HYDROLOGICAL PROCESSES
Hydrol. Process. **22**, 3373–3385 (2008)
Published online 20 March 2008 in Wiley InterScience
(www.interscience.wiley.com) DOI: 10.1002/hyp.6931

Effect of water table drawdown on peatland dissolved organic carbon export and dynamics

M. Strack,[1]* J. M. Waddington,[2] R. A. Bourbonniere[2,3] E. L. Buckton,[2] K. Shaw,[2]
P. Whittington[4] and J. S. Price[4]

[1] *Department of Geography, University of Calgary, Calgary, AB, Canada*
[2] *School of Geography and Earth Sciences, McMaster University, Hamilton, ON, Canada*
[3] *Environment Canada, National Water Research Institute, Burlington, ON, Canada*
[4] *Department of Geography, University of Waterloo, Waterloo, ON, Canada*

Abstract:

Peatlands play an important role in the global carbon cycle, and loss of dissolved organic carbon (DOC) has been shown to be important for peatland carbon budgets. The objective of this study was to determine how net production and export of DOC from a northern peatland may be affected by disturbance such as drainage and climate change.

The study was conducted at a poor fen containing several pool–ridge complexes: (1) control site–no water table manipulation; (2) experimental site–monitored for one season in a natural state and then subjected to a water table drawdown for 3 years; (3) drained site–subjected to a water table drawdown 9 years prior to monitoring. The DOC concentration was measured in pore water along a microtopographic gradient at each site (hummock, lawn and hollow), in standing water in pools, and in discharge from the experimental and drained sites. The initial water table drawdown released ~3 g of carbon per square metre in the form of DOC, providing a large pulse of DOC to downstream ecosystems. This value, however, represents only 1–9% of ecosystem respiration at this site. Seasonal losses of DOC following drainage were 8–11 g of carbon per square metre, representing ~17% of the total carbon exchange at the experimental study site. Immediately following water table drawdown, DOC concentrations were elevated in pore water and open water pools. In subsequent seasons, DOC concentration in the pool declined, but remained higher than the control site even 11 years after water-table drawdown. This suggests continued elevated net DOC production under lower water table conditions likely related to an increase in vegetation biomass and larger water table fluctuations at the experimental and drained sites. However, the increase in concentration was limited to initially wet microforms (lawns and hollows) reflecting differences in vegetation community changes, water table and soil subsidence along the microtopographic gradient.

KEY WORDS biogeochemistry; carbon cycling; DOC; peatland; hydraulic conductivity; drainage; climate change; wetland

Received 3 January 2007; Accepted 10 October 2007

INTRODUCTION

Northern peatlands play a significant role in the global carbon cycle and sequester approximately one-third of the global soil carbon pool (Gorham, 1991) primarily through peat accumulation in anoxic soils (Clymo, 1984). Whereas many studies suggest that carbon continues to accumulate in peat soils (e.g. Waddington and Roulet, 1996; Bubier *et al.*, 1999; Asada and Warner, 2005), most do not account for hydrologic losses of carbon. Such losses can be important when determining carbon storage in peatlands (Billett *et al.*, 2004) and may be increasing (Freeman *et al.*, 2001; Worrall *et al.*, 2003).

Dissolved organic carbon (DOC) is operationally defined as organic carbon that is smaller than 0·45 µm in diameter (Thurman, 1985). DOC is found in concentrations ranging from 3 to 400 mg l^{-1} in natural peatlands and averages 30 mg l^{-1}, whereas DOC export from peatlands is 5–40 g m^{-2} $year^{-1}$ (Thurman, 1985; Moore, 1998). Moreover, the proportion of peatland area within a watershed is related to DOC export (Koprivnjak and Moore, 1992; Mattsson *et al.*, 2005) and DOC loss may be important for determining peatland carbon balances (Moore, 1998; Billett *et al.*, 2004). In addition, DOC plays many important ecological and geochemical roles both within peatlands and in downstream ecosystems, affecting acidity, nutrient availability, metal mobility and light penetration in aquatic habitats (Steinberg, 2003).

DOC export from peatlands occurs in two stages: (1) the production of DOC and (2) export. High net production in peatlands is due to the largely anaerobic soil conditions (e.g. Moore and Dalva, 2001). Increases in primary productivity can result in higher concentrations of peatland DOC (Freeman *et al.*, 2004). DOC export is controlled by site hydrology (Hinton *et al.*, 1997; Fraser *et al.*, 2001; Pastor *et al.*, 2003) and is generally greater from peatlands with higher measured discharge (Fraser *et al.*, 2001; Freeman *et al.*, 2001). Storm events also play a large role in DOC export, causing flushing of DOC-rich water from peatlands to downstream ecosystems (Hinton *et al.*, 1997). Thus, any disturbance altering factors affecting DOC production or hydrology could potentially change the quantity and chemistry of exported DOC.

* Correspondence to: M. Strack, Department of Geography, University of Calgary, Calgary, AB, Canada. E-mail: mstrack@ucalgary.ca

Northern peatland water tables may be lowered by drainage (e.g. Moore and Roulet, 1993; Price, 2003) or climate change (Roulet *et al.*, 1992). Many peatlands are drained for forestry, agriculture or peat extraction (Moore, 2002; Joosten and Clarke, 2002). This process involves the construction of ditches, thereby increasing both peatland discharge (Van Seters and Price, 2002) and DOC export. Also, lowered water tables will initially cause surface subsidence as a result of peat compression (Price and Schlotzhauer, 1999; Price, 2003). This, in turn, decreases hydraulic conductivity and specific yield and increases the magnitude of water table fluctuations (Price and Schlotzhauer, 1999; Price, 2003; Whittington and Price, 2006). Through time, continued subsidence occurs as a result of peat oxidation (Schothorst, 1977; Price and Schlotzhauer, 1999; Price, 2003).

Under climate change, persistent water table drawdown may result in the maintenance of the water table at a level, relative to the surface, that is similar to pre-drainage conditions as a result of continued peat subsidence (e.g. Whittington and Price, 2006). In addition, a change in the vegetation community may occur as observed in naturally (Foster *et al.*, 1988) and artificially drained peatlands (Minkkinen *et al.*, 2002; Strack *et al.*, 2006). If this results in increased vegetation productivity then it may enhance production and export of DOC (Freeman *et al.*, 2004). Thus, long-term DOC dynamics are influenced by ecological as well as physical and hydrological changes. This complex interplay of processes leads to questions regarding the applicability of short-term drought studies when considering climate change scenarios.

Therefore, the aim of this study was to compare the concentration of DOC in pore water and surface water, and DOC export between a natural peatland, a peatland with newly imposed drainage and water table drawdown, and a peatland with the water table lowered 9 years prior to the study. Specifically, the objectives were (1) to determine the effect of peatland drainage on DOC export during the initial ditching, subsequent two growing seasons and following 9 to 11 seasons of persistent water table drawdown, (2) to determine the impact through time of persistent water table drawdown on DOC dynamics in peatland pore water and open water pools, (3) to determine whether changes in pore water DOC dynamics induced by water table drawdown varied between peatland microforms, and (4) to investigate ecohydrological controls, such as water table and vegetation productivity, on peatland DOC dynamics. The results will provide insight regarding the interaction between hydrology and ecology for controlling peatland DOC dynamics, provide information about the potential response of peatland DOC dynamics to climate change, and provide an estimate of DOC exported via peatland drainage.

MATERIALS AND METHODS

Study area

The study site is a poor fen fragment (see Kellner *et al.* (2004)) with peat depth of 0·8–1·5 m underlain by clay mineral soil located in St-Charles-de-Bellechasse, Québec (46°40′N, 71°10′W). The 30-year normal (1971–2000) precipitation and temperature from May to September for the region are 590 mm and 15·5 °C respectively (climate data available from Environment Canada at http://climate.weatheroffice.ec.gc.ca). The study was carried out between 2001 and 2004, during which time 2001 and 2002 were drier than normal and 2003 and 2004 were close to the long-term average (Table I).

