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Ecohydrological change following rewetting of a deep-drained northern raised bog

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Abstract

Restoration of degraded peatland ecosystems (by rewetting) is undertaken to bring back key ecosystem services. However, the restoration process can have a range of ecohydrological effects, due to the associated physical and biogeochemical disturbance. In the case of northern peatlands drained by large and deep ditches, the rewetting effects are relatively unknown. The raised bog Grande plée Bleue (1,500 ha) is one of the largest pristine bogs in the St-Lawrence lowlands in North America; however, it contained an old (>60 years), 750 m long, 3.5 m deep, and 8 m wide ditch. Rewetting of the area affected by the ditch was carried out by the construction of six dams at 40 cm elevation intervals and felling of all trees (with diameter at breast height >10 cm) within 30 m. Water table was restored to levels similar to intact bog reference sites, only at elevation differences up to 17 cm from the nearest lower dam, while rewetting did not affect pore-water chemistry. Five to 6 years post-rewetting, the cover of both pioneer mosses, and late successional mosses (*Sphagnum*) had not changed significantly compared with pre-rewetting. This may have been due to the presence of dense shrub cover. For more effective ecohydrological restoration, dams should be spaced at smaller elevation intervals (e.g., every 20 cm of elevation or less), to allow recovery of water table along the entire length of the ditch, and vegetation introduction using the moss layer transfer technique may accelerate *Sphagnum* recruitment, especially in the few first metres from the ditch.

KEYWORDS
ditch blocking, peatland restoration, *Sphagnum*, water chemistry, water table level

1 | INTRODUCTION

Globally, peatlands provide a range of vital ecosystem services including climate regulation, nature conservation, and water regulation (Bonn, Allott, Evans, Joosten, & Stoneman, 2016), despite accounting for a small percentage of all land cover (2.84%; Xu, Morris, Liu, & Holden, 2018). However, pressure from climate change, land management, and industrial development are all threats to their long-term sustainability (Joosten & Clarke, 2002; Poulin, Rochefort, Pellerin, & Thibault, 2004). In eastern Canada, most especially in the

Nomenclature for vascular plants: Marie-Victorin (1995) lowlands of the St. Lawrence River, significant losses and disturbances of peatlands have been found during the last decades (Pellerin & Poulin, 2013). These disturbances are largely related to agricultural and forestry practices, as well as urbanization and peat extraction. Drainage (by excavation of ditches) is a common operation used in any of these contexts; it is used to lower the naturally high water table level (WTL) found in peatlands (Holden, Evans, Burt, & Horton, 2006).

Drainage has a detrimental effect on several peatland functions (Préfontaine & Jutras, 2017; Zeitz, 2016). The lowering of the water table, which is related to drainage depth (e.g., Sarkkola et al., 2010), causes vegetation change as key moss species decline (e.g., *Sphagnum*). These can be replaced with more generalist plants, such as shrubs (Laine, Vasander, & Laiho, 1995). The drier conditions nearby the ditches can also encourage the growth of trees (Murphy, Laiho, & Moore, 2009). This can result in a loss of habitat for insects and birds (Grégoire Taillefer & Wheeler, 2010; Wilson et al., 2014). Vegetation change, that is, loss of *Sphagnum*, feedbacks on the formation of peat and the carbon sequestration function of peatlands. Additionally, as the peatland dries out, surface peat can begin to oxidize; thus, previously sequestered carbon can be lost to the atmosphere in the form of CO_2 and to watercourses as dissolved organic carbon (DOC; Rowson et al., 2010). Thus, there can also be impacts of drainage on water quality, through increased acidity, DOC, and nutrient concentrations, which can occur in drained peatlands and affect the water quality of receiving watercourses (Lundin & Bergquist, 1990; Macrae, Devito, Strack, & Waddington, 2013).

In recent years, the societal value placed on peatland landscapes in terms of the specialist habitat they provide along with other ecosystem services has led to the restoration of many drained peatlands (Bonn et al., 2016; González & Rochefort, 2019). Here, the primary aim is to raise the water table (by damming of ditches) to establish hydrological conditions similar to that of intact peatlands (rewetting), which can promote the re-establishment of specialist peatland vegetation. Materials used for dam creation, include peat, plastic tiles, wood, and concrete (Armstrong et al., 2009), and the design is often governed by the size of ditches: deep, wide ditches may require large wooden or concrete structures to support the mass of water, although blocking huge ditch systems with peat in tropical peat swamp forest is practised in Indonesia (Ritzema, Limin, Kusin, Jauhianen, & Wosten, 2014).

Rewetting can result in a range of ecohydrological effects, which can be related to the length of the period under drainage and intensity or depth of the drains (Laine et al., 1995; Parry, Holden, & Chapman, 2014). Rewetting effects may include changes to hydrology, water chemistry, and vegetation. As the water table rises, this can impact upon redox conditions and pore-water nutrient and carbon concentrations (Fenner, Freeman, Hughes, & Reynolds, 2001; Strack et al., 2008), with direct consequences on microbial cycling and emission of CO_2 and CH_4. Over a slightly longer time scale, vegetation succession should develop towards more specialist peatland communities (Hancock, Klein, Andersen, & Cowie, 2018).

However, there is some evidence to suggest that rewetting effects (e.g., on water chemistry) and success in raising the water table may be related to drain depth (Armstrong et al., 2009, 2010). Indeed, deep drained peatlands may not behave in the same way upon rewetting as shallow drained peatlands (Ritzema et al., 2014). However, most published studies of rewetting are on restoration of shallow drained (<1 m) peatland; in the United Kingdom, many peatland drains are around 50 cm deep (Holden et al., 2017; Holden, Gascoign, & Bosanko, 2007), whereas in cutover peatlands in Canada, drains are typically 70–100 cm deep (Price & Schlotzhauer, 1999; Rochefort, Quinty, Campeau, Johnson, & Malterer, 2003). In tropical peatlands, for example, in Indonesia, there are some restoration schemes involving blocking of large drainage canals several metres deep, with one study reporting partial success (Ritzema et al., 2014). Globally, there are few reports of rewetting effects from deep drained (>1 m) peatlands (Chimner, Cooper, Wurster, & Rochefort, 2017). Therefore, there is currently a lack of knowledge on the ecohydrological effects of rewetting of deep-drained (>1 m) peatlands.

Our experimental study was carried out in an ecological reserve soon to be given a protected status; thus, there were strong incentives to carry out restoration before obtaining the final status of a conservation site. This study here reports on the ecohydrological effects of rewetting of a large deep-drained ditch (750 m long, 3.5 m deep, and 8 m wide) on a raised bog in Quebec. First, we demonstrated the effect of trees on the water table, in a paired transect study, as part of the justification used for tree removal as a rewetting measure in our study. We then specifically measured the response of the water table, pore-water chemistry, and vegetation cover, from 1 year prior to rewetting, until 6 years post-rewetting.

2 | METHODS

2.1 | Study site

The Grande plée Bleue (GPB) bog is a 1,500 ha raised bog (Figure 1) located in Quebec, eastern Canada (46°46′56″N, 71°02′51″W; Lavoie, Colpron-Tremblay, & Robert, 2012). The GPB bog is one of the largest pristine, relatively untouched bogs in the St-Lawrence lowlands, and the Quebec government is in the process of giving it the status of "ecological reserve" (Hugron, Landry, Raymond, & Marcoux, 2013), equivalent to the level of protection of category Ia of the IUCN (Dudley, 2008). However, a major threat to the ecological integrity of the site was the presence of an old (>60 years, pre 1960), 750 m long, 3.5 m deep, and 8 m wide ditch (Figure 1; Hugron et al., 2013). Therefore, the authorities decided to undertake hydrological and ecological restoration of the area affected by this ditch before the site receives its official status.

