Wetlands (2017) 37:773–785
DOI 10.1007/s13157-017-0913-5

ORIGINAL RESEARCH

Groundwater-Driven Wetland-Stream Connectivity in the Prairie Pothole Region: Inferences Based on Electrical Conductivity Data

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Received: 14 November 2016 / Accepted: 11 April 2017 / Published online: 20 April 2017
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Abstract This study examined the potential for electrical conductivity (EC) to serve as an indicator of groundwater-driven wetland-stream connectivity in the Prairie Pothole Region. Focus was on the Broughton's Creek Watershed (Manitoba, Canada) where thirteen wetlands and a creek were monitored in 2013–2014. A connectivity index (CI), computed by incorporating EC data in a hyperbolic solute export model, identified a potential for both shallow and deep groundwater-driven wetland-stream connectivity to occur, although shallower connections were rarer. Both raw EC and CI values were strongly correlated to wetland volume capacity, indicating the importance of storage and flow generation processes for wetland-stream connectivity potential. The proposed CI was instrumental in reaching that conclusion, making it a simple yet physically-based metric of wetland behavior that should be tested in multiple environments to confirm or infirm its validity.

Keywords Wetland-stream connectivity · Prairie Pothole Region · Groundwater · Electrical conductivity

Introduction

Glaciated Prairie landscapes across the north-central United States and the central plains of Canada are densely populated with depressional wetlands (Winter 1989; Tiner 2003) ranging in size from 1 m^2 to 100 km^2 (Hayashi et al. 2003; Van der Kamp and Hayashi 2009). Reducing the frequency and magnitude of flood waves (e.g., Brunet and Westbrook 2012) and retaining nutrients are some of the ecosystem services provided by those sloughs or Prairie potholes (LaBaugh et al. 1998) that drive efforts for their conservation and restoration (e.g., Yang et al. 2008, 2010). Wetland drainage has significantly reduced the density of pothole wetlands (PWs) in the Canadian portion of the Prairie Pothole Region (PPR) (National Wetlands Working Group 1988), and concern about the impacts of drainage on regional hydrology has led the Canadian provinces of Alberta and Manitoba to develop policies aimed at restoring and retaining wetland function. Successful policy implementation however requires an understanding of how wetland characteristics influence their connectivity (or lack thereof) to downstream waters, which remains a significant knowledge gap (Cohen et al. 2016).

PWs are critically important for watershed hydrologic, sedimentological, chemical and biological connectivity. Hydrologic connectivity, in particular, is often defined as the degree to which water can move, unimpeded, from source areas to a watershed outlet (e.g., Pringle 2003). PWs

Electronic supplementary material The online version of this article (doi:10.1007/s13157-017-0913-5) contains supplementary material, which is available to authorized users.

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frequently act as disconnective features due to their "geographically isolated" character – i.e., the fact that they lack visible surface inlets or outlets, are completely surrounded by uplands (Tiner 2003), and are part of drainage systems with only intermittent connection (LaBaugh et al. 1998). The prevailing paradigm is that under normal conditions, i.e., in response to the 1-in-2 flood, most PWs do not contribute surface water to streams (Stichling and Blackwell 1957). Maps of non-contributing areas have been created for the PPR to identify landscape areas that act as closed basins and are isolated from main hydrographic networks (Godwin and Martin 1975; PFRA-Hydrology-Division 1983; Martin 2001). However, under wet conditions, the water storage capacity of PWs can be exceeded, creating temporary surface connections towards downgradient streams when spilling occurs (Leibowitz and Vining 2003; Winter and LaBaugh 2003; Spence and Woo 2006; Shaw et al. 2012). Such temporary surface connections are rare outside of the spring freshet period, notably because of the high infiltration capacity of soils outside of PWs, and high evapotranspiration rates that prevent overland flow from travelling over long distances (Hayashi et al. 2016). Narrow ditches, which were used historically to drain PWs, have also modified the configuration of some formerly closed basins which can now contribute surface runoff downstream under normal conditions (Leibowitz and Vining 2003). Several authors (e.g., Rains et al. 2016) have called for better quantification of the frequency, magnitude, timing, duration, and rate of water fluxes from PWs to downgradient waters, and several recent studies have focused on high-magnitude, low-frequency surface-water driven connectivity between wetlands and streams (e.g., "fill and spill" events; Phillips et al. 2011; Shaw et al. 2012; Pomeroy et al. 2014). However, less attention has been directed to lower magnitude but potentially higher frequency groundwater-driven connectivity, likely due to the difficulties associated with quantifying groundwater movement. Like most depressional wetlands, PWs can have groundwater recharge, discharge or flow-through functions, with the prevalence of one function over the others depending on landscape position and geologic setting (LaBaugh et al. 1998; Van der Kamp and Hayashi 1998; Hayashi et al. 2016). Groundwater is therefore a critical pathway via which soluble materials are transported between PWs and other waterbodies along local, intermediate and regional groundwater flowpaths (Toth 1999). In the case of intermediate and regional groundwater flowpaths, the establishment of connectivity between a given PW and a stream is determined by the comparison between flowpath distance and hydraulic conductivity, and hence it is timescale-dependent (Winter and LaBaugh 2003). Although it is difficult to say how often groundwater connections are activated, the fact that they are space and time-dependent makes it clear that PW dynamics should be characterized within a connectivity gradient (Leibowitz and Vining 2003). To that end, groundwater fluxes in and out of depressional wetlands have been estimated as a residual term in the water budget (e.g., Labaugh 1986; McLaughlin and Cohen 2013); although that method is sometimes inaccurate given the errors associated with the measurement of precipitation, streamflow, and evapotranspiration (LaBaugh et al. 1998). This raises the question of whether basic water chemistry data, including pH, conductivity or oxidation-reduction potential, could be used to infer water fluxes between PWs and streams.

Among the variables listed above, electrical conductivity (EC)[1] can potentially provide strong insights into groundwater-driven wetland-stream connectivity due to known salinity patterns across glaciated Prairie regions and their documented shift in response to climate cycles (Mushet et al. 2015; LaBaugh et al. 2016). Using water chemistry to infer the degree of groundwater influence on wetland dynamics was suggested decades ago (e.g., Boelter and Verry 1977; Ingram 1983; Siegel 1988), based on the assumption that the concentration of dissolved solids in groundwater is much higher than that in precipitation. Salinity concentrations for groundwater flowing through clayey-silty tills usually increase from recharge areas to discharge areas, a consistent pattern due to the weathering of carbonate and sulfide minerals in the till and the dissolution of these minerals in Prairie soils (Rózkowski 1969; Cherry et al. 1971; Grisak et al. 1976; Hendry et al. 1986; Arndt and Richardson 1989; Keller et al. 1991; Arndt and Richardson 1992, 1993). PWs located in groundwater recharge areas are therefore less saline than those located in groundwater discharge areas (Sloan 1972; LaBaugh et al. 1987). Salinity is also affected by seasonality and extreme weather events, with concentrations that can decrease significantly in wet periods or spike in dry periods due to deflation (wind erosion) or evapoconcentration (Rózkowska and Rózkowski 1969; Winter and Rosenberry 1995; LaBaugh et al. 1996). However, no standard EC-based index exists to infer wetland-stream connectivity. The overall goal of the current study was therefore to examine the potential for EC to serve as an indicator of wetland-stream groundwater-driven connectivity in a landscape where intact and human-altered PWs co-exist. Three specific research objectives were pursued, namely: (i) characterize the spatiotemporal variability of EC in surface and subsurface water in a typical PPR landscape; (ii) propose an EC-based connectivity index to evaluate groundwater-driven wetland-stream interaction; and (iii) examine whether wetland EC concentrations and inferred wetland-stream interactions relate predictably to landscape characteristics.

