhanced subsurface connectivity post-drainage. Such results underline the importance of monitoring drainage ditches downslope of fully drained wetlands to better understand their impact on watershed dynamics, a component usually lacking from wetland hydrology studies. Some open-water wetland characteristics such as their area, storage volume, ConArea, and ratio between wetland area and ConArea, had a strong influence on surface water storage dynamics. However, the lack of temporally consistent correlations between wetland behavioural metrics and landscape characteristics stressed the importance of assessing wetland–landscape interactions under different antecedent wetness conditions to better identify nonstationary processes. The highlighted relationships between landscape characteristics and specific components of wetland hydrological behaviour could be useful for developing scientific criteria towards selecting priority wetlands for conservation and restoration. For example, knowledge regarding the fact that consolidated wetlands are more susceptible to spillage could be important if proven true for a larger portion of the PPR. The lack of correlation between landscape characteristics and drained wetland behaviour also hinted that hydrologic predictability decreases as a result of wetland loss, which has important

implications for flood forecasting in the PPR. Further research is, therefore, needed to confirm or reject some of the storage dynamics versus landscape factor relationships established in the current study.

## **2.6 Acknowledgments**

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### **Chapter 3: Event-Based Analysis of Wetland Hydrologic Response in The Prairie Pothole Region**

## **Abstract**

Previous studies in the Prairie Pothole Region (PPR) mainly assessed wetland hydrologic function at seasonal, annual and decadal scales. However, no study looked at wetland dynamics in response to individual rainfall events, especially not while considering prairie pothole wetlands (PWs) across a gradient of alteration. The present study aimed to investigate: (1) the most important metrics needed to characterize the spatial variability of wetland hydrologic response to rainfall-runoff events; (2) the temporal variability of individual wetland hydrologic response; (3) the spatial and temporal variability of wetland-stream interaction; and (4) the temporal persistence of various spatial controls on individual wetland hydrologic response characteristics. High-frequency water level data was collected over two years for ten intact, three consolidated, and seven fully drained PWs, as well as a creek located in southwestern Manitoba, Canada. The hydrologic response of the studied PWs to individual rainfall-runoff events was characterized using a range of metrics. Several data analysis methods were used, including principal component analysis, graphical assessments of wetland-stream hysteresis dynamics, and correlations analyses between wetland response metrics and spatial characteristics. Results showed that wetland alteration status (i.e., drained versus intact, open-water wetlands) plays an important role in explaining differences in the event-scale hydrologic behaviour of PWs. Climatic and antecedent storage conditions also had a strong influence on the hydrologic responses of PWs during individual rainfall-runoff events and appeared to override the influence of spatial controls such as wetland area, volume or catchment area. Antecedent storage also seemed to be the driving factor of wetland-stream interactions. A lack of persistent correlations between wetland spatial characteristics and response metrics was observed and suggested nonstationary wetland hydrological behaviours and controls, a conclusion that has significant implications for wetland classification and modelling.

**Keywords:** rainfall-runoff events, Prairie Pothole Region, wetland hydrologic response, wetland-stream interactions, spatial wetland characteristics

### 3.1 Introduction

Event-based analysis of stream hydrographs is commonly used for investigating dominant runoff generation processes in a watershed (Gou et al., 2019; Kirchner, 2019). Specifically, different event hydrograph metrics – related to the magnitude and timing of hydrograph response – can be used to infer watershed dynamic storage and release mechanisms (Blume et al., 2007; Kirchner, 2019). For instance, event peak flow rates have been documented across many watersheds and linked to catchment area (Hannah, et al., 2000). Response lag times have also been reported as important. Notably, the lag to peak flow, defined as the time elapsed from the beginning of the rising limb to the occurrence of the peak flow rate, is a good indicator of a watershed's ability to delay its response to a rainfall event (Beven, 2012; Hannah et al., 2000; Te Chow et al., 1988). As for recession rates, they describe the rate of change in flow during the receding limb of a hydrograph and have been linked to watershed spatial characteristics such as depressional, surface and subsurface storage (Hannah et al., 2000; Te Chow et al., 1988). Seasonal and annual average values of hydrograph metrics have also been correlated to watershed spatial characteristics, making it possible to estimate the former as a function of the latter. For example, Yadav et al. (2007) showed that rainfall-runoff response characteristics such as the runoff ratio and the slope of the flow duration curve of ungauged watersheds can be expressed as a function of watershed topography (i.e., slope), subsurface geology and soils characteristics. Buttle (2006) also argued that rainfall-runoff responses can be explained by three first-order controls: typology, topography, and topology. Further, the event-based analysis of hydrographs – sometimes in combination with chemographs – has highlighted the importance of nonlinear relationships – either in the form of input-output thresholds or input-output loops. Loop-like behaviours have been, notably, described as hysteretic relationships, whereby a given variable exhibits significantly

different dynamics on the rising and falling limbs of a hydrograph (Gharari and Razavi, 2018; Zuecco et al., 2016). Several studies have looked at examples of hysteretic relationships between streamflow and other hydrological variables (e.g., antecedent storage, travel times, water level, etc.) during individual rainfall-runoff events (Andermann et al., 2012; Burt et al., 2015; Davies and Beven, 2015; Fovet et al., 2015; Gharari and Razavi, 2018; Shook and Pomeroy, 2011; Valkama and Ruth, 2017; Zuecco et al., 2016). Other studies have rather looked at the hysteretic behaviour between streamflow and water quality variables such as stream nitrate and total phosphorus concentrations (Lloyd et al., 2016; Valkama & Ruth, 2017), bedload and suspended sediment concentrations (Mao et al., 2014; Landers and Sturm, 2013; Valkama & Ruth, 2017). Overall, event-based hydrograph analysis approaches have been successfully used to conceptualize stream and watershed processes. However, they have yet to be applied to smaller, yet important watershed features such as wetlands, especially in landscapes which are dominated by such features.

While wetlands occupy less than 10% of the global landmass, they are immeasurably important for the ecosystem and socio-economic services they provide (Mitsch & Gosselink, 2007). Indeed, wetlands are known to play critical roles in mitigating floods (EPA, 2015; Gleason et al., 2011; Wang et al., 2010), reducing downstream nutrient concentrations (Brunet et al., 2012) and sediment loading (Preston et al., 2013), and contributing to groundwater recharge (Bullock and Acreman, 2003; EPA, 2015; Gleason et al., 2007; Hayashi et al., 2016; Wang et al., 2010; Yang et al., 2010). Extensive wetland loss has taken place around the World in the last 50 years, and especially so in archetypical landscapes such as the Prairie Pothole Region (PPR) (Van Meter and Basu, 2015). The PPR is a vast – and hydrologically, biogeochemically and ecologically important – area in North America, spanning Canada and the United States, where numerous

topographic depressions or small basins left by glacier retreat have created a dense network of pothole wetlands (PWs), small lakes and streams (average density: 13 basins/km<sup>2</sup>; range: 1–195 basins/km<sup>2</sup>) (U.S. Fish and Wildlife Service, 2010). Those densities are, however, changing since rapid agricultural expansion in the PPR has led to significant surface water drainage activities (Johnston, 2013). Historically, draining small PWs required minimum effort and farmers did it for the convenience of agricultural operation (Van Meter & Basu, 2015). However, in some cases, multiple small PWs are drained into one large one (i.e., a consolidated wetland), an attempt to compensate the loss of small wetlands by creating a large one (Haque et al., 2018; McCauley et al., 2015). PWs, therefore, offer a unique opportunity to study the hydrological behaviour of small watershed features that exist along a continuum of alteration, from intact to consolidated and fully drained, and their impact on stream dynamics.

PWs are completely surrounded by uplands without continuous or well-defined surface connections to streams (Cohen et al., 2016; Leibowitz, 2003, 2015; Mushet et al., 2015; Tiner, 2003). Despite their generally small surface area (< 1,000 m<sup>2</sup>), the hydrological dynamics prevailing in individual PWs are thought to exert a significant influence on overall watershed hydrology (Cohen et al., 2016; Hayashi et al., 2003; Mushet et al., 2015;). While wetland hydrological dynamics can be described using several variables (EPA, 2015; Mitsch & Gosselink, 2007), pond water level (hereafter simply referred to as “water level”) is typically relied upon because it is easy to monitor. Park et al. (2014) and Hayashi et al. (2016) identified precipitation and evapotranspiration as key controls on pond permanence (Hayashi et al., 2016). In addition to those two variables, pond water level fluctuations are also known to depend on lateral water inputs to the wetland, especially in response to rainfall-runoff events (Hayashi et al., 2016; Shook 2013). During and after such events, PWs that belong to wetland complexes can be inter-connected and

connected to streams through overland (i.e., via fill-spill events) and shallow subsurface flow. Individual wetland hydrological behavior is also controlled by static landscape properties such as wetland geometry, geographic position, and alteration status (Haque et al., 2018; Hayashi et al., 2016). For instance, fill-spill events are possible for intact, partially drained and consolidated wetlands (McCauley et al., 2015; Shaw et al., 2012) but no longer exist in the case of fully drained wetlands. Fully drained wetlands can, nevertheless, contribute water to downstream systems by generating surface and shallow subsurface flow (Haque et al., 2018).

It should be noted that while numerous studies have characterized individual or cumulative wetland hydrologic function in a watershed context (e.g., Bullock & Acreman, 2003; EPA, 2015; Gleason et al., 2007; Haan & Johnson, 1968; Hayashi et al., 2003; Hayashi et al., 2016; Huang et al., 2011; Huang et al., 2013; Hubbard & Linder, 1986; Johnston et al., 1990; Lindsay et al., 2004; McAllister et al., 2000; McLaughlin et al., 2014; Tiner, 2003; Wang et al., 2010; Yang et al., 2010), they have almost exclusively done so over intermediate to long timescales. In the PPR, in particular, recent examples include the assessment of wetland hydrologic function, through field data or models, at seasonal and annual scales (Ali et al., 2017; Brannen et al., 2015; Euliss and Mushet, 1996; Golden et al., 2014; Haque et al., 2018; Hayashi et al., 2016) as well as decadal scales (Ehsanzadeh et al., 2016; Hayashi et al., 2016). One major gap, in the current literature, is the assessment of individual wetland hydrological dynamics over short timescales, namely specific snowmelt or rainfall events. In a manner similar to streams that can exhibit a wide range of flow responses to snowmelt or rainfall, subdued or sharp water level fluctuations can also occur in wetlands in general and in PWs, in particular, as a function of precipitation intensity, antecedent water storage conditions and other factors (Haque et al., 2018). The emergence of easily deployable and relative inexpensive water level loggers now allows for the high-frequency

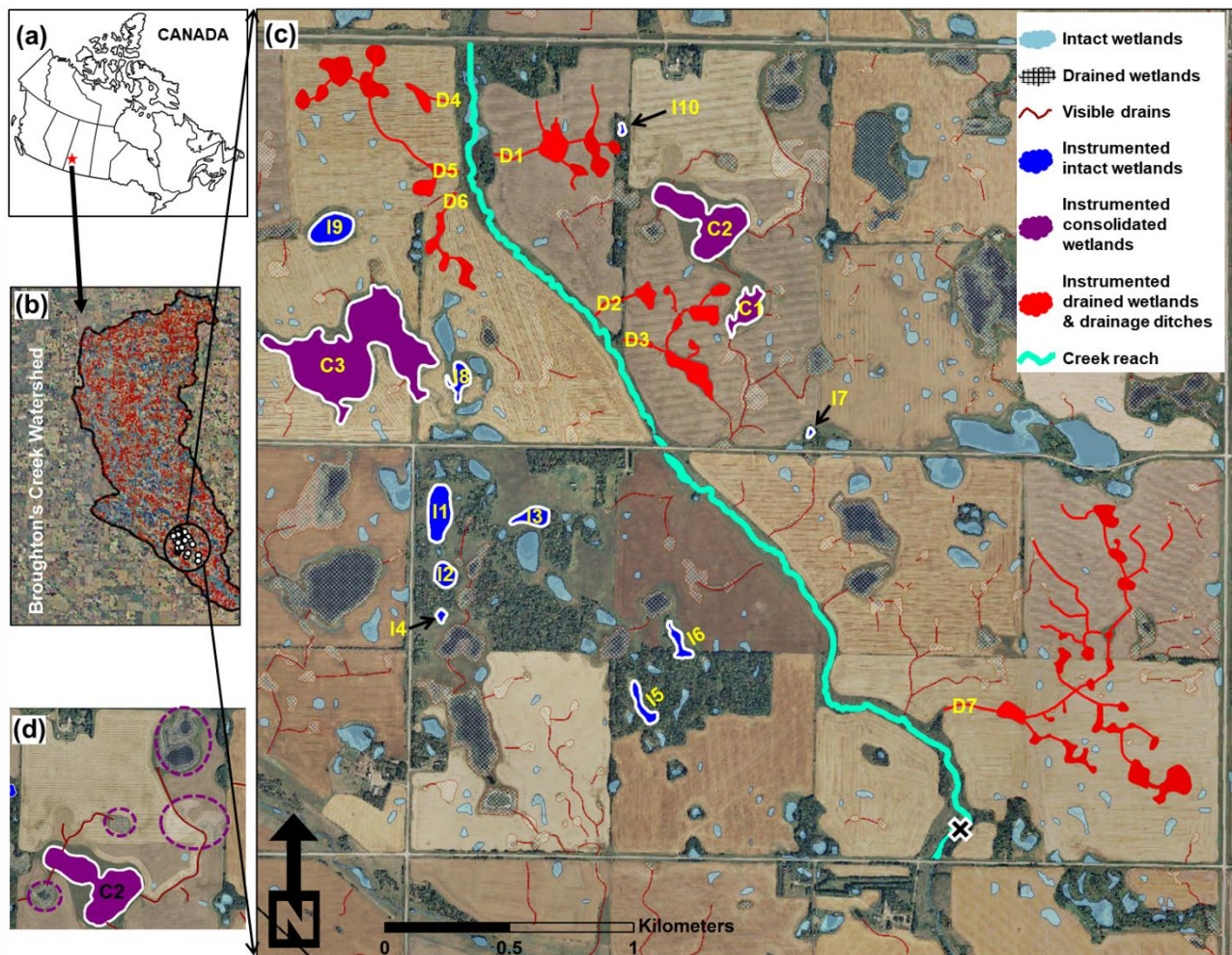
monitoring of wetland water level fluctuations and the capture of both short-lived and prolonged wetland hydrologic responses to precipitation. One potential way to exploit that high-frequency information is to estimate metrics from wetland water level hydrographs, in a manner similar to what is traditionally done with stream hydrographs, to assess the magnitude and timing of wetland hydrologic response to individual precipitation-runoff events. This type of high-frequency information also provides an opportunity for inferring wetland-stream interaction by comparing the timing of wetland and stream hydrologic response to precipitation-runoff events, notably via the assessment of hysteretic relationships. The wetland hydrology literature published to date does not report on event-specific wetland-stream dynamics, especially not while considering PWs across a gradient of alteration status (e.g., intact, consolidated, fully drained). Focusing on such short timescales is especially important in the context of land use and climate change, as some regions around the World, including the PPR, are expected to face increased frequencies of extreme, but short-lived rainfall-runoff events (Khaliq et al., 2015; Pfahl et al., 2017). Extreme rainfall-events were, historically, rare in the PPR characterized by a semi-arid to sub-humid climate and as a result, the hydrologic response of PWs to these events is poorly understood. The present study, therefore, aims to rely on high-frequency pond water level, ditch water table and stream water level data, combined with a detailed landscape analysis, to investigate: (1) the most important metrics needed to characterize the spatial variability of wetland hydrologic response to rainfall-runoff events; (2) the temporal variability of individual wetland hydrologic response; (3) the spatial and temporal variability of wetland-stream interaction; and (4) the temporal persistence of various spatial controls on individual wetland hydrologic response characteristics.

## **3.2 Methods**

### **3.2.1 Study Site Description**

The Broughton's Creek is a tributary of the Little Saskatchewan River and is located in southwestern Manitoba, Canada (Figure 3-1a). It drains an area of approximately 252 km<sup>2</sup> (Figure 3-1b). The Broughton's Creek Watershed (BCW) can be described as a typical PPR landscape which was created between 20,000 to 12,000 B.C due to the retreat of the Assiniboine glacial lobe (Wang et al., 2010; Yang et al., 2010). This area has numerous potholes, sloughs, and lakes and is described as a hummocky till plain. Most soils in the watershed are well-drained Newdale association soils (i.e., mainly Orthic Black Chernozems). However, some near-stream locations also have soils from the Dorset, Drokan, Eroded Slope Complex and Jaymar associations (MAFRI, 2008; Wang et al., 2010). According to the U.S. soil taxonomic classification, the Udic Boroll subgroups have similar properties as soils from the Newdale, Dorset, and Jaymar associations (Soil classification working group, 1998). Most of the land in the BCW is used for agricultural purposes (71.4%), followed by rangeland (10.8%), wetland (9.8%), forest (4%) and other uses (4%). Detailed landscape change detection studies have shown that between 1968 and 2005 approximately 21% of the wetland area in the watershed has been lost (i.e., those are, currently, fully drained wetlands that no longer have an open water component). However, 70% of the wetland area has been impacted when we account for all hydrological alterations, including partly drained wetlands and farmed wetlands (Wang et al., 2010).

For continuous monitoring of wetland hydrology, thirteen open water PWs and seven drainage ditches were selected on both sides of a 5 km creek reach in the BCW (see Figure 3-1c for the exact locations). Out of the thirteen open water PWs, ten are intact PWs (i.e., hydrologically undisturbed wetlands) and the remaining three are consolidated PWs. According to Ducks



**Figure 3-1:** (a) Location of the Broughton's creek watershed within Canada. (b) Creek reach and surrounding land under study. (c) Instrumented wetlands in the Broughton's creek watershed. I: Intact wetland, C: Consolidated wetland, D: Drained wetland(s) and associated ditch(es). The black cross shows the creek water level measuring location. (d) Example of consolidated wetland, C2; dashed circles indicate historic wetlands that have been drained into C2.

Unlimited Canada's (DUC) wetland inventory established for the BCW, there were no morphological changes to the intact PWs in the last 60 years. However, multiple PWs were drained to form a consolidated PW (see example in Figure 3-1d). Drainage ditches have visible surface

connections to the creek reach and are typically located downslope of fully drained PWs. Drainage ditches were selected based on the same previously mentioned wetland inventory dataset, which includes the location of historic PWs and confirms their role in diverting water from the drained PW basins to the creek reach. As indicated by Haque et al. (2018), over time these ditches have accumulated a lot of eroded materials, and overland flow through them is rare except during heavy rainfall.

A series of wetland spatial characteristics (Table 3-1) were calculated for all studied PWs (i.e., open water and fully drained) using the wetland inventory dataset provided by DUC. For open water PWs, DUC used the V-A-H method (Pomeroy et al., 2010) to calculate storage volume since this method can estimate volume below the standing water from LiDAR data. However, for drained PWs, the Surface Volume Tool in ESRI ArcGIS was used to calculate volume from LiDAR data. The areas of the open water PWs range from  $7.4 \times 10^2$  to  $1.5 \times 10^5$  m<sup>2</sup>, while their storage volumes range from  $7.4 \times 10^1$  to  $4.4 \times 10^4$  m<sup>3</sup>. The drained wetlands associated with the instrumented ditches had areas and storage volumes ranging from  $7.8 \times 10^2$  to  $1.3 \times 10^4$  m<sup>2</sup> and  $1.5 \times 10^2$  to  $7.0 \times 10^3$  m<sup>3</sup>, respectively. Euclidean distances estimated between each open water PW and the study creek reach range from  $4.7 \times 10^2$  to  $1.0 \times 10^3$  m, whereas ditch lengths for the drained wetlands range from  $9.3 \times 10^1$ - $2.2 \times 10^2$  m. Catchment areas for the open water and drained PWs range from  $6.0 \times 10^3$ - $3.1 \times 10^5$  m<sup>2</sup> and  $9.8 \times 10^3$ - $9.0 \times 10^4$  m<sup>2</sup>, respectively. Some spatial characteristics such as the incremental contributing area (ConArea) and the total drainage area (TotDrainArea) are only relevant for the consolidated open water PWs and drained PWs. The incremental contributing area of a consolidated PW was calculated by summing up the catchment areas of all the drained PWs that pre-existed the consolidated PW.

**Table 3-1:** Overview of wetland spatial characteristics for open water (i.e., current) and drained (i.e., historic) prairie pothole wetlands.

I: intact wetlands; C: consolidated wetlands; D: drained wetlands.

Name	Abbreviation	Valid for	Description (units)	Range
Area	Area	I & C D	Surface area of current or historic wetland (m <sup>2</sup> )	7.4×10 <sup>2</sup> -1.5×10 <sup>5</sup> 7.8×10 <sup>2</sup> -1.3×10 <sup>4</sup>
Perimeter	Peri	I & C D	Edge length of current or historic wetland (m)	1.0×10 <sup>2</sup> -2.8×10 <sup>3</sup> 1.0×10 <sup>2</sup> -6.6×10 <sup>2</sup>
Storage volume	Vol	I & C D	Storage volume of current or historic wetland (m <sup>3</sup> )	7.4×10 <sup>1</sup> -4.4×10 <sup>4</sup> 1.5×10 <sup>2</sup> -7.0×10 <sup>3</sup>
Bottom elevation	Elev	I & C D	Elevation of wetland or ditch bottom above sea level (m)	5.2×10 <sup>2</sup> -5.3×10 <sup>2</sup> 5.2×10 <sup>2</sup> -5.3×10 <sup>2</sup>
Spill level	SL	I & C	Height of water (relative to stilling well bottom) at which wetland will spill (m)	6.1×10 <sup>-1</sup> -1.2×10 <sup>0</sup>
Catchment area	CatArea	I & C D	Natural basin area (or drainage area) of current or historic wetland (m <sup>2</sup> )	6.0×10 <sup>3</sup> -3.1×10 <sup>5</sup> 9.8×10 <sup>3</sup> -9.0×10 <sup>4</sup>
Catchment to wetland ratio	Cat2Area	I & C D	Ratio of catchment area to surface area for current or historic wetland (-)	1.8×10 <sup>0</sup> -4.2×10 <sup>1</sup> 3.6×10 <sup>0</sup> -1.3×10 <sup>1</sup>
Catchment to volume ratio	Cat2Vol	I & C D	Ratio of catchment area to storage volume for current or historic wetland (m <sup>-1</sup> )	6.2×10 <sup>0</sup> -5.6×10 <sup>2</sup> 6.8×10 <sup>0</sup> -6.7×10 <sup>1</sup>
Catchment perimeter	CatPeri	I & C D	Edge length of current or historic wetland catchment (m)	4.1×10 <sup>1</sup> -4.8×10 <sup>3</sup> 6.6×10 <sup>2</sup> -2.1×10 <sup>3</sup>
Shortest distance to creek	Distance	I & C	Euclidean distance between wetland center and nearby creek (m)	4.7×10 <sup>2</sup> -1.0×10 <sup>3</sup>
Ditch length	Length	D	Length of drainage ditch routing water from historic wetland to nearby creek (m)	9.3×10 <sup>1</sup> -2.2×10 <sup>2</sup>
Incremental contributing area	ConArea	C D	Increase in current or historic wetland catchment area due to human alteration, i.e., drainage (m <sup>2</sup> )	0.0×10 <sup>0</sup> -1.2×10 <sup>5</sup> 0.0×10 <sup>0</sup> -6.2×10 <sup>4</sup>
Incremental contributing perimeter	ConPeri	C D	Edge length of current or historic wetland incremental contributing area (m)	0.0×10 <sup>0</sup> -2.4×10 <sup>3</sup> 0.0×10 <sup>0</sup> -1.8×10 <sup>3</sup>
Total drainage area	TotDrainArea	I & C D	Sum of wetland catchment area and wetland incremental contributing area for current or historic wetland (m <sup>2</sup> )	6.0×10 <sup>3</sup> -3.8×10 <sup>5</sup> 9.8×10 <sup>3</sup> -1.2×10 <sup>5</sup>
Total drainage area to catchment area ratio	TotDrain2Cat Area	I & C D	Ratio of wetland total drainage area to wetland catchment area for current or historic wetland (-)	1.0×10 <sup>0</sup> -1.5×10 <sup>0</sup> 1.0×10 <sup>0</sup> -2.3×10 <sup>0</sup>

Similarly, the TotDrainArea of a consolidated PWs was estimated as the sum of its original catchment area and its incremental contributing area.

### **3.2.2 Hydrometric and Climate Data Collection**

A capacitance-based water level logger (Odyssey<sup>TM</sup>, Dataflow Systems) was installed at the downstream end of the creek reach (i.e., south-end) and in each open water wetland using stilling wells, to record hourly surface water level fluctuations. For each of the drainage ditches, a one-meter-deep water table well was installed to house the same kind of logger for measuring hourly shallow water table fluctuations. Continuous monitoring of the creek reach, open water wetlands and ditches was done during the 2013 and 2014 open water seasons (i.e., April to October).

The closest Environment and Climate Change Canada weather station (Brandon A, climate ID: 5010481, World Meteorological Organization (WMO) ID: 71140) is located about 30 km south-west from the study location; other researchers relied on the data from that station and found it representative of BCW weather (Wang et al., 2010; Yang et al., 2010). Climate data pertaining to air temperature and daily rainfall amounts were extracted from the historical record available for that weather station. Mean air temperature was similar in both study years (~13.5°C in April-October 2013 and 2014), whereas total open water season precipitation was higher in 2014 (i.e., 523.4 mm) compared to 2013 (i.e., 402.8 mm). However, total open water season precipitation in both years was higher than the long term (1960 – 2014) average (i.e., 344.2 mm) in that region.

