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The impact of ditch-blocking on fluvial carbon export from a UK blanket bog

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ABSTRACT

We investigated the effects of ditch-blocking on fluvial carbon concentrations and fluxes at a five-year, replicated, control-intervention field experiment on a blanket peatland in North Wales, UK. The site was hydrologically instrumented, and runoff via open and blocked ditches was analysed for dissolved organic carbon (DOC), particulate organic carbon (POC), dissolved carbon dioxide (CO_2) and dissolved methane (CH_4). DOC was also analysed in peat porewater and overland flow. The hillslope experiment was embedded within a paired control-intervention catchment study, with three years of pre-blocking and six years of post-blocking data. Results from the hillslope showed large reductions in discharge via blocked ditches, with water partly redirected into hillslope surface and subsurface flows, and partly into remaining open ditches. We observed no impacts of ditch-blocking on DOC, POC, dissolved CO_2 or CH_4 in ditch waters, DOC in porewaters or overland flow, or stream water DOC at the paired catchment scale. Similar DOC concentrations in ditch water, overland flow and porewater suggest that diverting flow from the ditch network to surface or subsurface flow had a limited impact on concentrations or fluxes of DOC entering the stream network. The subdued response of fluvial carbon to ditch-blocking in our study may be attributable to the relatively low susceptibility of blanket peatlands to drainage, or to physical alterations of the peat since drainage. We conclude that ditch-blocking cannot be always be expected to deliver reductions in fluvial carbon loss, or improvements in the quality of drinking water supplies.

INTRODUCTION

All peatlands export carbon via their drainage networks, as dissolved organic carbon (DOC), particulate organic carbon (POC) and dissolved inorganic carbon (DIC), the dominant form of which in acidic bogs is dissolved CO_2. This 'waterborne' carbon export represents an important term in the peatland carbon balance (Roulet et al., 2007; Dinsmore et al., 2010; Koehler et al., 2011), particularly for peatlands (such as blanket bogs) with a large excess of precipitation over evapotranspiration. Previous studies show that much of this flux is subsequently mineralised in the aquatic system and emitted as CO_2 (Dinsmore et al., 2010; Billett and Harvey, 2013; Evans et al., 2016). Evidence also suggests that peatland drainage can lead to increased DOC loss, such that aquatic carbon may act as an indirect pathway for anthropogenic greenhouse gas (GHG) emissions (Evans et al., 2016), and this term has therefore been included in international emissions reporting methods for peatlands (IPCC, 2014).

High concentrations of DOC and associated water colour also present problems for water treatment, due to the energy and financial costs of DOC removal (Jones et al., 2016), and the production of potential harmful disinfection by-products during the chlorination process. These problems have been exacerbated by increases in surface water DOC concentrations since the 1980s, affecting Northern Europe and parts of North America (Monteith et al., 2007). These trends, which extend beyond peatlands to upland and semi-natural catchments more generally, are now broadly recognised as a response to declining levels of acidifying deposition (Monteith et al., 2007; Evans et al., 2012; SanClements et al., 2012) and thus probably represent ecosystem recovery towards a higher-DOC natural baseline (Valinia et al., 2015). Nevertheless, there is widespread concern that DOC increases have been intensified by peatland management activities such as drainage, heather burning, over-grazing and plantation forestry (Ramchunder et al., 2009; Parry et al., 2014). This perception has contributed to a number of water industry-linked initiatives aimed at improving raw water quality by re-wetting drained blanket bogs (e.g. RPSB 2011; Anderson et al., 2011; Grand-Clement et al., 2013). While there is reasonable evidence to suggest that peat drainage does indeed increase lateral DOC export, and that re-wetting can reduce these losses (Evans et al., 2016; Menberu et al., 2017), much of this evidence derives from studies of continental raised bog and fen systems, rather than UK blanket bogs, for which findings have been more equivocal. Here, some studies have reported clear effects (Wallage et al., 2006), but others have observed only limited impacts (Armstrong et al., 2010; Gibson et al., 2009; Turner et al., 2013), indicating that caution may be needed regarding the water quality outcomes of peat restoration (Armstrong et al, 2010). Furthermore, most studies were relatively short-term experiments or snapshot surveys, and few incorporated a full replicated control/intervention experimental design, or substantial pre-intervention baseline data.

Here, we describe the results of a replicated, hillslope scale ditch-blocking experiment at a drained blanket bog in Wales, UK. The experiment was designed to evaluate multiple ecological responses to peatland re-wetting including vegetation (Green et al., 2017), hydrology (Holden et al., 2017) and greenhouse gas emissions (Green et al., 2018). In this study, we aimed to establish the effects of ditch-blocking on the concentrations and fluxes of waterborne carbon at the ditch, hillslope and headwater catchment scales. Our hypothesis, based on previous work in other peatlands, was that ditch-blocking would reduce the overall loss of waterborne carbon from the system.

2. MATERIALS AND METHODS

2.1. Study sites

The study was carried out on the Migneint blanket bog, North Wales, UK (52.97°N, 3.84°W). The hillslope-scale experiment, described previously by Green et al. (2017) and Holden et al. (2017) lies on a north-facing slope approximately 500 m above sea level, with a total drainage area of 19300 m^2. It is drained by a set of 12 parallel ditches (numbered 1 to 12 from east to west, Figure 1) which are believed to have been dug during the 1980s with the aim of enhancing grazing quality for sheep production. Ditches run approximately downslope, with a mean spacing of 16 m (range 11-26 m), slope 4.5° (3.9-5.1°), length above sampling points 99 m (84- 107 m) and (pre-blocking) ditch depth 60 cm (30-90 cm). No ditches were found to intersect underlying mineral soil. All ditches were partially overgrown but hydrologically functional at the start of the study. Based on surface elevation data (0.5 m resolution areal LiDAR survey, National Trust unpublished data), mean ditch catchment area was 1610 m^2 (range 1070-2290 m^2). Mean measured peat depth was 1.3 m (range 0.5- 2.5 m). Vegetation comprised a typical blanket mire community of *Calluna vulgaris*, *Eriophorum vaginatum* and *Sphagnum* spp. The peat overlies Cambrian mudstones and siltstones (Lynas, 1973), which provide alkaline buffering of streams at baseflow, and are locally connected to the ditch network.

Measurements at the hillslope experiment began in October 2010. In February 2011, eight of the ditches were blocked using two different methods, with the four remaining ditches retained as (drained) open controls. Ditches were assigned to treatments by first placing them into four groups based on measured pre-blocking discharge, then randomly assigning one ditch from each subset to the three treatment/control categories. The two ditch-blocking methods used were 'damming' and 'reprofiling'. For the dammed treatment, peat dams were constructed at regular intervals along the ditch using peat extracted from nearby 'borrow pits', creating a sequence of moderately deep pools behind each dam. For the reprofiling treatment, vegetation was removed, the ditch base compressed, the ditch partially infilled with peat, and vegetation replaced. This treatment also involved the construction of peat dams, but pools tended to be shallower (mean 40 cm; Peacock et al., 2013) or absent due to the partial infilling of the ditch. Above all dams, small channels were created with the intention of channelling water back onto the bog surface rather than along the original drainage line.