Vegetation on the GPB bog is typical of an ombrotrophic peatland and consists of *Sphagnum fuscum* on hummocks, of *S. rubellum*, and of *S. magellanicum* on wetter lawns, whereas *S. angustifolium* is present in hollows. *Carex oligosperma* and *Eriophorum vaginatum* are found in some areas among the widespread ericaceous coverage, dominated

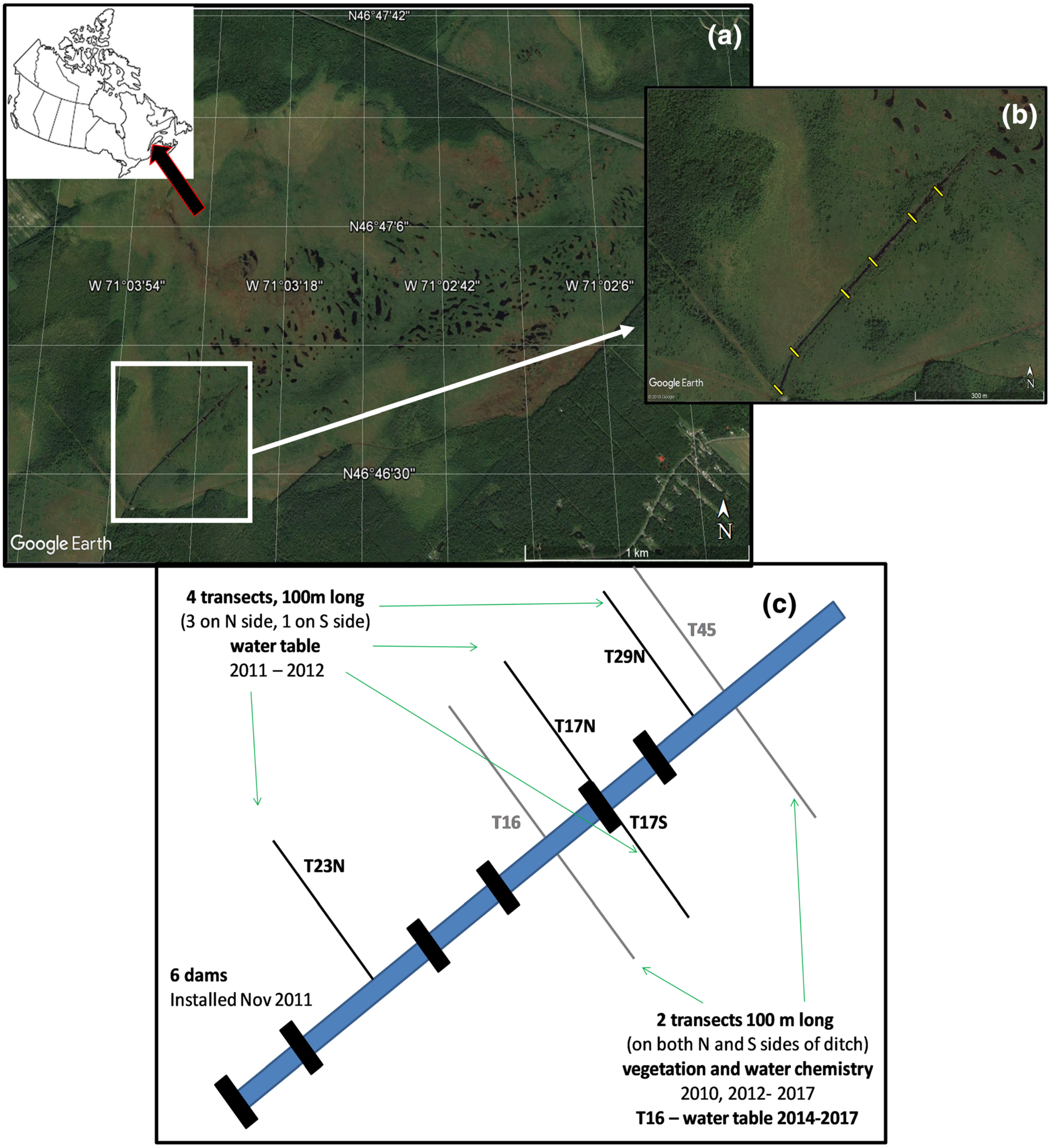

FIGURE 1 (a) The Grande plée Bleue (GPB) bog and (b) the large-sized ditch (yellow lines show position of the dams following rewetting). (c) Schematic of GPB drainage ditch, dams, and monitoring transects. The original water table level transects (T23N, T17S, T17N, and T29N) are named by elevation difference (cm) to the closest lower dam and N or S, whereas the other transects (T16 and T45) that cover both north and south sides are only named by elevation difference (cm) to the closest lower dam

by *Rhododendron groenlandicum*, *Kalmia angustifolia*, and *Chamaedaphne calyculata*. Thickets of black spruces (*Picea mariana*) or larches (*Larix laricina*), short in height (<5 m), small in diameter (<10 cm), and slowly growing (highest stems >150 years old) can be found sporadically on the site. However, within 25 m of the ditch, high and large trees were found in much greater abundance and diversity. This treed strip was characterized by an overstory of fast growing larches (10 to 15 m high, 20 to 40 cm diameter, and <50 years old) and black spruces (8 to 12 m high, 15 to 25 cm diameter, and <60 years old) crowning a dense, smaller sized (<8 m) paper birch (*Betula papyrifera*) understory. Tree stand density, evaluated as a basal area (i.e., the area covered by tree stems, measured from diameter at breast height [DBH]), could reach up to 28 m^2 ha^{-1} within 25 m of the ditch (M = 10.2 m^2 ha^{-1}; n = 12), but it never reached more than 3 m^2 ha^{-1} in the pristine part of the bog (M = 0.7 m^2 ha^{-1}; n = 12). Bryophyte cover under these treed surfaces was found to be lower than 11%, whereas in the pristine part of the peatland, this coverage averaged 70% (Hugron et al., 2013). Peat depth on the open peatland site varied between 0.5 to 4.5 m in places (Lavoie et al., 2012).

Mean annual temperature at the Lauzon meteorological station (<3 km distance; 46°45′18″N, 71°05′18″W), during the study period (2010–2017), was 4.8°C, whereas mean total annual precipitation was 1,104 mm, where 266 mm was provided in the form of snow (Government of Canada, 2018). During the study period, total annual precipitation ranged from 1,472 mm (2011) to 772 mm (2017).

2.2 | Rewetting activities

Rewetting of the raised GPB bog was initiated by installing dams on the deep ditch in November 2011 (Figure S1). For peatland restoration, it is generally recommended to install dams along ditches, every 30 to 40 cm of elevation change (Kozulin, Tanovitskaya, & Vershitskaya, 2010) to prevent erosion. Following this guidance, a topographic survey was carried out in spring 2011, and using the slope of the peat surface, it was calculated that six dams would be required to block this ditch at 40 cm elevation intervals, across an elevation change of 2.4 m (Figure 1c). The cost of building these dams was a factor in the choice of 40 cm intervals in this study. Two types of dam structure were constructed from larch lumber and metal pins during summer 2011. The first type used a 2.4 m high and 2.4 m wide hollow square box offering support to a 2.4 m high and 9.8 m wide solid lumber palisade. The second type was made of two palisades, 9.75 m large and 2.4 m high, attached together by 3 m long pieces of wood. The wooden structures were constructed firmly in a nearby parking lot and transported by helicopter directly next to the identified locations along the ditch in September 2011. Two lightweight tracked excavators were used to install the lumber structures, from November 8 to November 10, 2011. The structures were deposited in trenches excavated across the ditch to precise levels. Peat, excavated upstream of each dam, was used to tightly fill the structures and create 45° sloped sides upstream and downstream of the structures. Dams were constructed slightly higher than the surrounding peat surface (which likely subsided under drainage), so that water only flowed around them (in the subsided zone) in exceptionally high water levels (spring).

The other rewetting activity was tree harvesting. It was first carried out along a 50 m long section of the ditch in summer 2010; all trees were clear-cut on both sides of the ditch (birch, larch, and black spruce were all cut), to provide a reference area to monitor the influence of trees on the water table. A water table monitoring transect (T23N) was installed directly in the middle of the harvest area. Additionally, in 2010, all birch along the entire ditch were harvested, because it was considered an invasive species whereas larch and black spruce were not. From May 2225, 2012, all remaining trees with DBH >10 cm were clear-cut along the entire ditch (within 30 m distance). Tree stems were cut into four-foot sections and left on site.

2.3 | Ecohydrological monitoring

Monitoring of WTL, pore-water chemistry, and vegetation was carried out from 2010 to 2017, using a coordinated approach of transects of monitoring points at varying distances from the ditch edge (Figure 2). Transects were named by elevation difference to the nearest lower dam.