To examine the effects of peatland drainage on DOC dynamics through time, three pool–ridge complexes were compared (Figure 1). The first did not undergo any water table manipulation (control, C). The second (experimental, E) pool–ridge complex was monitored for one season (2001) and then subjected to a water table drawdown with a ditch connecting the pool to the larger drainage network. The third pool–ridge complex (drained, D) underwent water table drawdown by the peatland operators in 1993. Sampling was conducted along the microtopographic gradient at each site, with the driest ridge locations referred to as hummocks, intermediate locations as lawns, low-lying areas as hollows and open water areas as pools. At D, the water table drawdown almost completely eliminated the open water area; flooding only occurred in the spring and after intense summer storms. Also, water table drawdown has resulted in observable changes to the vegetation community at E and D. In general, hummocks experienced a reduction in moss cover, lawns

Table I. Climatic data from Québec City (30 km from the St Charles-de-Bellechasse research site) during the four seasons of the study and the 30-year normal (1971–2000) for the same location. Precipitation gives monthly totals and temperature, monthly mean

		May	June	July	August	September	Seasonal
2001	Precipitation (mm)	58·4	103·1	72·2	105·7	93·8	433·2
	Temperature (°C)	13·5	17·6	17·7	18·8	14·0	16·3
2002	Precipitation (mm)	107·6	67·7	63·4	11·8	107·7	358·2
	Temperature (°C)	9·5	14·9	19·8	19·4	15·6	15·8
2003	Precipitation (mm)	100·6	97·4	129·6	153·2	85·0	565·8
	Temperature (°C)	10·8	17·1	18·4	18·1	15·3	15·9
2004	Precipitation (mm)	122·9	117·0	168·8	71·6	115·6	595·9
	Temperature (°C)	10·5	14·9	19·5	17·7	13·7	15·3
Normal	Precipitation (mm)	105·5	114·2	127·8	116·7	125·5	589·7
	Temperature (°C)	11·2	16·5	19·2	17·6	12·5	15·5

experienced an increase in vascular plant cover, and hollows/pools experienced an increase in moss and vascular plant cover relative to C (Table II). For more information on vegetation changes, see Strack *et al.* (2006), Strack and Waddington (2007) and St-Arnaud (2007).

Across the site there is a gradient of approximately 50 cm per 100 m, with C at the highest elevation followed by E and then D. Hydrology and DOC dynamics were monitored at all sites between July 2001 and August 2004, with measurements generally carried out between May and September of each study season.

Dissolved organic carbon concentration

DOC concentration was determined in surface and pore water at each site and in discharge from E and D. Pore water samples were collected from piezometer nests installed throughout the peat depth and along the microtopographic gradient at each site. One nest of piezometers was placed at a hummock, lawn and hollow microform at each site. Samples were collected monthly throughout the study period; however, in 2001, only lawn areas were sampled and no pore water samples were collected from D. All sites had piezometers with 20 cm slotted intakes centred at depths 25, 50 and 75 cm below the surface with 100 cm also present at E, and 100 and 125 cm also present at C due to greater peat depths. Below the piezometer intake was a reservoir with a volume of at least 100 ml to collect water for determination of DOC concentration. Prior to sampling, the reservoir was pumped dry and allowed to refill. Fresh pore water was then collected in a clean Nalgene bottle after first rinsing the bottle with a small amount of sample. Some piezometers refilled quickly and could be sampled within 1 day of pumping, whereas others refilled more slowly (particularly at depth) due to the low hydraulic conductivity of the peat. If at least 50 ml of sample could not be collected after 3 days, then the DOC concentration was not determined for that location.

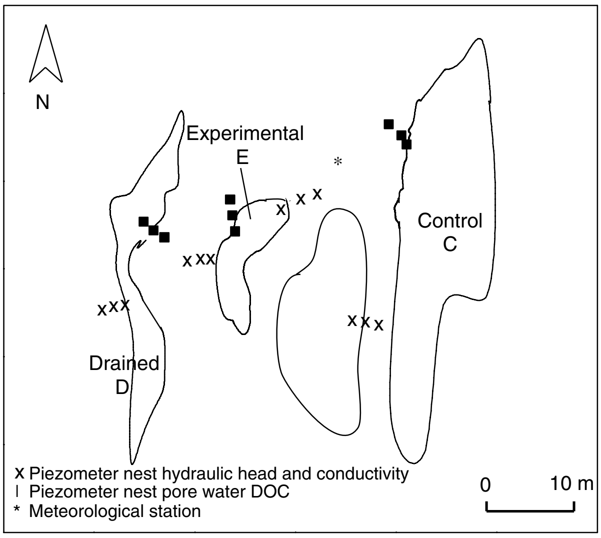

Figure 1. St Charles-de-Bellechasse experimental fen showing the extent of the hollow area at each of the C, E and D sites. Piezometers used for pore water DOC sampling and determination of subsurface flow are identified. The unlabelled hollow is an additional natural pool–ridge area used to assess influx of water into and between natural zones

Table II. Average vegetation cover at each site along the microtopographic gradient in 2004 following 3 and 11 seasons of water table drawdown at the experimental and drained sites respectively

Site	n	Microform	Cover, average ± SD (%)	
			Vascular	Moss
Control (C)	7	Hummock	51 ± 21	83 ± 30
	19	Lawn	25 ± 20	74 ± 38
	29	Hollow/Pool	13 ± 8	7 ± 18
Experimental (E)	13	Hummock	65 ± 26	59 ± 42
	18	Lawn	48 ± 31	61 ± 38
	21	Hollow/Pool	53 ± 31	6 ± 19
Drained (D)	16	Hummock	60 ± 27	43 ± 36
	12	Lawn	75 ± 23	62 ± 35
	23	Hollow/Pool	51 ± 24	71 ± 29

Surface water from the pools and drainage outflow pipes was sampled every 2 to 3 weeks over the 2002 and 2003 field seasons and monthly in 2001 and 2004. Nalgene bottles were rinsed with pool or discharge water and then a sample was collected.

All samples were filtered first with a 1·5 μm glass-fibre filter (Whatman 934-AH) followed by a 0·7 μm glass-fibre filter (Whatman GF/F) and were then refrigerated and transported to the National Water Research Institute where they underwent chemical analysis. The definition of dissolved is most often made using 0·45 μm pore-size membrane filters, but polymeric filters often bleed DOC into samples. The use of pre-rinsed glass-fibre filters eliminates bleed, but compromises the definition of dissolved somewhat because these filters are characterized as having nominal rather than exact pore sizes. The GF/F filters (Whatman) used in this study were also used by others in the field (e.g. Qualls and Haines, 1991, Mattsson *et al.* 1998, Brooks *et al.*, 1999) and have a nominal pore size of 0·7 μm. Filtered samples from this study held for archival purposes at 4 °C remained stable showing no flocculation of colloids during storage for up to 3 years.

DOC concentrations were determined using an Apollo 9000 Total Carbon Analyzer (Techmar-Dohrmann) employing platinum on aluminium oxide catalyst at 900 °C. Samples were routinely diluted using EPure (Barnstead) water to ensure they fell within the calibration range (carbon concentration [C] = 0–20 mg l^{-1}). Samples were acidified with 20% phosphoric acid just prior to analysis and were then sparged for 5 min with O_2 carrier gas to remove dissolved inorganic carbon. Each sample was injected six times and the four nearest replicates were averaged by the Apollo software. Within each analysis run of 70 vials, a calibration set, five blank and five check standards were analysed along with the samples. Blank and drift correction curves were constructed by regressing the deviations from the true values of blank and check standards respectively against vial sequence number, and these curves were applied to the initial results on each sample, which were then finally corrected

DOI: 10.1002/hyp

for dilution. Standard deviations for replicate DOC measurements fell between [C] = 0·03 and 1·46 mg l^{-1} with an average standard deviation of [C] = 0·4 mg l^{-1}.

Inter-microform dissolved organic carbon flux

In order to determine the flux of DOC between sites and from hummock to hollow microforms within each site, water flow was computed according to Darcy's law and multiplied by pore water DOC concentrations:

$$F = [\mathrm{DOC}]KA\ \mathrm{d}h/\mathrm{d}l \qquad (1)$$

where [DOC] is the average concentration of DOC in 2004 for pore water at 50 cm depth, K is the geometric mean of hydraulic conductivity throughout the peat profile, A is the cross-sectional area of flow, and dh/dl is the hydraulic gradient from hummock to hollow. One additional transect of piezometer nests was installed in 2004 from hummock to pool at the upslope and downslope edge of each site. Hydraulic conductivity was determined on several occasions throughout 2004 at lawns at 50, 75 and 100 cm depths using bail tests (Freeze and Cherry, 1979). The hydraulic gradient was determined as the difference in absolute water table position between the hummock and hollow across the horizontal distance of the piezometer transect. The cross-sectional area of flow was estimated based on the perimeter of the lawn zone at each site multiplied by an average peat depth of 1·4 m, 1·2 m and 0·8 m at the C, E and D respectively. Uncertainty in this flux was determined according to Bubier *et al.* (1999) using a first-order Taylor series and assuming that uncertainty in the parameters was independent. Uncertainties in K and [DOC] were based on the standard deviation of measurements taken throughout the study season, whereas the uncertainties for A, dh and dl were estimated as 10%, 0·2 cm and 0·5 m respectively.

