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Effectiveness of restoration of a degraded shallow mountain fen after five years

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SUMMARY

As a contribution to improving understanding of the mechanisms and relationships that exist within shallow peatland ecosystems, we report the results of monitoring five-years' recovery after restoration of a forestry-drained sloping rich fen site in the Central Sudetes in south-west Poland. Over the last 100 years, drainage ditches installed for forestry management purposes have affected the hydrology, soil and vegetation of this site. Spruce stands were present until 2010, when restoration started with blocking of ditches and clearcutting/removal of trees. The main objective of our study was to determine the effects of these restoration activities on aspects of hydrology and soil condition. We hypothesised that the five years following restoration could provide sufficient time to improve: 1) water table level and water quality, and 2) the physical and chemical properties of the organic soil. Restoration had a positive effect on water table level as early as two years after implementation of drain blocking and reduction of tree cover. However, five years was not sufficient time to reverse the decline in water quality. The concentrations of labile carbon forms in water, as well as water colour (Abs^{400}), were similar in 2010 and 2015. Due to peatland rewetting and peat swelling a decrease in peat bulk density was observed. However, this outcome was identified only in the bottom organic soil horizons. This, in turn, affected the thickness of the peat layer and the altitude of the soil surface. Five years of recovery is insufficient to stop the mineralisation of organic matter, as indicated by lower TOC/TN values, slightly higher concentrations of labile forms of carbon, and the W_1 index of secondary transformation.

KEY WORDS: ditch blocking, forestry, Norway Spruce, organic soils, water table level

INTRODUCTION

Peatlands occupy a relatively small percentage (~3 %) of the Earth's surface. Nonetheless, they are globally important ecosystems (Gorham 1991), fulfilling manifold ecological functions such as carbon accumulation (Chimner & Cooper 2003, Armstrong *et al*. 2010), water retention (Holden *et al.* 2004, Strack *et al*. 2008), biodiversity maintenance (Tousignant *et al*. 2010) and global climate regulation (Gorham 1991). Their functions depend on various processes, both large-scale (*e.g.* water table level, evapotranspiration, nutrient runoff) and small-scale (*e.g.* capillary flow of water in soil, soil water retention) (Waddington 2008).

In the last century, an increase in both direct and indirect human impacts on peatlands was observed (Heller & Zeitz 2012, Labaz & Kabala 2016, Drewnik *et al.* 2018). Peatlands were utilised mainly for agriculture (*e.g.* Kalisz *et al*. 2015, Lamers *et al*. 2015, Glina *et al*. 2016c, Lipka *et al.* 2017), peat extraction (*e.g.* Farrell & Doyle 2003, McCarter & Price 2013) and forestry (*e.g.* Ojanen *et al*. 2013, Glina *et al*. 2016a). All of these land uses require drainage to lower the water table (Laiho & Pearson 2016, Chimner *et al.* 2018, Drewnik *et al.* 2018). Drained peatlands are of global concern because of their altered soil physical and water conditions (Schimelpfenig *et al*. 2014), disturbed carbon cycle (Strack *et al*. 2008), degrading soil (Glina *et al*. 2016a, 2016c) and vegetation cover (Tousignant *et al.* 2010), and increased fire risk (Mangan *et al.* 2012, Glina *et al*. 2017). It is crucial to avoid further loss of carbon and to prevent fragmentation of these important ecosystems (Anderson & Peace 2017).

Peatland restoration has expanded in scale and scope since the late twentieth century (Parry *et al.* 2014a). The key event was implementation of the Ramsar Convention - the framework for conservation and wise use of wetlands - in 1975. This document was subsequently expanded by the addition of several thematic resolutions and, finally, the Guidelines for Global Action on Peatlands were published in 2002 (Ramsar Convention Contracting Parties 2002). Peatland restoration has now become one of the most popular topics in ecological research and the subject of numerous publications (Grand-Clement *et al.* 2015).

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The characteristics of peatland following restoration may provide a set of essential information (Grand-Clement *et al*. 2015), in particular about the release of dissolved organic carbon, the transformation of soil physical properties, and vegetation change. The effects of restoration are discussed primarily in relation to extracted (cutover) *Sphagnum*-dominated bogs in the Boreal zone (*e.g*. Sottocornola *et al*. 2007, Wilson *et al*. 2011, McCarter & Price 2013) and Central European lowlands (*e.g*. Poschlod *et al*. 2007, Zerbe *et al*. 2013). Other publications focus on the outcomes of restoring peatlands that were previously drained for agriculture (*e.g*. Cooper *et al*. 1998, Chimner & Cooper 2003, Schimelpfenig *et al*. 2014) or forestry (*e.g*. Mälson *et al*. 2009, Haapalehto *et al*. 2011, Koskinen *et al*. 2017, Nicia *et al*. 2017), in both lowlands and mountains. The effects of restoration on shallow fens have received less attention, resulting in only a small amount of information in the literature (Grand-Clement *et al*. 2015) even though, on average, 78 % of the mires in Central European countries belong to this specific ecosystem type (Bragg & Lindsay 2003). There is a need for better understanding of the mechanisms and relationships that exist within shallow peatland ecosystems, and especially in restored sloping fens. For this reason, the attention focused on such peatlands should be increased (Chimner *et al*. 2016).

In Poland, forestry-drained mountain peatlands are located mainly in the Sudetes range in the south-west of the country (Glina *et al*. 2016a). This type of land use was particularly common in the Central Sudetes at the turn of the 19th and 20th centuries (Glina *et al*. 2017). Vast peatland complexes were drained by the open-channel method (Kabała 2015) to allow planting of Norway spruce (*Picea abies*), which currently dominates 83 % of the stands (Gałka *et al.* 2014). Long-term forestry management subsequently caused multidirectional changes in the Central Sudetes peatlands (Bogacz and Roszkowicz 2010, Bogacz *et al*. 2012, Glina *et al.* 2016a). For that reason, in the year 2010 the local authorities in co-operation with the ecological organisation “Lubuski Klub Przyrodników” started a restoration programme on several peatland areas in the Stolowe Mountains National Park, including a unique area of sloping rich fen (Sienkiewicz & Wójcik 2012). This type of peatland is considered to be one of the most species-rich ecosystems (Mälson *et al*. 2009, Lamers *et al*. 2015), and thus an important target for restoration. However, methods for restoring drained sloping peatlands are limited and poorly tested (Schimelpfenig *et al*. 2014), due to numerous technical constraints including difficulties with heavy machinery use and site access (Joosten & Clarke 2002).