[1] Specific conductance, electrical conductance and electrical conductivity are terms that are functionally synonymous and often used interchangeably. Here we decided to use the term electrical conductivity for measures that were corrected to constant temperature of 20°C for comparison across seasons.

Study Site and Data Collection

The 260 km^2 Broughton's Creek Watershed (BCW) is located in south-western Manitoba, Canada (Fig. 1a) on a hummocky till plain with numerous potholes and small lakes. Soils throughout the watershed are mainly Orthic Black Chernozems (Udic Borolls in the U.S. soil classification), and land uses consist of agriculture (72%), rangeland (11%), wetland (10%), forest (4%) and others (3%). Between 1968 and 2005, nearly 6000 wetland basins, or 70% of the total number of PWs in the watershed, have been either degraded or totally lost due to drainage for agricultural expansion (Yang et al. 2008, 2010). Yang et al. (2010) and Dumanski et al. (2015) suggest that climate change and

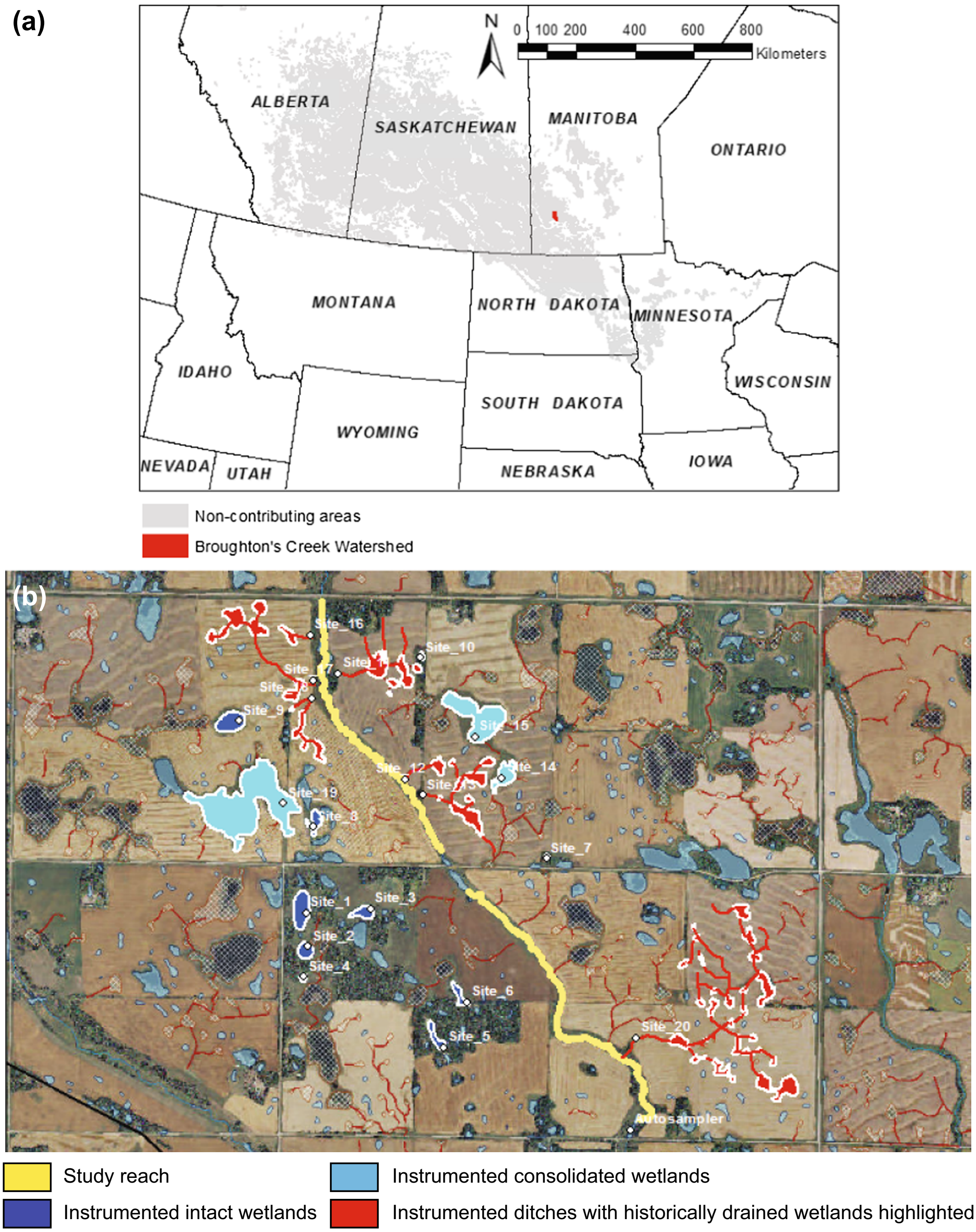


Fig. 1 **a** Location of the Broughton's Creek Watershed. **b** Aerial view of study area (image courtesy of Ducks Unlimited Canada) with wetland, ditch and creek ("Autosampler") sampling locations. On panel (**b**), north is up and the width of each section of land is a quarter mile

drainage activities have contributed to increases in peak discharge, water yield, and phosphorus export in the region.