Since water generally flowed from hummock to hollow on the upslope edge of each site and hollow to hummock on the downslope edge, the net flux of DOC was determined as the balance of these and expressed on an area basis relative to the estimated area of the hummocks and hollow/pool at each site. Uncertainty in the net flux was determined as the square root of the sum of the variances of the DOC input and output.

Hydrology and dissolved organic carbon export

Water table position was monitored manually during all study seasons in 2·5–3 cm i.d. PVC wells installed systematically throughout the study sites. In 2001, water table was measured continuously with a counterbalanced pulley on a potentiometer at a central meteorological station. This was connected to a CR21X data logger (Campbell Scientific, Alberta, Canada) measured each minute and averaged every 20 min. In 2002–2004, the water table was measured continuously at lawns at each site with counterbalanced pulleys and potentiometers connected to CR10X data loggers (Campbell Scientific). RDS (Remote Data System) wells were also used at each site to measure water table position in 2004.

Discharge from the experimental and drained sites were funnelled into horizontal PVC pipes which provided outflow to the drainage ditches. Discharge was measured several times a week from 2002 to 2004 at these outflow pipes for the experimental and drained sites. Flow was collected into a graduated container over a timed interval and discharge was calculated. The reported values represent averages of three measurements. DOC export was determined for E in 2002–2004 and for D in 2003 and 2004. To determine export, manual discharge measurements were regressed against water table position and this relationship was used to estimate daily discharge. DOC concentrations were interpolated between measurement dates and multiplied by calculated discharge rates to determine daily export values (modified from Hinton *et al.* (1997)). Owing to limited discharge data for D in 2002, export was not determined for this season. Since export was determined as the product of discharge and DOC concentration in the discharge water, uncertainty was approximated as (Devito *et al.*, 1989)

$$S_{\mathrm{E}}^2 = Q^2 S_{\mathrm{DOC}}^2 + [\mathrm{DOC}]^2 S_Q^2 + S_{\mathrm{DOC}}^2 S_Q^2 \qquad (2)$$

where S_{E} is the standard deviation of the export estimate, S_{Q} is the standard error of the water table–discharge relationship, S_{DOC} is the standard deviation of discharge DOC concentrations, [DOC] is the average DOC concentration in the discharge and Q is the average daily discharge.

The catchment area for each pool (C, E and D) was determined based on peat surface topography. Since surface water discharge was of interest, catchment area was determined based on surface topography, as it has been shown that, within patterned peatlands, little surface flow occurs when surface pools are disconnected (e.g. Quinton and Roulet, 1998), which was the case throughout study period. Although some subsurface water movement occurs (as determined above) and may contribute to surface water outflow, its magnitude is generally less than 10% of surface discharge. Also, since surface discharge ceases soon after standing water disappears, subsurface flow likely plays a minimal role in surface water loss. The estimated catchment size for each of the sites is ~900 m^2. The total export value (grams of DOC-C) for the season was divided by the catchment area size to obtain export on an area basis.

Since discharge was not determined during the initial drainage at E in 2002, the mass of DOC lost per unit area was determined by multiplying the depth of water lost (~20 cm) by peat porosity (~0·9 according to Baird and Gaffney (1995)) and average DOC concentration measured in the drainage water.

Statistical analysis

Differences in water table position between sites and DOC concentrations between sites and microforms were assessed with one-way analysis of variance (ANOVA) with Tukey pairwise comparisons at a family error rate

DOI: 10.1002/hyp

of 0·05. Regression lines between DOC concentrations and environmental variables were fitted using the least-squares method and were considered significant when $p < 0.05$. All analyses were completed in Minitab 14·1 (Minitab Inc.). Because all piezometers could not be sampled on each sampling date due to low recharge rates at some locations, sample size for statistical analysis varied between microforms and sites. In all cases, microform averages are based on greater than six samples, and in general the sample size was greater than 12.

RESULTS

Water table position

Based on manual measurements at lawns in 2001 (prior to water table manipulation at E), average plus/minus standard deviation water table position was significantly deeper (ANOVA, $p < 0.05$) at C (-9.5 ± 3.2 cm, $n = 23$, where a negative value indicates below the soil surface) than the E (-5.2 ± 2.5 cm, $n = 23$). D had a water table of -14.0 ± 1.9 cm ($n = 17$) that was significantly lower than both C and E. Following water table drawdown on day 161 of 2002, the water table dropped rapidly at E and remained significantly deeper than C throughout the remainder of the study (Figure 2). Fluctuations in water table position were related to precipitation events; however, the range of these fluctuations varied between sites. In 2004, the range for water table position during the growing season was 11·6 cm, 14·1 cm and 16·9 cm at C, E and D respectively (see also Whittington and Price (2006)).

Pore water dissolved organic carbon concentrations

Average plus/minus standard deviation pore water concentrations were 18.7 ± 3.8 mg l^{-1}, 24.4 ± 9.5 mg l^{-1} and 28.1 ± 12.0 mg l^{-1} at C, E and D respectively ($n = 61$, 69, 41 at C, E and D respectively) during the first year of the study. Thus, pore water concentration during this time was significantly lower at C than at E, which was lower than at D (ANOVA, $p < 0.05$). Because DOC concentrations were initially significantly higher at E than at C, it is difficult to evaluate changes in pore water concentration related to water table drawdown statistically. However, in all seasons following drainage, average E pore water concentrations were elevated above their initial levels and similar to those at the drained site (Figure 3).

In a natural state (C and E prior to water table drawdown), pore water DOC concentration varied significantly along the microtopographic gradient, this being highest at hummocks, intermediate at lawns and lowest at hollows (ANOVA, $p < 0.05$). There was no consistent pattern of DOC concentrations with depth at any microform; however, DOC concentration increased with depth at E and D hollows in 2002. This pattern was much less apparent in 2004 (Figure 4). Following water table drawdown (E after water-table drawdown and D), there was no consistent pattern for DOC concentration between microforms.

At E there were few significant changes in DOC concentration before and after water table drawdown. The only statistically significant changes were at −50 cm at lawns and hollows, which had higher concentrations in the first season after water-table drawdown compared with initially. In general, there was a trend for pore water DOC concentrations to decline at the hummock and increase at lawn and hollow following water table drawdown (Figures 3 and 4).

Inter-microform dissolved organic carbon flux

Geometric mean hydraulic conductivity in the peat profile declined from approximately 10^{-4} cm s^{-1} at C to 10^{-4}–10^{-5} cm s^{-1} at E and 10^{-5}–10^{-6} cm s^{-1} at D. Water table drawdown resulted in subsidence of the peat surface, with a greater change in peat volume occurring in low-lying areas than in neighbouring hummocks. This uneven subsidence resulted in steeper hydraulic gradients from the hummock to the pool at the E and D. The combined effect of these changes on site properties resulted in decreased fluxes of water between hummocks and pools from about 2 m^3 day^{-1} at C to 0·3 m^3 day^{-1} at E and 0·06 m^3 day^{-1} at D. The change in total fluxes of DOC between microforms followed the hydrological changes.

Despite the fact that more DOC moved between microforms at C, the inputs and outputs of DOC were nearly equivalent, whereas more water and DOC entered the pool at E than left from the downslope edge, resulting in a source of DOC-C to the pool of 5 g m^{-2} over the study season (Table III). Owing to the low hydraulic conductivity at D, very little carbon moved between microforms.

As a result of the overall gradient of the peatland, approximately 4 g m^{-2} DOC-C were transferred from the C to E, whereas ∼0·3 g m^{-2} DOC-C moved between E and D during the 2004 study season. Comparing the flux of carbon between C and a neighbouring natural pool, 0·4 g m^{-2} DOC-C moved between natural pools over the same period.