Catchment-scale water quality data were obtained from an ongoing paired-catchment monitoring study of two peat streams, the Afon Ddu and the Nant y Brwyn. The Afon Ddu includes the experimental hillslope, has an area of 1.59 km^2, and an altitudinal range of 455-503 m. The total length of ditches in the catchment (estimated from analysis of LiDAR data) was 32.5 km, with approximately 50% of the catchment area located within 10 m of a ditch. All ditches apart from the short sections of open control ditch at the experiment were blocked by reprofiling in 2011. The Nant y Brwyn, located 2.5 km from the Afon Ddu, has a catchment area of 1.57 km^2 above the sampling point, an altitudinal range of 400-490 m, and a mean peat depth of 1.2 m (Cooper, 2013). Total ditch length was estimated at 25.7 km, with 25% of the catchment within 10 m of a ditch. A very small proportion of the ditch network (~3 km) was blocked in March 2012, with the rest remaining open for the duration of the study.

2.2. Hydrological measurements

Hydrological measurements are described in detail by Holden et al. (2017). Briefly, each experimental ditch was instrumented with a 22.5° v-notch weir and a WT-HR 1000 water-level recorder (TruTrack, Christchurch, New Zealand) logging at 15-minute intervals, used to estimate discharge from each ditch based on a standard weir equation, and calibrated against discharge measurements made by recording time to fill large containers of known volume. Water flowing over or close to the peat surface (henceforth referred to as 'overland flow') was intercepted by inserting polyvinyl chloride soffit boards 3-5 cm into the peat, starting mid-way between adjacent ditches, which routed flow to a weir box near the ditch for discharge estimation (we did not analyse these data directly, but used results from the analysis of Holden et al., 2017, to support interpretation). Water-table data were obtained from four manual dipwells per ditch, located 2 m east and 1, 2 and 3 m west of each ditch. The 2 m dipwells were installed prior to ditch-blocking, and 1 and 3 m dipwells after blocking in June 2011. Water-table depths were recorded relative to the ground surface on each sampling visit, and converted to annual means per dipwell (see Table S3 of Holden et al., 2017). An automatic weather station (AWS; Vantage Pro2, Davis Instruments, Hayward, USA) was installed near the experimental site for the duration of the study, recording rainfall, air and soil temperature, wind speed and solar radiation.

2.3. Water sampling

Sampling of ditch waters at the hillslope experiment commenced in October 2010, with samples collected directly from water flowing over the v-notch weirs. Three sets of samples were collected before ditch-blocking in February 2011, after which sampling continued approximately monthly (but with slightly higher frequency during the growing season, and lower frequency in winter) for four years, up to the end of February 2015. On each sampling occasion, pre-washed polyethylene sample bottles were pre-rinsed with water from the sampling site, then fully filled. 125 ml samples were collected for solute analysis, and 500 ml samples for POC analysis. From December 2010 onwards, additional samples were collected for dissolved gas analysis using the headspace method (Green et al., 2014). During the project, it became clear that one of the ditches (Ditch 4, in the dammed treatment) was affected by an alkaline spring discharging directly above the v-notch weir, which was associated with anomalously high pH, alkalinity, DIC, POC and dissolved gas concentrations. In July 2011 an additional sampling point was established in the next pool upstream of the weir, above the influence of the spring, which was used to represent Ditch 4 in the analysis of DOC, pH and alkalinity. However an insufficient number of samples were collected for POC or dissolved gas analysis from the upstream pool, and (given the highly anomalous concentrations observed in the spring-influenced weir pool) Ditch 4 was omitted from the analysis of these determinands.

Bulked samples for porewater analysis were collected from groups of 2-3 piezometers installed 2-3 m west of each ditch, sampling passively at a depth of 10 to 15 cm. Overland flow was sampled using crest-stage tubes, comprising polypropylene tubes sealed at both ends, with holes slightly above ground level to collect water moving over the peat surface. Crest-stage tubes were situated in groups of 2-3, located 2 and 4 m either side of each ditch, and samples were bulked for analysis. Porewater sampling commenced in January 2011, with samplers removed before and reinstated after ditch-blocking. Overland flow sampling commenced in July 2011, five months after ditch-blocking. Due to hydrological variability it was not always possible to collect a full set of porewater or overland flow samples. All samples were returned immediately to the laboratories of the Centre for Ecology and Hydrology (CEH) in Bangor, filtered where necessary (see below) and stored in the dark at 4 °C until analysed.

Monitoring of the Afon Ddu and Nant y Brwyn began in 2002, as part of the CEH Carbon Catchment programme. For this study, regular monthly samples collected since January 2008 were used, providing three years of baseline data prior to ditch-blocking at the Afon Ddu. Samples were collected on the same day (usually within an hour) and returned to CEH Bangor as above. Sampling of both sites has continued, and data up to February 2017 (six years since blocking) were analysed.

2.4. Chemical analysis

DOC was analysed using the non-purgeable organic carbon (NPOC) method (Findlay et al., 2010); samples were filtered through Whatman 0.45 µm cellulose nitrate filters, acidified (pH < 3), sparged with oxygen to remove any inorganic carbon, and analysed using an Analytical Sciences Thermalox Total Carbon analyser (Analytical Sciences, Tewkesbury, UK). DOC concentrations were calculated using a seven point calibration curve (plus quality control sample), with additional standards to check for drift, and several samples (1-3 per run) duplicated to check for reproducibility. Each sample was injected five times, and the result accepted if the coefficient of variation was less than 3%. Analysis of DOC on samples from the Afon Ddu and Nant y Brwyn catchments followed the same method on a Formacs^{HT} (Skalar, Breda, Netherlands) analyser at the CEH Lancaster Laboratory.

Water pH was measured on unfiltered samples, and analysed by titration using a 0.01N H_2SO_4 solution on a Metrohm 888 Titrando (Metrohm, Herisau, Switzerland). Electrical conductivity (EC) was determined using a Jenway 4320 conductivity meter (Jenway, Staffordshire, UK). For ditch samples only, alkalinity was measured by Gran titration, using the Metrohm analyser. POC was measured on ditch samples by passing 500 ml of deionised water through 0.7 µm Whatman GF/C filters, which were placed in a furnace at 550 °C for three hours, then weighed when cooled. 500 ml of sample was passed through the same filters, which were then dried for three hours at 105 °C, cooled and weighed, placed in the furnace for a three hours at 550 °C, and weighed again when cooled. The mass difference between the last two phases provides the mass of particulate organic matter, of which 50% was assumed to be carbon (Francis, 1990). Headspace CO_2 and CH_4 samples were analysed on a Perkin

Elmer Clarus 500 gas chromatograph system (Perkin Elmer, Seer Green, UK) or an Agilent Varian 450 GC (Agilent place, Santa Clara, USA), fitted with a flame ionisation detector (GC-FID).