### **3.2.3 Rainfall-Runoff Event Delineation and Response Metrics**

Daily average water levels were calculated from the hourly recorded data for all the open water wetlands, drained wetlands and creek reach. Average values were used simply to avoid

dealing with regular, diurnal fluctuations or cycles in pond water level that occur irrespective of rainfall-runoff events, this as a result of the strong evaporative signal in the region. We used wetland fullness values (i.e., we divided the wetland water depth by the wetland spillage level) for open water wetlands as in Haque et al. (2018). Using wetland fullness values allowed us to normalize water level data for comparing wetlands with each other. For reasons related to data availability, the focus of this study was solely on rainfall-runoff events and snowmelt events were not considered. The start and end dates of rainfall-runoff events were identified from the creek data only, i.e., by manually matching daily rainfall records and daily creek water level timeseries. Based on the creek response, a total of 17 rainfall-runoff events were identified across the 2013-2014 open water seasons (Table 3-2). For each studied wetland, surface water level fluctuations or water table fluctuations were examined across the whole duration of each event. A creek or wetland response was identified as the presence of a rise in water level (i.e., departure from pre-event water level), followed by a peak and then a decrease. Responses of the studied PWs during individual rainfall-runoff events were compared with the respective creek water response during each of the identified rainfall-runoff events (Figure 3-2). Almost all the studied PWs responded noticeably during each of the identified rainfall-runoff events. The exceptions were events #5 and #6 where more than 50% of the studied wetlands did not respond to rainfall (Table 3-2). Because of this, these two events were excluded from further analysis. A series of hydrologic response metrics were estimated for each of the studied PWs and each of the rainfall-runoff events (Table 3-3). Similar to traditional hydrograph analysis for streams, metrics such as the maximum water level (i.e., MaxWL), final water level (i.e., FinalWL) and water level percentage change (i.e., WLPercChange) were estimated to get a sense of wetland storage capacity. Metrics like the lag to maximum water level (i.e., LagToMaxWL), water level rise rate (i.e., WLRiseRate) and water

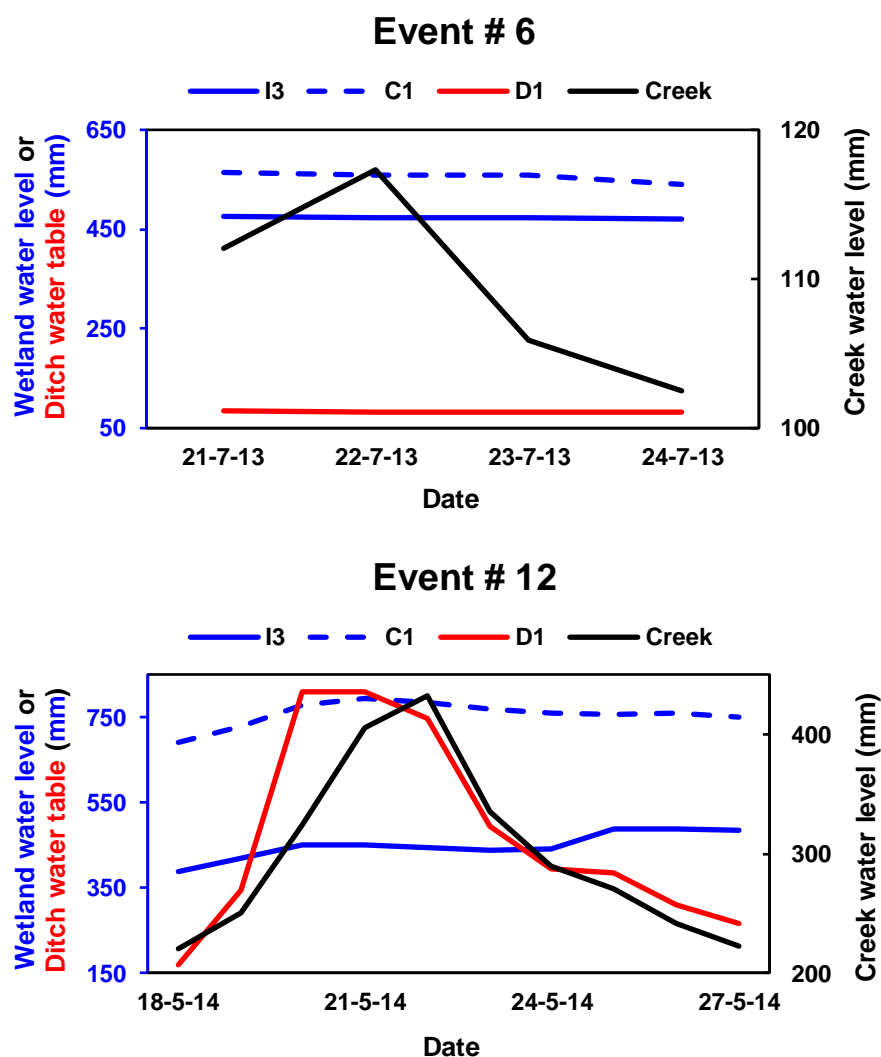
level recession rate (i.e., WLRecessRate) were also estimated to understand the temporal dynamics of storage change for individual PWs during a rainfall-runoff event. In total, thirteen different hydrologic response metrics were computed (Table 3-3) for each of the studied PWs and for each rainfall-runoff event.

**Table 3-2:** Rainfall-runoff events identified from stream hydrographs, and corresponding responses of monitored open water and drained wetlands (PWs). TR = Total event rainfall. ARI = average (event) rainfall intensity.

Event #	TR (mm)	ARI (mm/day)	% of PWs that responded to the event	% of PWs that did not respond to the event (Site ID)
1	20.6	5.2	100	0
2	12.7	6.4	95	5 (I7)
3	22.4	7.5	100	0
4	95.4	10.6	100	0
5	28.6	4.1	35	65 (I1, I2, I3, I4, I5, I6, I7, I8, I9, I10, C1, C2, <b>D4</b> )
6	8.4	8.4	40	60 (I3, I4, I5, I6, I7, I8, I9, C1, <b>D1, D3, D6, D7</b> )
7	7.2	3.6	95	5 ( <b>D5</b> )
8	16.6	8.3	100	0
9	26.0	6.5	90	10 (I5, I6)
10	9.4	4.7	100	0
11	26.6	6.7	100	0
12	47.4	15.8	100	0
13	19.2	9.6	100	0
14	47.0	15.6	100	0
15	102.6	20.5	100	0
16	20.4	19.0	100	0
17	28.8	9.6	100	0

**Table 3-3:** Hydrologic response metrics estimated for each wetland in association with each rainfall-runoff event. ARX = Antecedent rainfall in the “X” days prior to the start of the rainfall event under consideration.

Event response metric	Description	Code name
Initial water level	Wetland fullness (for open water wetland) or ditch water table height (for drained wetland) at the beginning of an event	InitialWL
Maximum water level	Peak wetland fullness (for open water wetland) or ditch water table height (for drained wetland) during an event	MaxWL
Time lag to maximum water level	Time elapsed between the beginning of the rainfall event and wetland fullness (for open water wetland) or ditch water table height (for drained wetland) during an event	LagToMaxWL
Difference between initial and max water level	Difference between peak and initial wetland fullness (for open water wetland) or ditch water table height (for drained wetland) during an event	DiffInitialMaxWL
Rate of water level rise	Rate of wetland fullness (for open water wetland) or ditch water table height (for drained wetland) rise during an event	WLRiseRate
Final water level	Wetland fullness (for open water wetland) or ditch water table height (for drained wetland) at the end of an event	FinalWL
Difference between max and final wetland fullness	Difference between peak and final wetland fullness (for open water wetland) or ditch water table height (for drained wetland) during an event	DiffMaxFinalWL
Water level recession duration	Time elapsed between peak wetland fullness (for open water wetland) or ditch water table height (for drained wetland) and the end of creek response during an event	RecessDura
Water level recession rate	Rate of wetland fullness (for open water wetland) or ditch water table height (for drained wetland) recession during an event	WLRecessRate
Water level percentage change	Percentage of wetland fullness (for open water wetland) or ditch water table height (for drained wetland) change compared to the initial value during an event	WLPercChange
Ratio of maximum water level rise to total event rainfall	Ratio between maximum wetland fullness (for open water wetland) or ditch water table height (for drained wetland) rise and total event rainfall	MaxRise/TR
Maximum water level rise to total event rainfall+AR1 ratio	Ratio between maximum wetland fullness (for open water wetland) or ditch water table height (for drained wetland) rise and the sum of total event rainfall and AR1	MaxRise/(TR+AR1)
Maximum water level rise to total event rainfall+AR7 ratio	Ratio between maximum wetland fullness (for open water wetland) or ditch water table height (for drained wetland) rise and the sum of total event rainfall and AR7	MaxRise/(TR+AR7)



**Figure 3-2:** Examples of creek response (i.e., stream hydrograph) and wetland response (surface and subsurface water level) during individual rainfall-runoff events.

### 3.2.4 Statistical Analyses

Since open water PWs were characterized using surface water levels and drained PWs were characterized using subsurface water levels, each research objective was addressed separately for open water and drained wetlands. To identify the most important metrics needed to characterize the spatial variability of wetland hydrologic response (i.e., research objective 1), principal component analysis (PCA) was performed. PCA, a statistical data reduction technique, identifies

the combinations of original variables (i.e., the principal components) that explain the maximum variation in a dataset (Jolliffe, 2011). In the present study, PCA was performed for each event, considering all the hydrologic metrics calculated across all the open water or drained PWs for that event. The hydrologic response metrics that contributed the most to the first and second principal components, i.e., had a loading score of 0.50 or higher for one of the first two principal components, were deemed to be the metrics that explain the maximum variation across the wetlands. The metrics that consistently (i.e., across multiple events) had high loading scores on the first two principal components were retained for further analysis. To address research objective 2 (i.e., to investigate the temporal variability of individual wetland hydrologic response), boxplots were created for the metrics that were retained after PCA. To address research objective 3 (i.e., to investigate the spatial and temporal variability of wetland-stream interaction), an analysis of hysteretic behaviors was done. Specifically, for each of the events and each studied PW, a scatter plot of wetland water level (x-axis) versus creek water level (y-axis) was built to examine wetland-stream hysteresis dynamics. Hysteresis curves were classified into five different types, namely: linear (i.e., no hysteresis), clockwise, counter-clockwise, eight-shaped and unclear. The percent time that each hysteresis type was observed for each wetland was then computed. To address research objective 4 (i.e., to investigate the temporal persistence of spatial controls on individual wetland hydrologic response characteristics), Spearman's rank correlation coefficients were calculated between wetland spatial characteristics (Table 3-1) and selected wetland hydrologic response metrics for individual rainfall events. Spearman's rank correlation coefficients were evaluated at the 95% significance level. Contrary to the commonly used Pearson correlation coefficient, the Spearman's rank correlation coefficient does not assume a normal distribution of the data and is used to assess both linear and nonlinear relations between variables (Sokal & Rohlf,

2012). The MATLAB Statistics and Machine Learning Toolbox, Release 2017b, was used for performing PCA and calculating the Spearman's rank correlation coefficients.

### **3.3 Results**

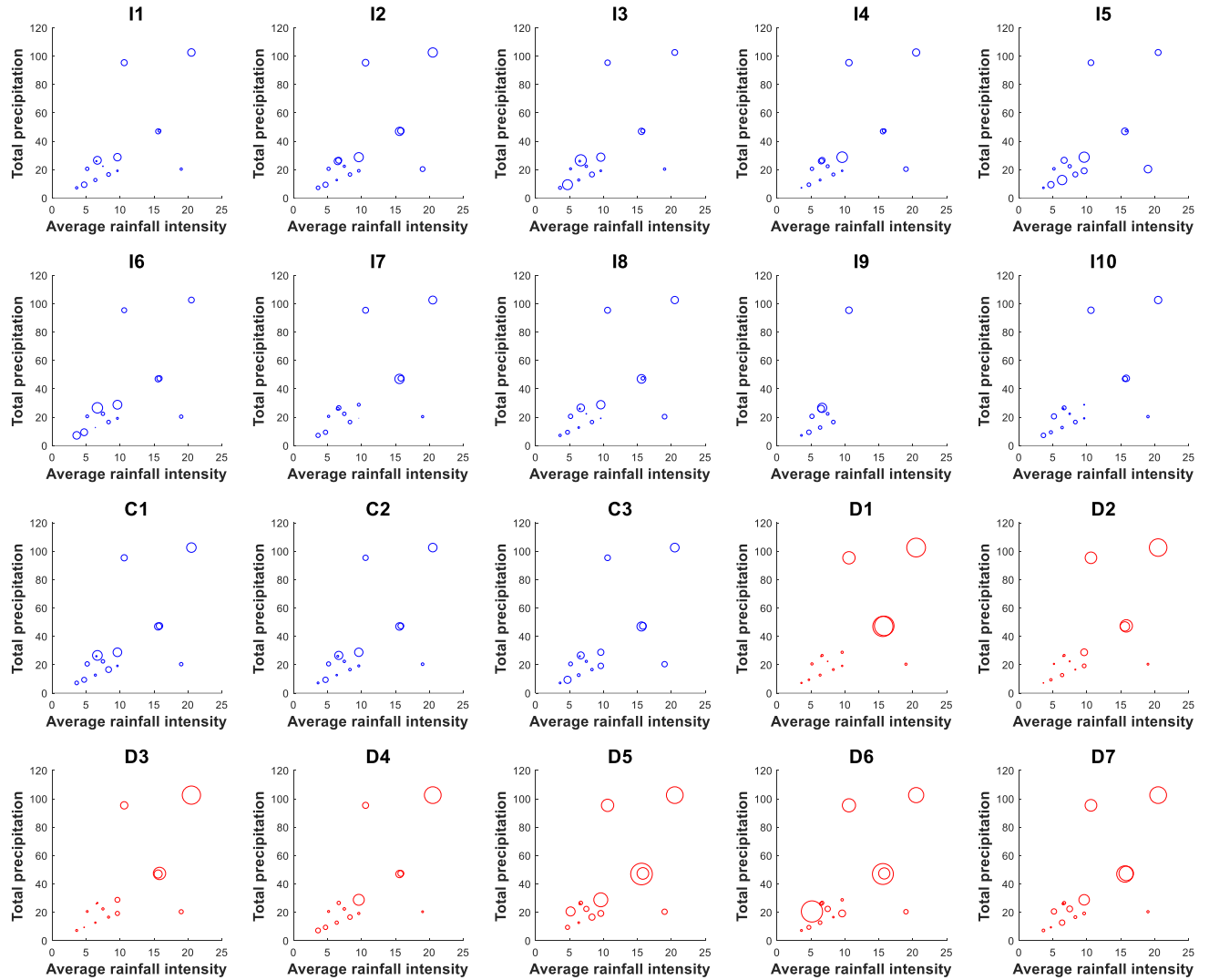
#### **3.3.1 Event Characteristics and Metrics of Wetland Hydrologic Response**

Total rainfall and average rainfall intensity were strongly positively correlated across all rainfall-runoff events, as shown graphically in Figure 3-3. The influence of those two rainfall event characteristics on hydrologic response metrics, however, appeared to differ slightly between open water and drained PWs. For example, when considering the water level rise rate during individual rainfall-runoff events, average rainfall intensity had more influence on the drained PWs compared to the open water PWs. Figure 3-3 notably shows that average rainfall intensities of more than 10 mm/day led to noticeable increases in the water level rise rate of drained PWs, whereas no significant change was observed above that same rainfall intensity for open water PWs (Figure 3-3). Simple descriptive statistics show that some hydrologic response metrics have greater spatial variability (i.e., vary more among the studied PWs) relative to others (Table 3-4). For example, metrics like WLRiseRate, WLRecessRate and WLPercChange had noticeably greater standard deviation values for both open water and drained wetlands, compared to other metrics. A few exceptions aside, drained PWs always had higher standard deviation and coefficient of variation values for all the hydrologic metrics, compared to open water PWs (Table 3-4). PCA showed that not all the calculated hydrologic response metrics were equally effective at capturing variation in wetland hydrologic behavior during individual rainfall-runoff events. Out of the 13 calculated hydrologic response metrics, only 8 had loading scores exceeding 0.50 on either the first or the second principal component for at least one rainfall-runoff event (Figure 3-4). On one hand, response metrics such as WLRiseRate, WLRecessRate and WLPercChange were able to capture

a large portion of the variation in hydrologic response for both open water and drained PWs for almost all the studied rainfall-runoff events. On the other hand, metrics such as LagToMaxWL, RecessDura, and MaxRise/TR were only able to capture a good portion of the variation in the hydrologic response of drained PWs (Figure 3-4). Hence, for further analyses, only selected metrics were retained, as they had loading scores exceeding 0.50 on either the first or the second principal component for at least one rainfall-runoff event, namely: LagToMaxWL, WLRiseRate, RecessDura, WLRecessRate, WLPercChange, MaxRise/TR, MaxRise/(TR + AR1), and MaxRise/(TR + AR7).

**Table 3-4:** Descriptive statistics for the hydrologic response metrics estimated across all instrumented wetlands and all rainfall events. Bold text refers to drained wetlands.

Response metric code name (unit)	Mean	Median	Minimum	Maximum	Standard deviation	Coefficient of Variation
Open water or <b>Drained</b> wetland						
InitialWL (-) or ( <b>m</b> )	0.51 <b>0.36</b>	0.50 <b>0.28</b>	0.03 <b>0.00</b>	1.12 <b>1.00</b>	0.30 <b>0.29</b>	58.41 <b>79.61</b>
MaxWL (-) or ( <b>m</b> )	0.58 <b>0.54</b>	0.56 <b>0.42</b>	0.06 <b>0.02</b>	1.47 <b>1.94</b>	0.32 <b>0.45</b>	55.20 <b>83.23</b>
LagToMaxWL (day)	2.78 <b>3.10</b>	2.00 <b>3.00</b>	1.00 <b>1.00</b>	9.00 <b>7.00</b>	2.05 <b>1.86</b>	73.68 <b>60.07</b>
DiffInitialMaxWL (-) or ( <b>m</b> )	0.07 <b>0.18</b>	0.04 <b>0.03</b>	0.00 <b>0.00</b>	0.41 <b>1.03</b>	0.08 <b>0.27</b>	116.37 <b>151.77</b>
WLRiseRate (mm/day)	24.04 <b>62.25</b>	16.17 <b>13.47</b>	0.01 <b>0.27</b>	104.73 <b>393.29</b>	22.94 <b>98.42</b>	95.42 <b>158.11</b>
FinalWL (-) or ( <b>m</b> )	0.53 <b>0.40</b>	0.53 <b>0.31</b>	0.04 <b>0.02</b>	1.22 <b>1.07</b>	0.30 <b>0.30</b>	57.21 <b>74.49</b>
DiffMaxFinalWL (-) or ( <b>m</b> )	0.05 <b>0.14</b>	0.03 <b>0.03</b>	0.00 <b>0.00</b>	0.32 <b>1.14</b>	0.05 <b>0.26</b>	107.41 <b>181.15</b>
RecessDura (day)	5.31 <b>4.76</b>	4.00 <b>4.00</b>	0.00 <b>0.00</b>	13.00 <b>13.00</b>	3.09 <b>3.70</b>	58.23 <b>77.85</b>
WLRecessRate (mm/day)	8.42 <b>18.07</b>	5.95 <b>6.80</b>	0.00 <b>0.00</b>	59.71 <b>103.51</b>	9.21 <b>23.78</b>	109.40 <b>131.60</b>
WLPercChange (-)	7.66 <b>20.08</b>	0.12 <b>2.20</b>	-29.08 <b>-54.34</b>	331.34 <b>307.84</b>	33.07 <b>55.25</b>	431.48 <b>275.16</b>
MaxRise/TR (-)	1.66 <b>3.61</b>	1.46 <b>1.91</b>	0.00 <b>0.04</b>	8.84 <b>20.27</b>	1.33 <b>4.11</b>	80.15 <b>113.65</b>
MaxRise/(TR+AR1) (-)	1.66 <b>3.61</b>	1.46 <b>1.91</b>	0.00 <b>0.04</b>	8.84 <b>20.27</b>	1.33 <b>4.10</b>	79.96 <b>113.69</b>
MaxRise/(TR+AR7) (-)	1.30 <b>2.81</b>	0.98 <b>1.38</b>	0.00 <b>0.01</b>	8.48 <b>18.72</b>	1.18 <b>3.41</b>	91.25 <b>121.41</b>



**Figure 3-3:** Scatter plots of average event rainfall intensity (mm/day) versus total event rainfall (mm). Circles represent individual events. The radius of the circles is proportional to the wetland fullness or ditch water table rise rate for the respective event; it provides a qualitative way of assessing the correlation – or lack thereof – between rainfall intensity, total rainfall and water level (surface or subsurface) rise rate.

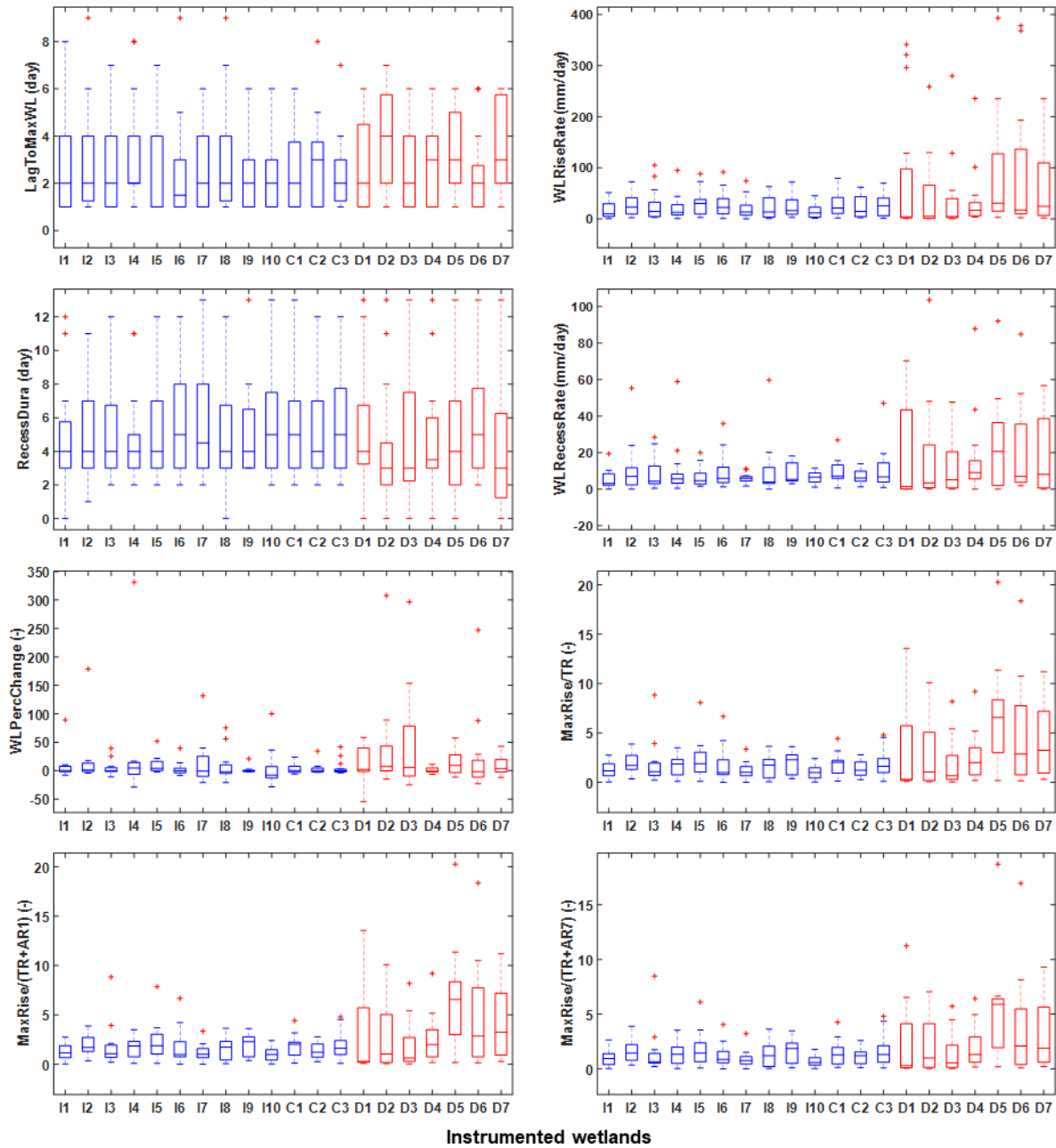
**Figure 3-4:** Wetland hydrologic response metrics with PCA loadings of more than 0.50 for either the first or the second principal component. Blue circles represent open water wetlands and red circles represent drained wetlands.

Event#	1	2	3	4	7	8	9	10	11	12	13	14	15	16	17	Total occurrences	
InitialWL																0/15	0/15
MaxWL																0/15	0/15
LagToMaxWL							●	●			●			●		0/15	3/15
DiffInitialMaxWL																0/15	0/15
WLRiseRate	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	15/15	15/15
FinalWL																0/15	0/15
DiffMaxFinalWL																0/15	0/15
RecessDura					●	●	●	●	●		●			●		0/15	7/15
WLRecessRate		●	●	●	● ●	●	●	●	●	●	●		●	● ●	●	2/15	13/15
WLPercChange	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	● ●	●	●	13/15	15/15
MaxRise/TR								●			●					0/15	2/15
MaxRise/(TR+AR1)								●			●					0/15	2/15
MaxRise/(TR+AR7)								●								0/15	1/15

### 3.3.2 Temporal Variability of Individual Wetland Hydrologic Response

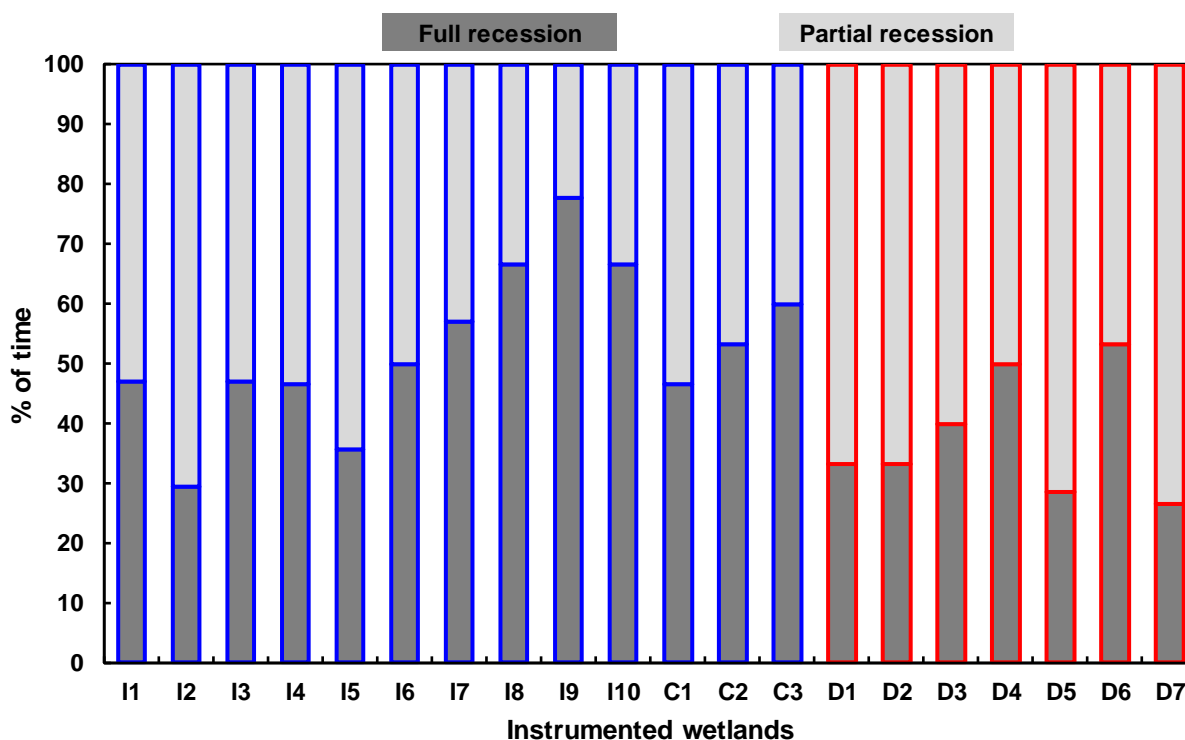
The metrics retained after synthesizing the PCA results were used to study the temporal variability of individual hydrologic responses across the studied PWs (Figures 3-4 and 3-5). When focusing on open water PWs alone, most of the metrics did not show considerable temporal variation (i.e., variation from one event to another, illustrated by the small height of the blue boxes on the boxplot figures) except for LagToMaxWL and RecessDura (Figure 3-5). However, focusing on the absolute values of the metrics revealed that LagToMaxWL had noticeably lower values (i.e., mostly between 1 and 4 days) compared to RecessDura (i.e., mostly between 3 and 8 days) for the open water PWs. Also, WLRiseRate (i.e., mostly less than 50 mm/day) had higher values compared to WLRecessRate (i.e., mostly less than 20 mm/day) (Figure 3-5). For individual open water PWs, WLPercChange had mostly positive values. There were no major differences between the MaxRise/TR and MaxRise/(TR+AR7) metrics in terms of absolute values or temporal variability (Figure 3-5).

The event response dynamics observed for drained PWs were different than those observed for open water PWs. Indeed, for most of the response metrics, drained PWs showed noticeably higher temporal variability compared to the open water PWs (i.e., red boxes were taller than blue boxes on the boxplot figures), suggesting that subsurface water storage dynamics were more variable, over the short-term (event-to-event scale), than wetland surface water storage dynamics. Except for the RecessDura metric, drained PWs usually had higher values for all the metrics compared to open water PWs (Figure 3-5). With very few exceptions (e.g., D2 and D7), in general, individual drained PW showed lower LagToMaxWL values compared to the RecessDura values. Similarly, WLRiseRate (i.e., mostly less than 150 mm/day) had considerably higher values compared to WLRecessRate (i.e., mostly less than 50 mm/day) (Figure 3-5). Drained PWs had



**Figure 3-5:** Spatio-temporal variability of selected wetland hydrologic response metrics. Blue-colored boxes represent open water wetlands (i.e., I1 to C3) while red-colored boxes represent drained wetlands (i.e., D1 to D7). Each box has lines at the lower quartile, median, and upper quartile values, while the whiskers beyond the box show the extent of the rest of the data. Outliers are shown as '+'. An outlier is a value that is more than 1.5 times the interquartile range away from the top or the bottom of the box.