Surface water dissolved organic carbon concentrations

The prior water table drawdown DOC concentration of surface water in pools was not significantly different between C and E, with average plus/minus standard deviation values of 16.2 ± 4.71 mg l^{-1} ($n = 12$) and 22.4 ± 6.1 mg l^{-1} ($n = 11$) respectively. During this period, average DOC concentration at the D was 25.5 ± 3.4 mg l^{-1} ($n = 5$), which was significantly higher (ANOVA, $p < 0.05$) than C but not E (Figure 2). During the first season after water-table drawdown, E pool concentrations increased substantially, averaging 40.5 ± 22.0 mg l^{-1} ($n = 8$), whereas C and D pool concentrations were on average 19.0 ± 2.1 mg l^{-1} ($n = 10$) and 30.0 ± 6.1 mg l^{-1} ($n = 5$) respectively. During this time, DOC concentration at E pool was significantly

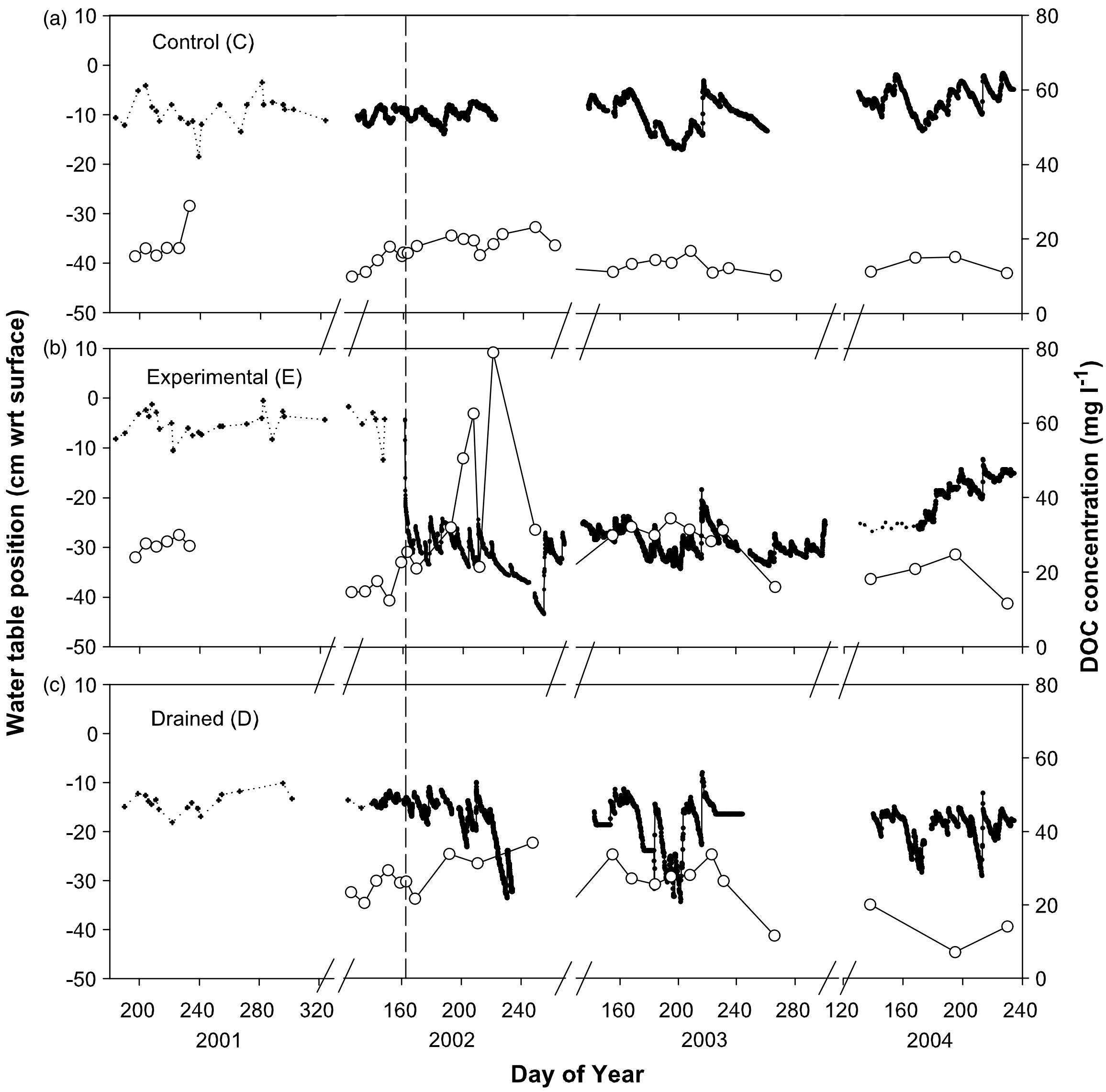

Figure 2. Water table position at lawns (solid symbols) and pool DOC concentrations (open symbols) throughout the study period. The vertical dashed line indicates the date of drainage at the E pool. The dotted line gives manual water table measurements during periods when continuous measurements were unavailable. Pool DOC concentrations in 2003 on day 121 were 12·0 mg l^{-1}, 19·4 mg l^{-1} and 18·0 mg l^{-1} at C, E and D respectively (not shown)

higher than C and D. Large variability observed in E pool DOC concentrations appear to be related to dilution by precipitation (Figure 2). By the second season, post-water-table drawdown (2003) E concentrations declined slightly but remained significantly higher than C. The average concentrations at C, E and D were $12{\cdot}6 \pm 1{\cdot}9$ mg l^{-1}, $27{\cdot}2 \pm 6{\cdot}6$ mg l^{-1} and $24{\cdot}2 \pm 8{\cdot}3$ mg l^{-1} ($n = 10$) respectively. During the third season after drawdown (2004) there were no significant differences between pools.

Discharge and dissolved organic carbon export

Throughout the 2 days following drainage at E, DOC concentration in discharge water was determined every few hours. The concentration increased from 17·3 to 21·7 mg l^{-1} over the first 3 h after drainage and declined to 20·7 mg l^{-1} over the following 2 days. Based on the water table decline, pool catchment area and peat porosity, it was estimated that $1{\cdot}62 \times 10^5$ l (180 mm) of water was lost during this initial drainage. Assuming that this water had an average concentration equivalent to the mean of all measurements taken during the first week following the initial ditch opening, i.e. 20·3 mg l^{-1}, an estimated DOC-C of $3{\cdot}7 \pm 1{\cdot}7$ g m^{-2} was lost during this period from E. During the remainder of the growing season (days 166–230), DOC-C loss from the E was $5{\cdot}2 \pm 1{\cdot}1$ mg m^{-2} (Table IV). Despite the fact that 2004 had the greatest amount of precipitation during the study (Table I), discharge was highest in 2003 (Table IV). This was primarily due to precipitation distribution. One large storm in August resulted in a large proportion of the discharge in 2003, whereas precipitation and discharge in 2004 were more evenly distributed. During these two growing seasons (days 128–230), DOC-C export from the E and D sites were similar, ranging between 8·4 and 11·3 g m^{-2} (Table IV).