2.5. Data analysis

2.5.1 Estimation of water fluxes

Catchment areas for each experimental ditch were initially defined from surface elevation, based on 0.5 m horizontal resolution LiDAR data as described by Holden et al. (2017). However, the analysis of discharge data by Holden et al. gave large (order of magnitude) variations in apparent areal mean runoff from different ditches. We therefore concluded that subsurface flow pathways – and thus catchment areas – could not be reliably predicted from surface topography at the small (1000-3000 m^2) scale of the individual ditch catchments, given the subdued topography and potential for subsurface flow. However areal mean runoff at the whole-hillslope scale, calculated as the sum of discharge for all twelve ditches, was consistent with measured precipitation (allowing for circa 15% evapotranspiration, and assuming negligible between-year storage) and with annual runoff estimates from well-defined nearby catchments, suggesting that source area could be reliably defined at this scale. We therefore assumed that areal runoff was uniform across the hillslope, and that the catchment area contributing flow to each ditch was directly proportional to the mean measured pre-blocking discharge in that ditch. From this, we were able to estimate 'apparent' pre-blocking catchment areas.

2.5.2. Analysis of chemical data

For each drainage ditch we calculated flow-weighted mean concentrations of each measured determinand as the sum of instantaneous concentration measurements, multiplied by the mean weir discharge for the day that each sample was collected, divided by the sum of daily mean discharge for all sample collection days. A single flow-weighted mean concentration per carbon form per ditch was calculated from the four years of post-blocking data (March 2011 to February 2015). Spatial relationships between chemical and hydrological variables were analysed by simple linear regression, taking mean values per ditch as data points. Total annual load of each carbon form (in kg C yr^{-1}) was calculated as the product of flow-weighted mean concentration and mean annual ditch discharge for the post-blocking period. Areal mean carbon fluxes (in g C m^{-2} yr^{-1}) were calculated as annual loads divided by topographically-defined and 'apparent' catchment areas.

Treatment effects on ditch carbon concentrations were analysed by repeated-measures ANOVA, using the AREPMEASURES procedure in GENSTAT version 18. For each sampling occasion we calculated mean concentrations per treatment as the mean of all replicates, with an associated standard error, as a basis for plotting and interpreting changes within and between treatments over time.

Porewater and overland flow DOC data were analysed in the same way as ditch data, but as there were no associated water fluxes for these sample, we could not calculate flow-weighted means or fluxes. Since we found minimal skewness in any of the DOC datasets, a simple mean of all samples (for the period July 2011 to February 2015) was used for comparison between treatments. To compare mean DOC concentrations in different water types we also calculated simple mean concentrations for the common period of ditch, porewater and overland flow sampling.

3. Results

3.1. Hydrological responses to ditch-blocking

A detailed assessment of hydrological responses to ditch-blocking is provided by Holden et al. (2017); here we summarise those responses most relevant to the analysis and interpretation of fluvial carbon data. Time-weighted average water-table depths (n=16 dipwells per treatment) during the post-treatment period were 11.0 cm (standard error ±1.4 cm) adjacent to open ditches, 7.7 cm (±0.7 cm) adjacent to dammed ditches, and 7.1 cm (±0.8 cm) adjacent to reprofiled ditches. This suggests that both re-wetting methods could have raised water tables by 3-4 cm. However, data from July 2010 to February 2011 for the subset of dipwells installed before ditch-blocking (n=8 per treatment) suggest that water tables were already slightly lower adjacent to the ditches that were not subsequently blocked (8.8 ±3.2 cm) compared to ditches that were subsequently dammed (7.1 ±2.3 cm) or reprofiled (6.8 ±1.8 cm). If we assume these differences reflected pre-treatment variations in water table across the site, the impact of ditch-blocking is reduced to 1.2 cm for the dammed treatment, and 2.1 cm in the reprofiled treatment.

Ditch discharge varied widely after blocking, with four-year mean discharge per ditch ranging from 11 to 5593 m^3 yr^{-1} (Table S1 in Holden et al., 2017). If we assume that the apparent catchment areas defined above remained unchanged after blocking, mean areal runoff for the four remaining open ditches increased to 2.38 m yr^{-1}, suggesting that the open ditches 'captured' some runoff from blocked ditches. There was variability among the individual open ditches, with runoff increasing in three (to between 2.51 to 3.66 m yr^{-1}), and decreasing in one (to 0.61 m yr^{-1}) (Table 1). Flow decreased in all eight blocked ditches, to 1.03 and 0.66 m yr^{-1} in the dammed and reprofiled treatments respectively. In Ditch 12, flow declined almost to zero (0.06 m yr^{-1}). Over the hillslope as a whole, mean ditch runoff after blocking was 1.35 m yr^{-1}, 34% lower than in the pre-blocking measurement period. Although the incomplete year of pre-blocking measurement makes direct comparison difficult, this decrease suggests that ditch-blocking led to an increased proportion of hillslope runoff bypassing the ditch network as seepage or overland flow, either between the ditches or along the natural topographic gradient towards the west of the site (Figure 1).

3.2. Ditch water chemistry

Volume-weighted mean ditch concentrations for the four years following ditch-blocking are shown in Table 1. All ditches were highly acidic (mean pH 3.82 to 4.04) with negative mean alkalinity (-119 to -87 µeq l^{-1}), indicating that DIC concentrations were zero. Electrical conductivity was consistently below 60 µS cm^{-1}. Ditch mean DOC ranged from 13.4 to 21.6 mg l^{-1}, with a mean of 17.7 mg l^{-1}. As noted above, concentrations of POC, dissolved CO_2 and dissolved CH_4 were highly anomalous at the spring-influenced Ditch 4 weir pool (around an order of magnitude higher than any other site for both POC and dissolved CH_4) and this site was excluded from further analysis. For the remaining 11 ditches, mean POC was 2.2 mg l^{-1} (range 0.6-5.1 mg l^{-1}), dissolved CO_2 2.4 mg C l^{-1} (1.5-4.7 mg C l^{-1}), dissolved CH_4 0.01 mg C l^{-1} (0.003-0.022 mg C l^{-1}) and total C 22.4 mg l^{-1} (16.4-28.5 mg l^{-1}). On average DOC contributed 80% of total C, POC 10%, dissolved CO_2 11%, and dissolved CH_4 0.04%.