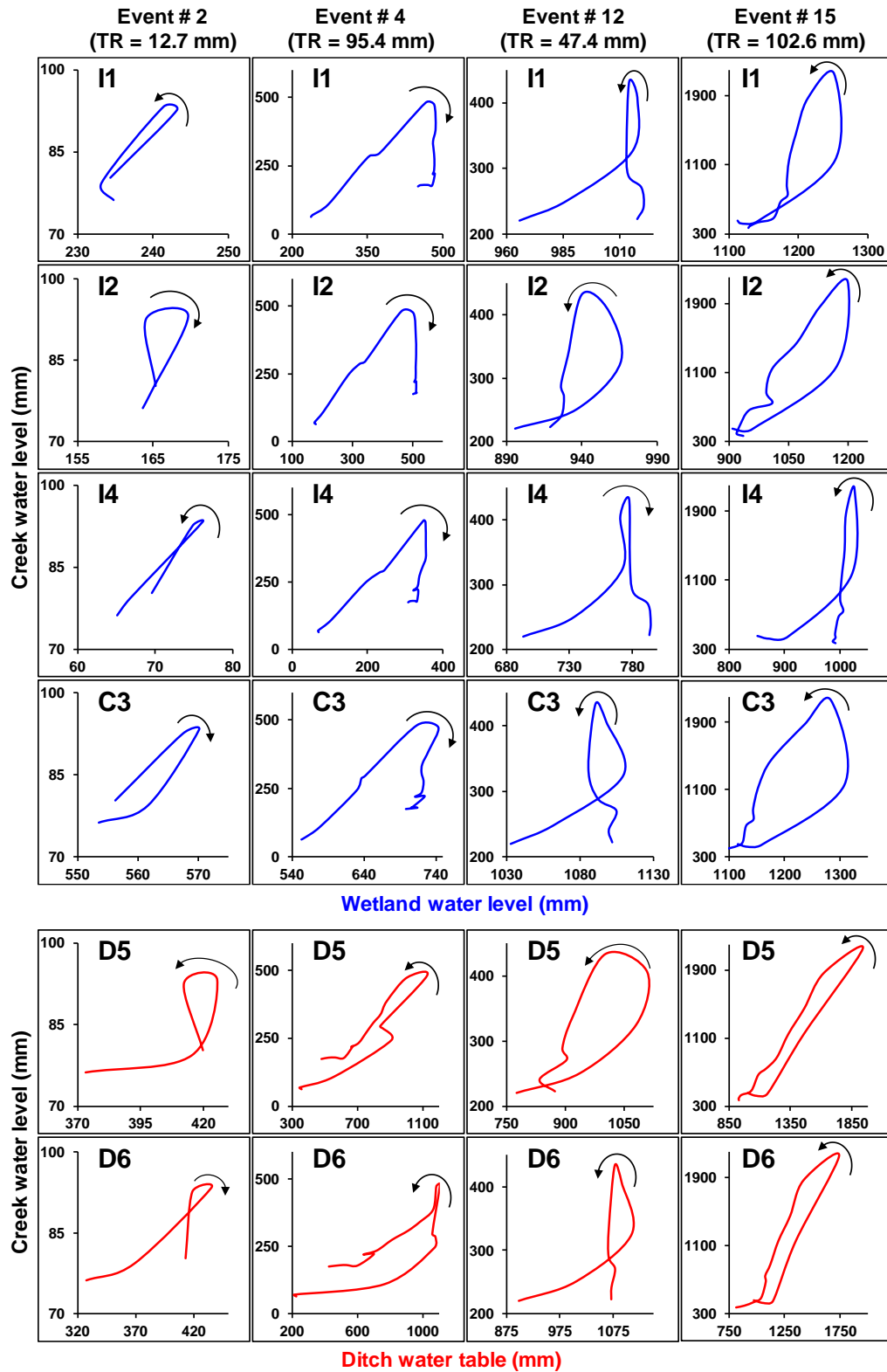
mostly positive WLPercChange. For individual drained PWs, there were no noticeable differences between metrics such as MaxRise/TR, MaxRise/(TR+AR1) and MaxRise/(TR+AR7); however, site D5 always had noticeably higher median values compared to the other drained PWs for these metrics. All drained PWs showed partial recession for most of the studied rainfall-runoff events, meaning that their final water table level (at the end of a rainfall-runoff event) was higher than the initial water table level (recorded before the event started). Conversely, more than half of the open water PWs showed full recession for most of the rainfall-runoff events (Figure 3-6).



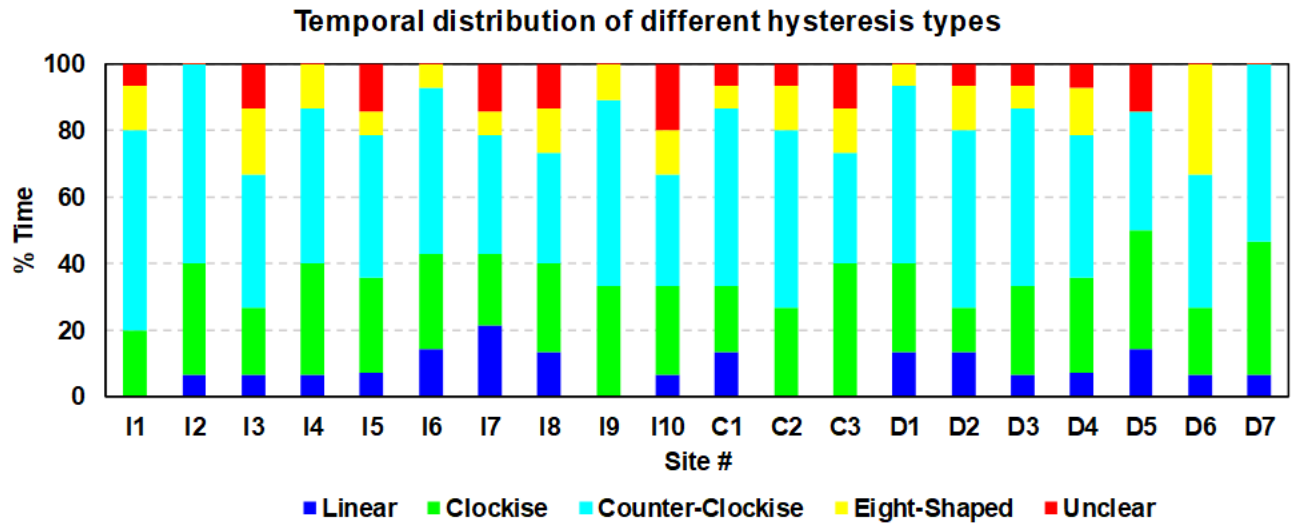
**Figure 3-6:** Spatio-temporal variability of recession dynamics observed across instrumented wetlands. “Partial recession” means that the final wetland water level (or ditch water table level) observed at the end of an event was noticeably higher than the initial value observed at the beginning of an event. “Full recession” rather means that the final wetland water level (or ditch water table level) was similar or lower than the initial value observed at the beginning of an event.

### **3.3.3 Spatial and Temporal Variability of Wetland-Stream Interaction**

There were considerable spatial and temporal variations in wetland-stream interaction patterns (i.e., hysteresis loop shapes) across the studied PWs (Figures 3-7 and 3-8). For example, during rainfall-runoff event #4, most of the open water PWs were associated with a wetland-stream clockwise hysteretic pattern, contrary to most drained PWs that were rather associated with a counter-clockwise pattern (Figure 3-7). Some rainfall-runoff events showed a lot of spatial variability, with individual open water wetlands having different hysteretic patterns (e.g., event # 12). Conversely, a single dominant hysteretic pattern could emerge across all studied PWs for some events, as was the case with the counter-clockwise pattern in event # 15 (Figure 3-7). The spatial variation (i.e., wetland-to-wetland variation) in hysteresis types was more pronounced for events that occurred following dry conditions and/or when precipitation duration was short, as shown in Figure 3-7 for event #2. In contrast, more spatial homogeneity in hysteresis types was observed for longer-duration precipitation events and/or with wet antecedent conditions, as evidenced in Figure 3-7 for event #15 (Table 3-2 and Figure 3-7). More importantly, when considering individual wetlands one at a time, hysteresis patterns were not temporally consistent (i.e., the same wetland was not consistently associated with the same hysteretic pattern for different rainfall-runoff events) (Figures 3-7 and 3-8). During the study period, although most wetlands switched between the five hysteresis types, the counter-clockwise type was the most common, regardless of wetland alteration status (Figure 3-8). Geographic proximity was not a good predictor of the spatial homogeneity or heterogeneity of hysteresis types: very different hysteresis patterns were sometimes obtained for wetlands located close to each other (e.g., I1 and I2, C1 and C2, or D5 and D6) (Figures 3-1 and 3-8).



**Figure 3-7:** Examples of hysteresis patterns observed for selected wetlands and events. TR = Total event rainfall.



**Figure 3-8:** Temporal distribution of different hysteresis types observed for each instrumented wetland across all 15 rainfall-runoff events under consideration. I = Intact wetland, C = Consolidated wetland, D = Drained wetland. Hysteresis types include: L = Linear (i.e., no hysteresis), CW = Clockwise, CCW = Counter-Clockwise, ES = Eight-Shaped, and U = Unclear.

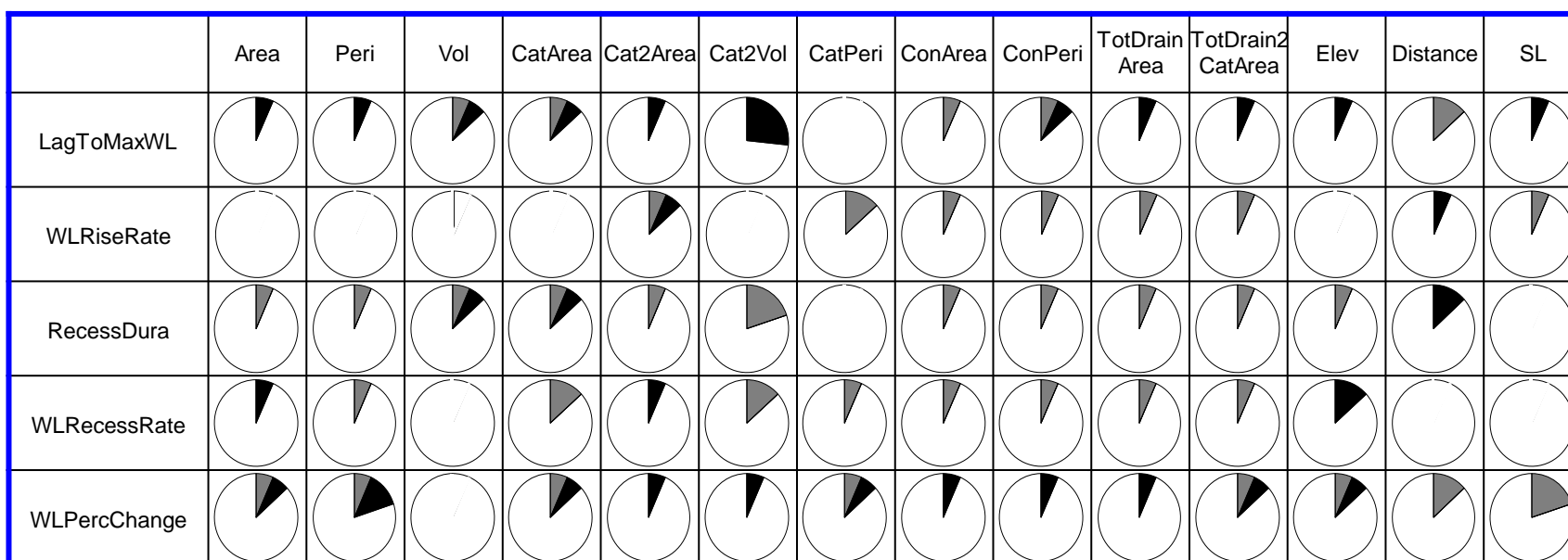
### 3.3.4 Temporal Persistence of Spatial Controls on Individual Wetland Hydrologic Response

In the context of this study, the phrase “temporal persistence” is used to refer to situations where correlations between wetland hydrologic response metrics and spatial characteristics are consistently detected across all monitored rainfall-runoff events. Such “perfect” temporal persistence would manifest through pie charts in Figures 3-9 and 3-10 that are entirely gray or entirely black with no white patches; however, as can be seen from those two figures, this situation was never observed. Most notably, when considering open water PWs, there was not a single event for which a statistically significant correlation could be detected between pairs of variables such as: LagToMaxWL versus CatPeri; WLRiseRate versus Area, Peri, Vol, CatArea, Cat2Vol or Elev;

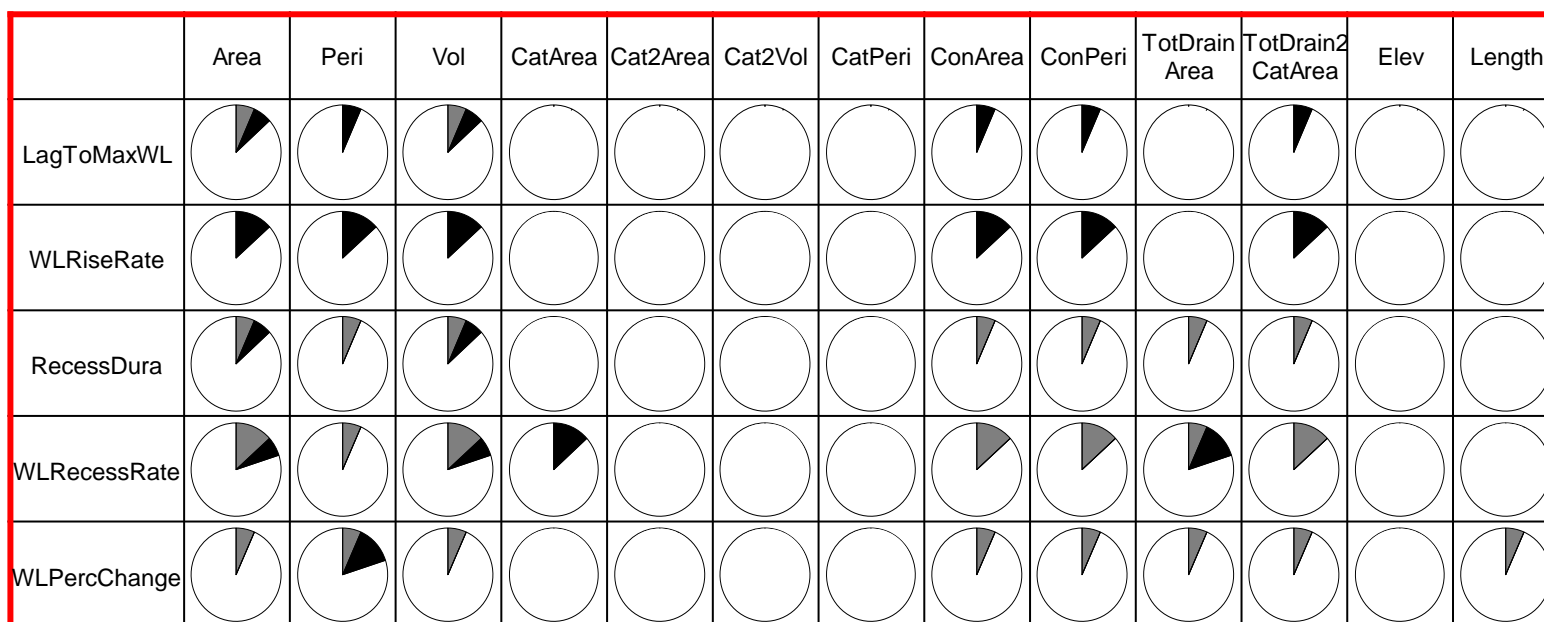
RecessDura versus CatPeri or SL; WLRecessRate versus Vol, Distance, or SL; and WLPerChange versus Vol (Figure 3-9). For pairs of variables other than the ones just listed, large white patches in the pie charts show that for the vast majority of events, no statistically significant correlation was detected between wetland hydrologic response metrics and spatial characteristics (Figure 3-9). Worthy of note are some spatial characteristics (i.e., Cat2Area, ConArea, ConPeri, TotDrainArea and TotDrain2CatArea) which were significantly correlated with key response metrics for a minority of events (Figure 3-9). For instance, LagToMaxWL was negatively correlated with Cat2Vol and positively correlated with Distance for about 30% and 20% of the monitored events, respectively. Conversely, RecessDura was positively correlated with Cat2Vol and negatively correlated with Distance for about 30% and 20% of the monitored events, respectively (Figure 3-9). WLRiseRate was positively correlated with CatPeri for about 20% of the monitored events. WLRecessRate was positively correlated with CatArea and Cat2Vol and negatively correlated with Elev for about 20% of the monitored events. WLPerChange was positively correlated with Distance and SL for about 20% and 25% of the monitored events, respectively (Figure 3-9). Some hydrologic response metrics did show both positive and negative correlations with some of the spatial characteristics (e.g., correlation between LagToMaxWL, RecessDura, WLPerChange and CatArea) (Figure 3-9). Thus, while none of the hydrologic response metrics could be described as showing temporally persistent correlation with any of the wetland spatial characteristics, spatial characteristics such as CatArea, Cat2Vol, CatPeri, Distance, and SL showed somewhat temporally persistent correlations for multiple hydrologic response metrics (Figure 3-9).

As was the case for open water PWs, temporally persistent spatial influences on hydrologic response were not detected for drained PWs (Figure 3-10). For instance, Cat2Area, Cat2Vol,

CatPeri, and Elev were not correlated with any of the response metrics (Figure 3-10). For some events, CatArea was correlated with WLRecessRate and Length was correlated with WLPercChange. However, for some monitored events in drained PWs, spatial characteristics such as Area, Peri, Vol, ConArea, ConPeri, and TotDrain2CatArea were significantly correlated with key response metrics. For example, WLRiseRate was negatively correlated with Area, Peri, Vol, ConArea, ConPeri and TotDrain2CatArea for about 20% of the monitored events. WLRecessRate was positively correlated with Area, Vol, ConArea, ConPeri, and TotDrain2CatArea and negatively correlated with CatArea and TotDrainArea for about 20% of the monitored events. WLPercChange was negatively correlated with Peri for about 20% of the monitored events (Figure 3-10).



**Figure 3-9:** Temporal persistence of spatial controls on selected wetland hydrologic response metrics for open water wetlands across all events. Each colored segment of the pie chart represents a percentage of monitored rainfall events for which the correlation coefficient between the column variable (wetland spatial characteristic) and the row variable (wetland response metric) was statistically significant (i.e.,  $p\text{-value} < 0.05$ ). Light gray: positive correlation; black: negative correlation. See Tables 3-1 and 3-3 for the descriptions of wetland spatial characteristics and wetland response metrics, respectively.



**Figure 3-10:** Temporal persistence of spatial controls on selected wetland hydrologic response metrics for drained wetlands across all events. Each colored segment of the pie chart represents a percentage of monitored rainfall events for which the correlation coefficient between the column variable (wetland spatial characteristic) and the row variable (wetland response metric) was statistically significant (i.e.,  $p\text{-value} < 0.05$ ). Light gray: positive correlation; black: negative correlation. See Tables 3-1 and 3-3 for the description of wetland spatial characteristics and wetland response metrics, respectively.

### **3.4 Discussion**

#### **3.4.1 Effective Metrics of Wetland Hydrologic Response**

One premise for the current study was that since stream hydrograph metrics provide useful information on watershed hydrologic function and help assess topographic and climatic controls on watershed hydrologic responses (Carey & Woo, 2001; Te Chow et al., 1988), a similar approach could be applied to individual wetlands to better understand wetland hydrologic function. One hypothesis for the current study was also that metrics that capture most of the variability that exists among wetlands are the most suitable – or effective – for characterizing and comparing individual wetland response. The high standard deviation values, in space (i.e., among wetlands), estimated for some metrics (e.g., WLRiseRate, WLRecessRate and WLPercChange, see Table 3-4) suggest that some hydrologic metrics may, indeed, be better suited than others for contrasting the hydrologic response of different PWs during rainfall-runoff events. The descriptive statistics reported in Table 3-4 and Figure 3-5 also indicate that wetland alteration status (i.e., drained versus open water wetlands) has a noticeable impact on wetland hydrologic responses: this was evidenced by the consistently higher standard deviation values (reflecting higher spatial and temporal variability) for response metrics associated with drained PWs, compared to those associated with open water PWs (Table 3-4 and Figure 3-5). Average event rainfall intensity appeared to have more influence on the WLRiseRate for drained PWs, compared to open water PWs (Figure 3-3), which further suggests a critical role of wetland alteration in modifying the degree to which a climatic control influences wetland hydrologic response.

PCA results were also useful in demonstrating that not all response metrics derived from hydrograph analysis are equally important for characterizing the variability of wetland hydrologic response. The stream-focused literature is rich in descriptions of how response metrics derived

from rainfall-runoff hydrographs can reflect either the magnitude or the timing of the hydrologic response (Beven, 2012; Hannah et al., 2000). Response magnitude shows how effective a drainage area can be in converting rainfall into runoff (Dingman, 2015), while response timing provides indirect information about the type of flow (i.e., surface vs subsurface flow) that dominates during a rainfall-runoff event (Tallaksen, 1995). In this wetland-focused study, metrics related to the magnitude of hydrologic response (e.g., WLRiseRate, WLRecessRate and WLPercChange) were found to be important to assess and compare wetland hydrologic response regardless of wetland alteration status. However, metrics related to the timing of hydrologic response (e.g., LagToMaxWL, RecessDura, and MaxRise/TR) appeared to be more important when assessing the variability in the hydrologic responses of drained PWs, and less so when considering open water PWs (Figure 3-4). The fact that metrics related to the magnitude of hydrologic response (i.e., ability to produce runoff and store water) were more commonly associated with high PCA loading scores (i.e., they captured a large portion of the variation in the hydrologic responses) may suggest that runoff generation processes from the land surrounding the wetlands play an important role in explaining the short-term, event-scale hydrologic behaviour of PWs. In the end, through the PCA, multiple metrics were needed to adequately capture the majority of the variation in hydrologic response among different wetlands, highlighting the importance of capturing the multiple facets of wetland hydrologic function, even at short timescales. Similar conclusions about the need for multiple metrics to capture the multiple facets of hydrologic response have been put forward in previous studies focusing on stream and watershed dynamics (e.g., Ali et al., 2010; Carey & Woo, 2001; Post & Jakeman, 1996; Ross et al., 2019).

### **3.4.2 Temporal Variability in Hydrologic Response**

For open water PWs, the lack of the temporal variability observed for the metrics related to the magnitude of hydrologic response (e.g., WLRiseRate, WLRecessRate and WLPercChange) suggests that flow generation mechanisms – leading to wetland water level change – were temporally consistent. The higher temporal variability observed in response magnitude for drained PWs, compared to that observed for open water PWs, strongly suggests that climatic variables exert a stronger influence on subsurface flow dynamics (i.e., beneath the ditches associated with drained PWs) compared to surface flow dynamics in open water PWs (Figure 3-5). All the studied PWs demonstrated considerable temporal variation for the metrics related to the timing of hydrologic response (e.g., LagToMaxWL, RecessDura). In the stream-focused and watershed-focused literature, response timing is usually deemed to be influenced by rainfall pattern and antecedent moisture conditions (Carey & Woo, 2001; Dingman, 2015). Along the same lines, in the present wetland-focused study, temporal variations in response timing metrics could indicate that climatic conditions had a significant influence on the short-term wetland storage dynamics, regardless of the alteration status of the PWs. It should be noted that for drained PWs, hydrologic response metrics were derived from data collected at the downstream extremity of the drainage ditches (i.e., close to where they connect to the Broughton's Creek, see Figure 3-1), as opposed to upstream where the data would have more directly reflected the conditions prevailing in the former basin of the drained PWs. In light of that, it may not be appropriate to compare the open water PW metrics and the drained PW metrics: the former are influenced by wetland catchment dynamics and ditch dynamics (i.e., the former encompass travel distances from the wetland catchment to the wetland basin, as well as travel distance along the ditch length between the wetland basin and the creek) (Haque et al., 2018). Higher values of WLRiseRate and WLRecessRate for drained PWs,

compared to open water PWs, could be an indication of the flashier nature of drained PWs and their associated subsurface flow dynamics (Figure 3-5). For all monitored PWs, higher LagToMaxWL and WLRiseRate values compared to the RecessDura and WLRecessRate values, respectively, suggest that the release of water from these wetlands was slower than the inflow of water towards them. These observations not only support the idea that open water PWs can effectively lower downstream flood peak by providing temporary storage of surface runoff during rainfall-runoff events, but also that open water PWs can contribute to stream baseflow during dry conditions (EPA, 2015; Hayashi et al., 2016; Wang et al., 2010; Yang et al., 2010).

Almost all drained PWs showed partial recession for most of the studied events, hinting at a prolonged contribution of subsurface flow after the rainfall-runoff event had ended, through the complex ditch drainage network (Figures 3-1 and 3-6). One alternative hypothesis might be that the partial recession of the drained wetlands is due to the proximity to the hyporheic zone of the adjacent stream, resulting in higher water tables after a rainfall-runoff event. About half of the open water PWs showed partial recession while the other half showed full recession for more than 50% of the studied events. An open water PW that showed frequent full recessions during the observed rainfall-runoff events could indicate that water levels rose quickly and fell quickly because they were only fed by local or proximal surface and subsurface water. Therefore, open water PWs may be part of the isolated drainage system, as described by other studies (Cohen et al., 2016; EPA, 2015; Haque et al., 2018; Leibowitz, 2003, 2015) and could also be part of a system that is connected through subsurface flow as outlined by Tiner (2003). Conversely, an open water PW with frequent partial recession dynamics (Figure 3-6) could indicate that the wetland spilled and merged with another (McCauley et al., 2015; Shaw et al., 2012), which is why its water level remains high after the event has ended. However, spilling does not necessarily mean that a wetland

is connected to a nearby stream, but rather that it is connected to another wetland that may or may not be connected to a stream (Leibowitz et al., 2016; Shaw et al., 2012). Alternatively, partial recession dynamics in open water PWs could indicate longer memory effects in these wetlands. Memory effects were also observed in other studies that looked at storage dynamics in PWs, and they can be explained by slower drainage due to the time delay (i.e., hysteresis) between rainfall-runoff and storage changes (Haque et al., 2018; Shook et al., 2015; Shook and Pomeroy, 2011).

### **3.4.3 Characterizing Wetland-Stream Interaction**

The presence of hysteresis (Figures 3-7 and 3-8) between wetland water level (or ditch water table) and creek water level indicates a time delay between wetland storage change and stream response and, hence, provides indirect information about the travel time of water from wetland to stream (Shook et al., 2015; Shook & Pomeroy, 2011). The predominance of the counter-clockwise hysteresis type (Figures 3-7 and 3-8) suggests that wetland water levels (i.e., surface water dynamics) or ditch water tables (i.e., subsurface water dynamics) responded before the creek water level during a rainfall-runoff event. As indicated by other studies, this scenario is potentially indicative of wetland-stream connectivity established via overland flow or shallow sub-surface flow during rainfall-runoff events (Haque et al., 2018; Shook et al., 2015; Shook & Pomeroy, 2011). However, while the counter-clockwise hysteresis type was predominant, it was not the only one observed. The temporal and spatial variations in wetland-stream hysteresis types (Figure 3-8) observed in this study suggest that a complex combination of factors might be influencing short-term wetland-stream interaction. Such factors include spatial wetland characteristics but also antecedent wetness conditions and precipitation duration (Hewlett & Hibbert 1967; Ross et al., 2019). For one of the major rainfall-runoff events monitored in this study (i.e., event # 4, Table 3-2, and Figure 3-7), all open water PWs showed a clockwise hysteresis pattern while drained PWs

showed a counter-clockwise pattern. These results suggest that creek water level responded before the surface water level of open water wetlands but after water tables located underneath the drainage ditches. Based on these observations, it is therefore unlikely that a connection between open water PWs and the creek was established via fill-spill or another overland flow generation mechanism. However, it is likely that ditches were connected to the creek through shallow subsurface flow (and potentially overland flow if the water table intersected the ground surface). For another major event that occurred in high antecedent moisture conditions (i.e., event # 15, Table 3-2, and Figure 3-7), all the studied PWs showed a counter-clockwise hysteresis pattern regardless of their alteration status, suggesting that they were responding before the creek and may have been connected to it through fill-spill or shallow subsurface flow. The fact that all the studied PWs could switch between opposite hysteresis dynamics (i.e., from clockwise to counter-clockwise from one rainfall event to the next) (Figure 3-7) indicates that the overall antecedent wetness (or storage) conditions and storage memory effects can exert a strong control on wetland-stream interactions, compared to the local landscape characteristics. The presence of more spatially homogeneous hysteresis dynamics across wetlands during wet periods (Figure 3-7) could reflect the fact that wetter conditions promote greater hydrologic connectivity, thus reducing the influence of individual wetland characteristics on wetland-stream interaction. Hence, in some situations, the influence of individual wetland characteristics on wetland-stream interaction may have been overcome by overall landscape conditions (e.g., connectivity). In contrast, hydrologic connectivity is less widespread following dry antecedent conditions and is highly dependent on the ability of individual wetlands to generate flow, an ability that is notably determined by the balance between wetland catchment area and wetland storage volume (Shaw et al., 2012).