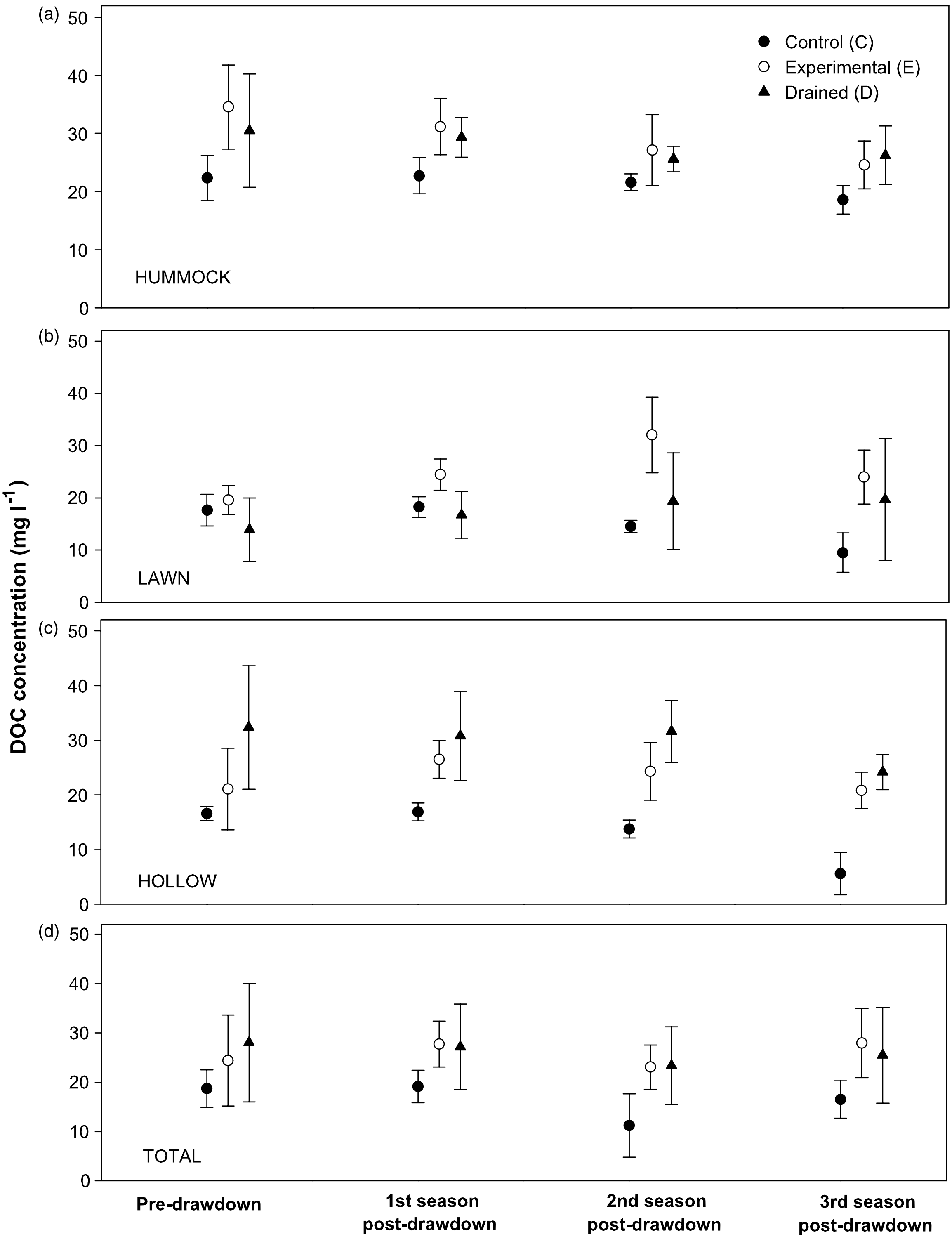

Figure 3. Seasonal average pore water DOC concentration at (a) hummocks, (b) lawns, (c) hollows and (d) average site conditions for C, E and D. Error bars represent one standard deviation. Sample size n varied between microforms and sites as not all sites could be measured on each sampling date due to low water tables or slow recharge rates. All means are based on greater than five values and 'total' values are the mean of greater than 30 samples at all sites

DISCUSSION

Effect of water table drawdown on dissolved organic carbon dynamics

Drainage and persistent water table drawdown resulted in higher DOC concentrations in surface water and pore water, and increased DOC export. Evidence from E reveals that these changes take place during the first season following water table drawdown, whereas D suggests that they persist for many seasons. Drainage increased DOC export due to an increase in growing season surface discharge from the peatland from near zero to a range of 2–6 mm day^{-1}. In addition, DOC export also increased due to increased DOC concentration in surface

DOI: 10.1002/hyp

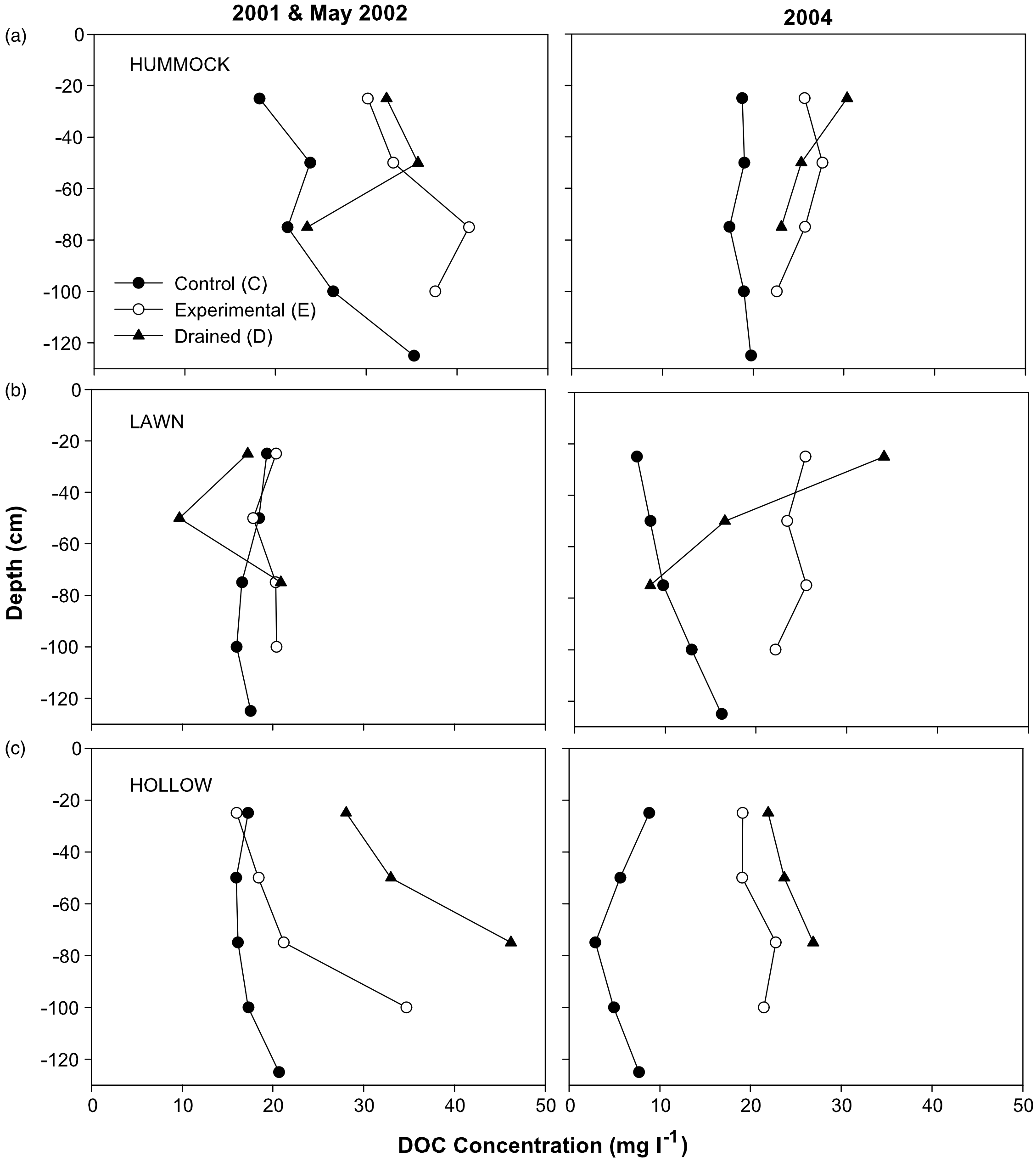

Figure 4. Profile of seasonal average pore water DOC concentrations before water table drawdown (2001 and May 2002) and during the third season after water table drawdown (2004) at (a) hummocks, (b) lawns and (c) hollows

and pore water. Our estimate of DOC export in surface water was limited to the growing season and we have missed export of DOC resulting from snowmelt and autumn discharge. Since snowpack depth should be similar across all sites and spring and autumn pool DOC concentrations are similar between C, E and D (Figure 2), then it is likely that DOC lost during this non-growing season period is similar among sites. Because we were interested in differences between C, E and D, the growing season was most important for understanding these differences; however, year-round monitoring of peatland DOC export would be desirable. Despite the fact that growing-season loss of DOC-C via surface water is within the range (5–40 g m^{-2} year^{-1}) reported for natural peatlands (Moore, 1998), it is clear that drainage has greatly increased DOC export at E and D relative to C.

Initially high pool DOC concentrations at E following drainage were likely caused by disturbance, which has been observed to enhance net DOC production (Lundquist *et al.*, 1999; Blodau and Moore, 2003). Increased concentrations may also result from changing the source of water to the pool. Following water table drawdown, uneven peat subsidence resulted in steeper hydraulic gradients, potentially increasing the input of higher DOC peat pore water to the pool.

During the second season after water table drawdown, DOC concentration in the E pool declined, but they remained similar to D and elevated above C. Elevated DOC concentrations were also observed in pore water following drainage, and this was maintained throughout the study period. Low concentrations in 2004 may be linked to the larger amount of precipitation (Table I)

Table III. Hydrologic parameters and inter-microform transfer of water and DOC at control (C), experimental (E) and drained (D) sites in 2004. Negative gradients d*h*/d*l* indicate a slope from the hollow to the hummock and negative values of DOC flux indicate a loss of carbon from the microform. Hydraulic conductivity K values are the geometric means of K throughout the peat profile. Hydraulic gradients are the average of measurements made twice weekly over the growing season

Location	K (cm s^{-1})	dh/dl	Subsurface discharge (l day^{-1})	DOC flux (mg m^{-2} day^{-1})	
				Hummock	Hollow
Control					
Upslope	3×10^{-4} $n = 8$	2×10^{-2} $n = 21$	2×10^{3}	-60 ± 200	20 ± 90
Downslope	6×10^{-3} $n = 5$	-1×10^{-2} $n = 21$	9×10^{2}	30 ± 100	-10 ± 50
Total				-30 ± 200	10 ± 100
Experimental					
Upslope	4×10^{-5} $n = 10$	8×10^{-2} $n = 21$	4×10^{2}	-70 ± 500	80 ± 200
Downslope	2×10^{-4} $n = 2$	-6×10^{-3} $n = 21$	1×10^{2}	30 ± 2000	-30 ± 700
Total				-40 ± 2000	50 ± 700
Drained					
Upslope	1×10^{-5} $n = 10$	7×10^{-3} $n = 21$	1×10^{1}	-4 ± 1	6 ± 1
Downslope	3×10^{-6} $n = 10$	-6×10^{-2} $n = 21$	3×10^{1}	2 ± 4	-2 ± 2
Total				2 ± 4	4 ± 2