For the current study, a 5 km-long creek reach was selected within the BCW. At the south (downstream) end of the reach, a battery-powered autosampler was used in 2013 and 2014 to collect composite streamwater samples every one (2014) or two days (2013). A capacitance-based water level logger was installed at the downstream end of the reach to record creek water level fluctuations every 15 min. An empirical relation with data collected at a downstream gauged location was used to convert water level measurements to discharge. On the lands adjacent to the reach, ten intact wetlands, three consolidated wetlands, and seven drainage ditches were monitored (Fig. 1b). While historical aerial photos reveal that the morphology of 'intact' wetlands was not modified by humans over the past 60 years, 'consolidated' wetlands result from two or more small wetlands that were re-routed to form a single, larger waterbody. Both intact and consolidated wetlands lack surface inlets and outlets. The seven ditch locations were selected based on current and historical maps showing that their role is to move runoff away from past wetland locations (that have since been drained) towards the creek reach under study. All monitored wetlands are (or were, before their modification) geographically isolated and thought not to contribute water to the creek in a 1-in-2-year flood (Martin 2001). Stilling wells (i.e., above-ground wells) equipped with capacitance-based water level loggers were deployed in the intact and consolidated wetlands to monitor stage fluctuations. Stage values were divided by each wetland average depth to obtain wetland fullness values (ranging from 0: dry wetland to 1: full wetland). Grab water samples were taken in all wetlands during 15 and 13 site visits in the 2013 and 2014 open water seasons (April–October), respectively, while subsurface water was collected below drainage ditches from nested piezometers installed at depths of 15, 45 and 60 cm. Here ditches were sampled because although their surface dynamics are ephemeral, they often appear to be wetter than their surroundings during field visits and were therefore assumed to be adequate sites for monitoring subsurface water flow paths. In total, 400 wetland water and ditch (subsurface water) samples and 338 creek water samples were collected over the study period. Upon collection, all samples were tested for electrical conductivity (EC) using a handheld water-quality pocket tester (Eutech Instruments Multi-Parameter PCSTestr™ 35). Daily climate data were obtained through nearby weather stations to assess the differences in antecedent conditions between sampling dates. The availability of 1 m–LiDAR data also allowed for the computation of a range of landscape characteristics, including wetland area and perimeter, storage volume capacity, area and perimeter of wetland catchment, catchment area to wetland area ratio, and wetland-to-stream flowpath distance (Table 1).

Data Analysis

To characterize the spatiotemporal variability of EC, maps showing wetland and subsurface water EC across a range of wetness conditions were built. Boxplots comparing EC concentrations in intact versus consolidated wetlands, at different depths below the drainage ditches and in the creek were also used, as well as scatter plots showing the co-evolution (or lack thereof) of wetland EC and creek EC for different months of the year. A log-log plot of creek EC versus creek discharge was produced to infer chemostatic, enrichment or dilution effects. Chemostatic refers to temporally invariant concentrations despite variable flow, while enrichment and dilution refer to concentrations that increase and decrease with flow, respectively (Godsey et al. 2009; Basu et al. 2010; Musolff et al. 2015).

To evaluate groundwater-driven wetland-stream interaction, an EC-based index of wetland-stream connectivity was developed based on the modification of the hyperbolic (or Hubbard brook) model often used for solute export (Johnson et al. 1969). Mathematical details can be found in electronic supplements (ES1). The connectivity index (CI) was formulated as:

$$\text{Connectivity index} = CI = \frac{C_{\text{Wetland}}}{C_{\text{Stream}}} - 1$$

where C is the EC value in stream or wetland water. The EC-based CI has a strong physical basis: indeed, high EC in streams can occur either when one of the source waters has high EC, or if mineral dissolution takes place while source water travels along slow-moving hydrological pathways, thus leading to higher EC as contact time of the water with the porous media increases. It must be noted that the concentration ratio ($C_{Wetland}/C_{Stream}$) does not measure actual wetland-stream connectivity but rather expresses a potential for it to occur if PWs intersect the groundwater contributing area to the stream; hence groundwater is the only end-member under consideration (see ES1). When $C_{Wetland} > C_{Stream}$ (or CI > 0), there is a potential for the wetland and the stream to be connected via shallow groundwater, provided that EC-rich wetland water is diluted while travelling along a shallow flowpath to the stream (Fig. 2, ES2). Such dilution could be the result of a rising water table (new soil water) and lead stream EC to be much lower than wetland EC. Regarding conditions when $C_{Wetland} < C_{Stream}$, two main processes can be invoked, namely streamwater evapoconcentration and mineral dissolution as wetland "source" water travels to the stream along deep groundwater flow paths. Given the strong dependence of evaporation on high air temperatures, stream and wetland water evapoconcentration (and the associated $C_{Wetland} < C_{Stream}$ conditions) should be highly seasonal in order to be deemed plausible. In the case of mineral dissolution, even though

Table 1 Landscape characteristics for the instrumented PWs

Site #	Wetland area (ha)	Wetland perimeter (m)	Wetland storage volume (m^3)	Catchment area (ha)	Catchment perimeter (m)	Catchment area to wetland area ratio (−)	Mean flowpath distance to stream (km)
1	1.78	556.68	7287.62	4.52	1210.00	2.54	13.62
2	0.69	301.52	2807.82	3.21	970.00	4.64	13.29
3	0.80	395.57	5025.33	4.92	1230.00	6.12	13.55
4	0.15	149.22	517.58	0.60	410.00	3.92	13.01
5	0.63	449.95	519.07	1.18	700.00	1.87	14.67
6	0.56	430.36	1646.34	2.57	950.00	4.60	14.71
7	0.07	105.26	74.64	1.26	630.00	16.82	13.76
8	0.74	508.41	2217.56	3.37	1180.00	4.55	14.18
9	1.67	483.80	2275.70	8.58	1770.00	5.14	16.24
10	0.10	130.45	77.96	4.36	1620.00	42.55	15.16
14	1.39	669.24	2447.65	5.90	1620.00	4.25	13.99
15	5.23	1223.11	27,588.98	26.14	3520.00	5.00	14.65
19	15.12	2823.39	43,926.51	31.27	4840.00	2.07	14.93

wetland water would initially have low EC (Fig. 2), its travel via slow-moving deep groundwater pathways could lead to the solubilisation of minerals (and hence, EC increase) on the way to the stream. Deep groundwater connectivity can therefore be associated with $C_{Wetland} < C_{Stream}$ ($CI \le 0$) as it is not the source (wetland) water that has high EC but rather the mineral dissolution process associated with long water travel times that results in high EC in streams. These hypotheses are examined in the current paper, with the recognition that the concentration ratio ($C_{Wetland}/C_{Stream}$) indicates a potential for shallow ($CI > 0$) or deeper ($CI \le 0$) wetland-stream connectivity depending on the depth at which PWs intersect the groundwater contributing area to the stream. To interpret the variability of the CI, descriptive statistics were computed and histograms produced. Kruskal-Wallis tests were also performed to assess the differences in timing (day of the year or DOY), wetland fullness and creek discharge between periods with shallow (positive CI) versus deep (negative CI) groundwater connectivity. The Kruskal-Wallis test was used here as it is non-parametric and does not rely on an assumption of normality for the data distribution (Sokal and Rohlf 1997). The null hypothesis was that the median values of DOY, wetland fullness and creek discharge are the same regardless of whether the potential for groundwater-driven wetland connectivity is via shallow versus deeper pathways. This null hypothesis was rejected when the probabilities associated with the Kruskal-Wallis test were smaller than 0.05.

Lastly, to examine the predictability of wetland EC concentrations and wetland-stream interaction as a function of landscape characteristics, Spearman's rank correlation coefficients (*rho*) were computed between EC concentration statistics, CI statistics and landscape characteristics. The Spearman's rank correlation coefficient was used as it does not assume normal distribution of the data nor the existence of linear relations between pairs of variables (Sokal and Rohlf 1997).