2.3.1 | WTL monitoring

WTL, defined as the distance separating the water table and the peat surface (lowest surface within a 2 m radius) was measured in wells (1.2- or 1.8-m long slotted plastic tubes, internal diameter = 3.8 cm), instrumented with water level loggers using capacitance technology (Odyssey™ water level logger; accuracy ±2 mm) or pressure transducers (HOBO U20; accuracy ±2 mm). Between June and November 2011 (pre-rewetting) and May and October 2012 (post-rewetting), WTL was monitored in four transects (T17N, T17S, T23N, and T29N; Figures 1 and 2). Transects were named by the elevation difference to the nearest lower dam in centimetres (the number in each transect name). Additionally, these four transects, which either had a north or

Year	2010	2011			2012	2013	2014	2015	2016	2017	2018
Hydrology	-	4 transects (T17N, T17S, T23N, T29N) with points 2,5,25,50,100 m				1 transect (T16) with points 2,5,25,50,100 m					
Water Chemistry	2 transects (T16, T46) with points 2 m and 100 m from ditch	-	D	C	2 transects (T16, T46) with points 2 m and 100 m from ditch						
Vegetation	2 transects (T16, T46) with points 2,5,10,25,50,100 m	-			2 transects (T16, T46) with points 2,5,10,25,50,100 m						

FIGURE 2 Monitoring timetable of hydrology (water table level), water chemistry, and vegetation. Grey shading indicates rewetting activities (D = dam installation, November 2011, C = clear-cutting, May 2012). The number in the name of each transect represents the elevation difference (cm) to the closest lower dam. Transects that only have a north or south arm are named N and S accordingly, whereas those that have both arms have no additional letter in the name

south arm, were named N or S accordingly. Each transect had WTL loggers at 2, 5, 25, 50, and 100 m from the ditch edge, thus allowing a comprehensive comparison of pre- and post-rewetting WTL. Additionally, these data were used to determine the effect of trees on WTL in 2011, as one transect in the middle of the 50 m clear-cut zone (T23N) was paired with another with dense tree cover (T17N). The loggers recorded WTL every 15 min, although due to the vast number of data generated, data were averaged to a 6-hr time step.

From 2014 to 2017 (between the months of May and November each year), WTL was monitored in one transect only (T16; Figures 1 and 2). Odyssey™ capacitance water level loggers were placed at 2, 5, 25, and 50 m from the ditch edge (50 m was already established as a "reference" distance for hydrology, which was not influenced by the drain), recording every 6 hr.

An automatic optic level and a staff ruler were used to measure the relative elevation of wells along the ditch. The difference between the relative heights of the lowest peat surface within a 2 m radius of each well and the top of the well was used to determine the peat-exceeding length of tube, for correction of raw WTL measurements. From November to early May each year, water level recorders were removed from the site to prevent damage to electronics or battery components caused by low winter temperatures (−35°C recorded in January 2012) and peat frost. Water level recorders were calibrated each year on site using pore-water, before installation.

2.3.2 | Water chemistry monitoring

Wells (slotted plastic tubes) were installed at 2 m (rewetted zone) and 100 m (reference zone) from the ditch edge for the monitoring of pore-water chemistry (Figure 2). Two transects (T16 and T45) were installed, each covering both the north and south sides of the ditch, giving a total of eight sampling wells (located in the centre of vegetation monitoring plots). Pore-water samples were collected in August 2010 (pre-rewetting) and annually from 2012 to 2017 (post-rewetting; every May) into plastic bottles using a syringe and plastic tubing.

Electrical conductivity (EC) and pH were measured in the field using a pH/EC/temperature meter (Hanna Instruments 98129), whereas samples for chemical analysis were frozen on return to the laboratory until analysis. Samples were analysed for Ca, Fe, K, Mn, Mg, Na, chloride (Cl^-), sulfate (SO_4^{2-}), ammonium (NH_4^+), nitrate (NO_3^-), phosphate (PO_4^{3-}), soluble phosphorus (P), and DOC. However, some parameters were not analysed every year (Table S1). Analysis of major and minor cations (metals) and P was carried out by ICP-OES using an Agilent 5110 SVDV, whereas Cl^-, SO_4^{2-}, NH_4^+, NO_3^-, and PO_4^{3-} were measured using a Quikchem 8500 Series 2 flow injection analysis system. DOC was measured using the high temperature catalytic combustion method (Shimadzu TOC V_{CSH}), and the limits of detection for all chemical analysis are given in Table S1.

2.3.3 | Vegetation monitoring

Vegetation monitoring was carried out on the same transects used for water chemistry monitoring (T16 and T45), although additional monitoring points were included in each transect (Figure 2). Vegetation was monitored at 2, 5, 10, 25, 50, and 100 m from the ditch edge at points marked with metal rods. At each monitoring point, a 12 m × 2 m plot was set up perpendicular to the transect, within which there were four circular quadrats, 70 cm in diameter, spaced equally apart (Figure 3). The percentage cover of vegetation strata (ericaceous, herbaceous, and bryophyte strata) was assessed by vertical projection in each quadrat. Here, all individual plant species were identified, and their percentage cover within the quadrats was estimated. Additionally, the percentage cover of different functional groups was calculated; these were pioneer mosses, late successional mosses, total ericaceous plants, total herbaceous species, and total trees (see Table S2 for classification of species). Functional groups were useful in describing the vegetation structure, although we acknowledge the caveat that some species within groups may have different moisture requirements, for example, ericaceous plants.

Vegetation monitoring was carried out during the growing season in 2010 (pre-rewetting) and annually from 2012 to 2017 (post-rewetting). In 2010, surveys were carried out in August and early September, whereas in all other years, surveys were carried out between May and August.

In 2010 and 2012, vegetation surveys were carried out at all monitoring points, whereas from 2013 to 2017, the monitoring effort

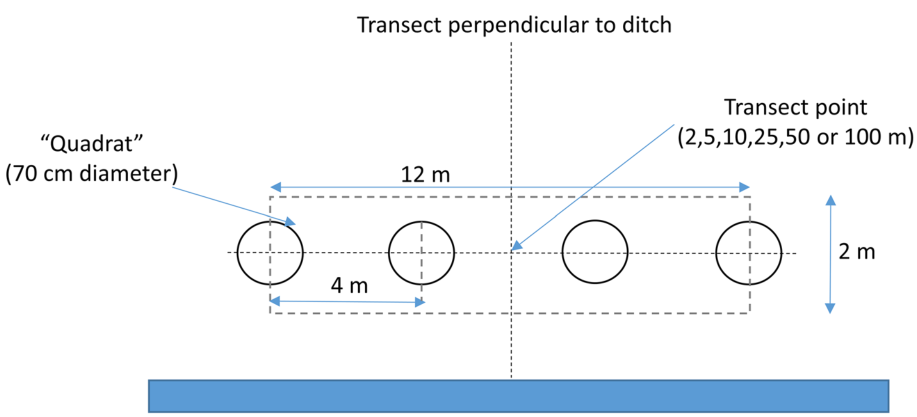

FIGURE 3 Method of vegetation monitoring at Grande plée Bleue

was reduced due for cost efficiency. In odd years from 2013 (i.e., 2013, 2015, and 2017), monitoring points at 5, 25, and 100 m from the ditch were surveyed, whereas in even years from 2014 (i.e., 2014 and 2016), monitoring points at 2, 10, and 50 m from the ditch were surveyed.

2.4 | Statistical analysis

All statistical analysis was carried out using RStudio, Version 3.3.3 (R Core Team, 2017). To answer our range of questions on the eco-hydrological effects of rewetting, we used a range of statistical techniques.

2.4.1 | Water table levels

WTL data were used to address three hydrology questions: first, the effect of trees on WTL; second, the short-term effect of rewetting (up to 1 year post-rewetting) on WTL; and third, the medium term (3–6 years) effect of rewetting on WTL.