Table IV. Calculated DOC export in surface water from 2002 to 2004 at experimental and drained sites. Surface water discharge was never observed at the control (C) site during the growing season; therefore, both discharge and export are assumed to be zero. Limited discharge measurements at the drained site in 2002 preclude export determination during that season. Values are given as average plus/minus standard error. Standard error for discharge and export are computed as described in the text

	Experimental (E)			Drained[a] (D)		
	Discharge (mm)	Average discharge [DOC] (mg l^{-1})	DOC-C export (mg m^{-2} day^{-1})	Discharge (mm)	Average discharge [DOC] (mg l^{-1})	DOC-C export (mg m^{-2} day^{-1})
Initial drainage (days 161–166)	180 ± 70	$20{\cdot}3 \pm 0{\cdot}4$ $n = 13$	617 ± 283	n.m.	n.c.	n.c.
Following drainage (days 167–230)	236 ± 271	$26{\cdot}6 \pm 1{\cdot}4$ $n = 15$	81 ± 116	n.m.	n.c.	n.c.
2002 growing season total (days 161–230)	416 ± 341	$23{\cdot}9 \pm 1{\cdot}0$ $n = 28$	127 ± 130	n.m.	30·2	n.c.
2003 growing season (days 128–230)	646 ± 700	$13{\cdot}8 \pm 1{\cdot}8$ $n = 7$	110 ± 105	342 ± 504	$22{\cdot}4 \pm 1{\cdot}8$ $n = 7$	104 ± 117
2004 growing season (days 128–230)	447 ± 354	$10{\cdot}7 \pm 0{\cdot}7$ $n = 4$	91 ± 38	330 ± 449	$14{\cdot}1 \pm 1{\cdot}4$ $n = 4$	82 ± 63

[a] n.m.: not measured; n.c.: not calculated.

resulting in dilution and flushing of DOC in surface water (Schiff *et al.*, 1998; Moore and Dalva, 2001; Tóth, 2002). In fact, according to Whittington (2005), seasonal values of precipitation minus evapotranspiration were −10 mm, 7 mm and 72 mm in 2002, 2003 and 2004 respectively; these are inversely correlated with average surface water DOC concentrations at C and D.

Despite the fact that higher DOC concentrations at E may be partially due to an increase in the inter-site and inter-microform transfer of water and DOC, evidence from D suggests that this enhanced subsurface flow is not persistent. Over time, subsurface flow is reduced by continued reductions in hydraulic conductivity caused by ongoing peat subsidence related to shrinkage and peat oxidation (Schothorst, 1977; Price, 2003). Moreover, enhanced hydraulic gradients may be minimized over time as peat is lost to soil respiration at hummocks and accumulated in hollows and pools via enhanced vegetation productivity (Strack *et al.*, 2006). Finally, the uncertainty in the absolute value of net subsurface DOC exchange is high, particularly at E, making it difficult to assert that the flow of dissolved carbon to hollows has been increased in the short term by water table drawdown. Therefore, maintenance of higher concentrations of DOC in pore water and surface water at E and D suggests that water table drawdown results in persistent elevated rates of net DOC production.

Controls on net dissolved organic carbon production

Shifts in the concentration of DOC result from changes in the balance between DOC production and consumption. Because we determined only concentration, we are unable to assess whether concentration changes result specifically from changes in production or consumption, but know only that net production of DOC has increased. This net production of DOC in organic soils may be controlled by temperature, soil moisture, soil solution chemistry, vegetation community and site hydrology (Moore *et al.*, 1998; Kalbitz *et al.*, 2000); however, the relative importance of these controls is not clear (Kalbitz *et al.*, 2000). Although several field studies have observed higher DOC concentrations in relation to higher temperatures (Moore, 1987; Bourbonniere, 1989; Moore and Dalva, 2001; Waddington and Roulet, 1997), water table drawdown had little effect on thermal regime in this study (see Strack *et al.* (2006)), suggesting that temperature is not the major factor influencing DOC concentrations following water table drawdown.

Water table position can also affect net production of DOC, as anaerobic conditions lead to less efficient decomposition of organic matter, increasing the production of water-soluble intermediate metabolites (Mulholland *et al.*, 1990; Kalbitz *et al.*, 1997). Water table drawdown exposes the upper peat profile to aerobic conditions, favouring CO_2 over DOC as a metabolic end product (Freeman *et al.*, 2004). However, following drainage, peat subsidence helped to maintain the water table closer to the surface than would be observed in a rigid soil (e.g. Whittington and Price, 2006). There is spatial variability in the response of DOC dynamics to water table drawdown because of variability in initial water table position and amount of peat subsidence between microforms. Following water table drawdown, DOC concentrations increased only at relatively wet microforms (lawns and hollows). Hummocks have experienced less peat subsidence, more water table drawdown (Whittington and Price, 2006) and reveal a trend of reduced pore water DOC concentrations (Figures 3 and 4). Since the water table at E and D hummocks is generally 30 to 40 cm below the surface, a large aerobic zone in the peat profile may result in the dominance of CO_2 as the end product of decomposition. This is supported by increases in ecosystem respiration observed at D hummocks (Strack *et al.*, 2006). The enrichment of DOC in lawn and hollow pore water is likely linked to larger water table fluctuations and a changing vegetation community (discussed below).

Aside from the importance of the water table position, water table fluctuation can also play an important role in DOC production. In a review of literature, Kalbitz *et al.* (2000) assert that rewetting following a dry period results in increased DOC concentrations in both field and laboratory studies. Microbial utilization of DOC is reduced during dry periods, and turnover of microbial biomass is enhanced upon rewetting, resulting in enhanced DOC release (Lundquist *et al.*, 1999). Also, several studies have observed that DOC production in wet soils decreases with time (Christ and David, 1996; Blodau and Moore, 2003) potentially resulting from a build up of DOC, which limits its continued production through the inhibition of anaerobic decomposition or DOC dissolution from an adsorbed phase. Water table fluctuations can flush DOC from stagnant soil horizons and may promote further DOC production. In the current study, the seasonal range of water table position increased with time after water-table drawdown (i.e. D has larger water table fluctuations than E). During periods of drought, water table is drawn down further at E and D than at C, and during subsequent precipitation events the water table rises more rapidly at the former sites (Figure 2). This different hydrological response results from decreased porosity caused by soil subsidence (Whittington and Price, 2006). These rapid wetting events through a larger area of unsaturated peat have likely increased net DOC production at E and D.

Additionally, increased vegetation productivity can increase net DOC production (Lu *et al.*, 2000; Kang *et al.*, 2001). In the current study, water table drawdown resulted in ecological succession at both E and D (St-Arnaud, 2007; Strack and Waddington, 2007). At E and D, the *Sphagnum* cover at hummocks was reduced, the sedge cover at lawns increased and the hollows were colonized by *Sphagnum* (Table II). Thus, vegetation productivity at hummocks was reduced, whereas lawns and hollows have become more productive at E and D (Strack *et al.*, 2006). These changes in vegetation productivity are mirrored by the trends in pore water DOC concentrations. This suggests that enhanced inputs of fresh organic matter by the changing vegetation community are partially responsible for the higher DOC concentrations observed across the experimental and drained sites following water table drawdown.

Dissolved organic carbon in the context of the carbon cycle

It is well known that the carbon balance of peatlands is often very close to neutral (Bubier *et al.*, 1999), with many sites storing carbon in some seasons and releasing carbon during others, often in relation to climatic conditions (e.g. Shurpali *et al.*, 1995; Joiner *et al.*, 1999). Thus, relatively small losses of carbon as DOC may be important in relation to the overall peatland carbon budget. For 2003 and 2004 at E, DOC export accounted for approximately 17% of the total carbon budget. Therefore, as has been found in other studies (Waddington and Roulet, 2000; Billett *et al.*, 2004), losses of carbon as DOC should be considered when determining net peatland carbon balance.