Here we report our findings from a monitoring study of this sloping rich fen over a period of five years following restoration. The main objective was to assess the impact of restoration activities (clearcutting of trees and shrubs, ditch blocking) on fen site conditions. We hypothesised that the five years following restoration could provide sufficient time to improve: 1) water table level and water quality; 2) the physical and chemical properties of the organic soils.

METHODS

Study area: location, genesis, vegetation and restoration works

The restored fen is located in the Stołowe Mountains of the Central Sudetes (Figure 1A). This mountain range in south-western Poland is composed mainly of upper Cretaceous sandstones, siltstones (mudstones) and claystones (Waroszewski *et al.* 2015). According to the Köppen-Geiger climate classification, the region is located in the humid continental climate zone, with warm summers (Kottek *et al*. 2006). For the six-year period 2010–2015, the mean annual air temperature of the region was 6.9 °C and the mean annual precipitation was 726 mm. Meteorological data were obtained from Słoszów and Kudowa stations, respectively.

Peatlands cover approximately 2.5 % (132 ha) of the Stołowe Mountains (Sienkiewicz & Wójcik 2012). The current peatland areas are only remnants of much larger complexes (Glina *et al.* 2016a) which have become degraded due to long-term drainage for forestry (Glina *et al.* 2017). As a result, it was impossible to identify an undisturbed peatland as a reference site for this study. The study site itself was a sloping (at 3–4 degrees) fen on the northern slope of the Skalniak ridge (840–850 m a.s.l.) with an inflow of mineral-rich groundwater (Glina *et al.* 2016b). This small (0.40 ha) rectangle-shaped fen is of late Holocene origin, and peat formation started here *ca*. 3320 BP (Glina *et al*. 2017). Throughout the late Holocene this area was covered by alder (*Alnus* spp.) stands with an undergrowth of vascular plants, particularly sedges (*Carex* spp.) until, around the year 1800, a Norway spruce (*Picea abies*) monoculture was planted (Gałka *et al*. 2014, Glina *et al*. 2017). More than 100 years of drainage (by ditches running parallel to the slope) connected with forestry management had affected the hydrology, soil cover and vegetation structure of the site. Spruce stands were present until restoration started in 2010.

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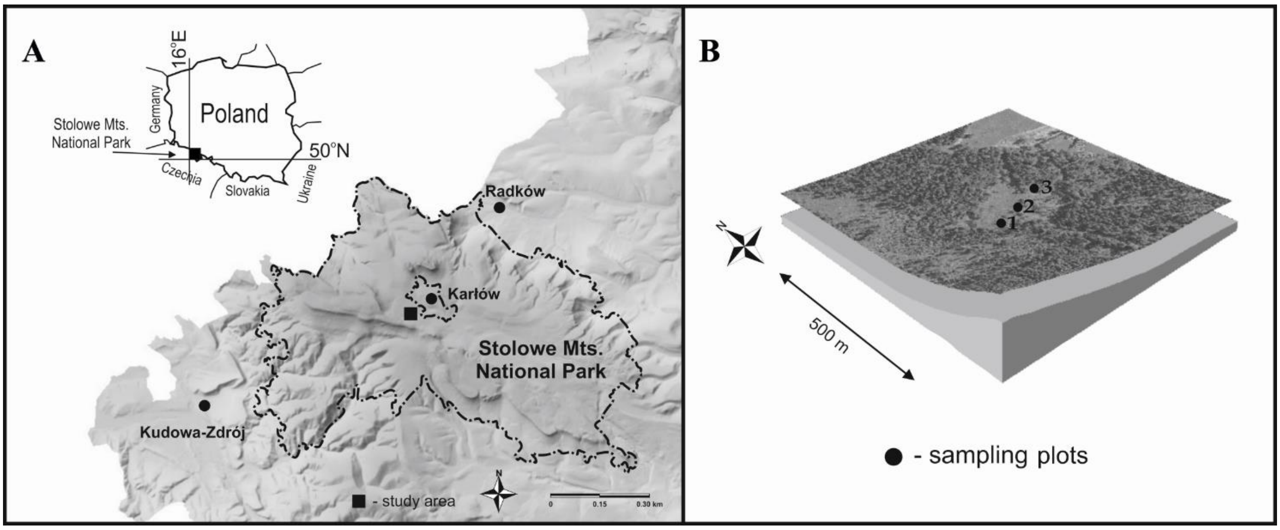

Figure 1. Location of the study area (A) and locations of the sampling plots within the research transect (B).

Restoration was conducted in two steps. In the first step the site was cleared of 'undesirable' grasses, shrubs and spruce trees (Figure 2A) and the resultant biomass (logging slash) was removed to reduce evapotranspiration, limit nutrient pools and avert acidification of the fen soils *via* coniferous tree remnants. The trees and shrubs were removed by clearcutting to allow peat-forming species to regenerate from the seed bank in the peat - which happened every September until the year 2012. In the second step, drainage ditches were blocked to further rewet the site. Two ditches were blocked by installing one single-wall wooden dam in each ditch (Figure 2B) in October 2010. The dams were made using hand operated tools, from wood acquired *in situ*. This restoration technique is particularly appropriate for sloping peatlands, where ditches do not naturally fill with peat or mineral sediment due to steep gradients and rapid surface runoff (Schimelpfenig *et al*. 2014).

A response of vegetation to the restoration actions applied was noticeable after two years (in 2013). After accomplishing the first step of restoration the surface of the peatland was free of trees and shrubs. Single clumps of *Carex paniculata* were visible and identifiable, especially in the upslope half of the site (see Figure 2C). The first changes in plant community structure were observed during the summer of 2013, when numerous peat-forming plant species such as *Carex paniculata, Carex flava, Carex remota, Carex effusus, Equisetum limosum, Equisetum fluviatile, Equisetum palustre* and *Scirpus sylvaticus* appeared or regenerated within the peatland. An especially noteworthy arrival was *Veratrum lobelianum* (Figure 2D), a characteristic species of damp habitats which is protected by law in Poland. By 2014, frequent alder (*Alnus incana*) seedlings were present. In the final year of observations (2015) the young alder trees were up to one metre tall and were growing mostly in the uppermost part of the fen (Figure 2E).