Repeated measures ANOVA showed no significant ($p < 0.05$) effects of ditch-blocking treatment on concentrations of DOC, POC, dissolved CO_2 or dissolved CH_4 in ditch water. This result was not altered by the inclusion or exclusion of the anomalous Ditch 4. We observed a positive correlation (R^2 = 0.54, p = 0.01) between ditch mean DOC and dissolved CH_4 concentrations, and a weaker correlation between POC and dissolved CO_2 (R^2 = 0.43, p = 0.03). No other significant correlations were observed between different carbon forms, or between any carbon form and any other measured water quality or hydrological variable (i.e. pH, alkalinity, EC, discharge).

Time series plots of mean DOC and dissolved CO_2 concentrations for each treatment on each sampling occasion are shown in Figures 2a and b. POC plots are not shown because very high concentrations during periods of low or zero flow dominate the instantaneous concentration data, but have little or no bearing on flow-weighted mean concentrations or fluxes. Dissolved CH_4 plots are omitted because concentrations were consistently very low. For both DOC and dissolved CO_2, the data show strong temporal coherence between sampling points, but little difference between treatments on any sampling occasion. For DOC there was some indication of higher peak DOC concentrations in the reprofiled ditches, but differences between treatment means are small compared to differences between individual ditch means within each treatment category (Table 1), implying that they are almost certainly a function of intrinsic variability rather than any effect of treatment.

We did observe high peak DOC concentrations in the summer of 2011, soon after ditch-blocking, which were not repeated in the two following years. While there was no effect of treatment, the entire site was subject to some disturbance by machinery as part of ditch-blocking activity during this time. However, there was no overall trend in concentrations over the full measurement period, and high concentrations were again observed during 2014. Mean soil temperatures for the June-September period (11.8 °C in 2011, 10.6 °C in 2012, 11.6 °C in 2013, 11.9 °C in 2015) correspond well with peak summer DOC concentrations in each year, suggesting that between-year variations may have been climate-driven, although with only four years of data we cannot draw firm conclusions regarding drivers. For dissolved CO_2, mean concentrations were (with the exception of one set of higher values recorded in April 2012) invariably below 5 mg C l^{-1} during the first two years of measurement, but exceeded this value for extended periods during the summers of 2013 and 2014. These higher values are consistent with warmer, dryer conditions during these years, compared to 2012, although meteorological factors appear unable to fully account for low observed CO_2 concentrations in summer 2011, when climatic conditions were similar to those in 2013-2014. Again, no treatment-related differences in dissolved CO_2 were apparent.

3.3. Porewater and overland flow DOC concentrations

Mean DOC concentrations in porewater, overland flow and ditch water for the period following ditch-blocking are shown in Table 2. Mean DOC concentrations were highest in porewaters, varying from 31 to 47 mg l^{-1}, but these variations appeared unrelated to treatment, and repeated-measures ANOVA showed no significant treatment effect ($p > 0.05$). Overland flow concentrations were consistently lower than porewaters and more uniform across the hillslope (range 22 to 28 mg l^{-1}), again with no significant treatment effect. Unweighted mean DOC concentrations in the ditches (range 20 to 28 mg l^{-1}) were similar to overland flow concentrations. Flow weighted ditch concentrations were lower, as DOC concentrations were typically highest during dry (summer) periods, and lowest during wet (winter) periods (Figure 2a). For the unweighted ditch data, mean DOC concentrations were highest in the reprofiled ditches and lowest in the open ditches, but differences were very small (< 3 mg l^{-1}), and disappeared when data were flow-weighted.

Time series of porewater and overland flow DOC concentrations, by treatment, are shown in Figure 3. Porewater DOC concentrations were highest (for all treatments) in summer 2011, declined over the following two years, but increased again in 2014. DOC concentrations were slightly higher in porewaters adjacent to blocked (particularly reprofiled) ditches compared to open ditches during the first six months after blocking, but treatment means then converged, with no apparent treatment effect thereafter. Overall, spatial and temporal variability in porewater DOC concentrations were high, and as it was not always possible to obtain porewater samples when conditions were dry we did not analyse these data in detail.

For overland flow, DOC concentrations were more coherent (i.e. showed the same temporal variation in all treatments) and very similar between the treatments (mean difference between treatment means at each sampling point was 12% for reprofiled versus open and 11% for the dammed versus open). Although concentrations generally declined from 2011 to 2013, peak concentrations during the warm summer of 2014 were higher than on any previous occasion.

A comparison of mean porewater, overland flow and ditch water DOC time series across all treatments (Figure 4) demonstrates strong similarities in both absolute concentrations and temporal patterns between overland flow and ditch water. Porewaters had consistently higher DOC in the first three years, but the magnitude of differences decreased with time such that porewater DOC only marginally exceeded ditch and overland flow DOC in 2014. Based on the mean DOC concentration of all twelve replicates on each sampling occasion, as shown in Figure 4, we observed a strong linear correlation between ditch and overland flow DOC concentrations ($R^2 = 0.76$), and weaker correlations between porewater and ditch DOC ($R^2 = 0.57$) and porewater and overland flow DOC ($R^2 = 0.53$) (all $p < 0.001$).

3.4. Hillslope-scale carbon fluxes

Calculated mean annual total carbon loads and areal mean fluxes for the post-restoration period are shown in Table 3. Loads of all carbon forms were highly variable, reflecting between-ditch variations in discharge (e.g. DOC load vs areal runoff $R^2 = 0.97$, $p < 0.001$). Consequently, the large reductions in ditch runoff in the two ditch-blocking treatments (60-80%, Table 1) led to correspondingly large reductions in all carbon loads (Table 2). For DOC, damming is estimated to have reduced DOC loss via the ditch by an average of 54%, and the reprofiling treatment by 74%. For total C fluxes, the corresponding reductions are 40% and 71%. These reductions are replicated in the areal mean flux estimates if we apply the pre-blocking, topographically-defined catchment areas. However, as discussed above, this assumption cannot be considered valid, because ditch-blocking undoubtedly altered flowpaths within and across the hillslope. Capture of flow from adjacent blocked ditches will have led to over-estimates of areal mean carbon fluxes from the remaining open ditches, and therefore also the magnitude of relative reduction in fluxes in the blocked ditches. If we instead assumed that areal runoff was uniform across the hillslope after blocking (i.e. that the flow down each ditch simply reflects its 'new' catchment area) we obtained more uniform estimates of areal mean C fluxes (Table 3) and no evidence of any flux reductions due to ditch-blocking (reflecting the absence of effects on concentrations noted above).