Results

Weather data showed that 2013 was a normal year while 2014 was wetter than normal – with larger amounts of snow water equivalent and rainfall (Fig. 3). In order for the EC-based CI to be used, wetland and streamwater samples needed to be collected in periods with no surface runoff. The two major peaks

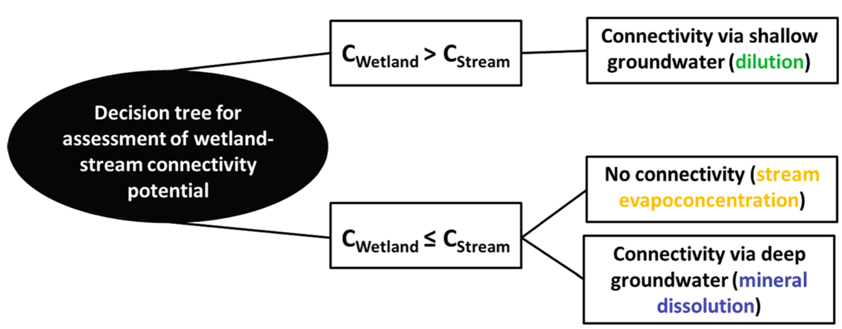


Fig. 2 Interpretation key regarding to use of stream and wetland EC data to assess wetland-stream connectivity potential

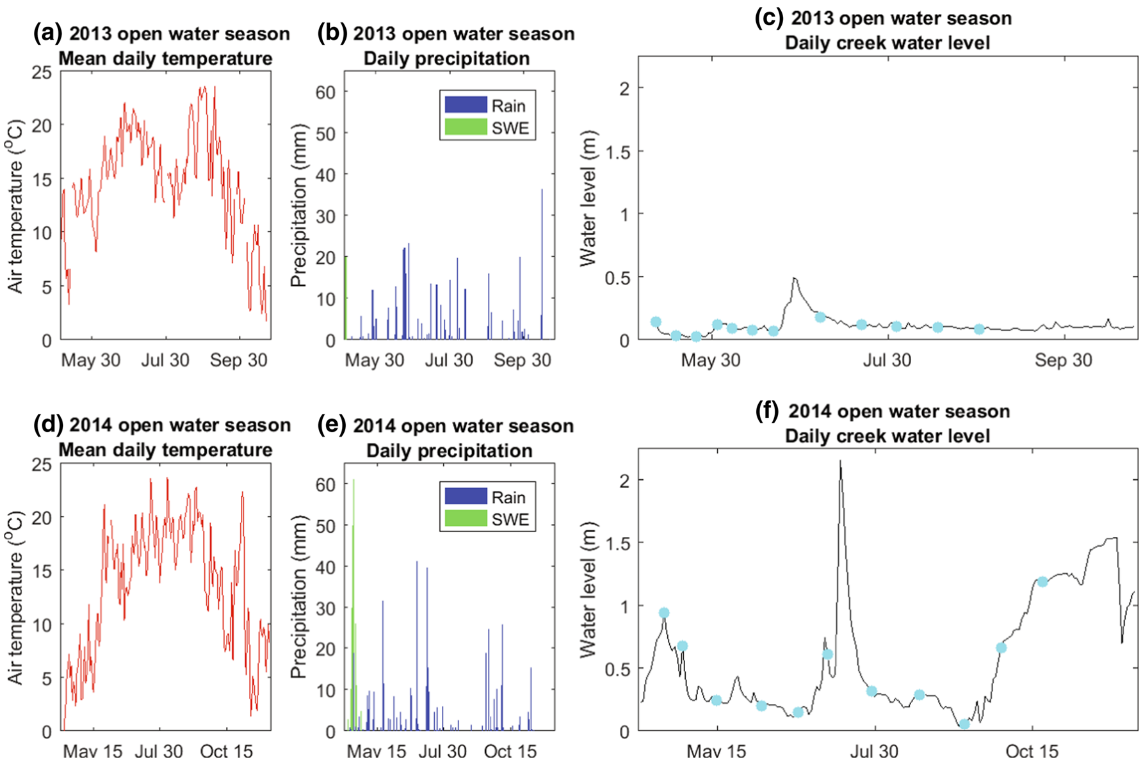


Fig. 3 Weather conditions and stream water levels during the 2013 and 2014 open water seasons. *Cyan dots* on panels (**c**) and (**f**) show water sampling dates considered for the current study

seen on Fig. 3c and f were associated with surface runoff and hence, samples collected during this period were discarded from the current analysis.

Spatiotemporal variability in water EC was present across the two monitoring years (Figs. 4 and 5). Subsurface water below the ditches draining a wetland complex (southeast portion of the study region) notably had EC ranging from a few hundred to more than 3000 μS/cm depending on antecedent wetness conditions (e.g., Fig. 4a, d). Temporal changes in wetland water EC seemed stronger in small wetlands (e.g., see three wetlands to the west of the creek in Fig. 4a through h). In general, subsurface water EC was more temporally variable than wetland water EC – as indicated by larger box and whisker extents in Fig. 5b, compared to Fig. 5a. Occasional surface runoff (or ditch ponded water) was the water source with lowest EC (Fig. 5b). A clear, nonlinear relation existed between streamwater EC and wetland water EC (Fig. 6). Seasonal differences were also clear (Fig. 6): both streamwater and wetland EC values were lowest in spring (below 1000 μS/cm in April at the onset of the spring melt), highest in dry summer conditions (e.g., August), and variable for the rest of the open water season. Month-to-month variability in streamwater EC dynamics was significant: the slope of the creek EC versus creek discharge relation in log-log space was sometimes close to zero (e.g., rightmost April points and July points on Fig. 7), which suggests chemostatic effects. The slope of that relation was however negative for the remainder of the time, indicating dilution effects.

CI values varied widely in time (DOY) and in space (across wetlands) (Fig. 8, Table 2). Some wetlands never had the potential to be connected to the stream via shallow groundwater, i.e., there was no sampling date with positive CI values for wetlands #4, 6, 7, 15. Wetlands #1, #2 and #3 – which are geographically close to one another (Fig. 1) but have very different wetland areas and storage volumes – had positive CI values for at least 58% of the sampling dates. The statistical distributions of the CI were moderately right-skewed for all wetlands (Fig. 8). All *p*-values associated with the Kruskal-Wallis tests were larger than 0.05, meaning that median values of DOY (or wetland fullness or creek discharge) were not significantly different when comparing periods with shallow (positive CI) and deeper (negative CI) groundwater connectivity potential.