To determine the effect of trees on WTL, data from paired transects T17N (uncut) and T23N (trees cut) from June 2011 to November 2011 were used, that is, the period prior to ditch blocking. Data from each transect, at each distance from the ditch edge (2, 5, 25, 50, and 100 m), were compared using a linear mixed effect model (function *lme* and package *nlme*, Pinheiro et al., 2016), with the factors cut/uncut and distance from ditch edge. The model included "date" as a random intercept with temporal autocorrelation accounted for using a first-order autoregressive structure (where successive observations are more closely correlated than those further apart). Due to a significant amount of missing data from the 5 m well on the cut transect (when WTL was lower than the bottom of the logger; −55.5 cm), WTL data were organized into bins. This strategy meant that all data could be included in the model. The WTL bins were either +10 to −20 cm (*good*), −21 to −50 cm (*low*), or lower than −51 cm (*very low*). Each bin was assigned an integer value, and WTL data were substituted with the appropriate integer.

To look at the 0- to 1-year effects of rewetting on WTL, data from the four transects of loggers (T17N, T17S, T23N, and T29N), installed in 2011 (pre-rewetting) and 2012 (post-rewetting) at distances 2, 5, 25, and 50 m from the ditch edge, were analysed. This dataset was analysed using a generalized least squares model (function *gls* and package *nlme*, Pinheiro et al., 2016), using fixed factors (pre-/post-rewetting, transect, and distance from ditch edge) with date as a random intercept (accounting for temporal autocorrelation using a first-order autoregressive structure). Again, there were some missing data from two wells (5 m well on both T17N and T17S), due to WTL being lower than the bottom of the logger (−55.5 and −62.0 cm, respectively). However, as this represented a small proportion of the overall dataset, data were not organized into bins but instead missing values were substituted with the logger depth as a conservative estimate (i.e., knowing that WTL was at least −55.5 and −62.0 cm on those occasions, these values were substituted into the appropriate missing data).

To answer the question on medium term effects of rewetting on WTL (3–6 years post-rewetting; 2014–2017), only one transect of water level loggers remained (T16; at distances 2, 5, 25, and 50 m both north and south of the ditch). Based on the results of the above short-term hydrology questions, the 50 and 25 m wells were the reference sites against which those closer to the ditch (rewetted sites; 2 and 5 m) were compared. This was carried out using a linear mixed effect model (functions *lme* and package *nlme*, Pinheiro et al., 2016) using fixed factors (treatment, e.g., rewetted or reference and years since restoration) with date as the random intercept (accounting for temporal autocorrelation using a first-order autoregressive structure).

2.4.2 | Pore-water chemistry

Pore-water chemistry data were analysed using principal response curves (PRCs). This is a multivariate method where the response variables, for example, water chemistry in a treatment are compared with that of a reference site over time (Poulin, Andersen, & Rochefort, 2013; van den Brink & Ter Braak, 1998). Thus, we could compare the overall temporal changes in water chemistry at rewetted sites (2 m from the ditch edge) against that of the reference sites (100 m from the ditch edge), to determine the effect of rewetting on water chemistry. The significance of the PRCs were then tested using Monte Carlo simulations (n = 999). We carried out PRCs on nine water chemistry variables, which were measured over the entire 7-year monitoring period (Ca, Mg, Fe, K, Na, sulfate [SO_4^{2-}], ammonium [NH_4^+], nitrate [NO_3^-], and phosphorus [P]).

2.4.3 | Vegetation

Vegetation data were also analysed using PRCs. Here, we split the vegetation data into rewetted and reference zones, to ensure a continuous time series of data (as individual monitoring points along the transect were only surveyed every 2 years from 2013). These zones were from 2 to 10 m (rewetted) and from 25 to 100 m from the edge of the ditch (reference). The zones were split this way as there was a clear difference in vegetation cover between the 10 and 25 m monitoring points, that is, the 10 m monitoring point had vegetation similar to those closer to the drain edge, whereas the 25 m point had vegetation cover similar to those further from the drain edge.

Vegetation data was analysed using PRCs, after being split into five functional groups, as there were too many species present to include in the analysis. These groups were pioneer mosses, late successional mosses, total ericaceous, total herbaceous, and total trees (-Table S2). This approach also allowed the main trends describing the functioning of the bog in terms of vegetation to be summarised. We then carried out univariate statistical analysis for each of the functional groups, which were influential on the PRC (i.e., scored >0.5; van den Brink & Ter Braak, 1999), to determine difference in cover between the reference and rewetted zones at the beginning (2010; pre-rewetting) and end of the study period (2016/2017; post-rewetting).

3 | RESULTS

3.1 | Effect of trees on WTL

Cutting trees caused a shallower WTL, which was significantly shallower in the cut transect than the uncut transect, at each distance from the ditch (F = 148, p < .001). The significant model interaction showed that the largest differences in WTL between the cut and uncut transects were measured in the first 25 m from the ditch edge; the greatest difference in mean WTL (up to 42 cm) was measured closest to the ditch (Figure 4a). At distances of 50 m or more, from the ditch edge, the difference in mean WTL between the transects was quite consistent (9.6–12.5 cm; Figure 4c), suggesting that the uncut transect was simply in a drier location.

The greatest difference in basal area between the transects was at distances up to 25 m from the ditch edge (Figure 4b). Therefore, there was a visible effect of both tree basal area and distance from the ditch on WTL in the uncut transect, whereas there was still a strong effect of distance from the ditch on WTL in the cut transect.

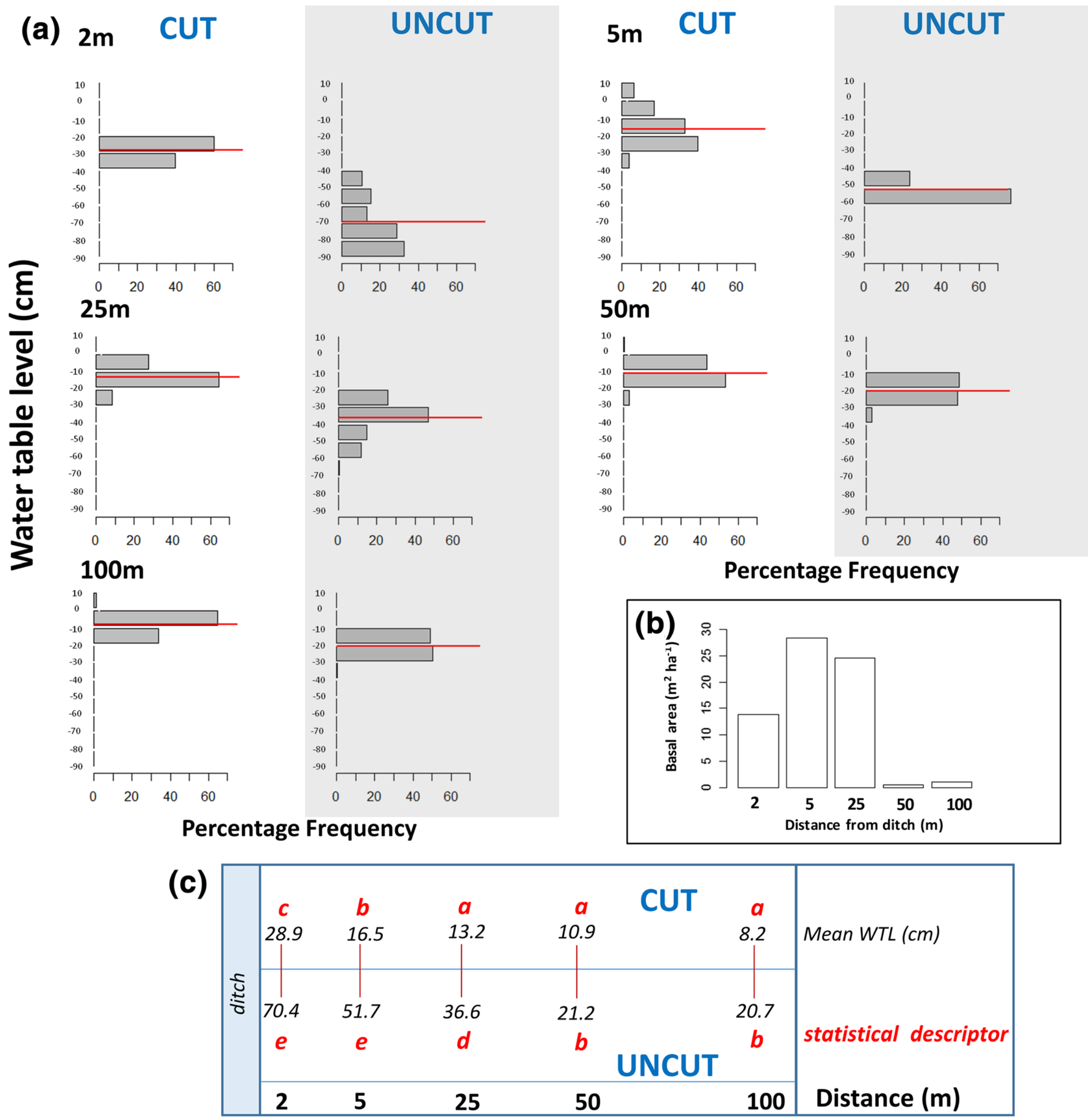

FIGURE 4 Frequency plots of water table level (WTL) in cut (T23N) and uncut (T17N) transects, at various distances from the ditch edge (a) from June to October 2011 (pre-rewetting); red lines show mean WTL for each treatment. Basal area of trees at each distance from ditch in uncut transect is plotted (b) (cut transect basal area = zero). Mean WTL and statistical letter descriptors for each distance in both transects (c), letters that are the same are not statistically different