3.5. Catchment-scale DOC concentrations

Long-term paired DOC measurements at the ditch-blocked Afon Ddu catchment and unblocked Nant y Brywn catchment are shown in Figure 5. During the three years before ditch-blocking at the Afon Ddu, the two streams exhibited similar concentrations at all times. After blocking, there was a brief peak in DOC at the Afon Ddu, coincident with the possible peak in porewater concentrations at the experimental hillslope. Subsequently, however, there has been no consistent offset between the sites. A scatter plot of measured DOC for the two sites before and after ditch-blocking at the Afon Ddu (Figure 6) shows a similar relationship during the two periods; a linear regression with a zero intercept on the pre-blocking data gave a strong correlation ($R^2 = 0.91$, $p < 0.001$) with a slope coefficient of 1.01 (standard error 0.026), while after ditch-blocking the relationship was slightly weaker ($R^2 = 0.73$, $p < 0.001$) with a slope coefficient of 0.99 (standard error 0.029). Although DOC concentrations were thus on average 2% lower at the Afon Ddu relative to the Nant y Brywn after ditch-blocking, 95% confidence intervals on the two regression coefficients had a large overlap (0.96 - 1.07 pre-blocking, 0.93 - 1.05 post-blocking), suggesting no significant change in the relationship between DOC concentrations in the two streams.

4. DISCUSSION

4.1. The contribution of different carbon forms to waterborne carbon fluxes

As in other carbon budget studies of Northern peatlands (e.g. Nilsson et al., 2008; Dinsmore et al., 2010), we found that DOC was the largest constituent of waterborne carbon export (80% of total ditch flux). The remaining flux comprised approximately equal amounts of POC and dissolved CO_2, whilst dissolved CH_4 was negligible, and DIC was assumed to be zero in the highly acidic conditions. POC fluxes were spatially and temporally variable, but high concentrations generally coincided with low or flows so did not contribute substantively to export. In areas of active peat erosion, there is strong evidence that POC export can be reduced by restoring hydrological function and vegetation cover (Worrall et al., 2011; Shuttleworth et al., 2015). However no ditches were observed to be actively eroding at the start of our study, and although weir pools construction could have triggered a flush of POC release, this was not evident in the data. The contribution of dissolved CO_2 to measured carbon concentrations varied considerably among ditches, possibly reflecting variations in the extent to which CO_2-enriched water emerging from the peat matrix was evaded to the atmosphere above the sampling point; dissolved CO_2 in peatland runoff tends to be evaded fairly rapidly, typically at locations of higher turbulence (Billett and Harvey, 2013). The very low observed concentrations of dissolved CH_4 are consistent with its very low solubility, and the observed correlation with DOC suggests this CH_4 is derived from the peat matrix. Gaseous emissions of CH_4 from pools created during ditch-blocking were found to be high (Green et al, 2018), but can be attributed to *in situ* production rather than lateral inputs.

4.2. Impacts of ditch-blocking on waterborne carbon concentrations

Over four years of measurement, we found no evidence of any systematic difference in DOC concentrations between blocked and unblocked ditches, in adjacent porewaters or in overland flow. Before-after comparison of paired catchments also did not demonstrate a change in DOC export during six years after one catchment was ditch-blocked. We did not observe treatment-induced changes in any other component of fluvial carbon loss in ditch waters, or any consistent differences between the two ditch-blocking methods. We did however find high temporal coherence in DOC among ditches (Figure 2a), porewaters (Figure 3a), overland flow (Figure 3b), and even different water types (Figure 4), suggesting that DOC is subject to strong hydrometeorological and/or biogeochemical controls at this scale. This coherence was also evident, albeit to a lesser extent, for dissolved CO_2 (Figure 2b).

The lack of apparent reduction in DOC concentrations following ditch-blocking is somewhat surprising, given that many studies have observed higher DOC concentrations in porewaters or runoff from drained peatlands, and/or reduced concentrations after re-wetting (e.g. Wallage et al., 2006; Höll et al., 2009; Urbanová et al., 2011; Frank et al., 2014; Haapalehto et al., 2014; Menberu et al., 2017). Overall, published data suggest that peatland drainage increases DOC concentrations by around 50%, and that re-wetting reduced concentrations by around 25% (Evans et al., 2016). However, only one of drained peatlands included in Evans et al. was blanket bog (Wallage et al., 2006), and of the five re-wetted blanket bogs included in this analysis, only two (Wallage et al., 2006; Armstrong et al., 2010) reported lower DOC concentrations, while the other three (O'Brien et al., 2008; Gibson et al., 2009; Turner et al., 2014) reported lower DOC fluxes only. The characteristics of blanket bogs that may account for a less pronounced response to ditch-blocking are discussed below.

Initial disturbance-induced increases in fluvial carbon concentrations also appear to have been minor. In the summer following ditch-blocking DOC concentrations were slightly elevated in reprofiled ditches (Figure 2a), porewaters (Figure 3a) and the ditch-blocked Afon Ddu (Figure 5). This increase could suggest a short-lived disturbance effect on DOC loss, consistent with some previous studies (e.g.

Glatzel et al., 2003). It is also possible that this disturbance (e.g. due to movement of machinery) affected the whole experimental hillslope, because porewater DOC concentrations were higher in all treatments following ditch-blocking than at any time thereafter. Declining DOC concentrations in porewater, overland flow and ditch samples from 2011 to early 2014 provide tentative support for this interpretation, but the absence of pre-treatment data limits the inferences that can be drawn. High DOC concentrations in all water types during the warm summer of 2014 also suggest that at least part of the previous downward trend was attributable to between-year climatic variations. In this context, the convergence of porewater, overland flow and ditch water DOC concentrations over the duration of the study does provide some evidence that porewater chemistry (at least) may have been impacted by disturbance effects. However, we cannot be certain whether such effects were attributable to the hillslope-scale impacts of ditch-blocking, or the small-scale impacts of piezometer installation.

Overall, we conclude that ditch-blocking did not (over a four-year period) reduce DOC concentrations in either peat porewater or runoff, and that any short-term DOC responses were – at most – minor in both magnitude and duration.