Field survey

The research transect consisted of three 4 × 4 m sampling plots which were laid out between two drainage ditches, approximately 10 m from each. The transect, like the ditches, ran parallel to the slope. Sampling plots were located as follows: Plot 1 (coordinates 16° 20' 25.2″ E, 50° 28' 06.1″ N) on the uppermost part of the slope, Plot 2 (16° 20' 23.9″ E, 50° 28' 06.3″ N) in the middle and Plot 3 (16° 20' 21.8″ E, 50° 28' 05.0″ N) on the lowermost part of the slope (Figure 1B).

Samples of soil and water were collected on two occasions, in September 2010 (just after accomplishing the first step of restoration) and in September 2015. Three replicate peat cores were extracted from each sampling plot using an "Instorf" peat corer (chamber 50 cm long, 5.2 cm in diameter) to obtain representative soil material for laboratory analysis and to measure the total thickness of the peat layer. Before sampling, the morphology of the studied soils was described according to the Guidelines for Soil Description (Jahn *et al.* 2006). The soils were classified on the basis of morphological features and physico-chemical properties according to the FAO-WRB classification (IUSS Working Group WRB 2015). Samples (70 in total) were collected from genetic soil horizons, using small metal rings.

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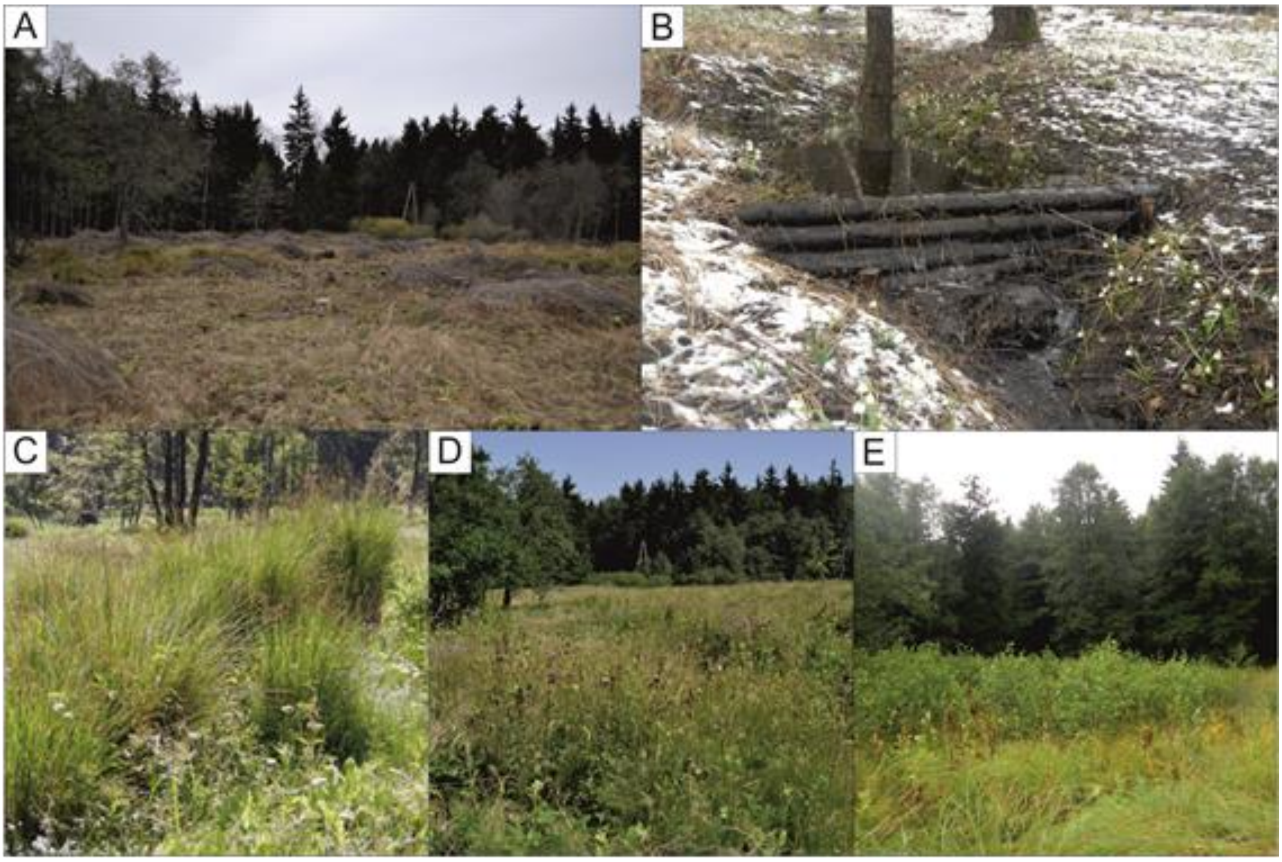

Figure 2. (A) Spruce branches, slash and trunks on the peatland surface (the effect of clearcutting trees) in September 2010. (B) Single-wall wooden dam in the drainage ditch, March 2011. (C) Regeneration of *Carex paniculata* within Plot 2, June 2013. (D) *Veratrum lobelianum* entering the fen, June 2014. (E) Young *Alnus incana* trees in the uppermost part of peatland (Plot 1), August 2015.

For monitoring of water table level and water sampling at each plot, dipwells constructed using perforated PVC pipes (10 cm diameter) were installed in boreholes that were backfilled with native soil. A single set consisted of two dipwells with filters, which were capped to prevent contamination. Water table level was monitored monthly from March to October (snow-free period) during the years 2010–2015 (Figure 3). Mixed water samples were collected in polyethylene bottles using a hand pump. Water pH was measured in the field using a CP-105 ELMETRON pH meter, and EC was measured using conductivity meter CPC-411. Soil and water samples were transported in a lightproof insulated box containing ice packs to ensure constant temperature.