### 3.4.4 Predictability of Spatial Controls on Wetland Hydrologic Response

For open water PWs, a negative correlation between Cat2Vol and LagToMaxWL and a positive correlation between WLRiseRate and CatPeri (Figure 3-9) suggest that wetlands with larger catchment areas and with smaller water storage volume tend to respond quickly during a rainfall-runoff event. Furthermore, Cat2Vol was positively correlated with RecessDura and WLRecessRate was positively correlated with both Cat2Vol and CatArea (Figure 3-9), suggesting that open water wetlands with proportionally larger catchment areas have slower and partial recession, with high water levels persisting long after major rainfall-runoff events. This could also indicate that these PWs have a longer storage memory as described in other studies (Shook et al., 2015; Shook & Pomeroy, 2011). For open water PWs, Distance was positively correlated with LagToMaxWL and negatively correlated with RecessDura (Figure 3-9), thereby hinting that the geographic location of a wetland plays an important role in determining wetland-stream interaction. The fact that WLPercChange was positively correlated with Distance and SL (Figure 3-9) also hints that wetlands located far from the stream and with higher spill levels are less likely to interact with the stream through the fill-spill mechanism. While these observations support conclusions made by others regarding wetland-to-wetland and wetland-to-stream connectivity via fill-spill (McCauley et al., 2015; Shaw et al., 2012), they also highlight the strong dependence of wetland-stream or wetland-wetland connectivity on distance, especially when there is no visible surface connection (i.e., drainage network) between wetlands, or between a wetland and a stream.

For drained PWs, magnitude metrics such as WLRiseRate were negatively correlated with Area, Peri, Vol, ConArea, ConPeri and TotDrain2CatArea (Figure 3-10). However, for similar hydrologic response metrics, open water PWs showed either a positive correlation or no significant correlation with the same spatial characteristics (Figure 3-9). Moreover, compared to open water

PWs, metrics associated with the timing of the hydrologic response (e.g., LagToMaxWL, RecessDura) for drained PWs showed opposite correlation or no significant correlation with wetland spatial characteristics (Figures 3-9 and 3-10). These results suggest that due to the physical alteration of historic wetlands, the hydrologic response mechanisms and/or the spatial controls of drained PWs have changed significantly compared to open water PWs. The fact that WLRecessRate was positively correlated with ConArea, ConPeri and TotDrain2CatArea (Figure 3-10) is consistent with the hypothesis that the incremental contributing area for drained PWs (due to the physical alteration of historic wetlands; see Table 3-1 and Figure 3-1) led to increased surface runoff and shorter storage memory. This hypothesis was advanced by Haque et al. (2018), who relied on annual and seasonal data to conclude that drained PWs, which have lost their water storage capacity, do not have any noticeable storage memory due to their physical alteration. While investigating hydrologic response metrics in different climatic and geographic settings, Ross et al. (2019) showed that the slope of a catchment can significantly influence stream hydrologic response metrics. However, the present study did not investigate the influence of slope parameters on wetland hydrologic response metrics for PWs.

The fact that some pairs of response metrics and spatial characteristics were positively correlated for some events and negatively correlated for others highlights the difficulty in identifying consistent – or temporally persistent – spatial controls on wetland dynamics. No single wetland characteristic was found to be significantly correlated to any wetland hydrologic response metric for more than 50% of the monitored events, suggesting that the wetland spatial characteristics evaluated here are not good predictors of the spatial variability of wetland hydrologic response, and that other characteristics should be considered. Significant correlations were observed during the events that occurred after the spring melt or following a wet period, and

during big events. This could mean that greater hydrologic connectivity – due to wetter conditions – enhances the influence of spatial characteristics on wetland hydrologic responses.

### **3.5 Conclusion**

This study focused on the hydrologic responses of PWs, the interaction between PWs and a nearby stream, and the temporal persistence of spatial controls on individual wetland hydrological response in the Prairie Pothole Region. Relying on high-frequency water level data with a detailed landscape analysis, this study aimed to enhance our understanding of wetland hydrologic function and its controls, especially at the timescale of individual rainfall events that is typically not documented in the existing literature. One major conclusion of this study is that wetland alteration status (i.e., drained versus open water wetlands) plays an important role in explaining the short-term, event-scale hydrologic behaviour of PWs. Climatic factors, which encompass not only rainfall event characteristics but also antecedent conditions, also had a significant influence on wetland hydrologic response, regardless of the alteration status of the PWs. However, the wetland alteration status appeared to modify the degree to which some climatic factors influence wetland hydrologic response. While a complex combination of factors appeared to influence short-term wetland-stream interaction, antecedent wetness (or storage) conditions and storage memory seemed to be the driving factors. The lack of temporally persistent correlations (i.e., from one rainfall event to another) between hydrologic response metrics and spatial characteristics either indicates that spatial controls on wetland hydrologic function vary greatly over short timescales, or that the spatial characteristics considered in this study are not good predictors of event-scale wetland hydrologic response. As the present study did not consider spatial characteristics related to catchment slope, land use pattern, soil types and wetland vegetation types,

further research is needed to either validate or invalidate the correlations observed in the Broughton's Creek Watershed.

### **3.6 Acknowledgments**

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**Chapter 4: Hydroclimatic Influences and Physiographic Controls on Phosphorus Dynamics  
in Prairie Pothole Wetlands**

## **Abstract**

While wetlands are known as long-term storages or sinks for contaminants, not all are equally effective at trapping phosphorus (P). The prevalence of P-sink behaviour in prairie pothole wetlands remains unclear, especially across gradients of human disturbance. The objectives of the current study were three-fold: (1) characterize the spatiotemporal variability of wetland hydrology and wetland water P concentration across a range of prairie potholes; (2) establish the propensity of different pothole wetlands to act as sources or sinks of P; and (3) assess the potential controls of climatic conditions, landscape characteristics, wetland soil physiochemical properties and local hydrology on source versus sink dynamics. Ten intact and three consolidated (i.e., drained) wetlands located in southwestern Manitoba, Canada, were monitored for water level fluctuations and water soluble reactive P (SRP) concentration over two years with contrasting antecedent wetness conditions. Soil cores were also collected to measure soil physiochemical properties such as the equilibrium phosphorus concentration (EPC). Water column SRP concentrations were compared to EPC values to infer the time-variable source versus sink behaviour of each wetland. Statistical analyses were then performed to assess whether the source versus sink behaviour of individual wetlands could be linked to their physiographic or hydrologic characteristics. Results show that some wetlands persistently acted as P sinks while others switched between source and sink behaviour. Persistent P-sink behaviour was more common with intact wetlands, as opposed to consolidated wetlands. Wetland soil texture, storage volume and short-term water level fluctuations appeared to control the source versus sink behaviour of individual wetlands. The dominant controls on P-sink behaviour identified under dry conditions were, however, different from those identified under wetter conditions. This study, therefore, highlights the importance of

considering the non-stationary nature of P-sorption dynamics and their controls, even at sub-annual timescales, in the Prairie Pothole Region.

**Keywords:** Wetlands, Prairie Pothole Region, local hydrology, phosphorus dynamics, climatic and landscape influences

## 4.1 Introduction

Nutrients such as nitrogen and phosphorus are of environmental concern in many landscapes as they promote eutrophication in wetlands, lakes and freshwater estuaries (Smith et al., 1999). Among the structural and non-structural land management strategies available to control nutrient sources and pathways, wetland conservation and restoration is sometimes advocated. Wetlands are often referred to as the “kidneys of a watershed” (Mitsch & Gosselink, 2007) due to their ability to retain excess water and nutrients and thus mitigate nutrient loading and the formation of eutrophication-driven algal blooms in downgradient systems (Fisher & Acreman, 2004; O'Geen et al., 2010). Several unknowns, however, remain regarding the nutrient retention capacity of wetlands: those unknowns relate to the role of wetland soil physiochemical properties, the particulars of sorption and desorption processes, and the influence of local hydrology in different wetland settings.

The current paper focuses on phosphorus (P), not only because P is an important macronutrient and one of the three major components of agricultural fertilizers (Reddy et al., 1999; Reddy & DeLaune, 2008), but also because previous studies have shown variability – in both space and time – in the P-retention potential of wetlands (Reddy et al., 1999; Fisher & Acreman, 2004; Vymazal, 2007; Wang et al., 2008; Hoffmann et al., 2009). Moreover, P has been identified as a key driver of harmful blooms in large lakes such as Lake Winnipeg (Schindler et al., 2016). P retention can occur through different biotic and abiotic processes such as assimilation by vegetation, periphyton and microorganisms, chemical precipitation and, most importantly, adsorption in wetland soil (Lockaby & Walbridge, 1998; Reddy et al., 1999). P retention in flow-through wetlands is typically inferred when P concentration in wetland inflows is larger than that in wetland outflows. While nutrient loading and wetland water residence time are important

determinants of wetland P retention (Fisher & Acreman, 2004), the physiochemical properties of wetland soil are commonly identified as most influential (Reddy et al., 1999; Reddy & DeLaune, 2008). For instance, fine wetland soil textures (i.e., silt, clay) can promote P retention, not only by slowing down pore water movement – due to the small hydraulic conductivity of the substrate – but also by providing more surface area to adsorb P – due to the composition of the soil particles (e.g., presence of iron and aluminum oxides that strongly adsorb P) and the larger amount of pore spaces available (Ellis & Stanford, 1988). Organic matter in wetland soil is known to facilitate long-term P storage (Mitsch & Gosselink, 2007) due to enhanced adsorption (Reddy et al., 1999). The impact of organic matter on P storage and release, however, differs among P species: organic matter restricts the release of soluble reactive P (SRP, which is mainly inorganic dissolved orthophosphate; Reddy & DeLaune, 2008), while it promotes the release of dissolved organic P (Wang et al., 2008).

When inflow and outflow data are not available, wetland P retention potential can be assessed by comparing P concentrations measured in wetland soil to those measured in the overlying wetland water column (hereafter simply referred to as “wetland water”). The equilibrium phosphorus concentration (EPC) indicates the ability of soil to adsorb P and corresponds to the concentration when adsorption equals desorption at the soil-water interface (Richardson, 1985; Pöthig et al., 2010). For a given wetland, when the wetland water P concentration is larger than the EPC value, P predominantly adsorbs to wetland soil and the wetland acts as a P sink. Conversely, when the wetland water P concentration is smaller than the EPC value, P is predominantly released from the soil into the water column and the wetland acts as a P source (Lane & Autrey, 2016). The equilibrium between adsorption and desorption is controlled by several factors such as the concentration of pore water P, the adsorption capacity of wetland soil

particles, the supply of additional P to the wetland, or the dilution of wetland water P by direct rainfall or lateral runoff inputs. Changes to that equilibrium – triggered by both biotic and abiotic factors – can lead to P storage or P mobilization (Casey et al., 2001; Malecki et al., 2004; White et al., 2004; Dunne et al., 2006; Reddy & DeLaune, 2008). Studies have shown that typically, P storage processes dominate in wetlands, which then act as P sinks (White et al., 2000; Reddy & DeLaune, 2008; Wang et al., 2013; Lane & Autrey, 2016). In some environments, wetlands can act either as long-term sources or as long-term sinks of P (Iqbal et al., 2006; Reddy & DeLaune, 2008). Individual wetlands can also switch from a P sink to a P source (or vice versa) when their trophic state changes, thus suggesting the existence of shifting, short-term dynamics as well, this at the seasonal or sub-seasonal scale (Cooper & Gilliam, 1987; Young & Ross, 2001). As for layering effects, they can make the assessment of source versus sink behaviour rather complex: Mukherjee et al. (2009) notably showed that in some cases, wetland surface soil may act as a source of loosely-bound P while deeper soil acts as a sink.

It is worth mentioning that studies targeting wetland P dynamics have mostly focused on riverine and riparian settings (i.e., flow-through floodplain wetlands), thus raising the question of whether their conclusions also apply to geographically isolated wetlands (Tiner, 2003) or upland-embedded wetlands (Neff & Rosenberry, 2018), namely non-floodplain wetlands such as prairie pothole wetlands (hereafter referred to as PWs). For instance, some of the controls on total P retention in riparian and riverine wetlands include channelization as well as the hydraulic gradient – and associated water travel time – between wetland inflow and outflow points (Knox et al., 2008; Wilcock et al., 2012). These conclusions are, however, not directly transferable to PWs which experience occasional spillage in wet years, are surrounded by uplands and do not have any inflow or outflow channels (Tiner, 2003) unless they are subject to ditching (Brunet & Westbrook, 2012).

Therefore, in most years, regardless of whether wetland sediments in PWs are acting as a sink or source, they will retain some P, if not all, in all forms by minimizing P transport via runoff. While controls on wetland total P retention have been identified in floodplain settings, controls on dissolved species such as SRP are less clear (Knox et al., 2008; Wilcock et al., 2012). This is problematic in the context of PWs, which receive P mainly through dry deposition and runoff (Hauer et al., 2002; Brunet and Westbrook, 2012) and are located in a region where dissolved P represents 80% or more of all P exported via surface and subsurface runoff (McCullough et al., 2012). Floodplain wetlands with periodic water logging tend to store organic P (Craft and Chiang, 2002), while prolonged water logging can create reducing conditions that increase soluble P in wetland water (Vanek, 1991). Those conclusions linking wetland permanence and P retention may be applicable to PWs, which are depressional wetlands prone to wetting and drying cycles (Detenbeck et al., 2002). Otherwise, according to Bridgham et al. (2001) and Casey and Klaine (2001), some of the most important factors for P release/retention in floodplain wetlands are soil particle size fractions and organic matter. Wetland soil properties in PWs are, however, variable based on surrounding land use (Martin & Hartman, 1987; Freeland et al., 1999): PWs surrounded by cropland receive more clay materials than PWs surrounded by forest or tall prairie grass (Preston et al., 2013). Furthermore, the loss of P from upland-embedded wetlands occurs through groundwater seepage, shallow lateral subsurface flow and surface spillage (Hauer et al., 2002), thus suggesting an important role of local surface and subsurface hydrology on the P-retention potential of PWs. PWs also exist across a range of alteration status, from “intact” when they have been mostly undisturbed by anthropogenic activities, to “cropped”, “consolidated” or “lost” when their partial or total drainage is done to facilitate agricultural expansion. The goal of the current paper was, therefore, to examine P-retention dynamics in the Prairie Pothole Region (PPR). The

focus was on a range of PWs of different sizes and different alteration status to evaluate the role of landscape setting and wetland drainage activities on wetland P retention potential. The chosen methodology relied on a combination of wetland-specific physiographic information, soil properties, biweekly water quality data and sub-daily hydrometric data to classify PWs according to their P storage and mobilization dynamics. Specifically, the research objectives were to: (1) characterize the spatiotemporal variability of wetland hydrology and water quality across a range of PWs; (2) establish the propensity of different PWs to act as sources or sinks of P; and (3) assess the potential controls of climatic conditions, landscape characteristics, wetland soil physiochemical properties and local hydrology on PW source versus sink dynamics.

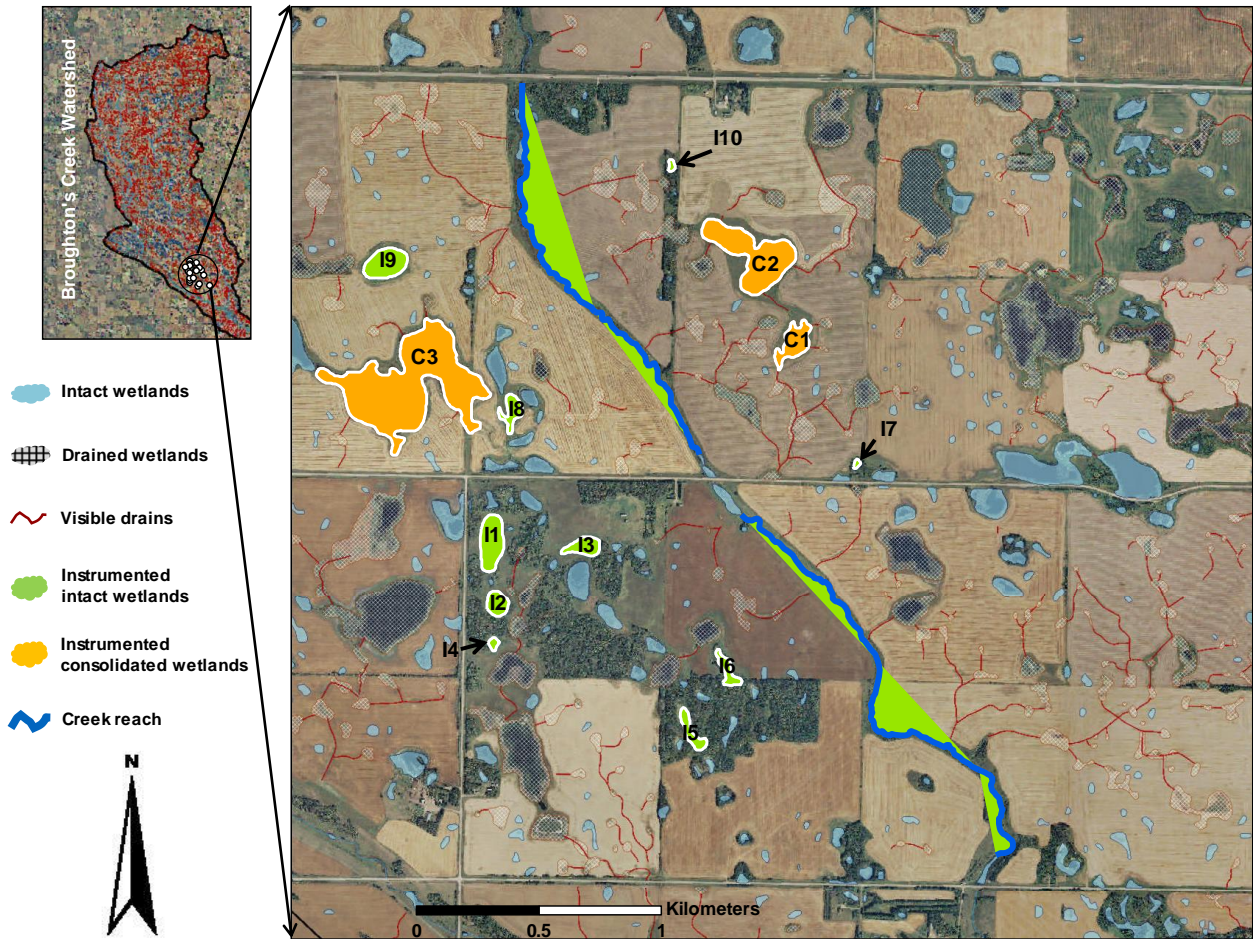
## **4.2 Methods**

### **4.2.1 Study Site Description**

Located in southwestern Manitoba, Canada, the Broughton's Creek Watershed (BCW) drains an area of about 252 km<sup>2</sup>. The main stream, Broughton's Creek, is a tributary of the Little Saskatchewan River that flows into the Assiniboine River which, in turn, flows into the Red River and finally into Lake Winnipeg. The BCW is part of the PPR with landscapes that comprise the surficial deposits left by the retreat of the Assiniboine glacial lobe from 20,000 to 12,000 B.C. (Wang et al., 2010; Yang et al., 2010). Geomorphologically, the area is broadly described as a hummocky till plain and contains numerous potholes, wetlands, and lakes. These water-holding depressions are located in commonly found hummocky moraines or near-level outwash plains and usually have a northwest-southeast orientation. Well-drained Newdale association soils (i.e., mainly Orthic Black Chernozems) are common throughout the watershed with solum depths ranging from 25 to 98 cm. These soils are usually well drained at shallow depths (i.e., <45 cm)

due to the presence of humus-rich A and N horizons. Other minor soils like Dorset, Drokan, Eroded Slope Complex and Jaymar associations are also found in the areas adjacent to streams (Wang et al., 2010). Newdale, Dorset and Jaymar associations are similar to the Udic Boroll subgroups in the U.S. Soil Taxonomic Classification (Soil Classification Working Group, 1998). Previous studies have categorized them as alkaline soils (Badiou et al., 2018; Ige et al., 2008). Land use in the BCW consists of agriculture (71.4%), rangeland (10.8%), wetland (9.8%), forest (4%) and others (4%). It has been estimated that between 1968 and 2005, about 6000 wetland basins (i.e., 21% of the total wetland area) have been fully lost or altered within the BCW due to drainage activities (Wang et al., 2010).

For continuous and targeted monitoring of wetland dynamics, ten intact and three consolidated PWs were selected along both sides of a 5 km study reach in the BCW (Figure 4-1). Based on a historic wetland inventory provided by Ducks Unlimited Canada, hydrologically undisturbed PWs were labelled as “intact”, whereas the term “consolidated” was used when two or more small PWs were drained (consolidated) into a single large one (McCauley et al., 2015). A set of spatial characteristics (Table 4-1) were calculated for the targeted PWs based on the current and historic wetland inventories provided by Ducks Unlimited Canada. The perimeters of the studied PWs vary from 0.1 to 2.8 km, while their areas and storage volumes range from 0.1 to 15.1 ha and from 0.0 to 4.4 ha-m, respectively. Spatial characteristics like the incremental contributing area (ConArea) and the total drainage area (TotDrainArea) were used to illustrate the impact of human alteration on consolidated wetlands. Notably, the incremental contributing area of a consolidated wetland is the sum of the catchment areas of the drained wetlands that existed prior to the drainage and consolidation. Consequently, the total drainage area for a consolidated wetland



**Figure 4-1:** Prairie pothole wetlands and creek reach under study within the Broughton's Creek Watershed. I = Intact wetland, C = Consolidated wetland.

is the sum of its own catchment area and its incremental contributing area. Conversely, for intact wetlands, the incremental contributing area is zero and hence the total drainage area is the same as the catchment area. All studied PWs are located at least 0.5 km (Euclidean distance) away from the study creek and lack surface channelized connections to it, thus fitting the definition of upland-embedded wetlands.

**Table 4-1:** Landscape characteristics estimated for each of the studied prairie pothole wetlands.

Std: standard deviation.

Name	Description (unit)	Code name	Average (Std)	Range
Area	Surface area of wetland (ha)	Area	2.2 (4.1)	0.1 – 15.1
Perimeter	Edge length of wetland (km)	Peri	0.6 (0.7)	0.1 – 2.8
Storage volume	Storage volume of wetland (ha-m)	Vol	0.7 (1.3)	0.0 – 4.4
Volume to area ratio	Ratio of wetland storage volume to wetland surface area (m)	Vol2Area	0.3 (0.2)	0.1 – 0.6
Catchment area	Natural basin area (drainage area) of wetland (ha)	CatArea	7.5 (9.7)	0.6 – 31.3
Catchment to area ratio	Ratio of wetland catchment area to wetland surface area (-)	Cat2Area	8.0 (11.0)	1.9 – 42.5
Catchment to volume ratio	Ratio of wetland catchment area to wetland storage volume (m <sup>-1</sup> )	Cat2Vol	69.1 (153.4)	6.2 – 559.3
Catchment perimeter	Edge length of wetland catchment (km)	CatPeri	1.6 (1.3)	0.4 – 4.8
Shortest distance to creek	Euclidean distance between wetland and nearby creek (km)	Distance	0.7 (0.2)	0.5 – 1.0
Incremental contributing area	Increase in wetland catchment area due to human alteration (ha)	ConArea	1.0 (3.3)	0.0 – 11.8
Incremental contributing perimeter	Edge length of wetland incremental contributing area (km)	ConPeri	0.2 (0.7)	0.0 – 2.4
Total drainage area	Sum of wetland catchment area and wetland incremental contributing area (ha)	TotDrainArea	8.5 (12.0)	0.6 – 37.9
Total drainage area to catchment area ratio	Ratio of wetland total drainage area to wetland catchment area (-)	TotDrain2Cat Area	1.0 (0.1)	1.0 – 1.5
Spill level	Water level at the position of stilling well when a wetland will spill (m)	SL	0.8 (0.2)	0.6 – 1.2
Bottom elevation	Elevation of wetland bottom above sea level (m)	Elev	529.6 (2.6)	526.0 – 534.0

#### 4.2.2 Climate, Hydrometric and Water Quality Data Collection

Stilling wells (i.e., above ground wells) equipped with capacitance-based water level loggers (Odyssey™, Dataflow Systems) were deployed in the intact and consolidated PWs to monitor fluctuations in surface water level at a 15-min frequency during the 2013 and 2014 open water seasons (i.e., April to October). It was assumed that during the rest of the year there was no P transport occurring through surface runoff or shallow groundwater flow, and likely no exchange of P with the water column above due to extremely cold temperatures and frozen water bodies. Bi-weekly surface water samples were collected in 2013 and 2014 from each of the studied PWs. To

ensure consistent access to the same sampling locations even during high-water conditions, samples were collected from the end of the wet-edge zone, just after crossing the emergent vegetation towards the center of each PW basin. Samples were filtered in the laboratory (0.45- $\mu$ m membrane filter) within 24 h of collection and subsequently analyzed for their SRP concentrations using a LaMotte SMART3 colorimeter. In total, 283 water samples were analyzed, and laboratory quality assurance and quality control procedures listed in the United States Environment Protection Agency Water Chemistry Laboratory Manual (U.S. Environmental Protection Agency, 2004) were followed. Filter blanks, field blanks, field duplicates and lab replicates were analyzed and no significant errors in sample collection or associated with the analysis process were detected.

Climate data (precipitation, air temperature) were obtained from an Environment Canada weather station (Brandon A, climate ID: 5010481, World Meteorological Organization (WMO) ID: 71140) located about 30 km south-west from the study site. This station was previously shown to be representative of weather in the BCW (Wang et al., 2010; Yang et al., 2010). Similar mean air temperatures ( $\sim 13.5$  °C) were experienced across the 2013 and 2014 open water seasons. Temperature data were used to estimate potential evapotranspiration (PET) using the Hamon equation (Haith and Shoemaker, 1987). The BCW received 402.8 mm of precipitation during the 2013 open water season, compared to 523.4 mm in 2014, making 2014 a slightly wetter year. The biggest difference between the two study years was in terms of antecedent wetness conditions: long-term (1960–2014) climate data showed that the pre-season moisture deficit (i.e., precipitation minus PET for the April–March period preceding each open water season) was larger than normal in 2013 and smaller than normal in 2014 (see Appendix B-1). Moreover, when average early-April soil moisture was examined, 2013 had smaller than normal values while 2014 had larger than

normal values (AAFC, 2017). Hence, 2013 and 2014 were considered to be dry and wet years, respectively.