4.3. Impacts of ditch-blocking on waterborne carbon fluxes

Total carbon loads exported via the ditches varied hugely, from < 1 to > 100 kg C yr^{-1}. Given the lack of concentration changes noted above, these variations are overwhelmingly attributable to variations in discharge between individual ditches, including the effects of ditch-blocking in directing flows away from (or between) ditches. Depending on whether we assumed variable or uniform hillslope runoff, we either obtained highly variable estimates of areal fluxes (e.g. 1 to 58 g C m^{-2} yr^{-1} for DOC) or rather uniform fluxes (18 to 29 g C m^{-2} yr^{-1}). In reality, both of these assumptions are likely to be wrong, since ditch-blocking undoubtedly *did* reduce the proportion of total runoff travelling down former ditch lines (Holden et al., 2017). True areal mean waterborne carbon fluxes in blocked ditches thus probably lie somewhere between the two extremes shown in Table 3. In addition, the observed reduction in areal runoff via the ditches at the whole-hillslope scale, when compared to the pre-blocking period, strongly suggests that a greater proportion of water left the site after blocking via surface or subsurface flow, particularly in areas adjacent to blocked ditches. Regardless of the assumptions made, however, we obtain average hillslope-scale fluvial C fluxes for the three years of post-blocking measurements of around 30 g C m^{-2} yr^{-1}.This is somewhat lower than the 44 g C m^{-2} yr^{-1} obtained for a Scottish peatland by Dinsmore et al. (2010), but with similar proportions of DOC, dissolved CO_2 and POC. The hillslope-mean DOC flux of 24 g C m^{-2} yr^{-1} is close to the average of published fluxes for temperate peatlands collated by Evans et al. (2016).

Despite the absence of observed changes in fluvial carbon concentrations following ditch-blocking, we did observe large reductions in flow down blocked ditches (by 60-80%), and in total measured runoff via the ditch network at the hillslope scale (by 32%). Larger reductions in flow down reprofiled ditches, relative to dammed ditches, is consistent with the (marginally) larger impact of ditch-blocking on mean water tables in this treatment. However, the consequences for overall fluvial carbon export are less clear-cut. Taken at face value, our results would imply a corresponding reduction in both DOC and total C fluxes down the ditch network of around 70%. This reduction would be consistent with the conclusions some other studies (e.g. Wilson et al., 2011; Gibson et al., 2009) that ditch-blocking substantially reduces DOC loss, even in the absence of a reduction in DOC concentrations, by reducing runoff via the ditch network. However, such a conclusion would be misleading, because it requires a hydrologically implausible reduction in overall peatland runoff. In upland Wales, and other cool oceanic regions where blanket peatlands form, evapotranspiration (ET) comprises only a small fraction of precipitation. At the Plynlimon research site in mid-Wales, ET from unforested catchments comprises 15% of precipitation, and even conversion to conifer forest only increases this proportion to 25% (Marc and Robinson, 2007). Thus there is negligible potential for peatland re-wetting to increase evaporative losses, and any water not leaving the site via the ditch network must do so via other flow pathways. These could include surface or near-surface flow between the ditches, or lateral flow from the site to the northwest following the underlying topography (Figure 1). Holden et al. (2017) found limited evidence of enhanced overland or near-surface flow between ditches, but given the practical challenges of capturing overland flow mean this possibility cannot be ruled out. Another possibility is that more water left the hillslope via unmeasured subsurface flow. Since mean water tables were around 7 cm adjacent to blocked ditches, and overland flow collectors did not intercept flow below 3-5 cm, this explanation appears plausible. Finally, as noted above, increased apparent areal runoff rates at three of the four remaining open ditches suggest that these ditches captured water that was no longer being transported via blocked ditches. However the 32% lower areal runoff at the whole-hillslope scale when comparing pre- and post-blocking data suggests (even allowing for over-representation of winter conditions in the pre-blocking dataset) that a considerable volume of water left the site as unmeasured surface or subsurface flow.

With regard to overall fluvial DOC fluxes, the striking similarity of measured ditch, overland flow and porewater DOC concentrations suggests that total DOC export from the hillslope is not sensitive to changes in water flowpath. During the initial post-blocking period (2011-2012), higher DOC concentrations in porewaters could (if not attributed to piezometer installation-related disturbance) imply that a transition from ditch flow to subsurface flow should have increased overall rates of DOC loss. This would be consistent with the small peaks in DOC concentration noted at this time. However from 2013 onwards, when DOC concentrations were almost identical in all three water types, any changes in hillslope hydrology would simply have transported the same quantity of DOC via different flowpaths. This could also account for the lack of any observed effects of ditch-blocking on DOC at the catchment scale (Figure 5-6), which integrates water and DOC exports from all flow pathways. This would also be consistent with the conclusions of Turner et al. (2013) that ditch-blocking effects decrease with increasing spatial scale.

4.4 Study limitations

While we observed few indications that ditch-blocking altered either concentrations or fluxes of fluvial carbon, we cannot exclude the possibility that this negative result reflected limitations with the experimental design, or the choice of study site. The use of small parallel ‘headwater’ ditches, whilst enabling replication that would not have been possible at a catchment scale, may have limited the extent to which each ditch was truly independent of adjacent ditches. For example, the remaining open ditches may have limited the effectiveness of re-wetting at adjacent blocked ditches, or peat near open ditches may have been partly re-wetted by adjacent blocked ditches. As demonstrated above, the remaining open ditches also captured some runoff displaced from blocked ditches. This ‘smearing’ of treatment effects, although not sufficient to negate the effects of ditch-blocking on ditch discharge or water-tables, may have reduced the magnitude of waterborne carbon responses.

In addition, by locating the experiment in a headwater area, we may have biased the study towards less effective ditches and wetter pre-restoration conditions. Although all ditches were functional before the start of the experiment, some were overgrown, and in general they were shallower than in downslope areas, where higher water flows had caused ditches to incise deeper into the peat. The magnitude of observed water-table increase following ditch-blocking was very small, and fluvial carbon responses to ditch-blocking may have been greater in deeper-drained areas. However the absence of clear impacts on DOC concentrations in the ditch-blocked catchment (incorporating large areas of deep ditches), compared to the reference catchment, suggests that the lack of responses at the hillslope scale are not wholly attributable to the design or location of the experiment.

Although longer than most other ditch-blocking studies (e.g. Gibson et al., 2009; Wilson et al., 2011; Turner et al., 2013), it remains possible that the limited duration of the study, and limited pre-blocking data, were still insufficient to detect long-term ecosystem adjustments to re-wetting. Previous work has shown that ditch-blocking does not immediately lead to the return of pre-drainage hydrological function (Holden et al., 2011), and carbon cycling may exhibit further lags relative to hydrological recovery if, for example, plant communities undergo long-term change. Armstrong et al. (2010) observed, in a survey-based study, that DOC concentrations were higher in *Calluna*-dominated areas compared to *Sphagnum*-dominated areas. During the study period, we observed little evidence of plant species change (Green et al., 2017), but more recent data do suggest some shift in dominance from *Calluna* towards *Eriophorum* and *Sphagnum* species (S. Carter, unpublished data) which could influence DOC loss. Again, however, we were unable to detect evidence for such a long-term change in six years of post-blocking paired-catchment data.