Wetland size/storage was the most dominant control on wetland water EC. For instance, mean, median and maximum wetland water EC were correlated to wetland storage volume ($0.62 \leq rho \leq 0.7$) (Table 3). Both the storage volume and the catchment area to wetland area ratio were positively correlated with the variability (i.e., coefficient of variation, standard deviation) of EC in individual wetlands (Table 3). The minimum EC value recorded in individual wetlands over the two monitoring years was negatively correlated to the flowpath distance to the stream (Table 3), while mean and median CI

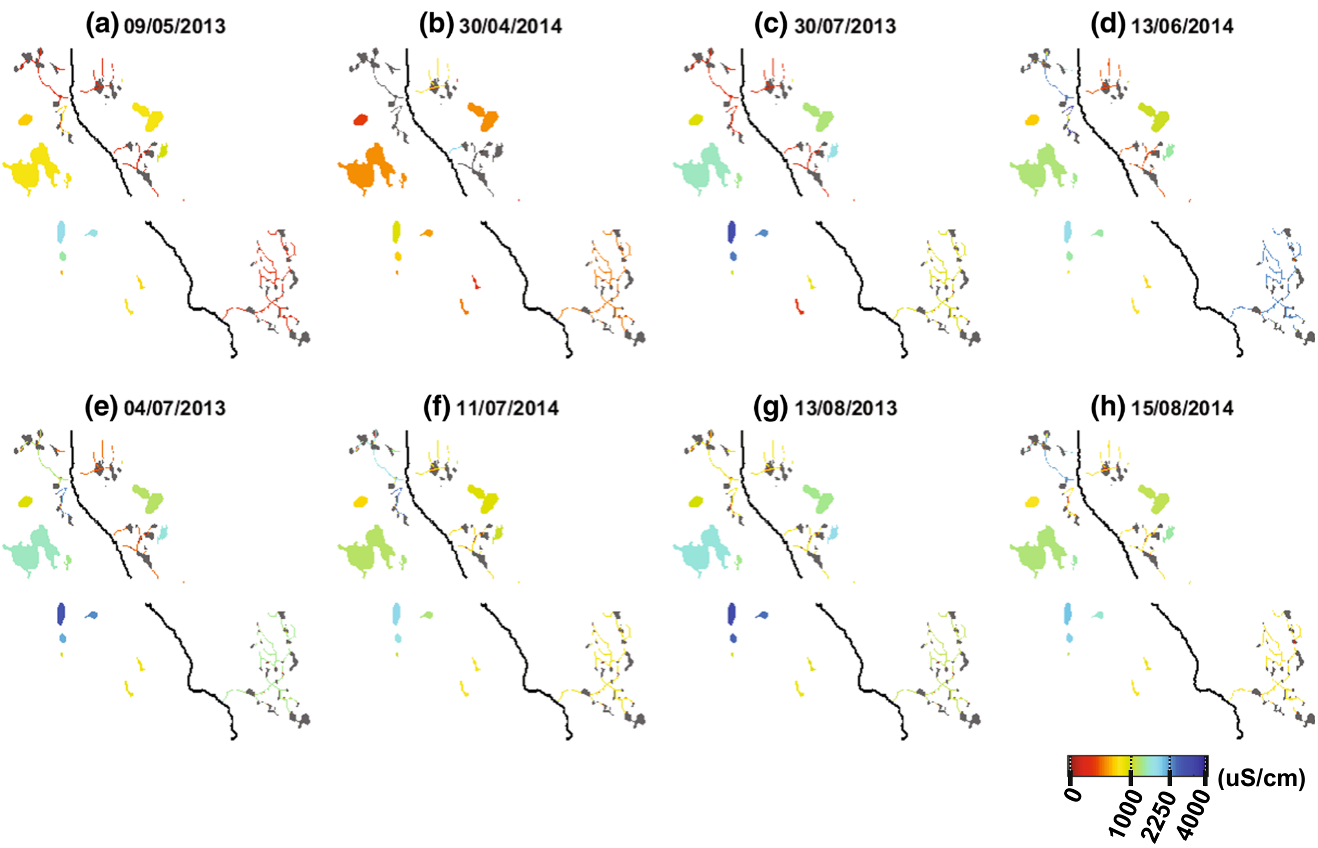
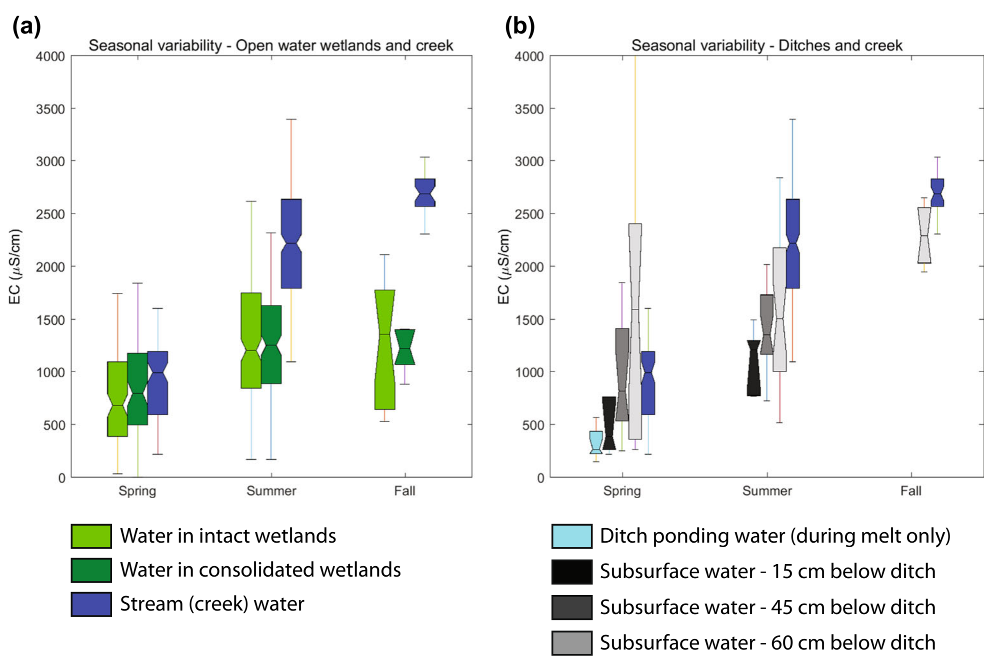


Fig. 4 Electrical conductivity (EC) in open water wetlands and below ditches (values averaged across the 15, 45 and 60 cm depths) for selected survey dates. **a** and **b**: spring melt; **c** and **d**: summer wet conditions; **e** and **f**: post-rainfall event; **g** and **h**: dry summer conditions. Note that the color-scale is nonlinear for better readability; the creek is shown in *black* while drained wetlands are in *grey*. For spatial scale and other spatial benchmarks, refer to Fig. 1b

values were positively correlated with wetland storage volume (Table 3). No landscape characteristic correlated with the percent time during which the potential for wetland-stream connectivity via shallow groundwater (CI > 0) was present.