Laboratory analyses

Soil

Bulk density (BD) was determined by drying 5 cm^3 sub-samples of peat to constant weight at 105 ºC, then dividing dry weight (g) by fresh sample volume (cm^3) (Chambers *et al*. 2011). The remainder of each soil sample was divided into two parts. One part was used in fresh (moist) condition to determine degree of decomposition by the half-syringe (fibre volume) method of Lynn *et al.* (1974), as well as secondary transformation of the soil horizons in terms of the water holding capacity index W_1 proposed by Gawlik (2000). The other part of the sample was dried at 105 °C then crushed in a mortar until it was homogeneous, and living plant remains were removed meticulously. The dried material was used to determine pH potentiometrically in a 1:2.5 soil:water solution (Kabala *et al.* 2016), total organic carbon (TOC) and total nitrogen (TN) content by catalytic dry combustion using a CNS VarioMax analyser, and the contents of hot water extractable carbon (HWC) (Sparling *et al.* 1998) and cold water extractable carbon (CWC) (Landgraf *et al.* 2006) using a Ströhlein CS-MAT 5500 analyser (after filtration *via* Whatman 0.45 membrane filters).

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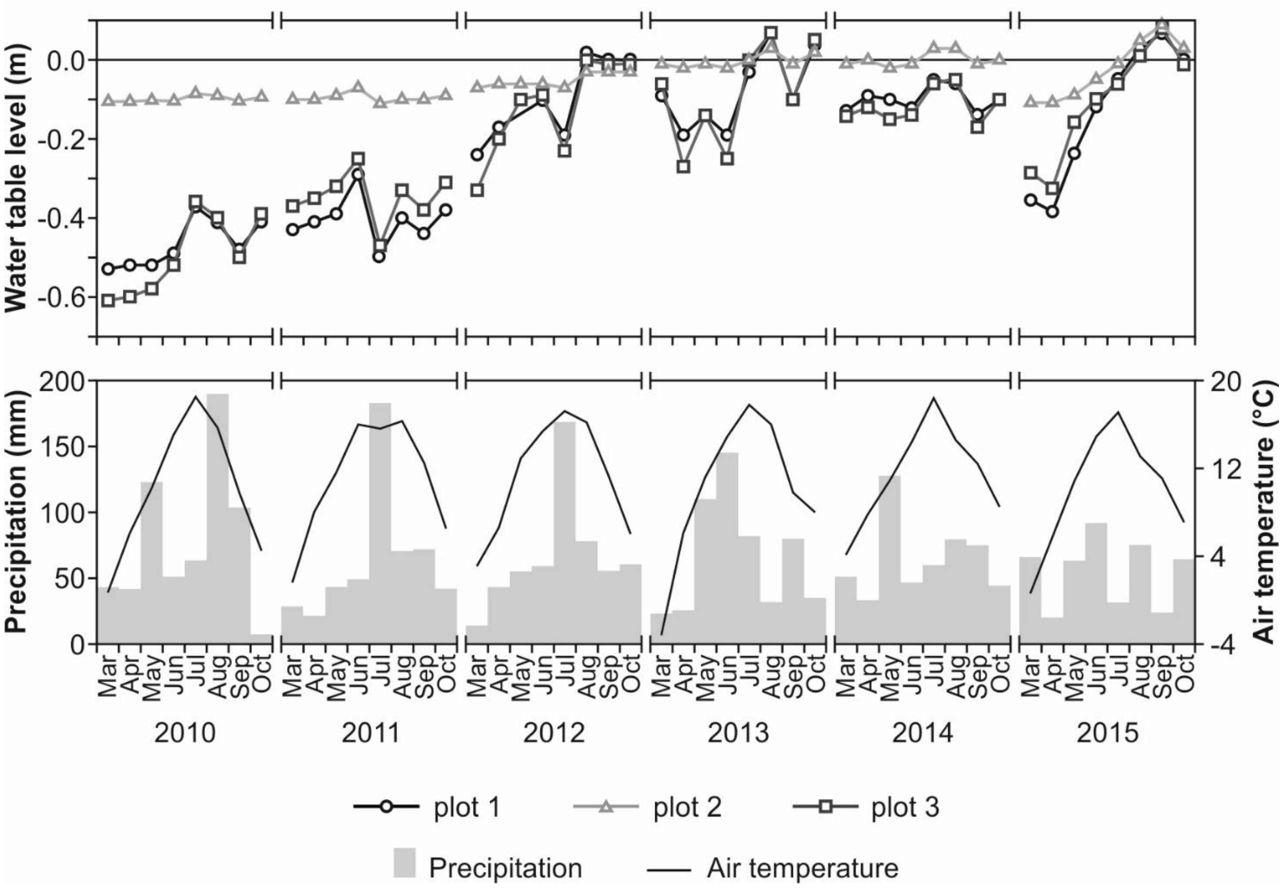

Figure 3. Weather conditions (temperature and precipitation) and water table level during the study period. Monthly precipitation amounts and mean monthly temperature were obtained from meteorological stations at Słoszów and Kudowa Zdrój, respectively.

Water

Water samples were filtered *via* Whatman 0.45 μm membrane filters within a day of collection. After filtration the absorbance at 400 nm (Abs^{400}) was measured in the laboratory using an Agilent Cary 60 UV-Vis Spectrophotometer (Wallage *et al.* 2006). Water samples for dissolved organic carbon (DOC) measurements were placed in 50 ml glass bottles and acidified with 150 μL of concentrated HCl to remove carbonates. The samples prepared in this way were stored in a refrigerator at 5 °C until analysis using a Ströhlein CS-MAT 5500 analyser with infrared detection of the evolving CO_2. Additionally, the following standard elemental composition of water was measured: concentrations of calcium (Ca^{2+}) and magnesium (Mg^{2+}) ions by atomic absorption spectrophotometry (AAS) after adding lanthanum to reduce anionic interference; and bicarbonate (HCO_3^-) content by titration with hydrochloric acid.

Basic statistical analysis (Pearson correlation coefficient) of the results was done using Statistica 12 software (StatSoft Inc., Tulsa, OK, USA).