Moreover, the rate of carbon cycling can vary greatly between microforms within a peatland due to varying environmental conditions. Several studies suggest that carbon accumulation is highest at hummocks, whereas hollows or pools are generally sources of carbon to the atmosphere (Moore, 1989; Belyea and Clymo, 2001). However, these studies generally ignore hydrologic fluxes of carbon, particularly between microforms. We have found that hummocks tend to lose carbon as DOC while pools gain DOC, and this may partially account for differences in carbon accumulation rates determined from soil–atmosphere gas fluxes alone.

Implications of drainage and climate change on peatland dissolved organic carbon dynamics

The initial drainage of the 900 m^2 E site released $3{\cdot}3 \pm 1{\cdot}5$ kg DOC over several days. In general, peatland drainage for peat extraction involves deeper drainage over much larger areas. According to Cleary *et al.* (2005), $\sim$760 ha of peatland in Canada entered into production for horticultural peat each year between 1990 and 2003. Considering the standard method of 30 m ditch spacing, the water table is lowered from its initial position by an average of 50 cm (Boelter, 1972; Hillman, 1992). Assuming a porosity of 0·9 and a DOC concentration (as observed in this study) of $\sim$20 mg l^{-1}, this drainage results in an increase in DOC export of 7×10^7 g of carbon per year, or $\sim$9 g of DOC per square metre per year in the affected catchments. This represents a substantial load to downstream ecosystems, delivered as a pulse, with the potential to alter their physical and chemical characteristics such as acidity, light penetration, and metal and nutrient availability (Steinberg, 2003). Although the percentage of peatlands affected by active drainage is small in Canada, the majority of peatlands has been affected in other countries. For example, Minkkinen *et al.* (2002) determined that 60% of Finnish peatlands have been drained for forestry with 70 000–80 000 ha undergoing drainage maintenance yearly. This has the potential to increase export of DOC by $1{\cdot}5 \times 10^9$ g of carbon per year.

Since the reduction in water table position in northern peatlands under climate change is expected to result from enhanced evapotranspiration rates, the water table drawdown induced via drainage in this experiment is not directly comparable to potential climate change scenarios. However, since DOC concentrations in surface and pore water were observed to increase in this study following water table drawdown and remain elevated despite the consistent loss of DOC in surface discharge, it is probable that DOC concentrations in peatland waters will also be increased under climate change. Moreover, changing hydraulic gradients, hydraulic conductivity and patterns of water table fluctuation resulting from peat subsidence would occur regardless of the mechanism of water table drawdown. The shift in the vegetation community along the microtopographic gradient also resulted from the drier site conditions. Thus, insight into DOC dynamics under a climate change scenario can be gained through comparison of E and D to C.

This study suggests that, if peatlands become drier, enhanced water table fluctuations and vegetation productivity will lead to higher DOC concentrations in surface and pore water, particularly at wet microforms. An increase in aerobic decomposition at dry microforms, such as hummocks, may lead to reductions in DOC pore water concentration in these zones. Thus, the initial hydrologic conditions and pattern of microtopography may influence the response of peatland DOC dynamics to climate change.

Higher DOC concentration in peatlands may help to maintain DOC export from these ecosystems, despite potential reductions in discharge resulting from higher evapotranspiration rates (Moore, 1998), or lead to increases in DOC export, as has been observed in Europe (Freeman *et al.*, 2001; Worrall *et al.*, 2003) and predicted in eastern Canada (Clair *et al.*, 2002). Whereas other studies have raised concerns that observed increases in DOC export indicate a destabilization of peatland carbon stocks, this study suggests that high DOC export may result from higher ecosystem productivity, potentially having little effect on net peatland carbon storage. An increase in export of DOC to downstream ecosystems will increase acidity and reduce light penetration depth (e.g. Schindler *et al.*, 1996). DOC can also act as a biogeochemical substrate (e.g. Wetzel, 1992); however, if climate change results in a shift in DOC quality, then its ability to act as substrate within the peatland and in downstream ecosystems may be altered (Glatzel *et al.*, 2003). Further investigation is needed to determine the effect of disturbance on peatland DOC quality.

ACKNOWLEDGEMENTS

We thank Karen Edmunston, Frank Dunnett, Erik Kellner, Matt Falcone, Leah Hartwig, Karola Tóth, Jamee DeSimone, and J. R. van Haarlem for their assistance in the field and laboratory. We also thank Nirom Peat Moss for site access. Comments from T. Moore, M. Waiser and an anonymous reviewer greatly improved this manuscript. This research was funded by NSERC (Canada) and Canadian Foundation for Climate and Atmospheric Sciences (CFCAS) grants to JMW, NSERC Julie Payette and CGS scholarships to MS and by the federal Inter-Departmental Panel on Energy Research and Development (PERD).

REFERENCES

Asada T, Warner BG. 2005. Surface peat mass and carbon balance in a hypermaritime peatland. *Soil Science Society of America Journal* **69**: 549–562.

Baird AJ, Gaffney SW. 1995. A partial explanation of the dependency of hydraulic conductivity on positive pore water pressure in peat soils. *Earth Surface Processes and Landforms* **20**: 561–566.

Belyea LR, Clymo RS. 2001. Feedback control of the rate of peat formation. *Proceedings of the Royal Society of London, Series B: Biological Sciences* **268**: 1315–1321.

Billett MF, Palmer SM, Hope D, Deacon C, Storeton-West R, Hargreaves KJ, Flechard C, Fowler D. 2004. Link land–atmosphere–

stream carbon fluxes in a lowland peatland system. *Global Biogeochemical Cycles* **18**: GB1024. DOI: 10·1029/2003GB002058.

Blodau C, Moore TR. 2003. Experimental response of peatland carbon dynamics to a water table fluctuation. *Aquatic Sciences* **65**: 47–62.

Boelter DH. 1972. Water table drawdown around an open ditch in organic soils. *Journal of Hydrology* **15**: 329–340.

Bourbonniere RA. 1989. Distribution patterns of dissolved organic matter fractions in natural waters from eastern Canada. *Organic Geochemistry* **14**: 97–107.

Brooks PD, McKnight DM, Bencala KE. 1999. The relationship between soil heterotrophic activity, soil dissolved organic carbon (DOC) leachate and catchment-scale DOC export in headwater catchments. *Water Resources Research* **35**: 1895–1902.

Bubier JL, Frolking S, Crill PM, Linder E. 1999. Net ecosystem productivity and its uncertainty in a diverse boreal peatland. *Journal of Geophysical Research—Atmospheres* **104**: 27683–27692.

Christ MJ, David MB. 1996. Dynamics of extractable organic carbon in spodosol forest floors. *Soil Biology and Biochemistry* **28**: 1171–1179.

Clair TA, Arp P, Moore TR, Dalva M, Meng FR. 2002. Gaseous carbon dioxide and methane, as well as dissolved organic carbon losses from a small temperate wetland under a changing climate. *Environmental Pollution* **116**: S143–S148.

Cleary J, Roulet NT, Moore TR. 2005. Greenhouse gas emissions from Canadian peat extraction, 1990–2000: a life cycle analysis. *Ambio* **34**: 456–461.

Clymo RS. 1984. The limits to peat bog growth. *Philosophical Transactions of the Royal Society of London, Series B: Biological Sciences* **303**: 605–654.

Devito KJ, Dillon PJ, Lazerte BD. 1989. Phosphorus and nitrogen retention in five Precambrian shield wetlands. *Biogeochemistry* **8**: 185–204.

Foster DR, Wright HE, Thelaus M, King GA. 1988. Bog development and landform dynamics in central Sweden and south-eastern Labrador, Canada. *Journal of Ecology* **76**: 1164–1185.

Fraser CJD, Roulet NT, Moore TR. 2001. Hydrology and dissolved organic carbon biogeochemistry in an ombrotrophic bog. *Hydrological Processes* **15**: 3151–3166.

Freeman C, Evans CD, Monteith DT, Reynolds B, Fenner N. 2001. Export of organic carbon from peat soils. *Nature* **412**: 785.

Freeman C, Fenner N, Ostle NJ, Kang H, Dowrick DJ, Reynolds B, Lock MA, Sleep D, Hughes S, Hudson J. 2004. Export of dissolved organic carbon from peatlands under elevated carbon dioxide levels. *Nature* **430**: 195–198.

Freeze RA, Cherry JA. 1979. *Groundwater*. Prentice-Hall: Englewood Cliffs, NJ.

Glatzel S, Kalbitz K, Dalva M, Moore T. 2003. Dissolved organic matter properties and their relationship to carbon dioxide efflux from restored peat bogs. *Geoderma* **113**: 397–411.

Gorham E. 1991. Northern peatlands: role in the carbon cycle and probable responses to climatic warming. *Ecological Applications* **1**: 182–195.

Hillman GR. 1992. Some hydrological effects of peatland drainage in Alberta boreal forest. *Canadian Journal of Forest Research* **22**: 1588–1596.