#### **4.2.3 Soil Sampling and Physiochemical Analysis**

One undisturbed soil core was collected from each of the studied PWs using a simple Piston Interface Corer (Fisher et al., 1992) from the same locations where surface water samples were collected. Soil cores were divided into two segments, a top portion (i.e., surface to 10 cm depth as previously done by Cohen et al. (2007) and Lane and Autrey (2016)) and a bottom portion (from 10 cm to ~25 cm depth). For each of the PWs, each core segment was mixed to homogenize and stored in a freezer until analyzed. A set of soil physiochemical properties (Table 4-2) were estimated in the laboratory, all within 12 months of collection. For both the top and bottom portions of each wetland soil core, a Malvern Mastersizer 2000 was used for particle size analysis, following standard procedures described in Sperazza et al. (2004). For the determination of organic matter, organic carbon, and inorganic carbon contents, sub-samples of the top and bottom core portions were taken from the preserved and homogenized samples and sent to a commercial laboratory (Agricultural and Food Laboratory, University of Guelph) for analysis. P-sorption isotherm analysis was done on sub-samples taken from the preserved and homogenized top portion of the cores only, following the standardized procedure recommended by the SERA-IEG 17 group (Graetz and Nair, 2009). One gram of air-dried soil sample was weighed into a 50-mL polyethylene bottle to which were added 25 mL of 0.01 M  $\text{CaCl}_2$  and a solution of potassium dihydrogen phosphate ( $\text{KH}_2\text{PO}_4$ ) containing one of the following concentrations of P: 0, 1, 2, 5 or 10 mg P  $\text{L}^{-1}$ . Three drops of chloroform were also added to inhibit microbial activity. The soil suspension was shaken at room temperature for about 24 h on a reciprocating shaker. After equilibration, the soil suspension was centrifuged, and the clear supernatant filtered through a 0.45- $\mu\text{m}$  membrane

**Table 4-2:** Physiochemical wetland soil properties inferred from the collected soil cores.

Description (unit)	Code name	Average (Range)
Equilibrium phosphorus concentration (mg P L <sup>-1</sup> )	EPC	0.2 (0.0 – 2.0)
Phosphorus equilibrium buffering capacity (L kg <sup>-1</sup> )	PEBC	197.4 (35.2 – 384.7)
Maximum phosphorus sorption capacity (mg P kg <sup>-1</sup> )	S <sub>max</sub>	108.5 (6.9 – 333.3)
Percentage of clay within the top 10 cm (%)	%Clay10cm	0.6 (0.2 – 1.4)
Percentage of silt within the top 10 cm (%)	%Silt10cm	57.4 (34.4 – 78.5)
Percentage of sand within the top 10 cm (%)	%Sand10cm	41.9 (20.0 – 65.3)
Percentage of clay below 10 cm (%)	%Clay>10cm	2.9 (1.0 – 9.9)
Percentage of silt below 10 cm (%)	%Silt>10cm	87.3 (81.6 – 94.2)
Percentage of sand below 10 cm (%)	%Sand>10cm	9.7 (4.0 – 17.4)
Percentage of organic matter within the top 10 cm (%)	%OM10cm	15.8 (7.9 – 27.8)
Percentage of organic matter below 10 cm (%)	%OM>10cm	4.8 (1.8 – 12.4)
Percentage of inorganic carbon within the top 10 cm (%)	%IC10cm	0.3 (0.0 – 1.1)
Percentage of inorganic carbon below 10 cm (%)	%IC>10cm	1.0 (0.0 – 3.6)
Percentage of organic carbon within the top 10 cm (%)	%OC10cm	9.3 (4.7 – 15.8)
Percentage of organic carbon below 10 cm (%)	%OC>10cm	2.9 (1.2 – 6.8)

filter. Concentration of P in the filtered solution was analyzed according to U.S. EPA Method 365.3 (U.S Environmental Protection Agency, 1979). Estimation of the soil maximum phosphorus sorption capacity (S<sub>max</sub>; mg P kg<sup>-1</sup>) and the equilibrium phosphorus concentration (EPC) was done by fitting a Langmuir model (Eq. (1)) to the P-sorption isotherm results (Graetz & Nair, 2009; Hongthanat, 2010; Bhadha et al., 2012; Sun et al., 2012; Dai & Hu, 2017):

$$\frac{C}{S} = \frac{C}{S_{max}} + \frac{1}{k \cdot S_{max}} \quad (\text{Eq. 1})$$

where C is the concentration of P in solution after 24-h equilibration (mg P L<sup>-1</sup>), S is the amount of P sorbed on the solid phase (mg P kg<sup>-1</sup>), and k is a constant related to the bonding energy (L mg<sup>-1</sup>). When the quantity C/S is plotted against C (with C on the x-axis), the slope of the linear regression line is equal to 1/S<sub>max</sub>. Since the intercept is equal to 1/(kS<sub>max</sub>), S<sub>max</sub> can be substituted into this equation to get the value for k. Rao and Davidson (1979) showed that at small P concentrations in soil solutions, the relationship between P in the solution phase (C) and P sorbed onto the solid phase (S) can be described by a simple linear equation (Eq. (2)):

$$S = KC - S_o \quad (\text{Eq. 2})$$

where the slope (K) is equal to the phosphorus equilibrium buffering capacity PEBC and indicates the ability of soil to sorb additional P from solution.  $S_o$  represents the P originally sorbed on the solid phase ( $\text{mg kg}^{-1}$ ) (Hongthanat, 2010).

#### 4.2.4 Graphical and Statistical Analyses

Raw surface water level timeseries for the studied PWs were transformed before analysis to normalize them for comparison. Daily wetland fullness was computed by calculating average daily wetland water depths and dividing them by the wetland spilling level (or depth): this was done so that a wetland fullness value more than 1 signals wetland spillage. The hydrologic behaviour of the studied PWs was also characterized in terms of overall storage regime and short-term storage change using a set of behavioural metrics listed in Table 4-3. Specifically, to assess wetland water storage dynamics, metrics such as the percent time a PW was dry (%Dry) or spilling/overflowing (%Spi) (Table 4-3) were computed. To evaluate the importance of short-term storage changes, metrics such as the percent time wetland water level was rising (%Ris), falling (%Fal) or remained unchanged (%NC) were also derived (Table 4-3). These metrics were computed on a year-specific basis, i.e., separately for the 2013 and 2014 data.

Regarding the first research objective, boxplots were constructed to show the temporal and spatial variation of wetland water SRP. To assess the relation between wetland hydrological behaviour and P dynamics, Spearman's rank correlation coefficients (noted as rho) were calculated and evaluated at the 95% significance level between wetland fullness and wetland water SRP concentration. Here the Spearman's rank correlation coefficient was used – rather than the more common Pearson correlation coefficient – because it does not assume a normal distribution of the data and can assess both linear and nonlinear relations between variables (Sokal & Rohlf, 2012).

To address the second research objective, for each of the studied PWs and for each sampling date, the status of “P sink” was inferred when the wetland water SRP concentration was greater than the soil EPC value; otherwise the “P source” status was inferred. The wetlands were then classified into two groups: the “sink” group (S) included PWs that never acted as a source during the 2013–2014 study period, whereas the “sink/source” group (SS) included PWs that switched between source and sink behaviour during the study period. To address the third research objective, Spearman's rank correlation coefficients were calculated and evaluated at the 95% significance level between variables related to P sorption (i.e., EPC, PEBC and  $S_{\max}$ ) and landscape characteristics, soil texture and carbon content variables. Kruskal-Wallis tests were also performed to assess whether the wetlands belonging to the S and SS groups were significantly different from one another, in terms of their landscape characteristics, hydrologic response characteristics, and soil physiochemical properties. The Kruskal-Wallis test was used here because it is non-parametric and helps evaluate two independent groups of same or different size for their similarity (Sokal & Rohlf, 2012). In the current application of this test, the null hypothesis was that the median value of each wetland landscape characteristic, hydrologic response characteristic or soil physiochemical property was similar for the S and SS groups, suggesting that the characteristic or property under consideration had no significant influence on P retention/release dynamics. The null hypothesis was rejected when the Kruskal-Wallis p-value (KWP) was less than 0.05. Kruskal-Wallis tests were performed separately on 2013 and 2014 data to assess the influence of climatic conditions, namely drier than normal and wetter than normal conditions, on wetland P retention and release dynamics and their controls. The MATLAB Statistics and Machine Learning Toolbox, Release 2017b was used for all statistical analyses.

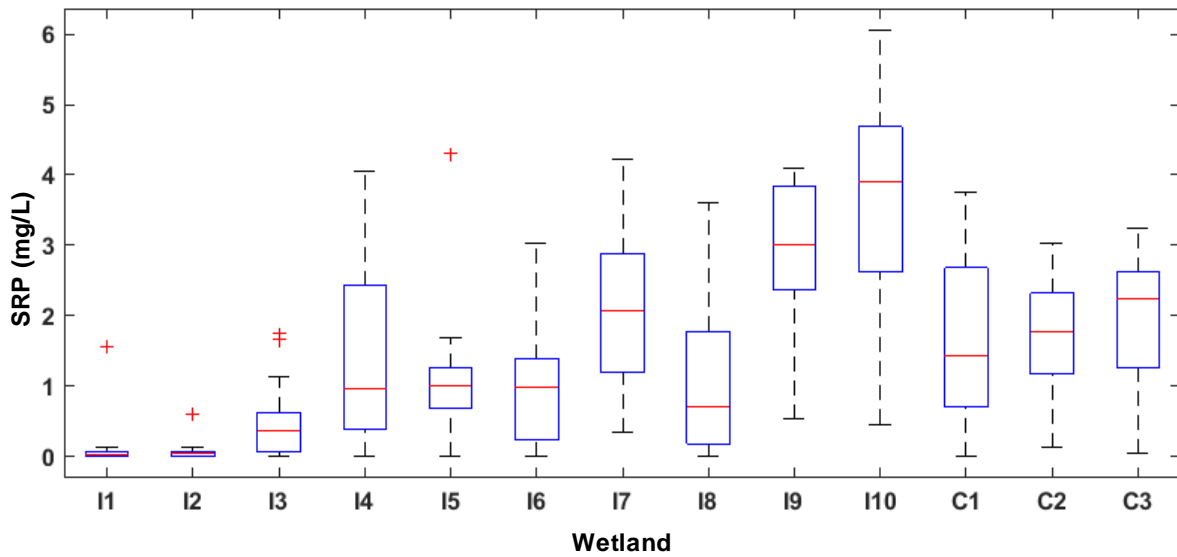
**Table 4-3:** Behavioral metrics used to compare local hydrological dynamics across the studied prairie pothole wetlands.

Properties	Description (Unit)	Code name	Average (Range)
<b>Storage regime</b>	Percent time during which wetland was dry (%)	%Dry	3.2 (0.0 – 24.0)
	Percent time during which wetland was spilling (%)	%Spi	7.6 (0.0 – 24.2)
	Percent time during which wetland was half-full (%)	%Hf	56.6 (0.1 – 100)
	Standard deviation of wetland fullness, used as an indicator of temporal variability (-)	StdWf	0.1 (0.1 – 0.2)
<b>Short-term storage change</b>	Percent time during which wetland water level was rising (%)	%Ris	76.9 (50.6 – 91.3)
	Percent time during which wetland water level was falling (%)	%Fal	20.2 (3.3 – 47.6)
	Percent time during which wetland water level did not change (%)	%NC	2.9 (0.8 – 9.1)

## 4.3 Results

### 4.3.1 Spatial and Temporal Variability of Wetland Hydrology and Water Quality

There was considerable temporal variation in the SRP concentrations observed in the water column of all studied wetlands with the exception of wetlands I1 and I2 (Figure 4-2). During the study period, SRP concentrations ranged from a minimum of 0.01 mg/L (equal to the detection limit of the colorimeter) to a maximum of 6.06 mg/L in the studied PWs (Figure 4-2). Spatial proximity was not a good surrogate for wetland water SRP concentration. For example, wetlands I1, I2, I3, and I4 showed very different SRP concentration ranges despite being geographically close to one another (Figures 4-1 and 4-2). Consolidated wetlands, however, showed similar SRP concentrations irrespective of their spatial location (Figure 4-2). As for local wetland hydrology, it varied greatly in time and space (Figure 4-3). Generally, wetland fullness values were larger in 2014 than in 2013 (Figure 4-3). About half of the studied wetlands showed statistically significant

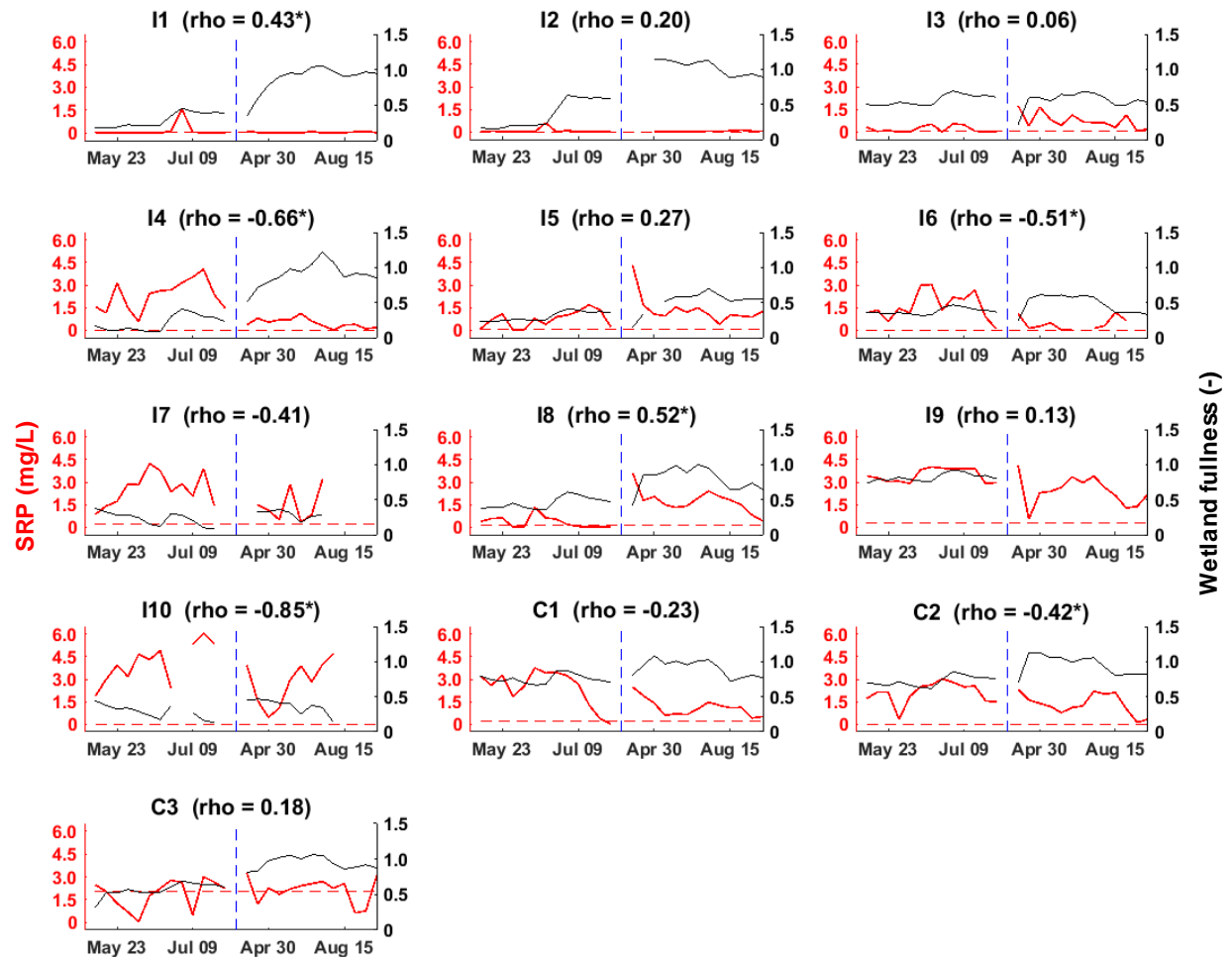


**Figure 4-2:** Variability of SRP concentrations in the water column of the studied wetlands. Each box has lines at the lower quartile, median, and upper quartile values, while the whiskers beyond the box show the extent of the rest of the data. Outliers are shown as red plus (“+”) signs.

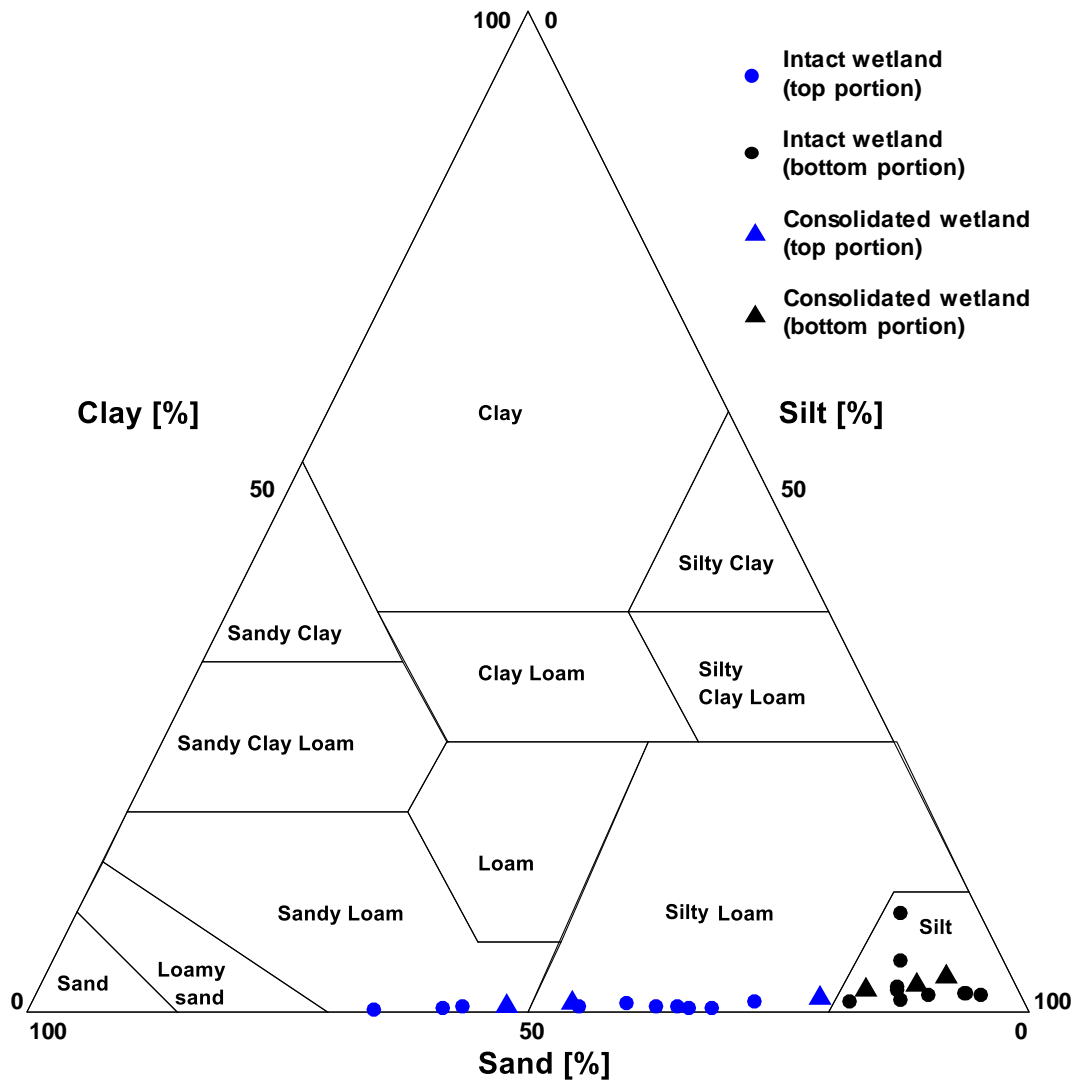
correlations between wetland fullness and SRP concentrations (Figure 4-3). The strength and direction of correlation between wetland fullness and SRP, however, varied spatially. For instance, although wetlands I1, I2, and I4 are geographically close (Figure 4-1), the SRP-wetland fullness correlation was positive for I1, negative for I4, and not statistically significant for I2. Similarly, SRP-wetland fullness correlations were very different for I5 and I6, and for C1 and C2, despite their relative geographic proximity (Figures 4-1 and 4-3).

#### 4.3.2 Wetland Soil Physiochemical Properties and P-Sorption Dynamics

Particle size analyses showed that the texture of the studied wetland soils was sandy loam, silty loam or silt (Figure 4-4). Although samples from the top portion of the soil cores fell into both the sandy loam and silty loam categories, samples from the bottom portion of the soil cores

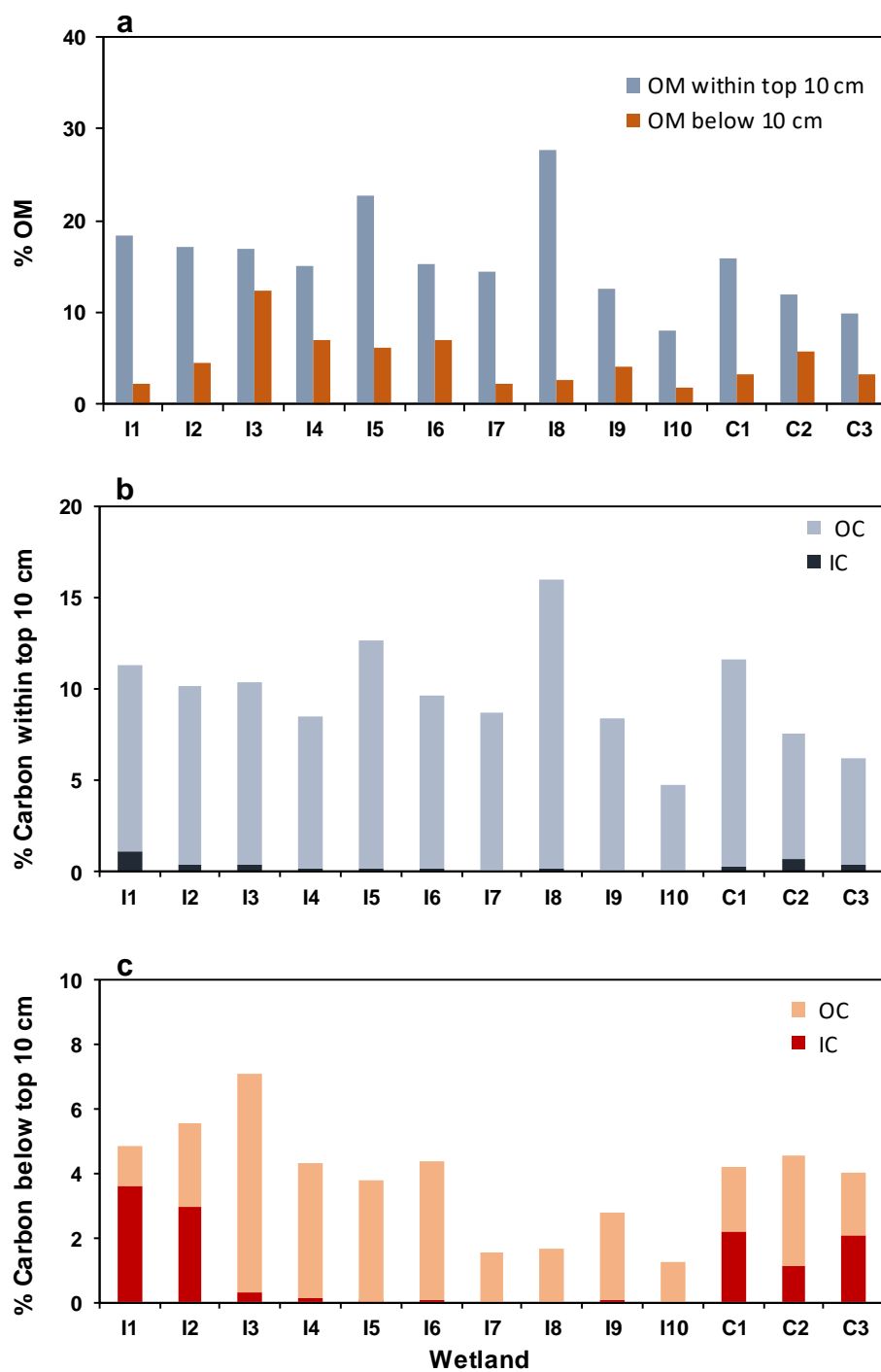


**Figure 4-3:** Timeseries of wetland fullness (daily data) and SRP concentrations (biweekly data) in each of the studied wetlands across the study period. Biweekly data points are linked to produce a SRP timeseries for easier visualization and comparison with the wetland fullness timeseries. The blue vertical dashed line separates 2013 data (left) from 2014 data (right). The red horizontal dashed lines represent the EPC value for each wetland. rho is the Spearman's rank correlation coefficient between wetland fullness and SRP concentrations for each wetland. ‘\*’ signals correlation coefficients that were statistically significant at the 95% level (i.e., p-value < 0.05). Missing data points on the timeseries signal the malfunction of a water level logger (e.g., wetland I9) or dry conditions.



**Figure 4-4:** Texture triangle showing the variability in wetland soil particle size among the studied wetlands.

were all identified as silt (Figure 4-4). Organic matter (OM) percentages within the top portion of the soil cores were larger than those in the bottom portion of the cores, and substantial spatial variability was present (Figure 4-5a). For instance, wetland I8 had the largest OM percentage in the top portion of its soil but also showed the largest difference in OM content between the top 10 cm and deeper layers.



**Figure 4-5:** Wetland-specific and depth-specific (a) organic matter; and (b) and (c) total, organic and inorganic carbon content estimated from the collected soil cores. OM: organic matter. OC: organic carbon. IC: inorganic carbon.

Conversely, wetland I3 had the largest OM percentage in the bottom portion of its soil and showed the smallest difference in OM content between the top 10 cm and deeper layers. (Figure 4-5a). In general, total carbon percentages (sum of organic and inorganic carbon) in the top soil core segment were larger than those in the bottom segment (Figures 4-5b and 4-5c). Regardless of depth, proportions of soil organic carbon (OC) were much larger than those of inorganic carbon (IC). The proportion of inorganic carbon increased with depth for all consolidated wetlands (C1, C2, and C3); however, such an increase was only observed for a minority of intact wetlands (e.g., I1 and I2) (Figures 4-5b and 4-5c).

P-sorption analyses revealed that more than half of the studied wetlands have EPC values equal or very close to zero mg/L (Table 4-2). The maximum EPC value ( $2.1 \text{ mg P L}^{-1}$ ) was estimated for wetland C3. Most of the studied wetlands had PEBC values larger than  $197.4 \text{ L kg}^{-1}$ , with a maximum value of  $384.7 \text{ L kg}^{-1}$  for wetland I2 and a minimum of  $35.2 \text{ L kg}^{-1}$  for wetland C3. Wetland I3 had the maximum  $S_{\text{max}}$  value of  $333.3 \text{ mg P kg}^{-1}$  while wetland I8 had the minimum value of  $6.9 \text{ mg P kg}^{-1}$ . However, almost half of the studied wetlands had  $S_{\text{max}}$  values exceeding  $100 \text{ mg P kg}^{-1}$  (Table 4-2). Intact wetlands had smaller EPC but larger PEBC and  $S_{\text{max}}$  values than consolidated wetlands. Average EPC, PEBC, and  $S_{\text{max}}$  values for intact wetlands were  $0.08 \text{ mg P L}^{-1}$ ,  $218 \text{ L kg}^{-1}$ , and  $123 \text{ mg P kg}^{-1}$ , compared to  $0.75 \text{ mg P L}^{-1}$ ,  $129 \text{ L kg}^{-1}$ , and  $60 \text{ mg P kg}^{-1}$ , respectively, for consolidated wetlands. Wetland C3 had the largest EPC but the smallest PEBC value, while wetland I2 had the smallest EPC value but the largest PEBC value.