RESULTS

Meteorological data, water table level and water chemistry

Mean growing season temperatures and precipitation sums for individual years of the study period were as follows: 10.1 °C and 616 mm in 2010, 11.1 °C and 468 mm in 2011, 11.1 °C and 474 mm in 2012, 10.1 °C and 498 mm in 2013, 11.4 °C and 473 mm in 2014, 10.1 °C and 372 mm in 2015. The wettest seasons were summers (June–August), while springs (March–May) were the driest, except in the year 2014 when the rainfall total for spring slightly exceeded the precipitation sums for summer and autumn (Figure 3).

Ditch blocking and reduction of evapotranspiration led to an increase in the mean growing season water table level during the five years of observations. A rise in water table level of up to ~ 40 cm was observed at Plots 1 and 3 in the second year following restoration (2012). The water table at Plot 2 was very close to the ground surface even

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before restoration treatments, and by the end of the study period (in August–October 2015), surface water was present at all of the plots (Figure 3).

pH and EC values had increased very slightly by the fifth year following restoration (Table 1). The water pH along transects increased from the margin towards the centre of the peatland. Bicarbonate, calcium and magnesium concentrations in peat water along the research transects were in the ranges: 157–179 mg L^{-1}, 53.2–60.7 mg L^{-1} and 2–3 mg L^{-1}, respectively, in 2010. The amounts of these elements were similar in 2015, and there was no noticeable trend. DOC concentrations in peat water were also similar in the years 2010 and 2015 (Figure 4). The same situation was observed in the case of water discolouration as defined by the Abs^{400} indicator; the average values ranged from 13.3 au m^{-1} in 2010 to 15.0 au m^{-1} in 2015 (Figure 4).

Soil morphology, classification and properties

Prior to restoration, the total thickness of peat varied amongst the sampling plots, within the range 54–80 cm. In 2015 an increase in peat thickness of up to 10 cm was recorded at Plot 3 (Table A1 in Appendix). These shallow organic soils consisted mostly of strongly decomposed sapric peat (less than 10 % undecomposed fibre) overlying the sandstone-siltstone bedrock. According to the FAO-WRB classification (IUSS Working Group WRB 2015), they belonged to the soil reference group Histosols, additionally described by various principal and supplementary qualifiers (see Table A1). In the

Table 1. Selected chemical properties of peat water (mean values and standard deviation).

Plot	pH		EC (μs cm^{-1})		HCO_3^- (mg L^{-1})		Ca^{2+}		Mg^{2+}	
	2010	2015	2010	2015	2010	2015	2010	2015	2010	2015
1	7.01 ±0.04	7.14 ±0.01	318 ±2.00	323 ±0.58	179 ±3.51	170 ±8.89	60.7 ±2.07	59.8 ±3.94	3.22 ±0.33	3.16 ±0.33
2	7.22 ±0.07	7.23 ±0.08	304 ±2.89	310 ±2.65	170 ±2.08	171 ±7.02	57.3 ±1.80	58.0 ±1.86	2.73 ±0.12	2.78 ±0.23
3	7.10 ±0.02	7.16 ±0.06	302 ±2.65	306 ±2.65	157 ±3.61	159 ±7.00	53.2 ±3.13	54.8 ±2.79	2.09 ±0.10	2.59 ±0.54

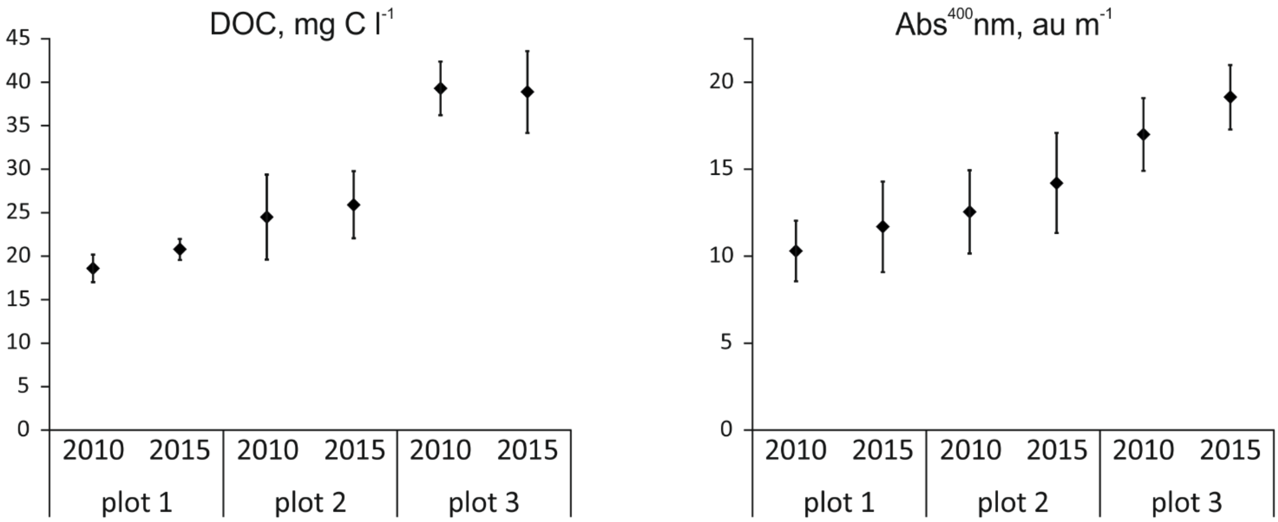

Figure 4. Average and standard deviation of DOC concentration (left-hand diagram) and absorbance (Abs) at 400 nm (right-hand diagram) in water. Comparison of data collected in 2010 and 2015.

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uppermost 20 cm of the soil profiles at Plots 1 and 3 a granular or weakly granular structure (peaty mursh) was observed. Five years after restoration the aggregate structure of this horizon at Plot 3 was looser, as a result of rewetting and disintegration of the aggregates.