Fig. 5 Seasonal variability of electrical conductivity (EC) in surface and subsurface water. Each box has lines at the lower quartile, median, and upper quartile values, while the whiskers show the extent of the remaining data (minimum and maximum). Outliers are not shown, but notches are drawn to provide a robust estimate of the uncertainty about the medians for box-to-box comparison



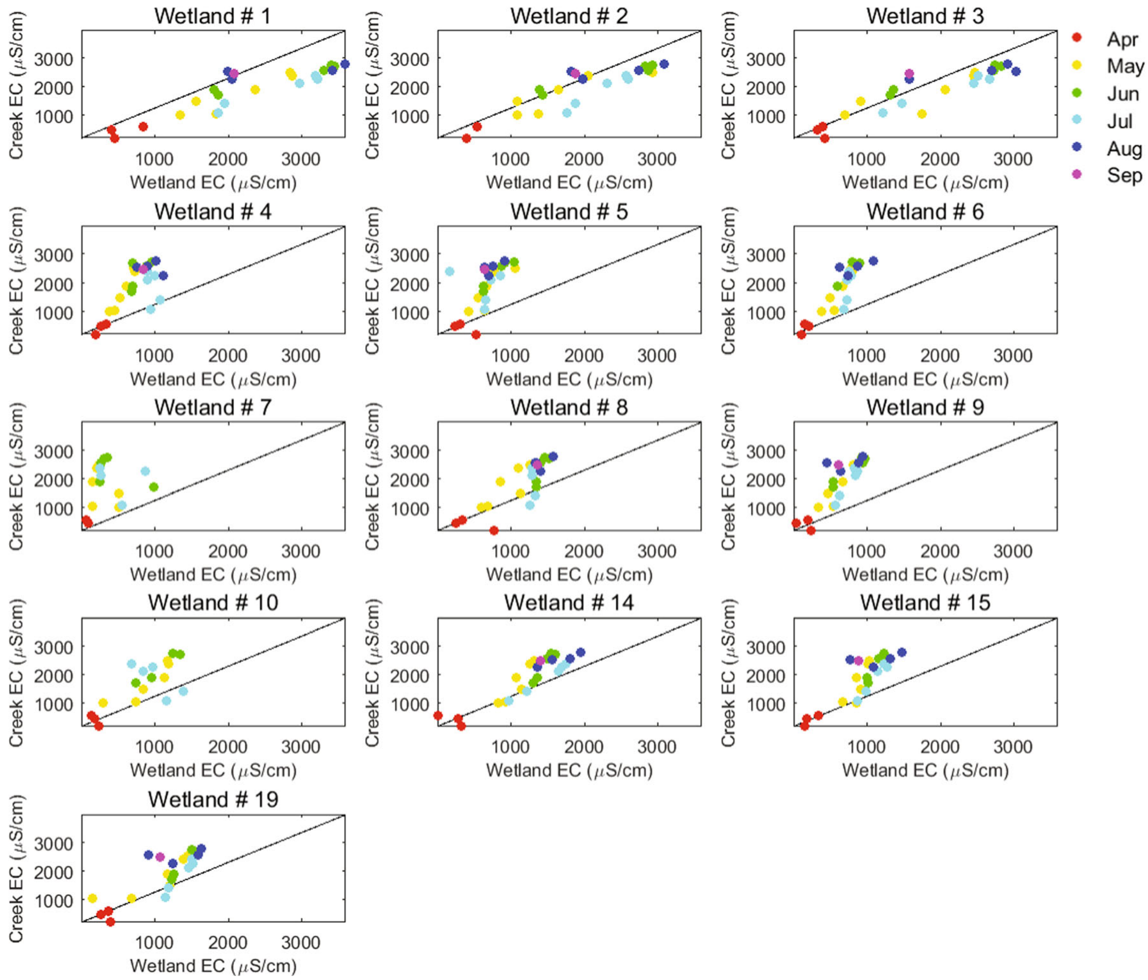


Fig. 6 Monthly variability in the relation between wetland and creek EC. On all plots, the *solid black line* is the 1:1 line

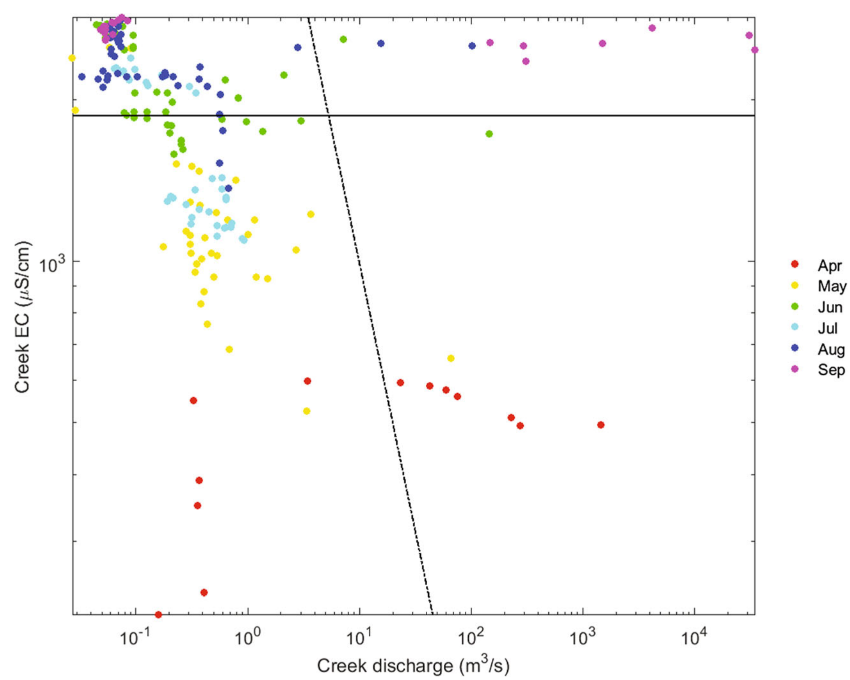


Fig. 7 Monthly variability in the relation between creek EC and creek discharge, in log-log scale. The *black solid* and *dashed lines* have slopes of zero and −1, respectively

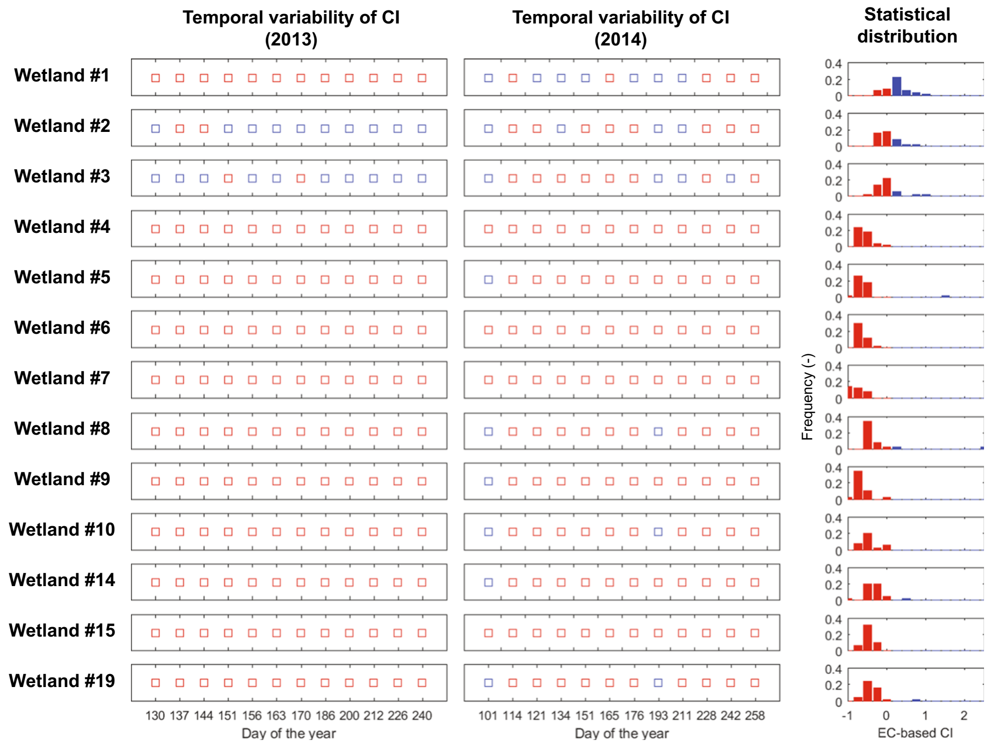


Fig. 8 Shallow versus deep groundwater-driven wetland-stream connectivity potential. For each row (or wetland), diagrams show, from left to right, the temporal variability of the CI in 2013 and 2014 and its overall statistical distribution across the study period. Negative and positive CI values are shown in *red* and *blue*, respectively