Hinton MJ, Schiff SL, English MC. 1997. The significance of storms for the concentration and export of dissolved organic carbon from two Precambrian Shield catchments. *Biogeochemistry* **36**: 67–88.

Joiner DW, Lafleur PM, McCaughey JH, Bartlett PA. 1999. Interannual variability in carbon dioxide exchanges at a boreal wetland in the BOREAS northern study area. *Journal of Geophysical Research—Atmospheres* **104**: 27663–27672.

Joosten H, Clarke D. 2002. *Wise use of mires and peatlands—background and principles including a framework for decision-making*. International Mire Conservation Group and International Peat Society: Saarijarvi, Finland.

Kalbitz K, Popp P, Geyer W, Hanschmann G. 1997. β-HCH mobilization in polluted wetland soils as influenced by dissolved organic matter. *Science of the Total Environment* **204**: 37–48.

Kalbitz K, Solinger S, Park J-H, Michalzik B, Matzner E. 2000. Controls on the dynamics of dissolved organic matter in soils: a review. *Soil Science* **165**: 277–304.

Kang H, Freeman C, Ashendon TW. 2001. Effect of elevated CO_2 on fen peat biogeochemistry. *Science of the Total Environment* **279**: 45–50.

Kellner E, Price JS, Waddington JM. 2004. Pressure variation in peat as a result of gas bubble dynamics. *Hydrological Processes* **18**: 2599–2605.

Koprivnjak J-F, Moore TR. 1992. Sources, sinks, and fluxes of dissolved organic carbon in subarctic fen catchments. *Arctic and Alpine Research* **24**: 204–210.

Lu Y, Wassman R, Neue H-U, Huang C. 2000. Dynamics of dissolved organic carbon and methane emissions in a flooded rice soil. *Soil Science Society of America Journal* **64**: 2011–2017.

Lundquist EJ, Jackson LE, Scow KM. 1999. Wet–dry cycles affect dissolved organic carbon in two California agricultural soils. *Soil Biology and Biochemistry* **31**: 1031–1038.

Mattsson T, Kortelainen P, David MB. 1998. Dissolved organic carbon fractions in Finnish and Maine (USA) lakes. *Environment International* **24**: 521–525.

Mattsson T, Kortelainen P, Raike A. 2005. Export of DOM from boreal catchments: impacts of land use cover and climate. *Biogeochemistry* **76**: 373–394.

Minkkinen K, Korhonen R, Savolainen I, Laine J. 2002. Carbon balance and radiative forcing of Finnish peatlands 1900–2100—the impact of forestry drainage. *Global Change Biology* **8**: 785–799.

Moore PD. 2002. The future of cool temperate bogs. *Environmental Conservation* **29**: 3–20.

Moore TR. 1987. Patterns of dissolved organic matter in subarctic peatlands. *Earth Surface Processes and Landforms* **12**: 387–397.

Moore TR. 1989. Plant production, decomposition, and carbon efflux in a subarctic patterned fen. *Arctic and Alpine Research* **21**: 156–162.

Moore TR. 1998. Dissolved organic carbon: sources, sinks, and fluxes and role in the soil carbon cycle. In *Soil Processes and the Carbon Cycle*, Lal R, Kimble JM, Follett RF, Stewart BA (eds). Advances in Soil Science. CRC Press: Boca Raton, FL; 281–292.

Moore TR, Dalva M. 2001. Some controls on the release of dissolved organic carbon by plant tissues and soils. *Soil Science* **166**: 38–47.

Moore TR, Roulet NT. 1993. Methane flux: water table relations in northern wetlands. *Geophysical Research Letters* **20**: 587–590.

Moore TR, Roulet NT, Waddington JM. 1998. Uncertainty in predicting the effect of climatic change on the carbon cycling of Canadian peatlands. *Climatic Change* **40**: 229–245.

Mulholland PJ, Dahm CN, David MB, DiToro DM, Fisher TR, Kögel-Knabner I, Meybeck MH, Meyer JL, Sedell JR. 1990. What are the temporal and spatial variations of organic acids at the ecosystem level? In *Organic Acids in Aquatic Ecosystems*, Perdue EM, Gjessing ET (eds). Life Sciences Research Report 48. Wiley: Chichester, UK; 315–329.

Pastor J, Solin J, Bridgham SD, Updegraff K, Harth C, Weishampel P, Dewey B. 2003. Global warming and the export of dissolved organic carbon from boreal peatlands. *Oikos* **100**: 380–386.

Price JS. 2003. Role and character of seasonal peat soil deformation on the hydrology of undisturbed and cutover peatlands. *Water Resources Research* **39**: 1241.

Price JS, Schlotzhauer SM. 1999. Importance of shrinkage and compression in determining water storage changes in peat: the case of a mined peatland. *Hydrological Processes* **13**: 2591–2601.

Qualls RG, Haines BL. 1991. Geochemistry of dissolved organic nutrients in water percolating through a forest ecosystem. *Soil Science Society of America Journal* **55**: 1112–1123.

Quinton WL, Roulet NT. 1998. Spring and summer runoff hydrology in a subarctic patterned wetland. *Arctic and Alpine Research* **30**: 285–294.

Roulet N, Moore T, Bubier J, Lafleur P. 1992. Northern fens: methane flux and climatic change. *Tellus Series B: Chemical and Physical Meteorology* **44**: 100–105.

Schiff S, Aravena R, Mewhinney E, Elgood R, Warner B, Dillon P, Trumbore S. 1998. Precambrian shield wetlands: hydrologic control of the sources and export of dissolved organic matter. *Climatic Change* **40**: 167–188.

Schindler DW, Curtis JP, Parker BR, Stainton MP. 1996. Consequences of climate warming and lake acidification for UV-B penetration in North American boreal lakes. *Nature* **379**: 705–708.

Schothorst CJ. 1977. Subsidence of low moor peat soil in the western Netherlands. *Geoderma* **17**: 265–291.

Shurpali NJ, Verma SB, Kim J. 1995. Carbon dioxide exchange in a peatland ecosystem. *Journal of Geophysical Research—Atmospheres* **100**: 14319–14326.

St-Arnaud C. 2007. *Dynamique de la végétation d'un fen pauvre face à une simulation de réchauffement climatique: réponses potentielles des tourbières boréales à sphaignes.* MSc thesis, Université Laval.

Steinberg CEW. 2003. *Ecology of Humic Substance in Freshwaters*. Springer: New York.

Strack M, Waddington JM. 2007. Response of peatland carbon dioxide and methane dynamics to a water table drawdown experiment. *Global Biogeochemical Cycles* **21**: GB1007. DOI: 10·1029/2006GB002715.

Strack M, Waddington JM, Rochefort L, Tuittila E-S. 2006. Response of vegetation and net ecosystem carbon dioxide exchange at different peatland microforms following water table drawdown.

Journal of Geophysical Research—Biogeosciences **111**: G02006. DOI: 10·1029/2005JG000145.
Thurman EM. 1985. *Organic Geochemistry of Natural Waters*. Martinus Nijhoff/Dr. W. Junk: Dordrecht.
Tóth K. 2002. *Dissolved organic carbon dynamics in a cutover and restored peatland*. MSc thesis, McMaster University.
Van Seters TE, Price JS. 2002. Towards a conceptual model of hydrological change on an abandoned cutover bog, Quebec. *Hydrological Processes* **16**: 1965–1981.
Waddington JM, Roulet NT. 1996. Atmosphere–wetland carbon exchanges: scale dependency on CO_2 and CH_4 exchange on the developmental topography of a peatland. *Global Biogeochemical Cycles* **10**: 233–245.
Waddington JM, Roulet NT. 1997. Groundwater flow and dissolved carbon movement in a boreal peatland. *Journal of Hydrology* **191**: 122–138.
Waddington JM, Roulet NT. 2000. Carbon balance of a boreal patterned peatland. *Global Change Biology* **6**: 87–97.
Wetzel RG. 1992. Gradient-dominated ecosystems—sources and regulatory functions of dissolved organic matter in freshwater ecosystems. *Hydrobiologia* **229**: 181–198.
Whittington PN. 2005. *The effects of water table draw-down on the hydrology of a patterned fen peatland near Quebec City, Quebec, Canada*. MSc thesis, University of Waterloo.
Whittington PN, Price JS. 2006. The effect of water table draw-down (as a surrogate for climate change) on the hydrology of a patterned fen peatland near Quebec City, Quebec. *Hydrological Processes* **20**: 3589–3600.
Worrall F, Burt T, Shedden R. 2003. Long term records of riverine dissolved organic matter. *Biogeochemistry* **64**: 165–178.