The range of bulk density (BD) values recorded was 0.14–0.54 g cm^{-3} (Table A2). Values determined before and five years after restoration were very similar except in the bottom horizons (Ha4 at Plot 1; Ha5 at Plots 2 and 3), where lower BD values were recorded in 2015 (Table A2). The water-holding capacity index (W_1) values showed large differences between different soil horizons in both 2010 and 2015 (Table A2, Figure 5). The highest W_1 index values were recorded in the uppermost (mursh) horizons at Plots 1 and 3 in both of these years. However, the W_1 index values (indicating degree of secondary transformation) generally increased over the five years. Some soil horizons (*e.g.* Ha2 and Ha3 at Plot 2) were assigned to Class 0 (none) in 2010 and showed initial or weak secondary transformation in 2015. The only reversal occurred in Horizon Ha2 at Plot 3, where the W_1 value decreased and the secondary transformation class changed from weak (in 2010) to initial (in 2015).

The peat was moderately to slightly acidic, with pH in the range 5.6–6.2 in both years of sampling. The range of TOC content was 125–426 g kg^{-1} in 2010 and 170–437 g kg^{-1} in 2015, while TN ranged from 5.50 to 30.2 g kg^{-1} (Table A3). TOC/TN was lowest (13.5–17.9) in the surface soil horizons and increased with depth. The content of potentially labile forms of soil organic carbon (HWC) ranged from 0.80 to 3.39 g kg^{-1} in 2010 and from 0.73 to 3.51 g kg^{-1} in 2015 (Table A3). The highest HWC concentrations were found in the surface soil horizons, and the lowest in horizons directly overlying the mineral bedrock (Table A3). Statistical analysis of the entire dataset for all soil horizons showed a significant positive correlation between HWC and the W_1 index (at $p = 0.05$, $r = 0.443$) and a significant negative correlation between TOC/TN and HWC (at $p = 0.05$, $r = 0.657$) (Table 2). CWC was decidedly lower than HWC, at 0.18–1.04 g kg^{-1} in 2010 and 0.23–1.22 g kg^{-1} in 2015. The highest variability of CWC was recorded at Plot 2 (Table A3). This soil property was significantly negatively correlated with BD (at $p = 0.05$, $r = -0.451$) (Table 2).

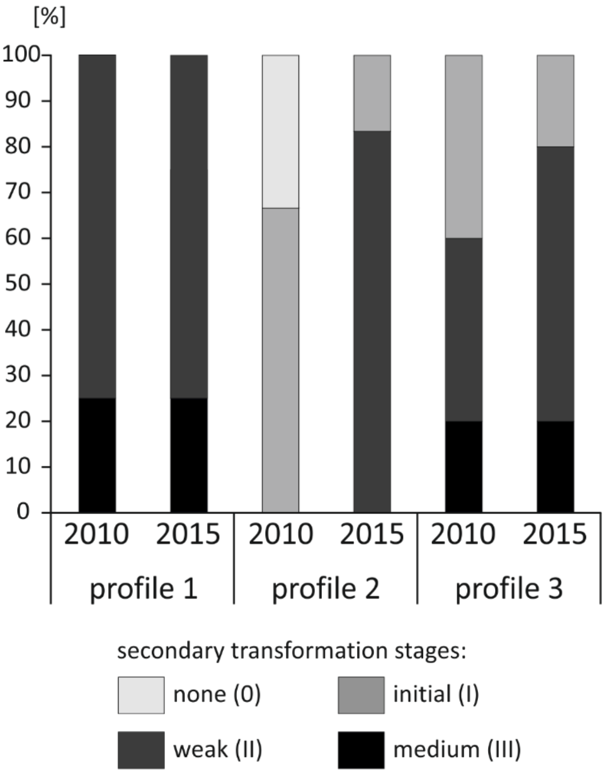

Figure 5. Secondary transformation stages of the studied soil horizons - percentage contribution (2010 and 2015 comparison).

Table 2. Pearson coefficients of correlations between selected soil properties (n = 70). Key: BD = bulk density; * = significant at p = 0.05.

	BD	W_1	HWC	CWC	TOC/TN
BD		0.123	-0.191	-0.451*	0.265
W_1	0.123		0.443*	-0.188	-0.219
HWC	-0.191	0.443*		0.502*	-0.657*
CWC	-0.451*	-0.188	0.502*		-0.362*
TOC/TN	0.265	-0.219	-0.657*	-0.362*	

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DISCUSSION

Effects of restoration on water quality

Hydrology is considered to be the most important factor affecting the proper functioning of peatland ecosystems (Rydin & Jeglum 2006, Waddington 2008). Moreover, change in water quality is one of the earliest noticeable effects of both degradation (Kalisz *et al.* 2015) and recovery (Haapalehto *et al.* 2011) of peatlands. A key indicator of changes is the concentration of labile carbon (DOC) in peat water (Koskinen *et al.* 2017), which results from destabilisation of the carbon stored in peat soils (Freeman *et al.* 2001). We hypothesised that a five-year restoration period should lead to a positive effect on both water table level and water quality. Favourable effects of restoration were observed only in relation to water table level, which was already very close to the ground surface (0.10–0.15 m below ground level) in the second year of recovery (2012). This is one of the most predictable consequences of peatland restoration (Wilson *et al.* 2011, Bogacz *et al.* 2012) and a good indication that hydrological restoration has been successful (Holden *et al.* 2011). A similar effect of restoration - a quick rise of the water table to almost ground level - has been reported in previously afforested fens in the USA (Haapalehto *et al.* 2011), Finland (Laine *et al.* 2011) and Poland (Nicia *et al.* 2017). Damming ditches and removing spruce trees definitely improved the hydrological conditions, although this was not the only determinant. In sloping fen peatlands fed mainly by seepage of groundwater, water table level can be strongly influenced at great distances beyond a ditch, as a result of the lateral flows of water during the whole season (Schimelpfenig *et al.* 2014). Despite the gradual decline in mean annual precipitation (Figure 3) during the study period, the water table rose due to the sustained inflow of groundwater from springs. A stable water table level indicates a low dependency on precipitation (McCarter & Price 2013).