3.2 | Short-term effects of rewetting on WTL (0–1 year post-rewetting)

There was an effect of rewetting on WTL in three out of four transects where the water table rose, post-rewetting in 2012 ($F = 383$, $p < .0001$). The greatest water table rise was measured when the transects were located immediately upstream of a dam (i.e., T17N and T17S, at 5 m distance and 17 cm elevation difference from a dam; Figure 5a). At T17N and T17S, the mean WTL at 2 m from the ditch edge increased by up to 90 cm post-rewetting, while up to 30 cm at 5 m from the ditch edge. At 25 and 50 m from the edge, there appeared to be no effect of rewetting.

In transect T29N (55 m distance and 29 cm elevation difference from a dam) following rewetting, mean water table only rose (mean increase 19.8 cm) at 2 m from the ditch edge, whereas in transect T23N (120 m distance and 23 cm elevation difference from a dam), WTL was significantly deeper following rewetting at each distance from the ditch edge (Figure 5b). As there were no effects of rewetting anywhere at 25 m from the ditch edge, we thus considered this distance to be part of the reference treatment for this study.

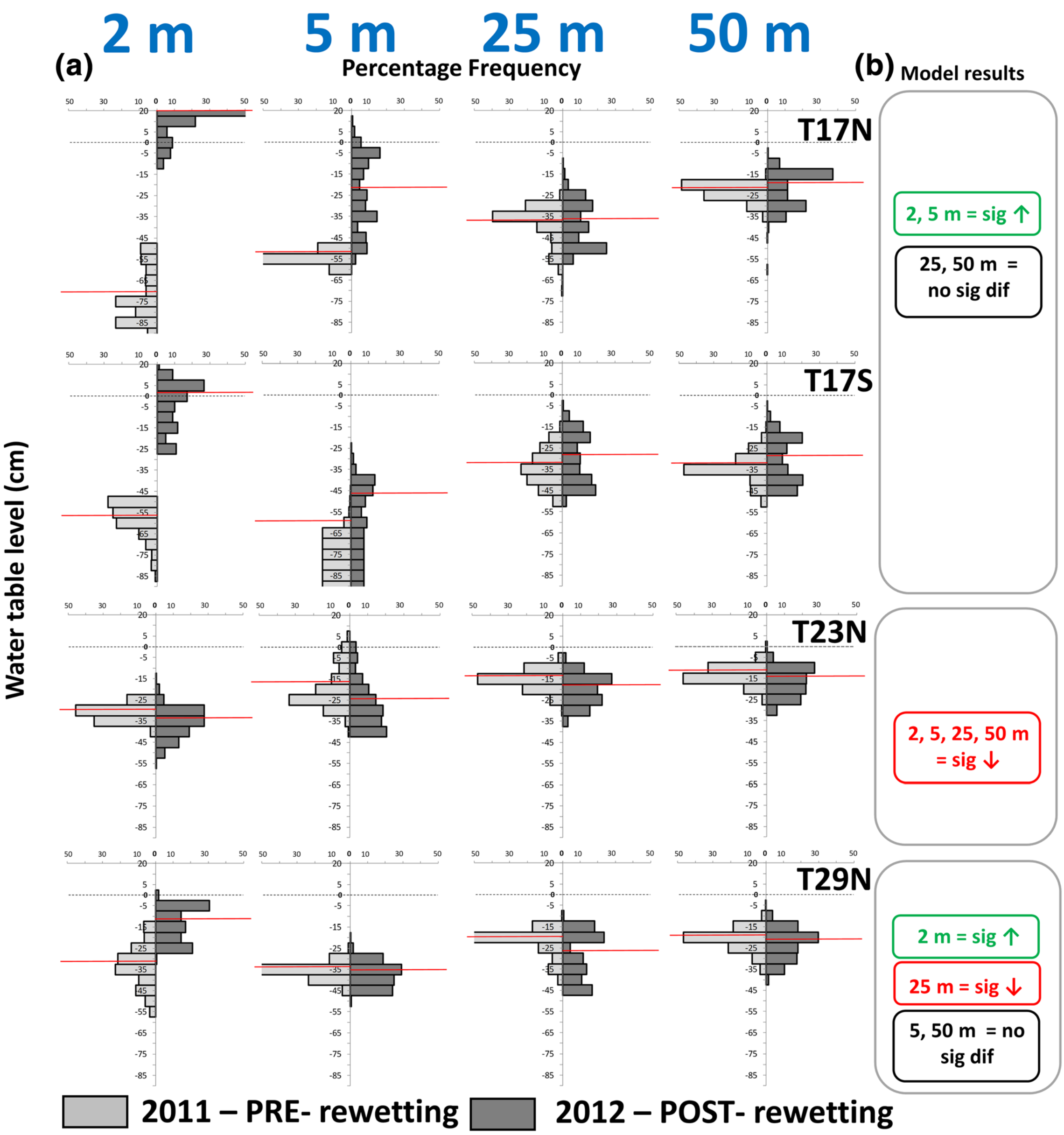

FIGURE 5 Frequency plots of water table level (WTL) pre-rewetting (2011) and post-rewetting (2012) in four monitoring transects (at varying elevations from the lower dam), at 2, 5, 25, and 50 m from the ditch edge. Red lines show the mean WTL for each treatment, whereas the dashed horizontal line (0 cm) represents the surface. On panel (a), transect T17N at 5 m from the ditch edge WTL was sometimes lower than the sensor (−56 cm) hence the flat shape of this distribution. Panel (b) shows the modelled three-way interaction results of differences in WTL pre- and post-rewetting, by each transect (elevation difference to dam) and by distance from ditch edge

3.3 | Medium-term effects of rewetting on WTL (3–6 years post-rewetting)

Water table from 3 to 6 years post-rewetting was significantly different (F = 313, p < .0001) between the rewetted sites (2 and 5 m away from the ditch edge) and reference sites (25 and 50 m away from the ditch edge), in terms of the absolute values; median WTL was shallower in the rewetted sites.

Each year, in the rewetted sites, WTL was 10 cm below the surface or higher around 50% of the time, whereas in the reference sites, this was only true for 20–30% of time. However, between years, the shape of the curves between the two treatments was mostly similar, suggesting that WTL behaved similarly in the rewetted and reference sites (Figure 6).

3.4 | Effect of rewetting on water chemistry

There were no overall changes in water chemistry following rewetting, as shown by the PRC, which suggested no significant temporal variation between the rewetted treatment and reference sites (p > .05). This precluded the need for any further univariate testing of water chemistry variables. On an individual basis, most water chemistry variables showed little change between pre- and post-rewetting, or between rewetted and reference sites, and were in a similar range to the regional mean concentration (Table 1). The only exception was sulfate (SO_4^{2-}), which was highest in the rewetted sites pre-rewetting (2010) and decreased post-rewetting. This pattern was also observed in the reference sites but to a lesser degree. In both rewetted and reference sites, there were interannual variations in water chemistry, with strongest concentration changes observed for major cations and anions (Ca, Mg, Na, K, SO_4^{2-}, and Cl^-) between years (Figure S2).