Discussion

Spatiotemporal Variability of Water EC in a Typical PPR Landscape

EC values observed across the study region (Fig. 4) are aligned with those reported by Barica (1975) and LaBaugh et al. (1998) – which ranged from a few hundred to 10,000 µS/cm for PWs in Manitoba. In this province, the EC guideline for water flowing through or adjacent to agricultural fields is 1500 µS/cm (Williamson 2011): it was frequently exceeded by the collected samples (i.e., in all seasons for subsurface water below drainage ditches; in summer and fall for surface water in intact and consolidated wetlands, see Fig. 5). Spatial patterns of EC were relatively consistent between 2013 and 2014, with the highest and lowest values almost always found in the same geographic areas (e.g., Fig. 3). Based on work by Sloan (1972) and LaBaugh et al. (1987), this suggests that major groundwater discharge (high EC) and recharge (low EC) areas do not vary much, interannually. Figure 6 shows a relation between creek EC and wetland EC, although the slope of that relation changes slightly not only between wetlands but also between months for individual wetlands, a clear indication of temporally variable flow generation mechanisms. Streamwater EC was generally higher than wetland EC, except in spring, indicating possible mineral dissolution in fall and winter and dilution in spring. The creek EC versus creek discharge relation (Fig. 7) also showcased changes between near-chemostatic behaviour – when similar concentrations are recorded despite highly variable discharge – and episodic behaviour – when both concentrations and discharge vary (Godsey et al. 2009; Basu et al. 2010), providing additional evidence of temporally variable water and salt export dynamics. The existence of dilution effects was confirmed by a pattern of decreasing stream EC with increasing stream discharge in April and May (Fig. 7), thus supporting some process assumptions highlighted in Fig. 2.

Table 2 Statistical summary of CI values for all PWs. CV: coefficient of variation

Site #	Mean CI	Median CI	CV of CI	% time with CI > 0
1	0.26	0.28	0.85	79.17
2	0.06	0.06	0.20	60.87
3	0.00	0.04	0.01	58.33
4	−0.56	−0.63	−3.18	0.00
5	−0.54	−0.64	−1.26	4.17
6	−0.63	−0.66	−6.51	0.00
7	−0.78	−0.87	−4.63	0.00
8	−0.26	−0.43	−0.42	8.33
9	−0.63	−0.66	−3.44	4.17
10	−0.45	−0.52	−1.78	11.11
14	−0.31	−0.30	−1.24	4.17
15	−0.47	−0.48	−3.53	0.00
19	−0.34	−0.38	−1.12	8.33

Table 3 Spearman's rank correlation coefficients between statistical summary parameters of EC concentrations and the CI and landscape characteristics. Min: minimum; Max: maximum; Std: standard deviation; CV: coefficient of variation. "n/s" signals pairs of variables for which the correlation coefficient was not significant at the 95% level

	Wetland area (ha)	Wetland perimeter (m)	Wetland storage volume (m^3)	Catchment area (ha)	Catchment perimeter (m)	Catchment area to wetland area ratio (−)	Mean flowpath distance to stream (km)
Min EC in wetland	n/s	n/s	n/s	n/s	n/s	n/s	−0.59
Max EC in wetland	n/s	n/s	0.67	n/s	n/s	n/s	n/s
Mean EC in wetland	n/s	n/s	0.70	n/s	n/s	n/s	n/s
Median EC in wetland	n/s	n/s	0.66	n/s	n/s	n/s	n/s
Std of EC in wetland	n/s	n/s	0.62	n/s	n/s	n/s	n/s
CV of EC in wetland	n/s	n/s	n/s	n/s	n/s	0.64	n/s
Mean CI	n/s	n/s	0.59	n/s	n/s	n/s	n/s
Median CI	n/s	n/s	0.70	n/s	n/s	n/s	n/s
CV of CI	n/s	n/s	n/s	n/s	n/s	n/s	n/s
% time with CI ≥ 1	n/s	n/s	n/s	n/s	n/s	n/s	n/s

Suitability of an EC-Based Index for Wetland-Stream Connectivity Assessment

Using EC as an indicator of active groundwater flow pathways is not new. Focusing on vernal pools, Leibowitz and Brooks (2008) suggested that specific conductance values exceeding precipitation values indicate groundwater contributions. While working on closed basin depressions in Alaska, Rains (2011) used several criteria to classify the inter-relation between wetlands and groundwater as "perched" or "flow-through", including a 20 μS/cm threshold. Hayashi et al. (1998) and Van der Kamp and Hayashi (2009) highlighted the role of groundwater flow direction in determining whether salts accumulate in a wetland or rather leach out of it. LaBaugh and Swanson (2004) also argued that EC can effectively be used to assess the relative position of wetlands on local groundwater flowpaths. Although the EC-based index of groundwater-driven wetland-stream connectivity suggested in this study cannot be used when surface runoff is present, it is robust given its derivation from a solute export equation – the hyperbolic model – and its reliance on parameters – the mean water residence time and end-member concentrations – that are physically based. Easiness of computation makes the CI an attractive metric that could be applied across wetland landscapes where the contrast in salinity between surface and subsurface water is important. The wetland-specific CI statistical distributions presented are also plausible: shallow groundwater-driven connectivity was never established for some wetlands and established only in wet conditions for others (Table 2). The absence of seasonal patterns in the occurrence of $C_{Wetland} < C_{Stream}$ (CI ≤ 0) (Fig. 8) makes it less likely for those occurrences to result from seasonally-driven streamwater evapoconcentration. As the hypothesis of mineral dissolution along deep groundwater pathways seems most plausible for CI ≤ 0, future studies should examine whether the absolute value of negative CI can be predicted based on the composition and heterogeneity of geologic layers, and can be used as a proxy for deep groundwater travel times between wetlands and streams.