Measured DOC concentrations in both years of observations were within the typical range for drained peatlands (10–60 mg L^{-1}) defined by Thurman (1985). Labile carbon concentrations in water, as well as water colour (Abs^{400}), were similar in both years of observation. Our findings are in line with those of Armstrong *et al*. (2010) and Wilson *et al.* (2011), who reported no change or a small decrease in DOC concentrations three years after ditch blocking. A similar restoration outcome has been observed in upland mires in the UK (Clark *et al*. 2008). However, this is contrary to the results of Kaila *et al*. (2016) and Koskinen *et al*. (2017) who found increased DOC concentrations in water 1–4 years after restoration of forestry-drained minerotrophic peatlands. Therefore, we can advocate ditch-blocking and clearcutting of trees to improve hydrological conditions. However, the impact on water quality will be different at every site, as a result of various controlling mechanisms (Clark *et al*. 2008) such as catchment characteristics (Bess *et al.* 2014, Wolf & Cooper 2015).

Effects of restoration on soil

Changes in water table level both prior to and after restoration affected the morphology and properties of the studied soils. Compaction of organic soils due to dewatering is well recognised in degraded peatland (*e.g.* Laiho & Pearson 2016), while in rewetted areas an increase in soil thickness (and surface altitude) associated with decreasing bulk density (BD) has often been reported (*e.g.* Anderson & Peace 2017). This is in line with our findings. The reduction in BD values observed in the bottom organic soil horizons can be associated with the expansion of peat material that led to the observed increase in thickness of the peat layer, which ranged from 2 cm at Plot 1 to 10 cm at Plot 3. We are aware that these results, based on discrete peat thickness measurements five years apart, may be slightly inaccurate. According to Parry *et al.* (2014b), a more robust approach would involve either manual probing or the installation of long rods anchored in the mineral material beneath the peat; however, such methods could also provide imprecise results. Because we took three replicate cores, we believe that our results are relatively reliable. Schmidt (1995) has reported rising of a fen peatland surface as an outcome of restoration after only two years of recovery.

The changes in structure of the surface soil horizon at Plot 3 were connected with rewetting and disintegration of aggregates. The consequence was finer structure in this layer. Thus, we assume that the weak aggregate structure in weakly murshified material may slightly lose its size and granular shape when subjected to permanent saturation. This assumption is supported by the reduction in W_1 index observed between 2010 and 2015 (Table A2). The changes in these soil physical properties indicate a positive effect of short-term restoration. However, our results are somewhat surprising when compared to those from other studies of sloping peatland restoration. For instance, Anderson & Peace (2017) observed a decrease in bulk density only after an interval of ten years had elapsed since restoration. Schimelpfenig *et al*. (2014) found that even 20 years

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was not a sufficient time period for recovery of soil properties, which they suggested may never be restored so that, instead, a new peat layer must form over the degraded peat.

The halting of carbon losses and re-establishment of carbon sequestration are some of the most frequently expected outcomes of peatland restoration (Strack *et al*. 2008, Poulin *et al*. 2012, Chimner *et al.* 2016). However, we found that the trend towards degradation caused by long-term forestry was not reversed in relation to organic matter transformation expressed as labile carbon forms content, TOC/TN values or the secondary transformation index W_1. Our observations have revealed that five years is insufficient time to restore these soil properties. The increased amount of labile carbon, especially the easily mineralisable CWC form, indicates that a substantial part of the soil carbon is still susceptible to efflux through fluvial pathways (Worrall *et al*. 2007). Moreover, the increase of HWC in the surface mursh horizons (in particular) might suggest an increase in microbial activity (Sparling *et al.* 1998) confirming ongoing mursh formation (Kalisz *et al.* 2015). Further transformation of organic matter is additionally confirmed by the lowered TOC/TN and increased W_1 (secondary transformation) values. Both of these indices are closely connected with the mursh-forming process which is typical for degraded organic soils (Kalisz *et al.* 2015, Glina *et al.* 2016a). In our study soils, surface mursh horizons were observed at Plots 1 and 3 (Table A1). Raising the water table did not result in a reduction of dissolved organic carbon in the soil or, thus, in the associated water. Discussion of this phenomenon is hampered by the lack of information about effects of restoration on sloping mountain fen soils. Available data chiefly relate to the magnitude of fluvial DOC export before and after restoration (*e.g*. Strack & Zuback 2012, Schimelpfenig *et al.* 2014, Kaila *et al*. 2016, Koskinen *et al*. 2017). However, in the lysimeter study presented by Schwalm & Zeitz (2015) it was found that raising the water table was not sufficient to reduce DOC in fen soils, and the crucial factors influencing DOC release were the degree of peat decomposition and soil pH. Such interesting findings should be taken into account when planning restoration of fen peatlands.

Effects of restoration on vegetation

Changes in vegetation structure due to forestry drainage progress more rapidly on fens than on bogs (Komulainen 1999). Dense tree stands cause shading and enhance the effects of drainage, so mechanical removal of trees and shrubs is recommended as best practice for restoration (Schumann & Joosten 2008). This also creates better conditions for growth and development of plant species with low competitive strength (Ryś 2011). To restore rich fen peatlands, it may sometimes be advantageous to apply various types of strategies to recreate conditions that facilitate establishment of the desired species (Mälson *et al*. 2009).

Positive feedback in terms of development of the desired vegetation within the studied fen proves that the methods used are sufficient to raise the water table and limit nutrient pools. In the first years after restoration, spontaneous recolonisation by sedges (*e.g*. *Carex paniculata*, *Carex flava)* and horsetails (*e.g. Equisetum limosum, Equisetum fluviatile)*, which are typical vascular plants for fens (Rydin & Jeglum 2006), was observed. Another good indicator of improved hydrological conditions was the appearance of *Veratrum lobelianum* (Melanthiaceae). This genus prefers full sunlight and wet soils (Mirek *et al*. 1995). Another important species which returned to the studied peatland is the grey alder *Alnus incana*, a typical tree species for poor to rich mountain fen peatlands (Edvardsson *et al*. 2016) which is common in western and central Europe (Lang 1952). Palaeoecological research has shown that the immediate vicinity of the study peatland was covered by riparian forest dominated by *Alnus* from *ca*. 3320 BP to 650 BP. Moreover, the results of macrofossil analysis have demonstrated that, while the uppermost parts of the peat profile of the study site are composed of *Carex* and *Equisetum* remains, the soil layer below 50 cm depth contains *Alnus* wood and bark. This confirms that *Alnus* stands formerly occurred actually on the study site (Glina *et al.* 2017).