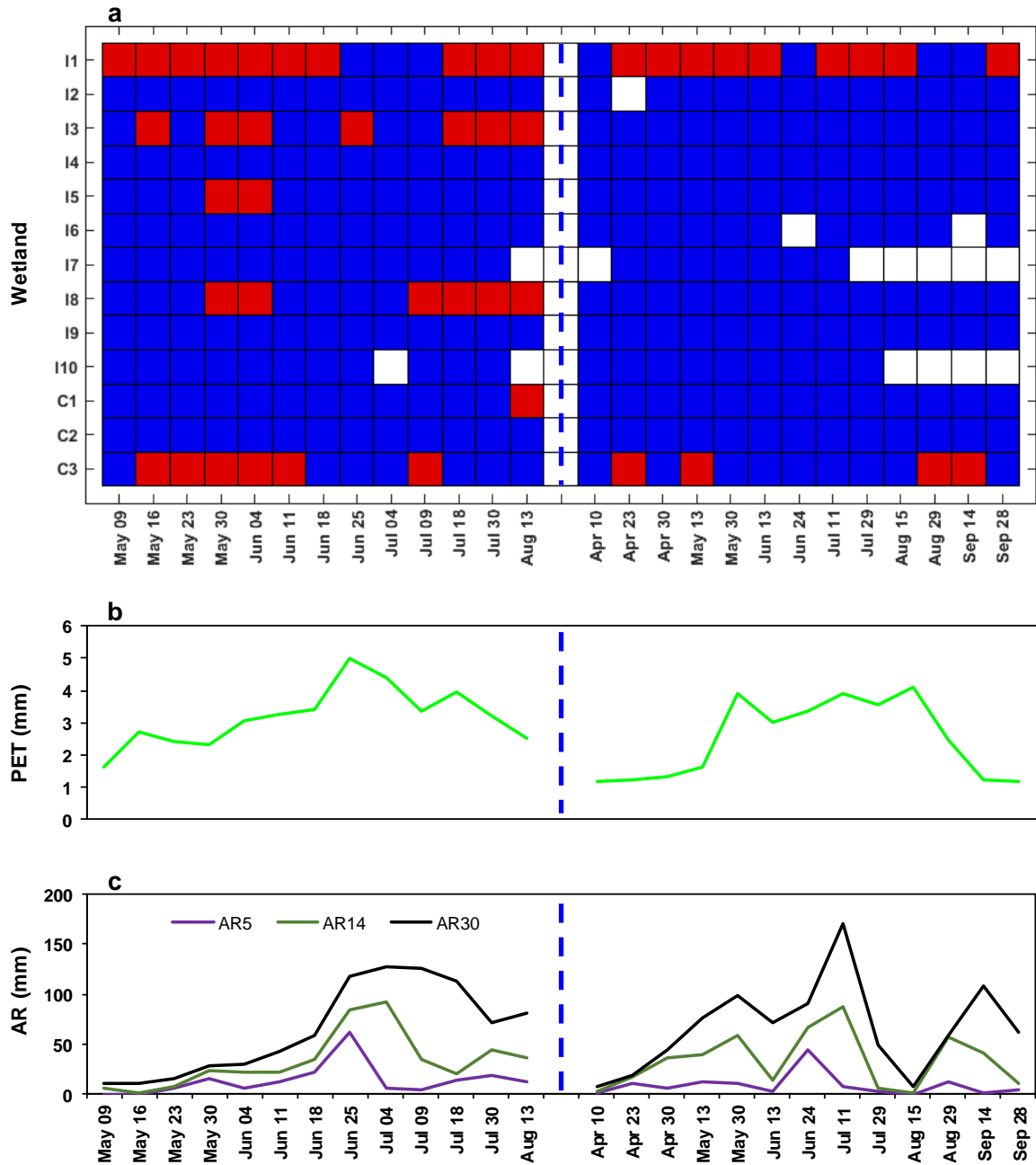
SRP concentrations in most of the intact wetlands (e.g., I2, I4, I6, I7, I9, and I10) and one consolidated wetland (C2) were always larger than their respective soil EPC value, indicating that they persistently acted as P sinks. All other studied wetlands switched between P-source and P-sink behaviour throughout the study period (Figures 4-2 and 4-6a). Interestingly, some wetlands

demonstrated switching behaviour in both study years (wetlands I1 and C3) while others only switched between P-source and P-sink dynamics in drier than normal conditions (wetlands I3, I5 and I8 in 2013). Wetlands associated with statistically significant negative correlations between wetland fullness and SRP (I4, I6, I10, and C2) always had SRP concentrations larger than their soil EPC values (Figures 4-2 and 4-6a). None of the instrumented wetlands acted solely as a P-source throughout the whole study period (Figure 4-6a). Regarding the role of climatic conditions during our study period, high potential evapotranspiration (PET) was associated with high antecedent rainfall (AR) conditions and vice versa (Figures 4-6b and 4-6c). “Switching” wetlands tended to act as P sinks during higher PET and AR conditions (e.g., on 25-June-13, 04-July-13, and 11-July-14) and as P sources at other times (Figures 4-6b and 4-6c).

#### **4.3.3 Potential Controls on Wetland P-Sorption Dynamics**

Spearman's rank correlation analyses revealed potential controls of landscape characteristics, wetland soil texture and carbon content on variables related to P-sorption dynamics. Here, only correlation coefficients significant at the 95% level are reported. Notably, the EPC of the studied wetlands showed positive correlation with %Clay10cm ( $\rho = 0.7$ ,  $p\text{-value} < 0.05$ ), whereas the PEBC showed positive correlation with Distance ( $\rho = 0.7$ ,  $p\text{-value} < 0.05$ ). No other landscape characteristics or soil physical properties showed significant correlation with EPC, PEBC, or  $S_{\max}$ .

A limited number of landscape characteristics, soil characteristics and metrics of hydrological behaviour showed statistically significant differences between wetland groups. Among soil physiochemical properties, only %OC10cm showed statistically significant differences ( $KWP = 0.03$ ) between the S and SS groups of wetlands during drier than normal conditions (i.e., 2013), with the former group (S) having smaller %OC10cm.



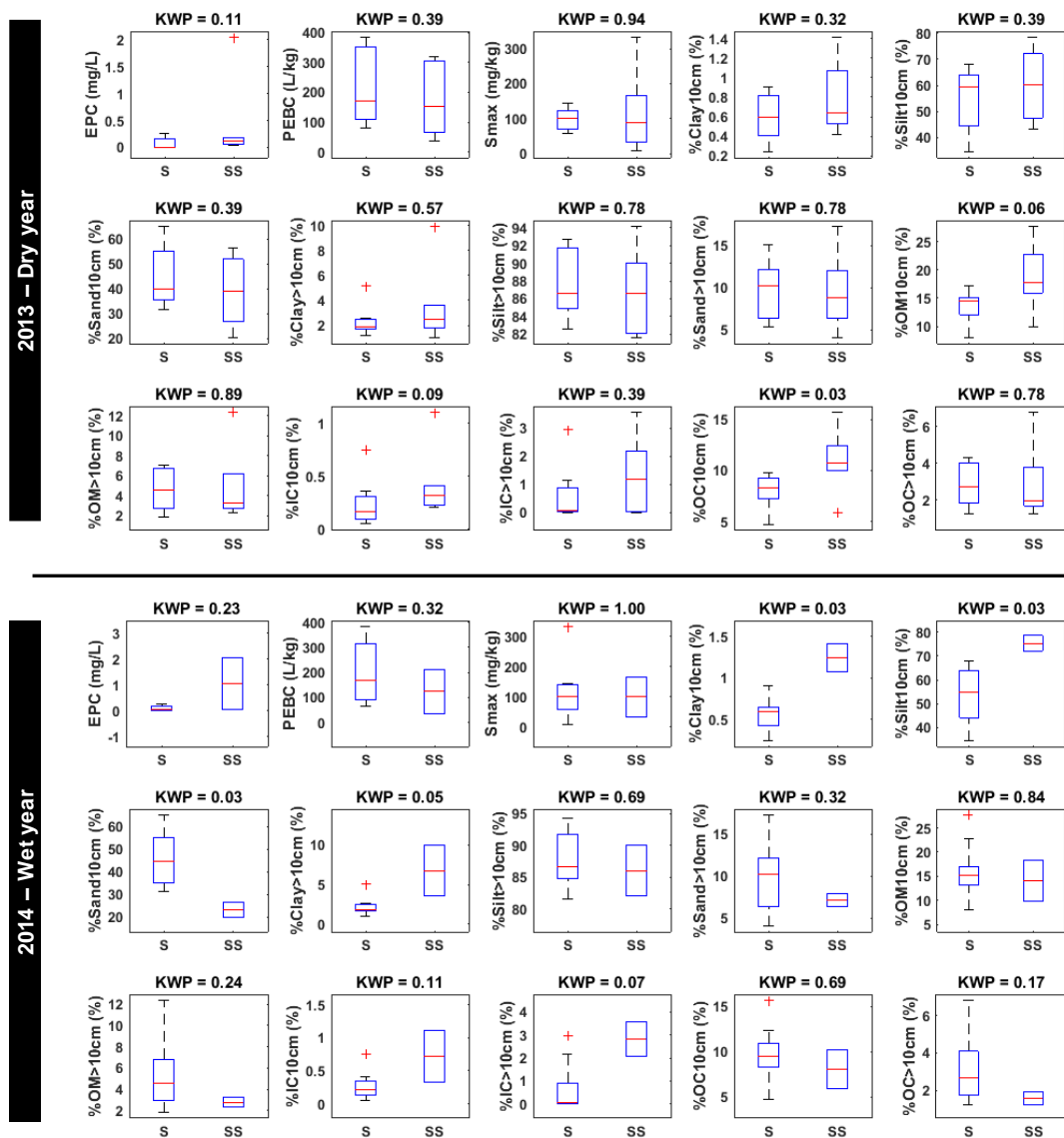
**Figure 4-6:** (a) Temporal variability of P-source (red cells) versus P-sink (blue cells) behaviour for each of the studied wetlands. White cells represent dry conditions or no data. (b) and (c): Temporal variation of daily potential evapotranspiration (PET) and antecedent rainfall (AR) across all sampling dates. ARX: cumulative rainfall for the “X” days prior to each sampling date. The blue vertical dashed line separates 2013 data (left) from 2014 data (right).

However, during wetter than normal conditions, %Clay10cm, %Silt10cm, and %Sand10cm were different ( $KWP = 0.03$ ) between the S and SS groups of wetlands (Figure 4-7), with wetlands in the S group having smaller %Clay10cm but larger %Silt10cm and %Sand10cm values than wetlands in the SS group (Figure 4-7). Metrics of wetland hydrological behaviour did not reveal any statistically significant difference between the S and SS groups (i.e.,  $KWP > 0.05$ ; Figure 4-8) during drier than normal conditions (i.e., 2013). Conversely, during wetter than normal conditions, the StdWf, %Ris, and %Fal metrics were different among groups ( $KWP = 0.03$ ; Figure 4-8): wetlands in the S group had smaller StdWf and %Ris values but larger %Fal values than wetlands in the SS group (Figure 4-8). During wetter than normal conditions (i.e., 2014), wetlands that persistently acted as P sinks (i.e., the S group) had larger Cat2Vol values ( $KWP = 0.03$ ) than wetlands that switched between P-sink and P-source behaviour (i.e., the SS group) (Figure 4-9). Other landscape characteristics did not show any significant difference between the S and SS groups of wetlands during the study period (Figure 4-9).

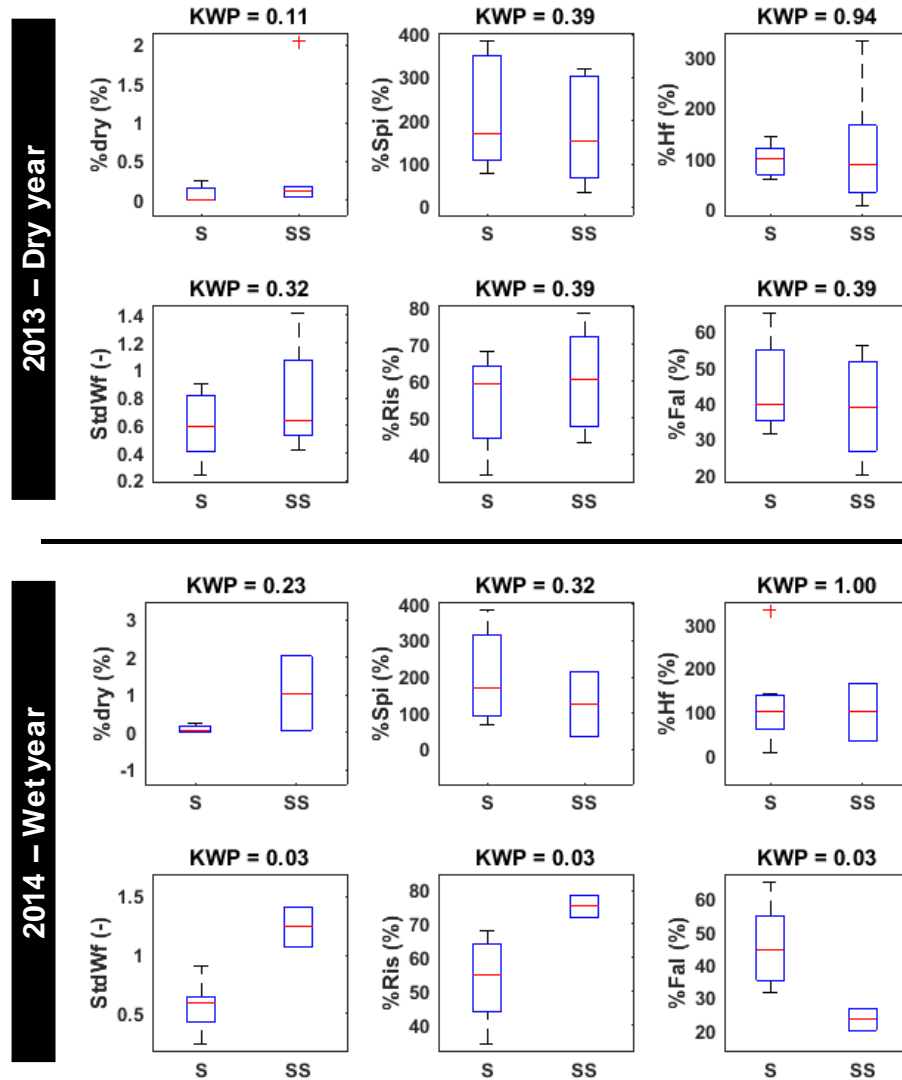
## **4.4 Discussion**

### **4.4.1 Linking Hydrology and Water Quality in Pothole Wetlands**

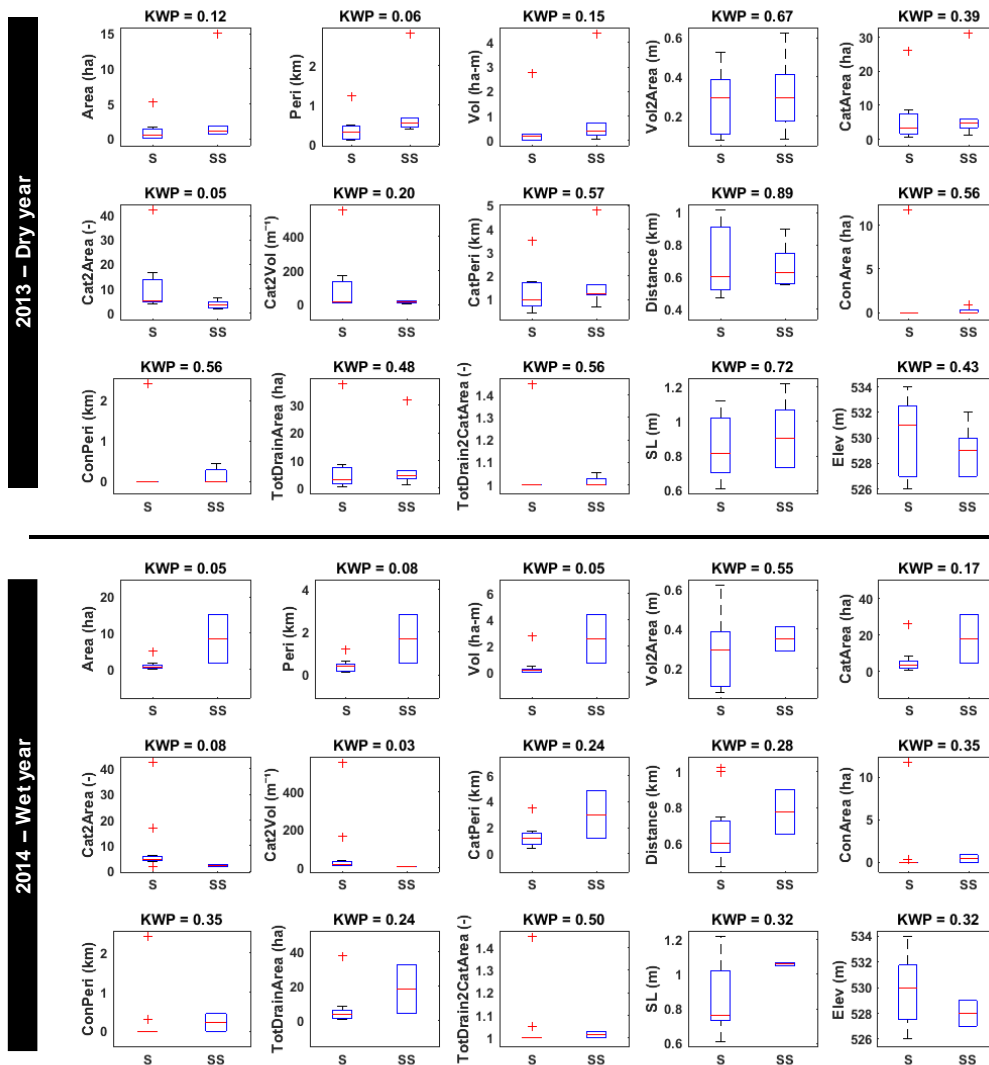
Both current-season precipitation and antecedent moisture conditions had significant influences on local wetland hydrology. Notably, larger wetland fullness values in all studied PWs in 2014 (Figure 4-3) can be attributed to the wetter conditions and larger amount of precipitation experienced that year. Irrespective of the year considered, consolidated wetlands were always “fuller” than intact wetlands. Spillage was also much more frequent in 2014 (wetland fullness values in excess of 1; see Figure 4-3) for both intact and consolidated wetlands. The variability of



**Figure 4-7:** Differences in soil physiochemical properties between wetlands that persistently acted as P sinks (i.e., S group) and wetlands that switched between P-sink and P-source behaviour (i.e., SS group) during the 2013 and 2014 open water seasons. Each box has lines at the lower quartile, median, and upper quartile values, while the whiskers beyond the box show the extent of the rest of the data. Outliers are shown as red plus signs. KWP: p-value associated with the Kruskal-Wallis test.



**Figure 4-8:** Differences in wetland hydrologic response characteristics between wetlands that persistently acted as P sinks (i.e., S group) and wetlands that switched between P-sink and P-source behaviour (i.e., SS group) during the 2013 and 2014 open water seasons. Each box has lines at the lower quartile, median, and upper quartile values, while the whiskers beyond the box show the extent of the rest of the data. Outliers are shown as red plus signs. KWP: p-value associated with the Kruskal-Wallis test.



**Figure 4-9:** Differences in landscape characteristics between wetlands that persistently acted as P sinks (i.e., S group) and wetlands that switched between P-sink and P-source behaviour (i.e., SS group) during the 2013 and 2014 open water seasons. Each box has lines at the lower quartile, median, and upper quartile values, while the whiskers beyond the box show the extent of the rest of the data. Outliers are shown as red plus signs. KWP: p-value associated with the Kruskal-Wallis test.

wetland water SRP concentrations across intact wetlands (Figure 4-2) suggests, indirectly, the influence of local and non-local landscape characteristics on the P dynamics prevailing in unaltered systems. Conversely, the similar ranges of variation observed for wetland water SRP in consolidated wetlands seem to downplay the influence of landscape position (Figure 4-2) and rather suggest that wetland alteration may result in a spatial homogenization of P dynamics. Hayashi et al. (2016) identified surface runoff as the main inflow to prairie wetlands that contributes to wetland water storage; it may lead to positive correlations between wetland fullness and SRP given the large nutrient content of surface runoff originating from agricultural land (McCullough et al., 2012). The temporally variable correlations, in terms of magnitude and direction, between wetland fullness and wetland water SRP, however, suggest more complex dynamics (Figure 4-3). One hypothesis might be that the studied wetlands do not receive P only through surface runoff (Johannesson et al., 2017) but also via other flow pathways such as shallow subsurface flow (Qiu et al., 2004). Large wetland water SRP concentrations despite small wetland fullness may be due to evapoconcentration effects, especially in relatively dry conditions (i.e., 2013). Negative correlations between wetland fullness and SRP concentration could also be an indication of dilution, when large volumes of low-SRP inflows to the wetlands occur.

#### **4.4.2 Characterizing the Spatial Variability of Wetland Soil Properties**

While the accumulation of sediment at the bottom of depressional wetlands is very common, the rate at which it occurs is influenced by proximal cultivation practices (Gleason & Euliss, 1998; Preston et al., 2013). Previous studies in North and South Dakota have notably shown that changes in agricultural practices in the prairie region over the last decade have increased the risk of soil erosion in the proximity of PWs (Johnston, 2013; Wright & Wimberly, 2013). However, in the Canadian portion of the PPR, the risk of soil erosion by wind, water and tillage is

considered very small (Lobb et al., 2016). Therefore, variation of soil texture between the top and bottom portions of the wetland soil cores (Figure 4-4) is likely due to changes in the depositional environment during the postglacial period. OM and OC contents in the top portion of the soil cores were larger than in the bottom portion of the soil cores (Figure 4-5), indicating that wetland vegetation is likely the primary source of organic materials to wetland soil and suggesting that the decomposition rate of wetland vegetation might be slower than the production rate (Thormann et al., 1999; Trites and Bayley, 2009). However, the spatial variation of organic matter content (Figure 4-5) is also impacted by environmental factors such as fluctuating water levels and drying cycles in individual wetlands (Thormann et al., 1999), both dynamics which are known to be especially important in PWs (Cook & Hauer, 2007; Goldhaber et al., 2014).

Studies documenting wetland soil physiochemical properties like EPC,  $S_{\max}$ , and PEBC for PWs, specifically, are rare. However, the EPC values reported in the current study (Table 4-2) are within the range of previously reported EPC values for palustrine forested and emergent marsh depressional wetlands (0.01–27.18 mg P L<sup>-1</sup>; Lane and Autrey, 2016), riparian and floodplain wetlands in the Lake Okeechobee Basin in Florida (0.03–1.28 mg P L<sup>-1</sup>; Reddy et al., 1995), wetlands located on alluvial deposits in Yangtze River Basin in China (0.0 – 0.06 mg P L<sup>-1</sup>; Wang et al., 2013), and laboratory experiments conducted on periodically flooded wetland soils (0.02–7.2 mg P L<sup>-1</sup>; Young & Ross, 2001). Similarly, the PEBC values reported in the current study (Table 4-2) are within the range of previously reported phosphorus buffering capacity values (which are akin to PEBC values) for palustrine forested and emergent marsh depressional wetlands (4.99 to 927.76 mg P kg<sup>-1</sup>; Lane & Autrey, 2016) and for riparian and floodplain wetlands in the Lake Okeechobee Basin (6.3 to 49.4 L kg<sup>-1</sup>; Reddy et al., 1995). As for the  $S_{\max}$  values reported in the current study (Table 4-2), they are also within the range of what Lane and Autrey (2016)

reported for palustrine forested and emergent marsh depressional wetlands (51.3–5115.0 mg P kg<sup>-1</sup>). Besides, in a study of over 170 wetlands in the southwestern U.S., Cohen et al. (2007) reported single point P-sorption index values (similar to  $S_{\max}$  as described by Reddy et al., 1998) ranging from 0.00 to 990 mg P kg<sup>-1</sup>. It is worth mentioning that laboratory procedures may have had an impact on our estimated P-sorption related properties. Notably, as we dried our samples prior to analysis and then “rewetted” them by adding P solutions to them, wet-dry cycles may have altered the P-sorption capacity of the collected soil due to particle aggregation (Tang et al., 2017), release of unreactive P bound to soil colloids/nanoparticles (Gu et al., 2018), or mineralization and change in the redox potential (Shenker et al., 2005). While drying soil samples for analyzing physiochemical properties such as EPC is a standard procedure (e.g., Reddy et al., 1995; Cohen et al., 2007; Graetz & Nair, 2009; Hongthanat, 2010; Bhadha et al., 2012; Sun et al., 2012; Lane & Autrey, 2016; Dai & Hu, 2017), we are aware of one group who chose to preserve soil samples at field moisture in the dark until analysis (e.g., Badiou et al., 2018). Regardless of whether wet or dry samples are used, ex-situ laboratory analyses are known to be error-prone (Palmer-Felgate et al., 2011). In the current study, we did not specifically quantify the effect of wet-dry cycles on P-sorption properties but assumed that any bias, if present, would be relatively constant across all wetland soil samples. For consistency, when comparing our results with those from other studies, we only reported soil sorption properties (i.e., EPC, PEBC and  $S_{\max}$  values) that were obtained using similar analysis procedures based on air-dried samples.

Spatial proximity did not always explain the variation of soil sorption properties (i.e., EPC, PEBC and  $S_{\max}$ ) among individual wetlands. As an example, I2 and I4 are located very close to each other (Figure 4-1) and showed similar sorption properties. However, other wetlands located very close to each other did not have similar sorption properties (e.g., I1 and I2, I5 and I6, C1 and

C2) (Figure 4-1). While the EPC just expresses the critical concentration at which adsorption and desorption processes balance one another, the PEBC provides an indication of how P inputs can be buffered by the system under study. Typically, wetland soils with small EPC values tend to have more P-sorption sites, hence their greater potential for buffering incoming P fluxes (Lane and Autrey, 2016). This was most evident from contrasting EPC and PEBC values for wetlands I2 and C3: the smallest EPC and largest PEBC values were observed for I2, and the exact opposite observed for C3. Similarly, the larger average EPC values but smaller average PEBC and  $S_{\max}$  values observed in consolidated wetlands, compared to intact wetlands, suggest that consolidated wetland soils have a weaker potential for buffering P inputs and are, therefore, more prone to release adsorbed P to the overlying water column.

#### **4.4.3 Classifying Wetlands According to Their P-Sorption Dynamics**

Figure 4-6a shows that 6 out of 10 of the intact wetlands were always acting as P sinks, while only one (out of 3) consolidated wetland (i.e., C2) acted as such. This result is likely due to the fact that the smallest PEBC and  $S_{\max}$  values were observed for two of the three monitored consolidated wetlands (see average figures provided in Section 5.3.2). Such an observation is, however, not unusual; Cohen et al. (2007) examined natural wetlands in different ecozones of the southwestern U.S. and concluded that those impacted through anthropogenic impairments (similar to the consolidated wetlands in the current study) had less P-sorption capacity than reference, minimally impacted wetlands.

As previously mentioned, the EPC value embodies a “threshold” according to which the direction of P flux (i.e., from soil to water or from water to soil) changes. However, due to the input and transport of P from different sources through different pathways (e.g., surface and subsurface flow; Qiu et al., 2004; Johannesson et al., 2017), the status of a given wetland as a P

source or a P sink is dynamic, in accordance with the temporal variation of wetland water column P concentrations (Figures 4-3 and 4-6a). For instance, water inputs to a wetland can lower (i.e., dilute) the existing wetland water column P concentrations below the EPC value, thus making a wetland act as P source despite having a small EPC value (e.g., I1; Figure 4-6a). The switching behaviour of some wetlands (e.g., I3, I5, I8, and C1) also highlights the importance of hydroclimatic conditions on P-sorption dynamics: the fact that some of the studied PWs fluctuated between being P sources and P sinks in 2013, but only acted as P sinks in 2014 (Figure 4-6a), could be an indication of greater P mobilization from different sources due to wetter than normal conditions in 2014, thus raising the wetland water column P concentration above the EPC value (Figures 4-3 and 4-6) and promoting P-sink behaviour. Wetland water P concentrations can also change without the input of external P but rather due to the remobilization of internal P: indeed, wetland soil usually has large P concentrations around the area where P-enriched water enters the wetland (Reddy & DeLaune, 2008), and this elevated soil P can act as a source of P within the wetland itself and contaminate less impacted areas elsewhere in the wetland through remobilization. Such remobilization can happen due to enhanced intra-wetland hydrologic connectivity between impacted areas and non-impacted areas, or because of changes in redox conditions in P-rich areas (Grunwald et al., 2006; Bostic et al., 2010). All of these mechanisms may be the root causes of the temporal variation in wetland water P concentrations and thus explain why some wetlands did not always have their SRP values exceed their EPC threshold (Figure 4-6a).