3.5 | Effect of rewetting on vegetation

The vegetation cover of the reference zone (25–100 m) over the 6-year study period was significantly different to that of the rewetted zone (2–10 m; F = 170.8, p = .001). By 6 years post-rewetting, the percentage cover of the main functional groups of vegetation in the rewetted zone appeared slightly closer to that of the reference zone than pre-rewetting (Figure 7). However, the PRC output showed no clear effects of rewetting on overall vegetation change in the rewetted zone, over the 6-year monitoring period.

In general, the rewetted zone had a lower cover of late successional mosses (mainly *Sphagnum*) and pioneer mosses than the reference zone but a higher cover of trees and ericaceous plants.

Using a before-after-control-impact model design (Stewart-oaten, Murdoch, & Parker, 1986; on the percentage cover of individual functional groups), the pioneer mosses and total ericaceous plants indicated a possible significant effect of rewetting on percentage cover (Figure 8a,d). However, on closer examination (least squares means comparison), only the reference zone for each of these two groups was different pre- and post-rewetting. Therefore,

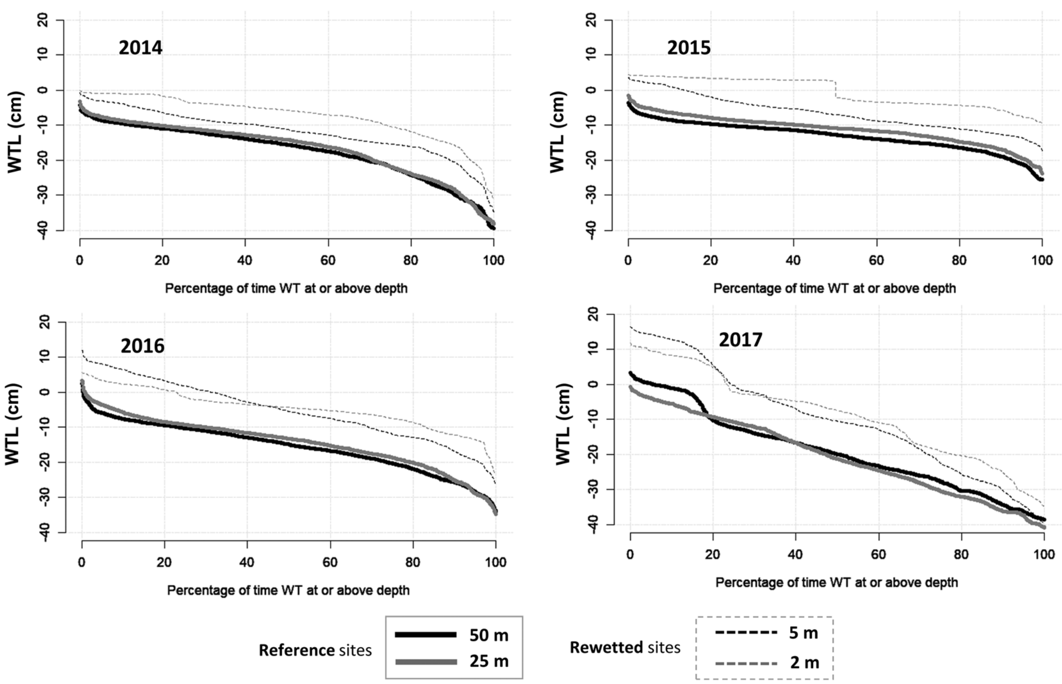

FIGURE 6 Cumulative frequency plot of water table level (WTL) in the medium-term post-rewetting (3–6 years post-rewetting) for each year from 2014 to 2017 in monitoring transect T16 at 2 and 5 m (rewetted treatments) and at 25 and 50 m (reference locations) from the ditch edge. Transect T16 has both north and south arms; data from both are combined in this figure

TABLE 1 Water chemistry results, mean ± (*SD*) pre- and post-rewetting in rewetted (2 m) and reference (100 m) zones, along with regional means for Quebec Province (Andersen, Rochefort, & Landry, 2011)

		Rewetted treatment (2 m)				Control points (100 m)				Regional mean	
		Pre-rewetting		Post-rewetting		Pre-rewetting		Post-rewetting			
		2010		2012–2017		2010		2012–2017		Regional	
Element		Mean	(*SD*)	Mean	(*SD*)	Mean	(*SD*)	Mean	(*SD*)	Mean	(*SD*)
Ca	mg L^{-1}	1.06	(0.83)	0.94	(1.27)	1.03	(0.85)	0.96	(1.01)	0.54	(0.54)
Mg	mg L^{-1}	0.26	(0.19)	0.23	(0.19)	0.25	(0.18)	0.21	(0.22)	0.19	(0.35)
Na	mg L^{-1}	0.86	(0.36)	0.63	(0.59)	0.82	(0.17)	0.51	(0.24)	1.83	(2.64)
K	mg L^{-1}	0.75	(0.40)	0.75	(1.41)	0.73	(0.14)	0.46	(0.59)	1.43	(1.05)
Fe	mg L^{-1}	0.22	(0.09)	0.33	(0.18)	0.20	(0.07)	0.25	(0.07)	0.23	(0.13)
Mn	mg L^{-1}	—	—	0.02	(0.02)	—	—	0.03	(0.04)	—	—
pH		3.09	(0.73)	4.18	(0.61)	3.35	(0.71)	4.16	(0.47)	4.00	(0.4)
EC	μS cm^{-1}	60	(16)	94	(78)	60	(26)	63	(28)	57	(25)
DOC	mg L^{-1}	—	—	52.8	(32.3)	—	—	41.6	(12.8)	—	—
Cl^-	mg L^{-1}	—	—	2.1	(1.8)	—	—	2.0	(1.1)	—	—
SO_4^{2-}	mg L^{-1}	20.2	(4.3)	11.4	(10.0)	14.0	(3.6)	9.1	(7.2)	14.1	(11.1)
NH_4^+	mg N L^{-1}	0.50	(0.29)	0.49	(0.36)	0.80	(0.41)	0.58	(0.67)	0.65	(0.47)
PO_4^{3-}	mg P L^{-1}	—	—	0.33	(0.82)	—	—	0.11	(0.12)	—	—
NO_3^-	mg N L^{-1}	0.08	(0.03)	0.03	(0.02)	0.05	(0.01)	0.01	(0.01)	0.04	(0.12)
Exchang-able P	mg P L^{-1}	0.29	(0.51)	0.34	(1.04)	0.10	(0.08)	0.16	(0.21)	0.17	(0.2)
Total *N*	mg N L^{-1}	—	—	0.96	(0.40)	—	—	0.71	(0.23)	—	—

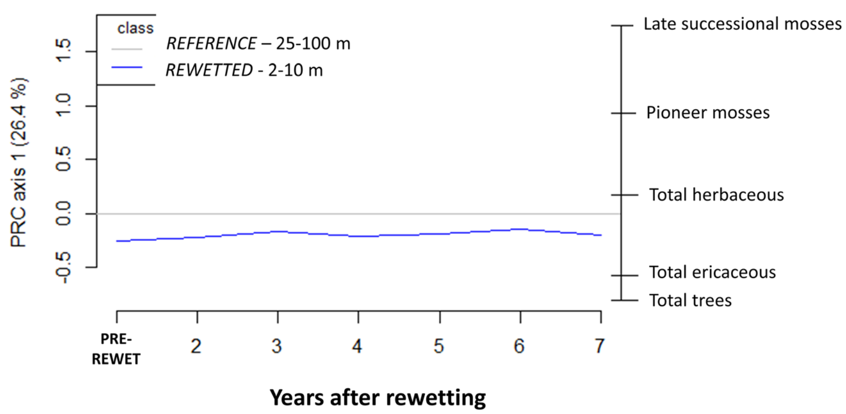

FIGURE 7 Principal response curve (PRC) of temporal vegetation changes between reference zone (25–100 m from ditch) and rewetted zone (2–10 m from ditch), using the main functional groups of vegetation in transects T16 and T46. The left hand axis represents overall deviation of the rewetted zone from the reference zone (represented by the zero line), expressed as a canonical coefficient on the first principal component axis (PC1). The right hand axis shows canonical coefficients for each of the vegetation functional groups. A more positive coefficient shows a stronger relationship with the curve, whereas a more negative coefficient suggests the opposite

rewetting had no significant effects on percentage cover of any of the vegetation functional groups within the rewetted zone.