4.5 Resilience of blanket bogs to drainage and ditch-blocking

As noted earlier, many previous studies have shown impacts of peatland drainage and re-wetting on DOC loss, but relatively few of these were undertaken on blanket peatlands. Blanket peatlands develop over undulating topography in areas of high rainfall, and possess distinct physical characteristics that enable peat to accumulate. In particular, they are characterised by rapidly deceasing lateral hydraulic conductivity with depth (Hoag and Price, 1995; Holden and Burt, 2003) restricting vertical water movement and favouring surface or near-surface flow. This, along with high rainfall, make blanket bogs inherently difficult to drain, which is why ditch densities tend to be higher (e.g. every 10-20 m at our site) than in other peatlands. Blanket bog data presented by Luscombe et al. (2016) suggest that water table drawdown often extends only a few metres from each ditch, so even in dense drainage networks some intervening areas may remain poorly drained. Our data, showing a mean water-table depth of only 11 cm adjacent to open ditches, support this interpretation. Analysis of LiDAR data from the Migneint area (Williamson et al., 2016) also suggests that, since ditches were dug in the 1980s, peat adjacent to many ditches has subsided by 10 cm or more, effectively following the water table down (Joosten, 2016). Although unquestionably an impact of drainage, this subsidence means that the volume of aerated peat has declined with time since drainage, and it is also probable that peat bulk density has increased, and hydraulic conductivity decreased, as a result of associated compaction. These changes in peat depth and structure may have restricted potential for re-wetting. Consistent with this, our data suggest (depending on the assumptions made) that ditch-blocking raised near-ditch water tables by just 1-4 cm. A similarly muted water table response to drainage and re-wetting was reported for a site in Northern England by Holden et al., (2011), although both drained and re-wetted blanket bogs in this study displayed different short-term hydrological behaviour compared to intact areas. At our study site, damming of ditches was clearly effective in reducing ditch flows and enhancing water movement across or within the peat, which in the long term may lead to larger changes in peatland hydrological and biogeochemical functioning.

Constraints on the amount of aerated peat in ditched blanket bogs may also help to explain differences in DOC response observed between studies in different regions. A comparison of DOC radiocarbon (^{14}C) measurements from intact, drained and re-wetted peatlands from the boreal to tropical climate zones (which included our study catchments) showed highly variable sensitivity to drainage-induced loss of DOC from deeper peat (Evans et al., 2014), with tropical peatlands showing the greatest sensitivity and blanket bogs the least. These differences are reflected in literature showing clear evidence of DOC response to drainage and re-wetting in boreal, continental temperate and tropical peatlands (Moore et al., 2011; Glatzel et al., 2003; Strack et al., 2008; Urbanová et al., 2011; Haapalehto et al., 2014; Menberu et al., 2017), but not in blanket peatlands (see above). One possibility, given the observed subsidence at our site, is that blanket bog drainage causes an initial flush of DOC, which dissipates as the peat surface subsides and ditches infill. Elsewhere, if ditches develop into erosional features and water-table drawdown is maintained or intensified, more sustained DOC increases and loss of old carbon via this pathway may occur (e.g. Evans et al., 2014; Stimson et al., 2017).

4.6. Conclusions: Implications for peatland restoration and drinking water quality

One of the major drivers of investment in UK peat restoration has been the desire of water companies to reduce treatment costs by reducing levels of DOC-related colour in water supplies. The challenges of treating upland-derived waters have increased as a result of multi-decadal rising trend in DOC concentrations affecting much of Northern Europe, and it is possible that these increases have been exacerbated by peatland drainage and other management-related disturbances. However, evidence that ditch-blocking can substantively reduce DOC losses remains limited and equivocal. Our results suggest that, in blanket bogs that were subject to limited drainage and/or have adjusted to past drainage, large and/or immediate reductions in DOC loss should not necessarily be expected. On the other hand, we also found no evidence that ditch-blocking led to short-term peaks in any component of the fluvial carbon flux, so the risk of detrimental impacts on water quality or waterborne GHG emissions appears low, and longer-term benefits remain possible. We conclude that, from a waterborne carbon perspective, blocking of ditches on blanket bogs represents at worst a 'no regrets' option. At best it may deliver modest improvements in drinking water quality and reductions in waterborne GHG emissions, particularly over longer periods as the ecosystem adapts to wetter conditions. Therefore improved water quality should be considered a potential co-benefit, rather than a primary objective, of peat restoration.

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Table 1. Mean ditch discharge and water chemistry for the four-year period following ditch-blocking. Ditch 4 POC, dissolved gas and total C are omitted due to the effects of groundwater spring inputs to the weir pool on measured concentrations; other determinands were measured on samples collected from an upstream pool and considered to be unaffected by groundwater inputs. Note that areal mean discharge values are based on pre-blocking catchment area estimates.

Ditch	Status	Mean discharge	Areal mean discharge	pH	Alkalinity	Conductivity	DOC	POC	Dissolved CO_2	Dissolved CH_4	Total C
		$m^3\ yr^{-1}$	$m\ yr^{-1}$		$\mu eq\ l^{-1}$	$\mu S\ cm^{-1}$	$mg\ l^{-1}$	$mg\ l^{-1}$	$mg\ C\ l^{-1}$	$mg\ l^{-1}$	$mg\ l^{-1}$
1	Reprofiled	608	0.36	3.82	-101	49.0	13.4	1.3	1.8	0.005	16.4
2	Open	5593	2.51	4.13	-90	38.0	20.2	3.3	1.6	0.011	25.2
3	Reprofiled	962	1.18	4.04	-107	53.5	15.9	1.7	2.1	0.004	19.7
4	Dammed	4371	1.02	3.87	-98	48.6	16.4	-	-	-	-
5	Dammed	2259	2.15	4.04	-106	56.4	18.8	5.1	4.7	0.008	28.5
6	Open	4897	2.75	4.04	-105	49.6	18.4	1.3	1.6	0.008	21.2
7	Open	4039	3.66	4.01	-108	54.9	15.7	0.7	1.7	0.003	18.2
8	Reprofiled	598	0.45	3.99	-116	58.5	21.6	3.3	3.5	0.010	28.5
9	Open	2087	0.61	3.96	-119	51.4	14.7	0.9	1.5	0.003	17.1
10	Dammed	1068	0.91	3.99	-119	60.1	19.8	2.1	3.3	0.008	25.1
11	Reprofiled	691	0.64	4.03	-107	58.5	16.9	3.5	1.8	0.004	22.2
12	Dammed	11	0.06	4.03	-87	50.1	20.9	0.6	2.4	0.022	24.0

Table 2. Mean DOC concentrations in porewater, overland flow and ditch flow for each of the experimental ditches in the four-year period following ditch-blocking. Flow-weighted ditch water data were collected from March 2011 (immediately after blocking) to February 2015 (52 sampling visits in total). Overland flow, porewater and unweighted ditch water means are based on the common period of measurement, from July 2011 to February 2015, to enable inter-comparison, and thus do not represent whole-year means.