The positive outcome of our restoration project after five years is the natural recolonisation by peat-forming plants which were important components of the habitat until spruce monocultures were introduced by humans almost 120 years ago (Glina *et al*. 2017). This probably stems from peatland rewetting and the presence of viable seed and spores in the soil. In other studies on mountain fens (*e.g*. Mälson *et al*. 2009, Haapalehto *et al*. 2011, Laine *et al*. 2011), authors have noted that timespans of 3–10 years are needed for peatland flora to respond to restoration. Our short-term (5-year) observations of flora gave quite similar but somewhat faster outcomes, in that initial changes were already observable 1–2 years after restoration. The complexity of hydrological patterns and processes within fen peatlands means that all restoration outcomes may not be visible within the same time frame, and this account of short-term effects in a mountain fen brings new findings to the current state of knowledge.

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Appendix

Table A1. Morphology and classification of the soil profiles at the three sampling plots. For ‘Structure’: A = amorphous, GR = granular, L = lumpy, F = fibrous, W = weak. For ‘Horizon boundary’: G = gradual, C = clear, W = wavy.

Soil horizon	Depth (cm)		Colour (moist)	Structure		Horizon boundary	Wood fragments	Material type
	2010	2015		2010	2015			
Plot 1 - Eutric Rheic Sapric Murshic Histosol (Lignic)								
Ha1	0–12	0–12	10YR 2/1	GR	GR	G	-	mursh
Ha2	12–25	12–25	10YR 2/2	GR	W-GR	CW	-	mursh
Ha3	25–40	25–40	10YR 3/4	A-L	A-L	G	+	sapric peat
Ha4	40–54	40–56	10YR 3/3	A-L	A-L	G	+	sapric peat
Plot 2 - Eutric Rheic Sapric Drainic Histosol (Lignic)								
He	0–7	0–7	10YR 2/1	A-F	A-F	G	-	hemic peat
Ha1	7–22	7–22	10YR 3/1	A	A	G	-	sapric peat
Ha2	22–37	22–37	10YR 3/2	A-L	A-L	G	+	sapric peat
Ha3	37–55	37–55	10YR 3/2	A-L	A-L	CW	+	sapric peat
Ha4	55–70	55–70	10YR 3/3	A	A	G	-	sapric peat
Ha5	70–80	70–85	10YR 4/3	A	A	G	-	sapric peat
Plot 3 - Eutric Rheic Sapric Murshic Histosol (Lignic)								
Ha1	0–10	0–10	10YR 2/1	GR	GR	G	-	mursh
Ha2	10–20	10–20	10YR 2/1	W-GR	GR-A	CW	-	mursh
Ha3	20–35	20–35	10YR 3/2	A-L	A-L	G	+	sapric peat
Ha4	35–50	35–50	10YR 4/4	A-L	A-L	G	+	sapric peat
Ha5	50–80	50–90	7.5YR 4/4	A-L	A-L	G	+	sapric peat

Table A2. Soil physical and physicochemical properties (mean values) of soil horizons.

Plot	Soil horizon	Bulk density (g cm^{-3})		Fibre content (%)		Index W_1		pH H_2O	
		2010	2015	2010	2015	2010	2015	2010	2015
1	Ha1	0.22	0.21	-	-	0.62	0.69	5.8	5.9
	Ha2	0.21	0.20	-	-	0.47	0.50	5.7	5.7
	Ha3	0.25	0.25	5	5	0.57	0.60	5.7	5.6
	Ha4	0.54	0.34	4	4	0.47	0.43	6.0	5.9
2	He	0.16	0.16	10	9	0.45	0.48	6.2	6.1
	Ha1	0.17	0.16	9	7	0.39	0.59	5.8	5.9
	Ha2	0.16	0.15	6	6	0.31	0.45	5.9	5.7
	Ha3	0.17	0.14	6	6	0.32	0.48	5.8	5.7
	Ha4	0.17	0.14	5	6	0.36	0.46	6.0	6.1
	Ha5	0.24	0.17	6	6	0.45	0.46	5.8	5.9
3	Ha1	0.22	0.20	-	-	0.62	0.64	6.0	6.0
	Ha2	0.20	0.18	-	-	0.49	0.45	5.8	5.8
	Ha3	0.17	0.17	4	5	0.42	0.56	5.7	5.6
	Ha4	0.14	0.16	5	5	0.41	0.55	5.8	5.8
	Ha5	0.26	0.19	8	7	0.49	0.49	5.9	5.8

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Table A3. Soil chemical properties (mean values).

Plot	Soil horizon	TOC (g kg^{-1})		TN (g kg^{-1})		TOC/TN		HWC (g kg^{-1})		CWC (g kg^{-1})	
		2010	2015	2010	2015	2010	2015	2010	2015	2010	2015
1	M1	346	358	19.3	23.8	17.9	15.0	2.39	2.51	0.43	0.39
	M2	315	338	18.2	22.8	17.3	14.8	1.78	1.97	0.41	0.40
	Ha1	232	290	11.6	16.5	20.0	17.6	1.32	1.44	0.18	0.23
	Ha2	125	170	5.50	9.10	22.7	18.7	0.88	0.92	0.22	0.40
2	He	401	394	28.7	26.5	14.0	14.9	2.64	2.82	1.02	1.22
	Ha1	404	407	26.9	30.2	15.0	13.5	2.15	2.13	1.04	1.21
	Ha2	418	422	24.4	28.6	17.1	14.8	1.65	1.77	0.85	0.84
	Ha3	397	435	18.9	25.7	21.0	16.9	1.52	1.58	0.81	0.84
	Ha4	408	437	20.6	19.1	19.8	22.9	0.98	1.04	0.52	0.57
	Ha5	336	398	13.0	15.5	25.8	25.7	1.12	1.17	0.56	0.57
3	M1	399	392	27.1	26.9	14.7	14.6	3.39	3.51	0.66	0.64
	H/M	395	410	26.7	24.7	14.8	16.6	1.38	1.91	0.41	0.54
	Ha1	405	420	26.6	21.4	15.2	19.6	0.80	1.08	0.20	0.37
	Ha2	426	431	23.4	19.9	18.2	21.7	1.04	0.73	0.25	0.59
	Ha3	335	316	15.8	13.1	21.2	24.2	1.01	1.06	0.23	0.47

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