In the reference zone, the cover of pioneer mosses decreased by ~5% between the pre-rewetting and 5- to 6-year post-rewetting monitoring, and there was negligible difference in the rewetted zone (Figure 8a). Total ericaceous plant cover increased slightly ~15% in the reference zone pre- and post-rewetting; however, the difference was not significant in the rewetted zone (Figure 8d). Although not significantly different, there was also an increase (~5%) in the cover of late successional mosses in the reference zone between the pre-

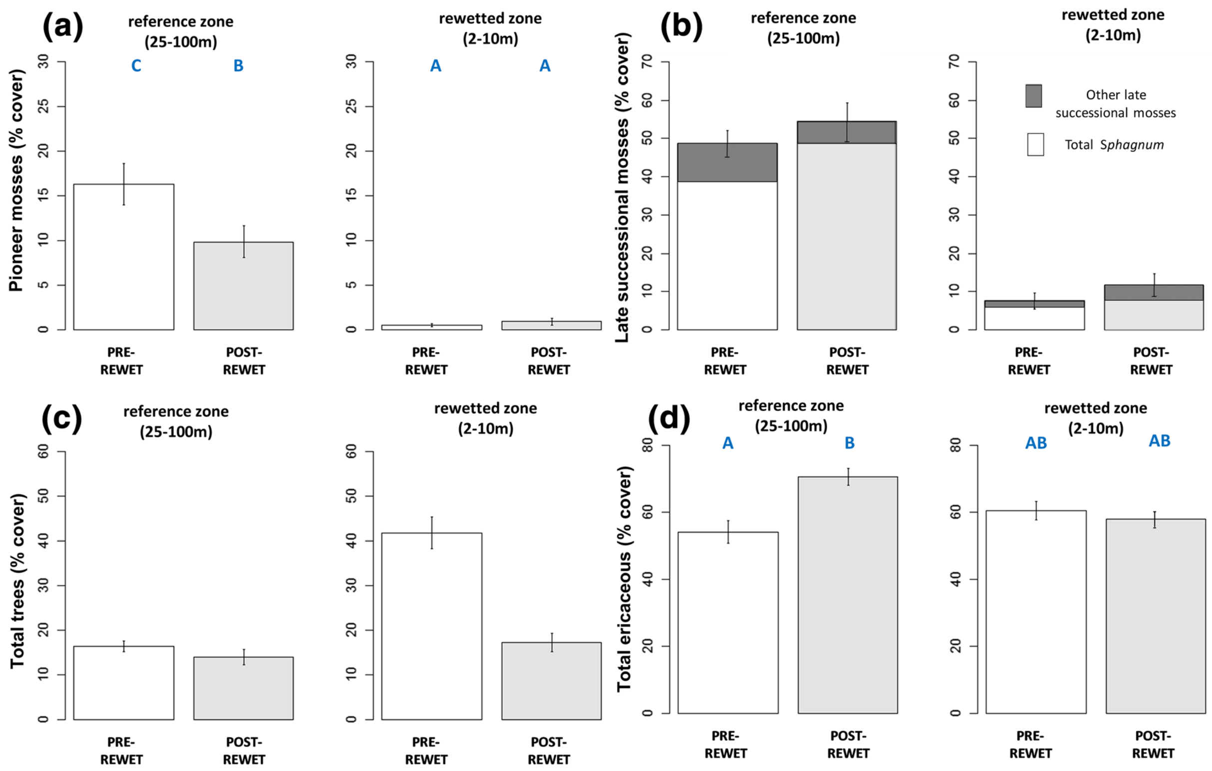

FIGURE 8 Percentage cover (mean ± *SE*) of the main functional groups of vegetation pre-rewetting (2010) and 5-6 years post-rewetting (2016–2017) in the reference and rewetted zones in transects T16 and T46. Only those functional groups scoring highly (>0.5 and <−0.5) on the PRC are included: (a) pioneer mosses, (b) late successional mosses, (c) total trees, and (d) total ericaceous. In panels (a) and (d), bars containing the same letters are not significantly different. Letters appear only in panels (a) and (d), as only here was there a significant treatment *time interaction

rewetting and 5- to 6-year post-rewetting monitoring (Figure 8b), which was mostly due to an (~10%) increase in *Sphagnum* cover in this zone.

4 | DISCUSSION

The ecohydrological changes following 6 years of rewetting on the GPB bog were mainly on WTL, as there were limited effects on water chemistry and vegetation cover. There was a rise in WTL post-rewetting in the first 5 m from the ditch edge, at locations within 17 cm elevation difference of a dam. At locations with a greater elevation difference to a dam, the effects of rewetting on water table extended a smaller distance from the edge of the ditch.

4.1 | Effect of trees on WTL

In the uncut transect, tree density was highest in the first 20–25 m from the ditch edge (up to 28 m^2 ha^{-1}; much higher than in the surrounding open bog 0–3 m^2 ha^{-1}), indicating that the deepening of the WTL caused by drainage, facilitated the establishment of trees (Ahti, 1987). Trees then continued to deepen the WTL, through their biological pump, where a high density of trees “drains” the peatland by evapotranspiration (Hökkä, Repola, & Laine, 2008).

Cutting trees led to a rise of the WTL as it “switches off” the biological drainage (Jutras, Plamondon, Hökkä, & Bégin, 2006). This was evident in our dataset, where in the densest treed part (5 m distance from the edge) we found differences of up to 35 cm in WTL between the cut and uncut transects. In the case for restoration and rewetting of the site, removal of trees appears a necessary measure as flooding alone might not be sufficient to kill the trees; in our case, all trees (of DBH >10 cm) were felled in 2012 based on this data.

4.2 | Effect of rewetting on WTL

Our results clearly showed an effect of rewetting on WTL, but this was strongest, where the monitoring transect was closest (in elevation) to a dam. Other studies have also found that proximity to a dam affected water table recovery following ditch blocking (Wilson et al., 2010).

In our study, the transects that had the smallest elevation difference (T17N and T17S) therefore had the greatest WTL increase (up to 30 cm, at 5 m from the ditch edge), but these effects did not extend as far as 25 m from the ditch edge (where there were no significant changes post-rewetting). However, others have measured the effects of rewetting by ditch blocking to extend 20 m from the ditch edge (Holden, Wallage, Lane, & McDonald, 2011).

WTL in transect T23N did not rise post-rewetting, possibly as this transect was located too far (120 m) from a dam to be affected by the ditch blocking, (even though the elevation difference was only 23 cm). Instead, mean WTL fell slightly in 2012, (1 year post-rewetting) likely due to climatic factors. Contrastingly, our results showed a significant rise in WTL (at 2 m from the ditch edge), post-rewetting in transect T29N, where the elevation difference between the dam and transect was 29 cm. Therefore, there are also some local factors that influence the effectiveness of rewetting.

In the medium term (3–6 years post-rewetting), the shallower WTL in the rewetted sites (2 and 5 m from the edge) compared with the reference sites (25 and 50 m from the edge) was in agreement with other medium length studies, which found that ditch blocking can maintain a high WTL for at least around one decade afterwards (Haapalehto, Vasander, Jauhiainen, Tahvanainen, & Kotiaho, 2011).

As the shape of the curves between the two treatments was similar each year, this suggests a similar response over time between the rewetted and reference sites. The inflection point in the cumulative frequency WTL curves (the point where the curve starts to bend) can represent the transition between the free draining depth and that controlled by evapotranspiration (Holden et al., 2011). Our data suggest that in the rewetted sites, free drainage can occur when the water table is at 5 cm or more above the surface, whereas in the reference sites, free drainage can occur down to 5 cm below the surface. Therefore, in the rewetted treatment, WTL is more often above the surface, for example, in spring and in autumn when evaporation is lower. Excessive inundation could potentially affect the recovery of mosses (Rochefort, Campeau, & Bugnon, 2002; Smoulders, Tomassen, van Mullekom, Lamers, & Roelofs, 2003) and may be due in part to peat subsidence around the ditch, during the period of drainage (Price & Schlotzhauer, 1999).