One drawback of the proposed CI is that it does not consider temporal lags in the establishment of groundwater-driven wetland-stream connectivity nor its duration. Indeed, the concentration ratio in the CI formula is based on EC values measured on the same day from both a wetland and a stream; it therefore illustrates the presence of a "current" wetland-stream connectivity potential. The ratio of same-day concentrations also assumes that the establishment of groundwater-driven wetland-stream connectivity leads to an "instantaneous" change in stream chemistry (i.e., change occurred in less than 24 h), which is an unrealistic assumption for some PWs located at great distances from the creek. The applicability of the proposed CI to stormflow conditions would be possible if data were available for EC concentrations in surface runoff: this could be achieved either via additional sampling in the field or by estimating EC in surface runoff via nonlinear regression, i.e., by fitting the hyperbolic solute export equation (see ES1) to creek EC and creek discharge data. Quantifying surface runoff EC would be especially important in dry periods when "salt rings" can form at the periphery of ponds (Nachshon et al. 2013): subsequent wet periods might lead wetlands to expand and flush these salts (LaBaugh et al. 2016), thus creating an increase in EC in wetlands during wet periods and making it possible for salt-rich surface water flow paths to exist. An additional weakness of the suggested CI is its reliance on EC concentration changes that are not process-specific. EC changes can indeed be seen as a response to several confounding factors, including precipitation timing, magnitude and intensity, evapotranspiration, the balance between surface runoff and infiltration, and potentially even biological activity. For instance, alkalinity generation through sulfate reduction in both wetlands and streams – when they act as storage zones – has the potential to influence EC (Heagle et al. 2007). This is especially true in the Prairies where the sub-humid climate gives rise to important hydrologic abstractions (e.g., evaporation, depression storage) and the relative importance soil frost and vegetation at wetland margins can impede or enhance depression-focused recharge (LaBaugh et al. 1998). During non-stormflow periods, the extent to which changes in wetland water EC are due to their position in groundwater flowpaths and not to biogeochemical transformations could be assessed by interpreting the CI in light of a wetland water mass balance and as well as a wetland solute (salt) mass balance – to investigate whether increases in salt masses are linked to increases or decreases in water volumes. Data collected weekly or biweekly for the current study were not sufficient to calculate such mass balances. The lack of statistical difference between CI values as a function of DOY, wetland fullness or nearby creek discharge may also be attributed to the fact that the sampling frequency used in this study was not fine enough to capture the timescales over which those variables influence groundwater-driven connectivity. High-frequency (sub-daily) timeseries of EC from other PPR sites would be critical to confirm or reject the validity of the CI proposed here, especially for moderately brackish and highly transient systems where salinity is likely affected by confounding processes. Alternatively, using a known conservative tracer such as chloride might help avoid confounding biological influences.

Predictability of EC and CI Values from Landscape Characteristics

Wetland storage volume and the catchment area to wetland area ratio were the only two landscape characteristics seemingly influencing wetland water EC (Table 3). This finding is aligned with conclusions from past studies which stated the relation between wetland water salinity and topographic position not to be statistically significant (e.g., Swanson et al. 1988; LaBaugh et al. 1998). Others have suggested that wetland water salinity might be related to wetland position within a fill and spill sequence, while acknowledging at the same time that surrounding land use might be more influential on wetland water EC than topography (Brunet and Westbrook 2012). Mean and median CI values were positively correlated to wetland storage volume, suggesting that large-capacity PWs tend to be associated with larger CI values than small-capacity PWs (Table 3). The negative correlation between minimum wetland water EC and the flowpath distance to the stream also supports the mineral dissolution hypothesis, whereby hydrologically distant PWs have lower EC and can only connect to the stream if they intersect deep groundwater pathways which will solubilize minerals and increase water EC while *en route* to the stream. The lack of statistically significant correlation between landscape characteristics and the percent time during which the potential for wetland-stream connectivity via shallow groundwater ($CI > 0$) was present is likely due to the fact that the current analysis did not allow the actual timing of physical transport to be quantified, but only hypotheses about it to be examined.

Despite the many confounding factors mentioned above, the CI offers an interesting alternative to practitioners interested in making structural connectivity assessments in landscapes with geographically isolated wetlands (GIW). Indeed, both the ecological and hydrological literature make a distinction between structural and functional connectivity assessments: the former focus on the physical adjacency of landscape elements that is thought to influence material (e.g., water, solutes) transfer, while the later focus on how spatial adjacency characteristics interact with temporally varying factors to lead to the connected flow of material (Bracken et al. 2013). Structural hydrologic connectivity therefore indicates potential water movement based, mostly, on physiography while functional hydrologic connectivity quantifies actual water movement. Based on these definitions, structural connectivity describes the necessary – though not sufficient conditions – for connectivity to occur based on landscape configuration. Hydrologic research to date has been successful at deriving measures of structural connectivity, for instance in the form of topographic indices, but much less so at dealing with data-hungry and effort-intensive methods to quantify functional connectivity (Bracken et al. 2013). Two issues arise in the context of groundwater-driven connectivity in GIW landscapes, namely the fact that: (1) functional elements such as water fluxes and travel times (Knudby and Carrera 2005) are difficult to quantify, and (2) topographic indices are not good predictors of subsurface water movement in flat or complex terrain. An index based on elements other than topography is therefore needed for structural connectivity assessments, and the EC-based CI could endorse that role. In addition to being neither data-hungry nor effort-intensive, the CI is an indicator of geological conditions since its values can be interpreted in terms of soil-water or rock-water contact time. Hence, while the CI does not allow the quantification of actual (functional) connectivity, it helps identify landscape regions where geological conditions suggest the existence of a porous medium-water contact that is necessary (though not sufficient) for connectivity to occur. Such a structural connectivity assessment would be critical in a management context to identify areas where the potential for groundwater-driven connectivity exists, thus helping to rationalize efforts and target critical areas where more detailed monitoring might be needed toward jurisdictional assessment.

Conclusion

This study aimed to quantify the potential for wetland-stream connectivity to occur, not via surface pathways as has been the focus of many previously published studies but rather via groundwater pathways. To that end, electrical conductivity (EC) was measured in wetlands and in a creek within a typical PPR landscape. A connectivity index (CI) computed from EC data identified a potential for shallow groundwater-driven wetland-stream connectivity to occur intermittently, while deep groundwater-driven connectivity was more common. And while the influence of several landscape characteristics on wetland dynamics was considered – including that of wetland size, catchment area and distance to the stream – only the wetland storage volume capacity was consistently correlated with wetland EC and CI values, highlighting the difficulty in identifying dominant physical controls on PW hydrological dynamics. Groundwater movement, in addition to surface spilling events, is therefore an important mechanism via which PWs can potentially connect to downgradient waters. The CI was instrumental in reaching that conclusion, and it is suggested it be tested in other environments and with tracers other than EC to confirm or reject its validity.

Acknowledgements Special funding was provided by Manitoba's Water Stewardship Fund and Environment Canada's Lake Winnipeg Basin Stewardship Fund. We acknowledge Mike Chiasson, Halya Petzold, Cody Ross, Samuel Bansah, Adrienne Schmall and Lauren Timlick for technical help. We are also grateful to Lyle Boychuk and Bryan Page for providing spatial data for the Broughton's Creek Watershed.

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