The influence of local hydroclimatic conditions on P-sorption dynamics is particularly important in the case of PWs because of their peculiar setting. Indeed, PWs do not have surface connections to stream networks and therefore, PET is a major component that controls their water

storage status (Hayashi et al., 2016). Particularly, high PET could decrease water storage (i.e., wetland fullness) and, consequently, increase the wetland water SRP concentration to a level above the EPC threshold. This may explain why, in the current study, all of the wetlands that showed negative Spearman's rank correlation coefficients between SRP and wetland fullness were the same that consistently acted as P sinks (Figures 4-3 and 4-6a). Also, even the “switching” wetlands (i.e., those belonging to the SS group) had a greater tendency to act as P sinks during high PET days (Figure 4-6). The fact that none of the studied wetlands always acted as a P source throughout the study period indicates the strong potential of PWs for P storage. However, the EPC, PEBC and  $S_{\max}$  values of intact wetlands and their dominant classification in the S group throughout the study period – spanning both dry and wet conditions – suggest that they have more potential than consolidated wetlands to act as long-term P sinks. It should be mentioned that the premise of the current study was that the main elements controlling the P-source or P-sink behaviour of a wetland were soil adsorption/desorption processes (Richardson, 1985; Hongthanat, 2010; Pöthig et al., 2010; Lane and Autrey, 2016), and our experimental set-up was chosen accordingly. As a result, our study may have underestimated the effects of chemical precipitation or assimilation by vegetation, periphyton and microorganisms on P-sorption dynamics.

#### **4.4.4 Identifying Controls on P Dynamics in Pothole Wetlands**

The current study highlighted some statistically significant correlations between soil properties and P-sorption variables which are not always in agreement with previous findings reported in the literature. For instance, previous studies showed that fine-grained wetland soils (i.e., silt and clay) have better P-retention capacity and smaller EPC values than other wetlands (Reddy et al., 1998; Hongthanat, 2010; Lane & Autrey, 2016). The combined presence of clay and organic matter is also thought to improve the P-buffer capacity of soils (Hongthanat, 2010; Lane

& Autrey, 2016). However, Dunne et al. (2006) indicated that the complex association of aluminum (Al) and iron (Fe) at the surface of organic matter can indirectly influence the P sorption capacity of soil. Nair et al. (2015) notably found that the role of organic matter in wetland soil P retention and release depends on the P saturation ratio (i.e., ratio of P to [Al + Fe]). No data regarding Al and Fe levels were available in the current study. Nevertheless, the positive correlation between %Clay<sub>10cm</sub> and sorption variables (i.e., EPC), and the lack of correlation between sorption variables and organic matter content, perhaps indicates that the affinity of P to clay and organic matter is different in pothole wetland settings. Interestingly, during dry conditions, the wetlands that consistently acted as P sinks (i.e., S group) had smaller levels of organic carbon in the shallow portion of their substrate (i.e., in top portion of the soil cores) than others. During wet conditions, the wetlands belonging to the S group distinguished themselves by their finer soil textures (Figure 4-7). Measuring soil Al and Fe content might, therefore, be crucial for better understanding P sorption dynamic in PWs.

In the current study, metrics of hydrologic behaviour did not show any difference between the S and SS groups of wetlands under drier hydroclimatic conditions (i.e., in 2013). However, during the wetter year (i.e., in 2014), the wetlands that showed little temporal variability in their short-term water level fluctuations (i.e., small StdWf value) and generally small water storage status (i.e., small %Ris but large %Fal values indicative of drying conditions) seemed to have a greater tendency to act as P sinks (Figure 4-8). Wetter conditions could have enhanced hydrologic connectivity between wetland complexes through subsurface flow (Hayashi et al., 2016), which is known to favour P adsorption in wetland soil (Mukherjee et al., 2009; Lane & Autrey, 2016). Therefore, wetter conditions may have promoted P-sink behaviour. Those results also highlight the fact that the relation between local wetland hydrology and P-sorption potential is non-

stationary. The positive correlation between the PEBC and the Distance metric indicates that in our study region, the wetlands located closer to the stream have a greater P-buffering capacity than others. This is aligned with conclusions by Daneshvar et al. (2017), according to which wetland size and proximity to stream increase P-retention capability. In the current study, none of the considered landscape characteristics showed any statistical difference between the S and SS groups of wetlands during dry conditions (Figure 4-9). However, during wetter conditions, wetlands with smaller volumes but larger catchment areas seemed to have more potential for P retention (Figure 4-9). Such results underscore that antecedent wetness conditions can dramatically influence the control – or lack thereof – of landscape characteristics on P-sorption dynamics. In a modelling study, Cheng and Basu (2017) found that wetlands with smaller surface area were more effective in retaining nutrients – such as phosphorus and nitrogen – than bigger wetlands, thus invoking a size effect. While the Kruskal-Wallis tests performed in the current study did not reveal statistically significant differences, in size, between wetlands belonging to the S and SS groups, intact wetlands were typically smaller than consolidated ones and also appeared to retain more P. Our results therefore partly agree with those of Cheng and Basu (2017), in that they hint at a potential combined influence of wetland geometry (size) and alteration status on P-sorption dynamics.

#### **4.5 Conclusion**

This study examined phosphorus dynamics in PWs by characterizing their water column SRP concentrations, along with their local hydrologic behaviour. Water column SRP concentrations were compared to soil variables related to P-sorption (i.e., EPC, PEBC, and  $S_{\max}$ ) to infer the temporal variability in the P-source versus P-sink behaviour of both intact and consolidated wetlands across a range of wetness conditions. Results showed considerable spatial and temporal variability of wetland water SRP concentrations, even among proximal wetlands.

The physical alteration of wetlands due to anthropogenic activities (e.g., wetland consolidation) was also found to impact the P-storage potential of PWs. However, this result needs to be interpreted with caution since only three consolidated wetlands were monitored. Our study highlighted the non-stationary behaviour of PWs with respect to P retention and uncovered a strong influence of wetness conditions on the prevailing P dynamics not only seasonally, but also on individual dates based on short-term weather patterns and local hydrology. Statistical analyses linking soil sorption properties and sink-source behaviour to wetland soil particle size, wetland soil carbon content, and landscape characteristics also suggested that P-sorption dynamics in prairie potholes may not be similar to those observed in other wetland settings. One critical take-home message of the current study is the need for future studies addressing the controls on P-sorption dynamics in the specific context of prairie potholes – to confirm or affirm the aforementioned results – and across a wide range of hydroclimatic conditions – to explicitly consider issues of non-stationarity. For example, here the source versus sink behaviour of individual wetlands was assessed based on bi-weekly SRP data, and while this approach provided critical information, it did not address the fact that the P-sorption status of individual wetlands can change within timespans that are much shorter than our chosen sampling frequency. To that effect, event-based analysis of wetland water P concentrations may be needed to enhance our understanding of P-sorption dynamics in the Prairie Pothole Region.

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## **Chapter 5: Synthesis and Conclusions**

## **5.1 Summary of Major Findings**

The benefits of wetland restoration and conservation efforts depend on the selection of wetlands using specific scientific criteria (White & Fennessy, 2005). Furthermore, the identification of scientific criteria for wetland conservation and restoration should be dictated by wetland functions, notably hydrological and biogeochemical functions in the context of the present thesis. Since the establishment of such criteria requires a clearer understanding of the effects of wetlands on watershed processes (e.g., connectivity between wetlands and streams), this thesis aimed at enhancing our understanding of these effects. The initial analyses presented in section 1.11 suggested the possible existence of some hydrological processes that may promote wetland-stream connectivity. The analyses reported in section 1.11 also suggested that both water level and nutrient concentration (i.e., SRP) data may be used to predict wetland-stream connections. However, section 1.11 only focused on stream data to predict wetland-stream connectivity and did not discuss or consider the role or function of the individual wetlands that likely cause some of the patterns observed in the stream data. Identifying specific wetland properties that govern wetland function and, therefore, influence wetland-stream interactions, was necessary and made possible through the consideration of additional, wetland-specific data in Chapters 2, 3 and 4. The motivation for focusing on individual wetland behavior – and the wetland and landscape properties that influence it – was the generation of new knowledge and the identification of metrics that may prove useful to select priority areas for wetland conservation and restoration.

Chapter 2 discussed the hydrological dynamics of individual wetlands at seasonal and annual scales across a continuum of alteration status (i.e., intact, consolidated and drained) using sub-daily hydrometric data. It looked at the importance of surface and subsurface water storage dynamics in the Prairie Pothole Region by documenting the fill-spill dynamics and water-table

dynamics prevailing in individual wetlands, and by comparing wetland hydrological behaviour with stream dynamics. This chapter also included a broad correlation analysis to investigate the link between seasonal and annual wetland hydrological behaviour and landscape characteristics. One major finding from Chapter 2 was that the physical alteration of wetland catchments (i.e., through the consolidation of wetlands) increases fill-spill events, while fully drained wetlands lose their water storage capacity and thus no longer have a noticeable water storage memory. Chapter 2 also highlighted the nonstationarity (i.e., lack of consistency in time) of dominant hydrologic processes affecting wetlands at seasonal and annual scales, and the lack of consistent control exerted by wetland characteristics on wetland behaviour across different seasons and different antecedent moisture conditions. Building on the nonstationarity of dominant hydrologic processes, Chapter 3 focused on the short-term hydrologic response of individual wetlands, during rainfall-runoff events, to better understand wetland-stream dynamics. Chapter 3 not only focused on the temporal and spatial variability of wetland hydrologic response to individual rainfall events, but also the temporal variability of wetland-stream interaction illustrated by hysteresis patterns, and the spatial characteristics that seemed to influence short-term wetland hydrologic response. One major finding of Chapter 3 was a demonstration of the high variability of wetland hydrological behaviour from one rainfall-runoff event to another. Chapter 3 also revealed that climatic conditions, including antecedent storage conditions, have a strong influence on the hydrologic behaviour of individual wetlands and can even override the influence of wetland geometry and landscape properties on wetland hydrologic response. Antecedent storage and storage memory seemed to be the driving factor for short-term wetland-stream interaction. Chapter 3 hinted that the hydrologic behaviour of prairie pothole wetlands is highly nonstationary and difficult to predict

based on the wetland geometry variables and the landscape properties that were considered in the current thesis.

The biogeochemical function of wetlands is another important element to consider when evaluating wetland restoration and was the topic of Chapter 4. Indeed, Prairie pothole wetlands usually act as sinks, as opposed to sources, for nutrients like phosphorus and act as long-term storage vessels (Hauer, et al., 2002; Whigham & Jordan 2003). However, wetland biogeochemical function in general – and phosphorus storage, in particular – can be heavily influenced by wetland hydrologic behaviour (Mitsch & Gosselink, 2007). Wetland-stream connections through surface (i.e., spilling) or subsurface hydrological processes can transport phosphorus from wetlands to nearby streams, depending on the combined hydrological and biogeochemical conditions of wetlands (Hauer, et al., 2002). Therefore, Chapter 4 focused on the spatiotemporal variability of phosphorus concentrations in Prairie pothole wetlands while investigating the potential controls (e.g., climatic conditions, landscape characteristics, wetland soil physiochemical properties, and local hydrology) on source versus sink dynamics. Results from this chapter indicated that intact wetlands are more likely to store phosphorus, compared to consolidated wetlands. Chapter 4 also underlined that along with antecedent moisture conditions, wetland soil texture, storage volume and short-term water level fluctuations can be used to predict the source versus sink behaviour of individual wetlands. One thing to note is that the source versus sink behaviours of wetlands alluded to in this thesis are defined based on purely geochemical criteria at the individual wetland scale. As discussed earlier, PWs do not have visible surface connection with nearby stream or wetlands. Therefore, the role of an individual wetland as a P source does not necessarily translate to watershed-scale concepts of source areas and contributing areas that determine watershed-scale water and nutrient export. Another result worthy of note comes from Chapter 4 and indicates that

soil with higher sand percentages are better for P retention. Although it sounds counter-intuitive, it is possible that some of the sediments that were analyzed were sand-sized but had a mineralogy that makes them different than classical sand particles, in that they are more reactive in relation to P retention. Data presented in the current thesis were collected in two different years, one considered a dry year (i.e., 2013) and the other a wet year (i.e., 2014), which allowed me to compare different climatic conditions. However, additional years of data or data from similar climatic conditions in both years might have led to different conclusions than the ones presented in the present thesis.

## **5.2 Candidate Metrics and Criteria for Wetland Conservation and Restoration**

As discussed in Chapter 1, floodwater retention is one key benefit targeted by wetland conservation and restoration. The ability of wetlands to promote floodwater retention depends not only on individual wetland hydrologic responses (e.g., storage dynamics) but also on wetland-stream interactions characterized across a range of temporal scales (i.e., annual, seasonal, event). Therefore, some of the behavioral metrics introduced in the present thesis to characterize wetland hydrologic function (i.e., Chapters 2 and 3) could form the basis of one criterion or multiple criteria for wetland conservation or restoration. For instance, metrics such as the percentage of time during which a wetland was dry (i.e., %WetDry), or the percentage of time during which a wetland was half-full (i.e., %WetHFull), could form the basis of criteria to identify priority areas for wetland conservation and restoration. For example, if one wanted to reduce peak flows during a flood event in a given watershed similar to the Broughton's Creek Watershed, then one could decide to conserve in priority all existing wetlands that have a %WetDry metric value that exceeds 50%. Along similar lines, restoration efforts could target, in priority, former or historical wetlands which were known or thought to have a %WetDry metric value over 50%. That 50% threshold could be

chosen to ensure that the wetlands that are conserved or restored in priority are those that have the ability to collect excess water – because they are dry – for more than 50% of the time during the open water season, especially after the spring when rainfall-induced flash floods are possible.

A similar approach could be taken for metrics such as %WetHFull, to come up a criterion or multiple criteria for selecting priority wetlands for conservation and, thereby, achieve watershed benefits related to flood attenuation. For example, one could choose to conserve intact wetlands and restore drained wetlands that have %WetHFull metric values lower than 50%, thereby ensuring that the wetlands chosen for priority conservation or restoration are those which have half of their storage capacity – or more – available to receive new water during at least 50% of the time. One logical concern to have, with such an approach, would be that high-frequency water level data are not widely available for a large number of wetlands and, therefore, values of the %WetHFull and %WetDry metrics would not be possible to estimate from field-based, wetland-specific data. One alternative could be to infer the values of those metrics using surrogate or proxy variables to which these metrics are correlated. In the present study, since the %WetHFull and %WetDry metrics showed consistent, statistically significant correlations with multiple wetland geometry variables and landscape characteristics (see Chapters 2 and 3, and section 5.3), there would be an opportunity to extrapolate the values of those metrics for all wetlands in the BCW, this by deriving and then applying a regression equation expressing the value of each metric as a linear or a nonlinear function of one or multiple spatial variables.

Other metrics such as the percentage of time during which the wetland fullness did not change (%WetNC), the percentage of wetland fullness change compared to the initial value during an event (WLPercChange), the time elapsed between the beginning of a rainfall event and maximum wetland fullness during an event (LagToMaxWL, or the time elapsed between peak

wetland fullness and the end of creek response during an event (RecessDura), represent the magnitude and timing of hydrologic responses of individual wetlands during rainfall-runoff events. They reflect the ability of individual wetlands to collect and store water, thus attenuating runoff during heavy precipitation events. Wetlands with higher WLPercChange values will be able to collect more water, whereas wetlands with higher RecessDura values will be able to store water for longer periods of time after rainfall events. Therefore, these metrics could also form the basis of criteria to identify priority areas for wetland conservation and restoration. For example, for a watershed like the BCW, one could rank open water and drained wetlands based on their WLPercChange during precipitation events (i.e., wetlands with higher WLPercChange values will receive higher ranks, and vice versa). Therefore, to reduce the flood peak during a heavy precipitation event, based on the ranking, one could prioritize to conserve the top 25% of intact wetlands and restore the top 25% of drained wetlands. As explained in Chapter 2 and 3, wetlands with higher LagToMaxWL and RecessDura values can store flood water longer due to their long storage memory. Hence, a similar ranking approach could be taken for metrics like LagToMaxWL and RecessDura to develop criteria for selecting priority wetlands. For example, to reduce flood intensity, one could prioritize the conservation of intact wetlands and the restoration of drained wetlands that have the highest values for either or both of those two metrics.

For biogeochemistry-related watershed benefits (e.g., nutrient control), intact wetlands appeared to be more suitable for P-sorption and more likely to act as P-sinks, compared to consolidated wetlands. As discussed in Chapter 4, P-sorption behavior is also linked to soil properties such as percentage of sand, silt or organic carbon. Wetlands that showed P-sink behavior have less than 10% organic carbon content and more than 30% sand within the top 10 cm of their bottom sediment (Figure 4-7). Therefore, if one wanted to achieve watershed benefits related to

the capture of excess phosphorus, one of the criteria could be to conserve and restore, in priority, intact wetlands that have less than 10% of organic carbon or more than 35% of sand within the top 10 cm of wetland bottom sediments. It should, however, be noted that wetland landscape characteristics did not show any statistically significant correlation with wetland P-storage variables or wetland soil properties, thus making it difficult to extrapolate across the whole BCW and prioritize different conservation and restoration scenarios.

In conclusion, one novel contribution of this Ph.D. thesis is undoubtedly the proposition of a wide range of metrics of wetland hydrological function and wetland biogeochemical function and the identification of the metrics which may offer the greatest promise for establishing conservation and restoration criteria. Indeed, while there is a great range of metrics that can be derived when high-frequency water level data are available, it is important to acknowledge that such data are still scarce in the vast majority of the PPR. Hence, for practical use, these metrics need to be estimated through other commonly available or accessible variables such as landscape characteristics. The metrics showing the greatest promise for establishing conservation and restoration criteria are, therefore, those that can not only be linked to tangible watershed benefits but can also be correlated with wetland geometry variables or landscape characteristics. This way, those metrics can be estimated across a large number of Prairie pothole wetlands. It should also be noted that establishing effective criteria necessitates that not only a key metric be identified, but also that a threshold value of that metric be determined, above and below which different management decisions (in terms of prioritization of conservation or restoration) should be recommended. While this thesis provides interesting information to start identifying candidate metrics, it does not provide quantitative information for determining what the optional threshold values of individual metrics should be in order to maximize watershed-wide benefits. Such an

optimization exercise would, however, be possible using a flexible modelling framework. It is worth mentioning that while the simple scenarios presented in this thesis chapter focus on one metric or one criterion only at a time, future studies should consider multiple factors simultaneously to assess the criteria to achieve certain wetland benefits. For example, wetlands that are dry most of the time – because they function as recharge wetlands – may offer benefits for attenuating floods but may present a threat to water quality by allowing the transport of surface pollutants to groundwater.

### **5.3 Implications for Wetland Conservation and Restoration in the Broughton's Creek Watershed**

Following up on section 5.2, the idea was to choose one metric of wetland hydrological function and then explore several hypothetical wetland conservation or restoration scenarios based on different threshold values of the chosen metric. Indeed, the results reported in Chapters 2 and 3 revealed that some hydrological response metrics showed consistent, statistically significant correlations with multiple landscape characteristics (see examples in Table 5-1). Thus, as alluded to in section 5.2, correlations established based on a subset of wetlands could be extrapolated for the whole BCW through the use of a regression equation, where a wetland geometry variable or a landscape characteristic acts as the independent variable and the chosen metric is the dependent variable which is being predicted. Once that extrapolation has been done, different threshold values of the chosen metric can be explored, as each threshold value would represent a different scenario of wetland conservation or restoration. Of course, one should exercise caution and consider the slight regional differences that exist within the PPR before extrapolating the results of the present thesis beyond the boundaries of the Broughton's Creek Watershed.

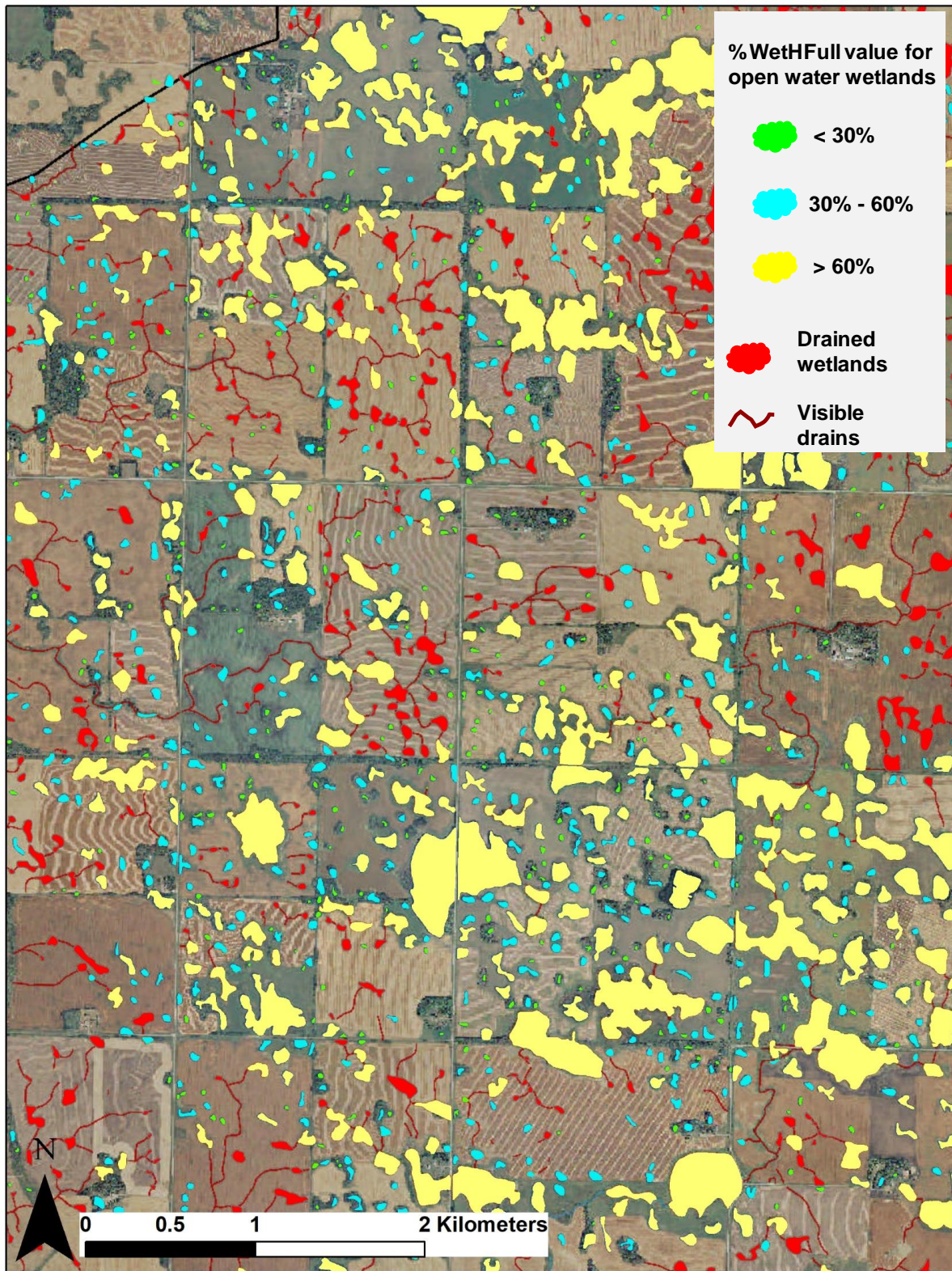
Here, solely for illustration purposes, an example was worked through with the %WetHFull metric as the dependent variable, and a wetland geometry variable as the independent variable. Using a regression analysis based on the wetlands for which high-frequency water level data were available, a relationship between %WetHFull and wetland area was established as follows (Eq. 3):

$$\%WetHFull = 22.315 \ln(\text{Area}) - 120.32 \quad (\text{Eq. 3})$$

**Table 5-1:** Examples of hydrologic response metrics and wetland landscape characteristics that were identified as significantly correlated (95% significance level) at the end of Chapters 2 and 3. For abbreviated wetland characteristics and response metric names, refer to Tables 2-1, 2-2, 3-1 and 3-3.

<b>Hydrologic response metrics</b>	<b>Landscape characteristics with significant correlation</b>
%WetDry	Area, Peri, Vol
%WetHFull	Area, Peri, Vol, ConArea, ConPeri, TotDrainArea, TotDrain2CatArea
%WetNC	Area, Peri, Vol, Elev, Cat2Vol, TotDrainArea,
LagToMaxWL	Cat2Vol, Distance
RecessDura	Cat2Vol, Distance
WLPercChange	Peri, Distance, SL

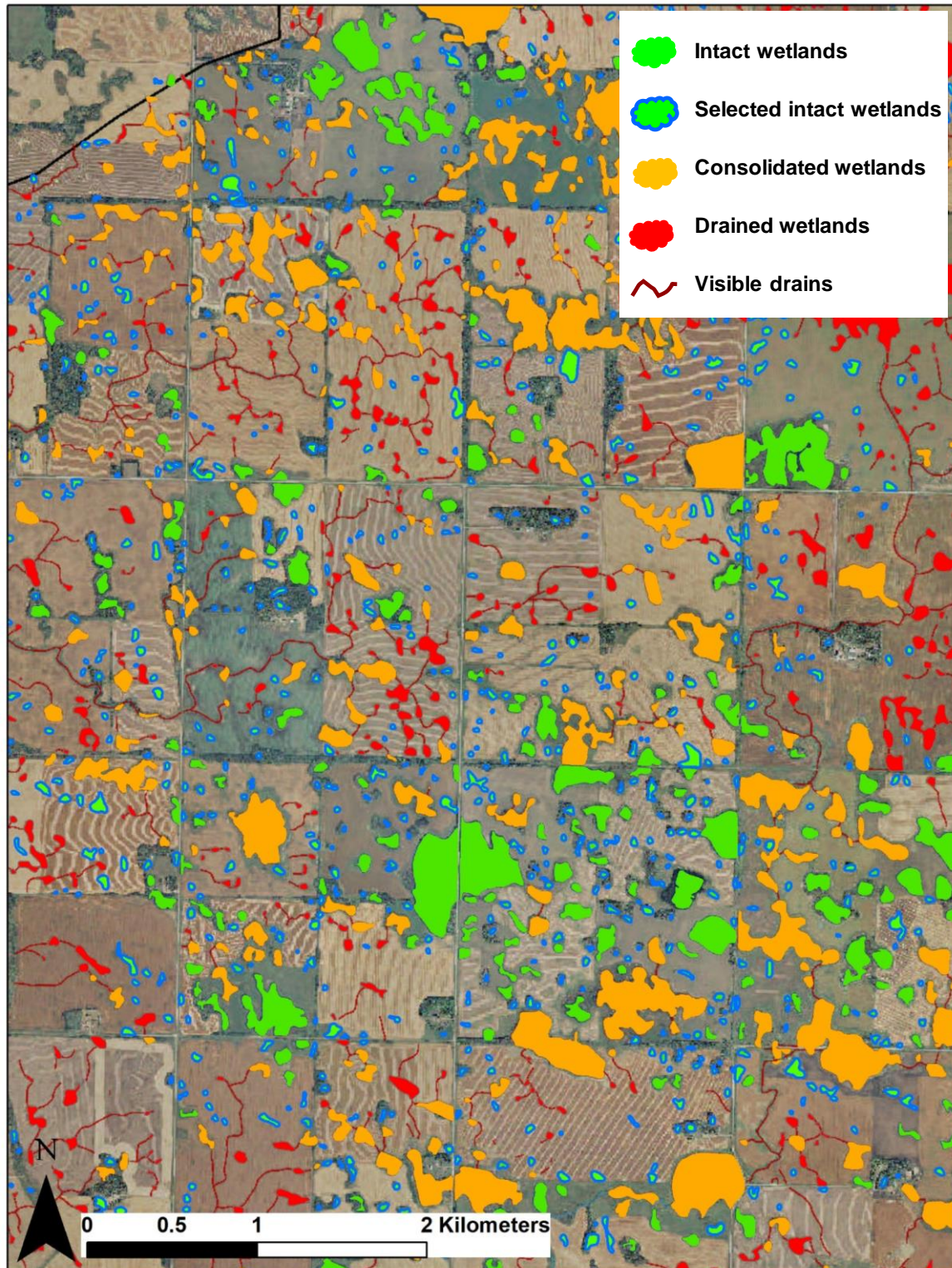
where Area is the surface area of each wetland, expressed in m<sup>2</sup>. That relationship was nonlinear and associated with a coefficient of determination, R<sup>2</sup>, of 0.53 – which indicates a moderately strong relationship. Based on the above equation (Eq. 3), values for %WetHFull metric were estimated for all the open water wetlands in the BCW. At the end of the extrapolation exercise, the average value for the %WetHFull metric across all the BCW intact wetlands was found to be 47.64%. For illustration purposes, the results of this extrapolation exercise for the northwest corner of the BCW are shown in Figure 5-1.