The WTL transect monitoring from 3 to 6 years post-rewetting had an elevation difference of 16 cm from the lower dam (at 65 m distance), suggesting that rewetting at the GPB was quite effective in maintaining a high WTL at this elevation difference (even if the hydrological functioning was not quite the same as at the reference sites). This was in agreement with the short-term WTL results (transect T17), where at an elevation difference of 17 cm from a dam, WTL recovered at 2 and 5 m from the ditch edge.

Therefore, the elevation difference which a dam can effectively rewet appeared to be around 17 cm for such a large ditch. However, the short-term results for T29 (WTL rise post-blocking at 2 m from edge) and T23 (no WTL rise post-blocking) suggest that both the elevation change and lateral distance are important factors along with local topography in determining the effectiveness of dams (Parry et al., 2014; Wilson et al., 2010). In general, it is important to consider the height of the dam (and maximum water level) in relation to the surrounding peat surface (Landry & Rochefort, 2012). The approach used at the GPB, of placing a dam every 40 cm in elevation change, was insufficient to fully rewet the area affected by the ditch and clearly more dams would be required for rewetting to be successful along the whole ditch length.

4.3 | Effect of rewetting on water chemistry

Contrary to other studies where rewetting (by both tree felling and ditch blocking) resulted in significant changes in water chemistry (Gaffney, Hancock, Taggart, & Andersen, 2018; O'Driscoll et al., 2014), we found no significant change following rewetting on the GPB bog. This was additionally surprising as even the presence of trees themselves on a bog can affect water chemistry compared with non treed locations (Ratcliffe et al., 2017). Thus, differences between rewetted and reference sites would also have been expected in 2010 in the pre-rewetting year, although we found little difference here also. The only notable change was a decrease in SO_4^{2-} concentrations in the rewetted sites, which was likely due to the reduction of SO_4^{2-} into less soluble states, following the post-rewetting water table rise (Adamson, Scott, Rowland, & Beard, 2001).

Our strategy was to measure the water chemistry that surface plants and bryophytes had access to, but a composite pore-water sample was collected (over a depth of 1 m) in order ensure a sample could be collected even if the water table was low in dry conditions. This may have slightly diluted the effects of rewetting, which can be stronger in shallow pore water (Gaffney et al., 2018). However, prior to rewetting, there was also little effect of drainage on the water chemistry, suggesting both drainage and rewetting had a limited effect on chemistry, perhaps as the ditch was a relatively small disturbance compared with the size of the GPB bog.

In terms of interannual variation, patterns in major cations and anions with lower concentrations observed in 2014, then maxima in 2015 may have been related to WTL, which was shallower in 2015 than 2014. Thus, wetter conditions may have resulted in a release of ions to solution (Kaila et al., 2016), although others have found decreased concentrations of these parameters following rewetting (Lundin, Torbjorn, Jordan, Lode, & Stromgren, 2017). Our sampling took place in May of each year; thus, we were unable to capture any seasonal variations in water chemistry. Concentrations of major ions are related to precipitation inputs and evaporation, for example, Ca and Mg concentrations may increase with evaporation over the summer season (Proctor, 2006). For other parameters, which are associated with temperature and decomposition cycles, for example, PO_4^{3-} and DOC, concentrations may also increase over the growing season (Muller, Chang, Lee, & Chapman, 2015).

4.4 | Effect of rewetting on vegetation

In the same transects where water chemistry was measured (65–75 m from a dam with 16–46 cm elevation difference), there was little vegetation recovery observed over the 6-year period. Even when one transect (T16) showed a recovery in water table, it appears to not have stimulated vegetation recovery. Indeed, the percentage cover of pioneer mosses and late successional mosses (which are mostly *Sphagnum* species) in the rewetted zone remained substantially lower than the reference zone.

Highly degraded peatlands (where vegetation cover has been totally stripped out, i.e., vacuum-harvested peatland) are successfully restored by rewetting and reintroducing vegetation (González & Rochefort, 2014). However, in the GPB setting, dispersion limitation was not expected to be a problem, as the entire rewetted site was surrounded by open bog (giving a diaspore source of 20 m or less).

For *Sphagnum*, a keystone species in bogs in the late successional mosses group, we expected natural recruitment (Rydin, Gunnarsson, & Sundberg, 2006). This meant that we expected *Sphagnum* spores to germinate on the bare peat surface that was present near the ditch (Campeau & Rochefort, 1997), underneath the ericaceous shrubs (Sundberg & Rydin, 2002), or for the *Sphagnum* present on the adjacent intact bog to slowly colonize (vegetatively) the rewetted peat (from the reference zone towards the ditch; Rydin et al., 2006).

Despite this, *Sphagnum* recruitment was prevented, which may have been as the rewetted peat was covered by a dense layer of litter (mainly leaves from the tree and ericaceous shrubs layer) that may have prevented germination of spores because litter remained dry even if the underlying peat was wet (humidity at the air-peat interface is a key requirement for *Sphagnum* spores; Ferland & Rochefort, 1997). Another factor, which can aid the germination of spores, is shade by vascular plants (or in post-harvested peatlands, a protective layer of straw mulch; Rochefort et al., 2003); however, as vascular plant cover was around 60% in both the rewetted and reference zones, this would not appear to have been preventative.

Additionally, the recovery of mosses may have been discouraged by changes in the peat chemistry near the ditch due to the long period of drainage. For example, in cutover peatlands, plants were phosphorus (P) limited to due to a lack of P produced by plant-microbe cycling (Andersen, Rochefort, & Poulin, 2010), and P limitation is one reason thought to prevent development of *Sphagnum* spores into new plants (Rydin et al., 2006). However, the pore-water nutrient data show that P levels were slightly higher in the rewetted zone than the reference zone, which suggests that P limitation should not be preventative.

Although *Sphagnum* regularly produces spores, their role in dispersal and regeneration is not well known (Rochefort, 2000). Here, the distance between the intact bog and the ditch may have been too great for spore dispersal (Sundberg, 2005). Similarly, there may not yet have been enough time for vegetative colonization from intact bog towards the ditch. Either way, our results show that vegetation reintroduction may have been necessary for successful re-establishment of *Sphagnum* on the GPB.

The only place where *Sphagnum* was observed recolonizing was in the open water on the edge of the ditch, either because *Sphagnum* was present prior to rewetting on the bottom of the ditch or conditions were adequate for spore germination, for example, poorly humified peat here provided a favourable substrate (Smoulders et al., 2003). Additionally, an (~10%) increase in *Sphagnum* cover in the reference zone was observed between pre-rewetting (2010) and 6–7 years post-rewetting. This appeared mainly due to an increase in cover around 25 m from the ditch. However, WTL was deeper here at 6–7 years post-rewetting, compared with earlier in the study, so does not appear directly due to hydrology.

4.5 | Conclusion and management implications

The construction of six dams at the GPB bog at elevation intervals of 40 cm and felling of all trees (with DBH >10 cm) for rewetting of the area affected by a deep ditch was not successful in restoring the full range of ecohydrological functions. Hydrology (water table) was restored within a limited elevation difference (of around 17 cm) of dams rather than along the entire length of the ditch and moss cover (pioneer and late successional species) remained as low as pre-rewetting; therefore, there was no developing trajectory for vegetation recovery in the 6-year period.

Two factors were identified, which could be addressed for future schemes. First, to restore the hydrology more fully, dams should be spaced at smaller elevation intervals (e.g., every 20 cm), to allow recovery of the water table along the entire length of the ditch. Even though the site was relatively flat, the current design (dams built at elevation intervals of 40 cm) means that dams were spaced too far apart to retain water effectively.

Second, inadequate conditions for spore germination or the presence of a high cover of shrubby vegetation prevented *Sphagnum* re-establishment. Therefore, the introduction of mosses, for example, using the moss layer transfer technique (Rochefort et al., 2003), may be required to help achieve levels of cover similar to intact sites for the area affected by the ditch, in combination with fully restoring WTL. This could accelerate *Sphagnum* establishment, although existing shrub vegetation may need to be first removed for this technique to work. The introduction of vegetation may help achieve a fuller ecohydrological recovery for the site in areas where water table is restored.

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DATA AVAILABILITY STATEMENT

Research data are not shared.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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