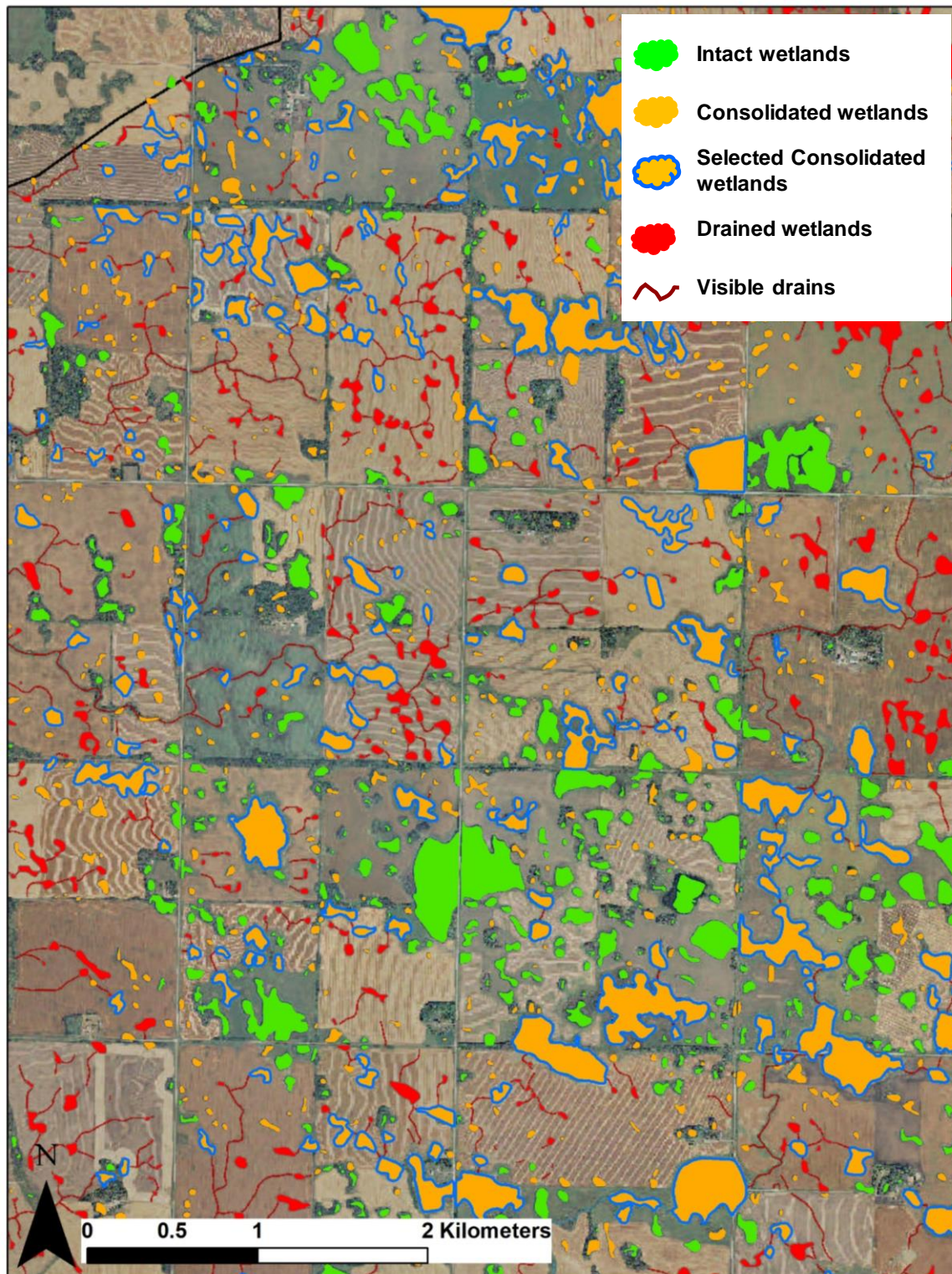
**Figure 5-1:** Spatial distribution of %WetHFull metric values for open water wetlands (i.e., intact and consolidated) in the northwest corner of the BCW.

One hypothetical conservation scenario could be to select, in priority, all intact wetlands in the BCW that have a %WetHFull metric value of less than 50%, in order to achieve watershed benefits related to the storage of floodwater. The outcomes of that particular scenario for the northwest corner of the BCW are illustrated in Figure 5-2. Most of the selected intact wetlands are relatively smaller, in size, compared to other intact wetlands: they are surrounded by agricultural land and far away from any stream network. This highlighted the fact that small wetlands pay an important role in providing watershed-scale benefits, and large individual wetland areas should not necessarily be targeted in priority for conservation. As discussed in Chapter 2, the monitored consolidated wetlands tended to be fuller than intact wetlands and also appeared more likely to spill during rainfall-runoff events (compared to intact wetlands). Therefore, one hypothetical restoration scenario could be to select, in priority, consolidated wetlands that have a %WetHFull metric value of more than 50% and restore them back to their “natural” or original status, in an attempt to increase the storage capacity of the landscape (see Figure 5-3 for the outcomes of that particular scenario). Selected consolidated wetlands for this restoration scenario are relatively larger, in size, compared to other consolidated wetlands. This seems to suggest that while small consolidated wetlands may remain in the landscape, larger ones need to be restored. Selected consolidated wetlands are located within agricultural areas, surrounded by drained wetlands and ditches and usually far away from any stream channels (Figure 5-3). Lastly, the same regression equation (i.e., Eq. 3) was also used to predict the %WetHFull value for drained wetlands in the BCW. Selecting drained wetlands with %WetHFull metric values of less than 50% could be another possible restoration scenario (see Figure 5-4). Drained wetlands selected for this scenario are typically associated with historic wetland complexes, surrounded by agricultural land and distributed across the whole BCW. Based on these conservation and restoration scenarios (Figures

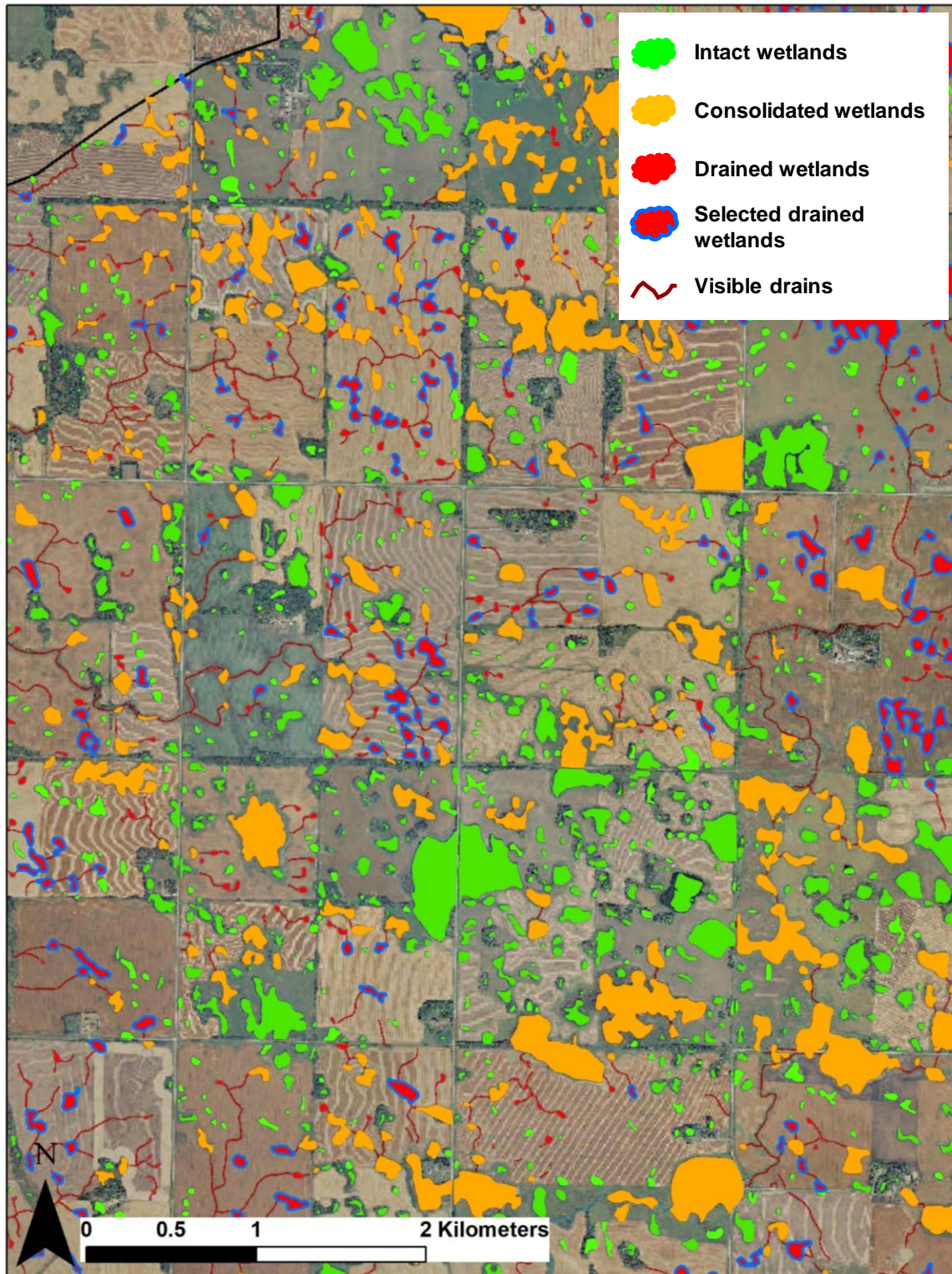
5-2, 5-3 and 5-4), one could suggest that priority be given to small intact wetlands for conservation and rather, larger consolidated wetlands and drained wetland complexes for restoration. It should be noted that the range of wetland geometry variables and landscape characteristics available for the 20 monitored wetlands described in Chapters 2, 3 and 4 was larger than the range of wetland geometry variables and landscape characteristics available across the whole BCW. Hence, while the hypothetical wetland conservation and restoration scenarios described in the present section were all based on spatial extrapolations from wetland area, it is worth reiterating that this Ph.D. research showed that wetland area is not the only surrogate measure of wetland function. Rather, other landscape characteristics were also found to play an important role for predicting wetland function and could be used for extrapolation purposes, provided that data on those landscape characteristics are widely available. It is important to reiterate that the scenarios presented in this chapter only considered certain wetland hydrological functions. The preservation of other critical wetland functions (i.e., ecological and/or biogeochemical) will require different scenario mapping, based on behavioural or system response metrics that are specifically related to those functions.



**Figure 5-2:** Outcomes of a hypothetical wetland conservation scenario, according to which intact wetlands that have %WetHFull metric values of less than 50% would be selected in priority. Results are shown only for the northwest corner of the BCW.



**Figure 5-3:** Outcomes of a hypothetical wetland restoration scenario, according to which consolidated wetlands that have %WetHFull metric values of more than 50% would be selected in priority. Results are shown only for the northwest corner of the BCW.



**Figure 5-4:** Outcomes of a hypothetical wetland restoration scenario, according to which drained wetlands that have %WetHFull metric values of less than 50% would be selected in priority. Results are shown only for the northwest corner of the BCW.

#### **5.4 Limitations of the Present Study and Recommendations for Future Studies**

While the reliance on high-frequency water level timeseries and bi-weekly water quality information does make this thesis data rich, the number of individual wetlands monitored was limited. Observations of about the effects of disturbance (e.g., consolidation) on wetland hydrology were also based on only three sites and were sometimes anecdotal, as opposed to being based on robust comparisons with high statistical power. Therefore, caution should be applied when extrapolating the findings from this thesis to a larger context. Additional datasets, if available, would have proven useful and helped confirm or infirm some of the hypotheses and interpretations put forward in Chapters 1 through 4. For instance, the availability of water table data below and in the vicinity of intact and consolidated wetlands, as opposed to below drainage ditches only, could have helped validate some of the assumptions regarding wetland-to-wetland and wetland-to-stream subsurface connectivity. Wetland vegetation surveys could have been useful to assess if evapotranspiration rates were responsible for the differences observed between intact and consolidated wetland fullness dynamics. The present thesis also did not consider spatial characteristics related to catchment slope, land use patterns, and wetland vegetation types when conducting correlation analyses to identify spatial controls on wetland hydrological and biogeochemical functions. The present thesis also did not consider the influence of vegetation growth on wetland hydrologic response during individual rainfall-runoff events. As for the source versus sink behavior of individual wetlands, it was only assessed based on bi-weekly SRP data and, therefore, negates the effects of P-sorption and mobilization processes that occur over timescales that are shorter than the two-week sampling interval. This thesis also only considered a single core per wetland to determine soil physiochemical properties and did not consider the role of wetland vegetation for P-sorption and mobilization processes.

In terms of future research avenues, simulation models have always been critical tools to assess wetland conservation and restoration scenarios. The wetland function metrics evaluated in this thesis could help refine the way simulation models are calibrated or validated towards assessing various scenarios for wetland conservation and restoration based on critical wetland functions. Indeed, modeling approaches to wetland restoration scenario evaluation have typically involved the over-simplification of wetland functions (Martinez-Martinez et al., 2014), and/or the placement of wetlands in positions where no wetlands were historically present (Newbold, 2005). These have limited the models' ability to predict potential, watershed-wide impacts of the restoration of pre-existing wetlands. Based on the data and correlation results presented here, future studies articulated around SWAT or other simulation models could be used for optimizing conservation/restoration scenarios based on physical criteria (such as the hypothetical ones explored in section 5.3) but also economic considerations. The datasets used in this thesis were not adequate to conduct cross-correlation analysis, thereby offering opportunities for future research to investigate the existence of temporal lags for the observed correlations. Investigating the role of soil Al and Fe content on P-sorption dynamics could be crucial for better understanding P-sorption dynamics in PWs and parameterize those processes into simulation models. Lastly, event-based analyses of wetland water P concentrations may be needed to enhance our understanding of P-sorption dynamics in the Prairie Pothole Region. Wetland drainage in Manitoba is regulated through different regulatory frameworks. From a policy development perspective, there is also a need to create links between typically used wetland classifications (such as Class 1 to 5), and a wetland function-based classification as evaluated through the function metrics in this thesis. Creating those links will help the management and implementation of regulations related to wetland drainage under different regulatory frameworks such the Water Rights Act.

## 5.5 References

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## **Appendices**

## **Appendix A:** Supplemental Materials Related to Chapter 2

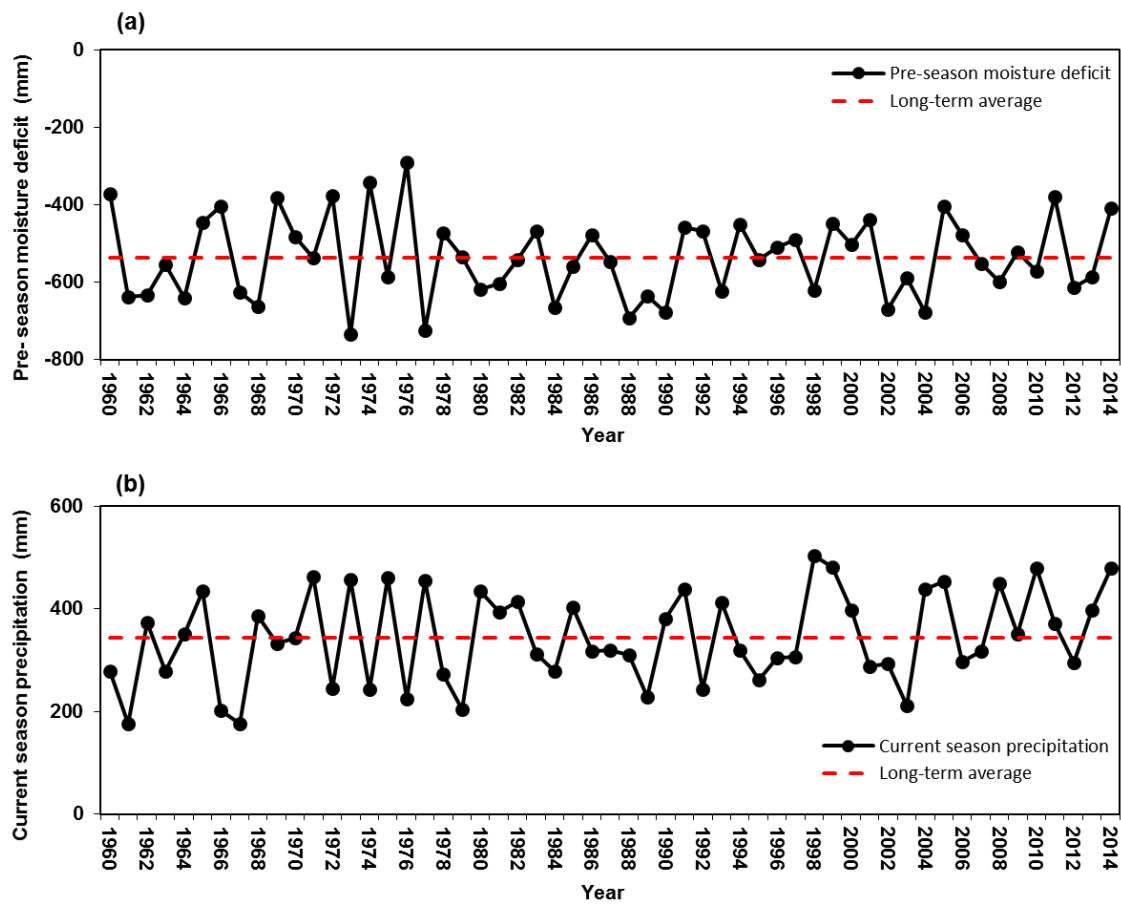
## **Abbreviations used in this appendix**

**I:** Instrumented intact wetlands

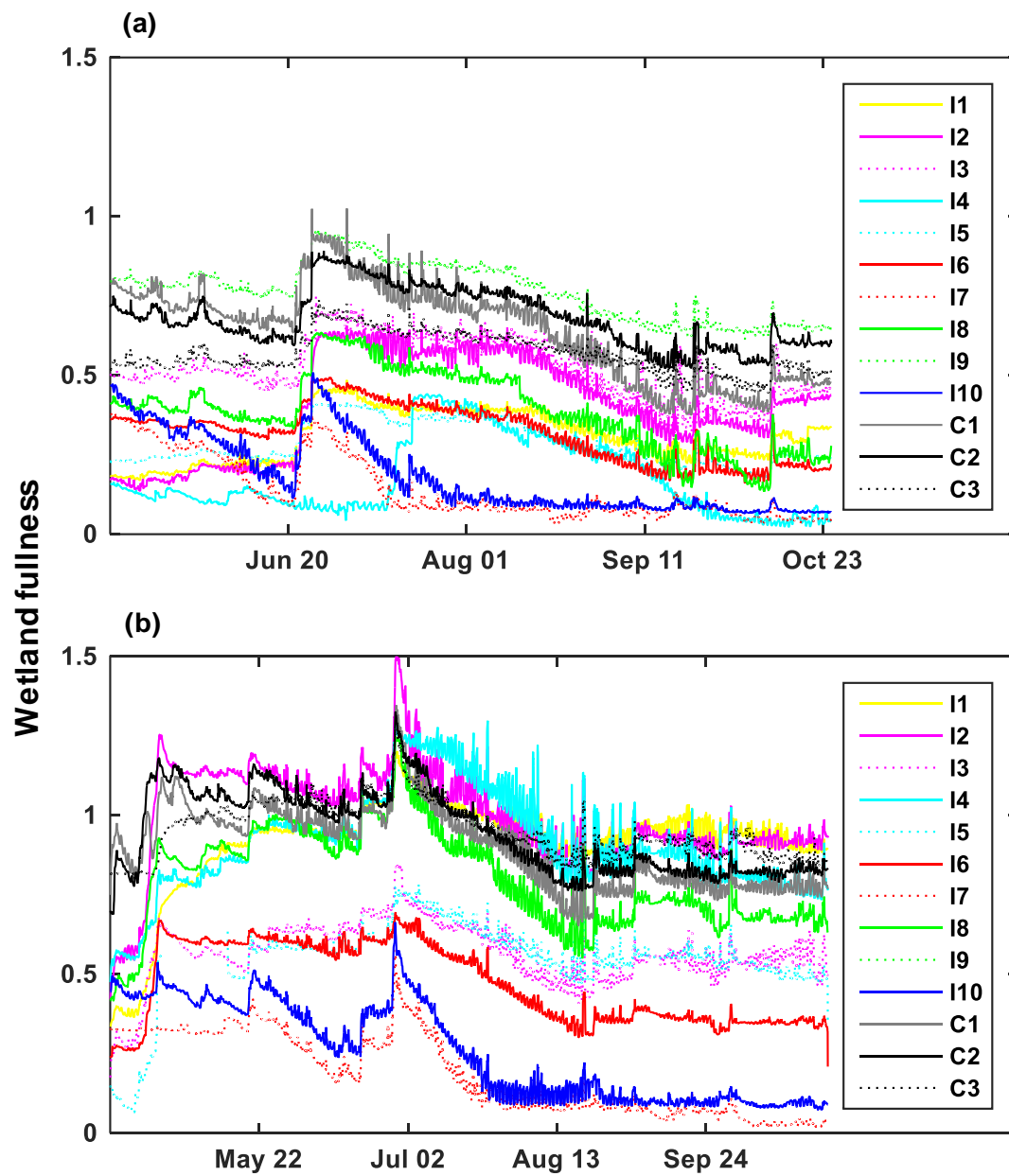
**C:** Instrumented consolidated wetlands

**D:** Instrumented drained wetlands and drainage ditches

**Appendix A-1:** Long-term historical data for (a) pre-season moisture deficit (computed as precipitation minus potential evapotranspiration, and (b) current season precipitation. In (a), pre-season values are computed between April 1<sup>st</sup> of the year preceding the one indicated on the x-axis, and March 31<sup>st</sup> of the year indicated on the x-axis. In (b), “Current season” refers to the open water season of the year indicated on the x-axis and ranges from April to October. To obtain the timeseries in panels (a) and (b), temperature and precipitation data were obtained from ENR (2017) (see manuscript for full bibliographic reference).



**Appendix A-2:** Hourly wetland fullness timeseries for (a) 2013, and (b) 2014.

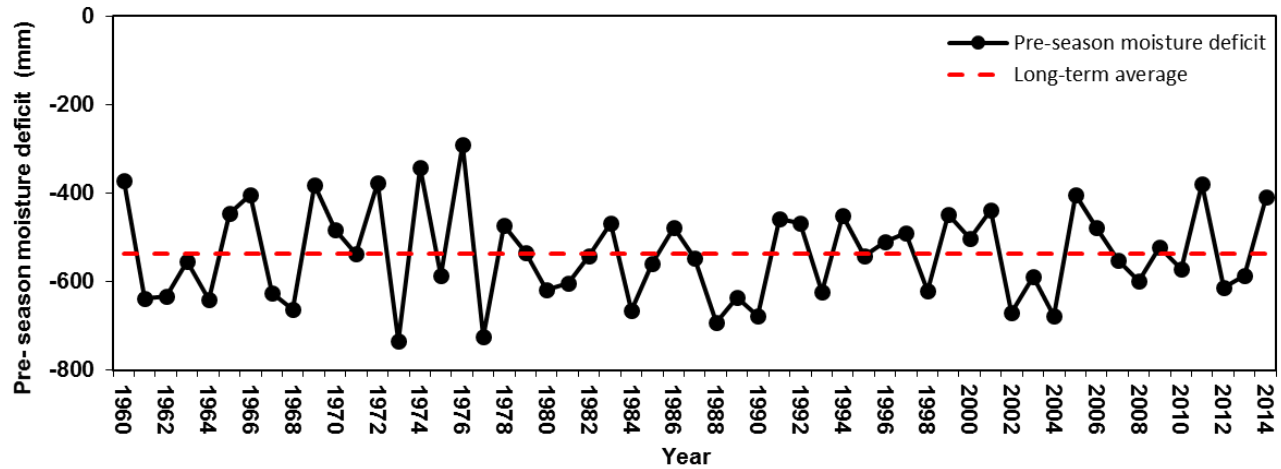


**Appendix A-3:** Spearman’s rank correlation coefficient values between creek water level and wetland fullness or ditch water table timeseries. The numbers “13” and “14” juxtaposed to season names (e.g., Summer13, Summer14) refer to the years 2013 and 2014, respectively. “X” signals a period for which no data were collected. Empty table cells indicate time periods for which the Spearman’s rank correlation coefficient was not statistically significant at the 95% level.

		Sites																			
		I1	I2	I3	I4	I5	I6	I7	I8	I9	I10	C1	C2	C3	D1	D2	D3	D4	D5	D6	D7
Time periods	2013	0.6	0.6	0.4	-0.2	0.5	0.3	0.1	0.3	0.4	0.1	0.3	0.4	0.5	0.4	0.6	0.4	0.4	0.4	0.4	0.4
	Spring 13	0.6	0.8	0.4	-0.2	0.5	-0.1	-0.3		0.2	-0.3	-0.2	-0.1	0.3	-0.4	-0.6	-0.2	0.5		0.2	-0.7
	Summer 13	0.7	0.6	0.7	-0.3	0.7	0.7	0.7	0.7	0.7	0.7	0.8	0.7	0.7	0.6	0.8	0.6	0.6	0.6	0.7	0.7
	Fall 13	0.3	0.3	0.2	-0.2	0.3	0.4	0.2	0.2	0.4	0.2	0.4	0.3	0.3	0.1	0.2		0.3	0.3	0.3	0.2
	2014	0.2	-0.1	0.0	-0.2	-0.1	-0.2	-0.3	-0.1	X	-0.3	-0.1	-0.1	-0.2	0.3	0.2	-0.2	-0.1	0.1	0.0	0.0
	Spring 14	-0.4	0.4	-0.1	-0.4	-0.4	0.4	0.5	-0.2	X	0.7	0.4	0.6	-0.4	0.6	-0.3	0.9	0.7	0.9	0.7	0.7
	Summer 14	0.5	0.4	0.4	0.3	0.4	0.3	0.1	0.4	X	0.2	0.4	0.4	0.4	0.7	0.5	-0.1	0.5	0.5	0.5	0.4
	Fall 14	0.3	0.3	-0.1	0.3	0.3			0.1	X	-0.1	0.2		0.3		0.4	0.2	-0.2	0.3	0.3	0.3

## **Appendix B:** Supplemental Materials Related to Chapter 4

**Appendix B-1:** Long-term historical data for pre-season moisture deficit computed as precipitation minus potential evapotranspiration. Pre-season values are computed between April 1<sup>st</sup> of the year preceding the one indicated on the x-axis, and March 31<sup>st</sup> of the year indicated on the x-axis. To obtain the timeseries temperature and precipitation data were obtained from [ENR \(2017\)](#) .



## References

ENR, (Environment and Natural Recourses, Government of Canada). (2017). Historical Data.  
 Environment Canada weather station (Brandon A, climate ID: 5010481, World  
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