		DOC (mg l^{-1})			
Ditch	Status	Porewater	Overland flow	Ditch (un-weighted)	Ditch (flow-weighted)
1	Reprofiled	29.7	25.3	23.9	13.4
2	Open	37.7	23.6	20.4	20.2
3	Reprofiled	41.4	22.6	22.3	15.9
4	Dammed	47.4	23.9	21.8	16.4
5	Dammed	36.2	22.5	23.0	18.8
6	Open	30.9	27.6	22.5	18.4
7	Open	46.0	24.1	22.7	15.7
8	Reprofiled	38.0	25.6	26.6	21.6
9	Open	42.4	23.0	23.9	14.7
10	Dammed	42.7	24.6	25.2	19.8
11	Reprofiled	42.1	22.3	28.4	16.9
12	Dammed	31.8	26.6	21.8	20.9
Mean of open		39.3	24.6	22.4	17.3
Mean of reprofiled		37.8	23.9	25.3	17.0
Mean of dammed		39.5	24.4	23.0	19.0
Mean of all		38.9	24.3	23.5	17.7

Table 3. Carbon exports during the four-year period following ditch-blocking, expressed as annual carbon loads (based on measured discharge from each weir); areal mean fluxes assuming variable hillslope runoff (based on apparent pre-blocking catchment areas, see text); and areal mean fluxes assuming spatially uniform hillslope runoff . Ditch 4 POC, dissolved gas and total C fluxes are omitted due to the effects of groundwater spring inputs to the weir pool on measured concentrations.

Ditch	Status	Annual carbon load (kg C yr^{-1})					Areal mean carbon flux (g C m^{-2} yr^{-1})									
							Assuming variable hillslope runoff					*Assuming uniform hillslope runoff*				
		DOC	POC	CO_2	CH_4	Total C	DOC	POC	CO_2	CH_4	Total C	DOC	POC	CO_2	CH_4	Total C
1	Reprofiled	8.1	0.8	1.1	0.00	9.7	4.8	0.5	0.6	0.00	5.9	18.0	1.8	2.4	0.01	22.2
2	Open	113.2	18.6	9.0	0.06	140.8	50.8	8.3	4.0	0.03	63.2	27.3	4.5	2.2	0.01	34.0
3	Reprofiled	15.3	1.7	2.0	0.00	19.4	18.7	2.0	2.5	0.00	23.2	21.4	2.3	2.8	0.01	26.6
4	Dammed	71.7	-	-	-	-	16.7	-	-	-	-	22.1	-	-	-	-
5	Dammed	42.4	11.4	10.5	0.02	67.0	40.3	10.9	10.0	0.02	61.2	25.3	6.8	6.3	0.01	38.5
6	Open	89.9	6.2	7.6	0.04	103.6	50.4	3.5	4.3	0.02	58.2	24.8	1.7	2.1	0.01	28.6
7	Open	63.4	3.0	6.9	0.01	73.0	57.5	2.7	6.3	0.01	66.4	21.2	1.0	2.3	0.00	24.5
8	Reprofiled	12.9	2.0	2.1	0.01	16.8	9.8	1.5	1.6	0.00	12.9	29.2	4.5	4.7	0.01	38.4
9	Open	30.6	1.9	3.2	0.01	36.4	8.9	0.5	0.9	0.00	10.4	19.8	1.2	2.1	0.00	23.1
10	Dammed	21.1	2.2	3.5	0.01	27.6	17.9	1.9	2.9	0.01	22.7	26.7	2.8	4.4	0.01	33.9
11	Reprofiled	11.7	2.4	1.3	0.00	15.3	10.8	2.3	1.2	0.00	14.3	22.8	4.7	2.5	0.01	30.0
12	Dammed	0.2	0.0	0.0	0.00	0.3	1.3	0.0	0.1	0.00	1.4	28.3	0.8	3.2	0.03	32.4
Open	**Mean**	**74.3**	**7.4**	**6.7**	**0.03**	**88.5**	**41.9**	**3.8**	**3.9**	**0.02**	**49.6**	**23.3**	**2.1**	**2.2**	**0.01**	**27.6**
	Std Deviation	*35.5*	*7.7*	*2.5*	*0.03*	*44.4*	*22.2*	*3.3*	*2.2*	*0.01*	*26.3*	*3.4*	*1.6*	*0.1*	*0.01*	*4.9*
Dammed	**Mean**	**33.9**	**4.6**	**4.7**	**0.01**	**31.6**	**19.0**	**4.3**	**4.4**	**0.01**	**28.4**	**25.6**	**3.5**	**4.6**	**0.02**	**34.9**
	Std Deviation	*30.5*	*6.1*	*5.4*	*0.01*	*33.6*	*16.1*	*5.8*	*5.1*	*0.01*	*30.3*	*2.6*	*3.1*	*1.5*	*0.01*	*3.2*
Reprofiled	**Mean**	**12.0**	**1.7**	**1.6**	**0.00**	**15.3**	**11.0**	**1.6**	**1.5**	**0.00**	**14.1**	**22.9**	**3.3**	**3.1**	**0.01**	**29.3**
	Std Deviation	*3.0*	*0.7*	*0.5*	*0.00*	*4.1*	*5.7*	*0.8*	*0.8*	*0.00*	*7.1*	*4.7*	*1.5*	*1.1*	*0.00*	*6.9*
All	**Mean**	**40.0**	**4.6**	**4.3**	**0.01**	**46.4**	**24.0**	**3.1**	**3.1**	**0.01**	**30.9**	**23.9**	**2.9**	**3.2**	**0.01**	**30.2**
	Std Deviation	*36.4*	*5.6*	*3.6*	*0.02*	*44.5*	*20.0*	*3.4*	*2.9*	*0.01*	*25.7*	*3.5*	*1.9*	*1.4*	*0.01*	*5.8*

FIGURE CAPTIONS

Figure 1. Map of the study site showing ditches, flow collection and measurement infrastructure, and topographically-defined catchments based on pre-blocking conditions. Letters after ditch number indicate treatment; O = open, D = dammed, R = reprofiled and dammed. OLF = overland flow.

Figure 2. Mean and standard error of a) ditch DOC concentrations and b) dissolved CO_2 concentrations by treatment on each sampling occasion, from October 2010 to February 2015. For the groundwater-influenced Ditch 4, dissolved CO_2 data were excluded from the dammed treatment because baseflow CO_2 concentrations were frequently an order of magnitude higher than all other ditches (DOC samples were collected from an upstream pool, above the area of groundwater influence).

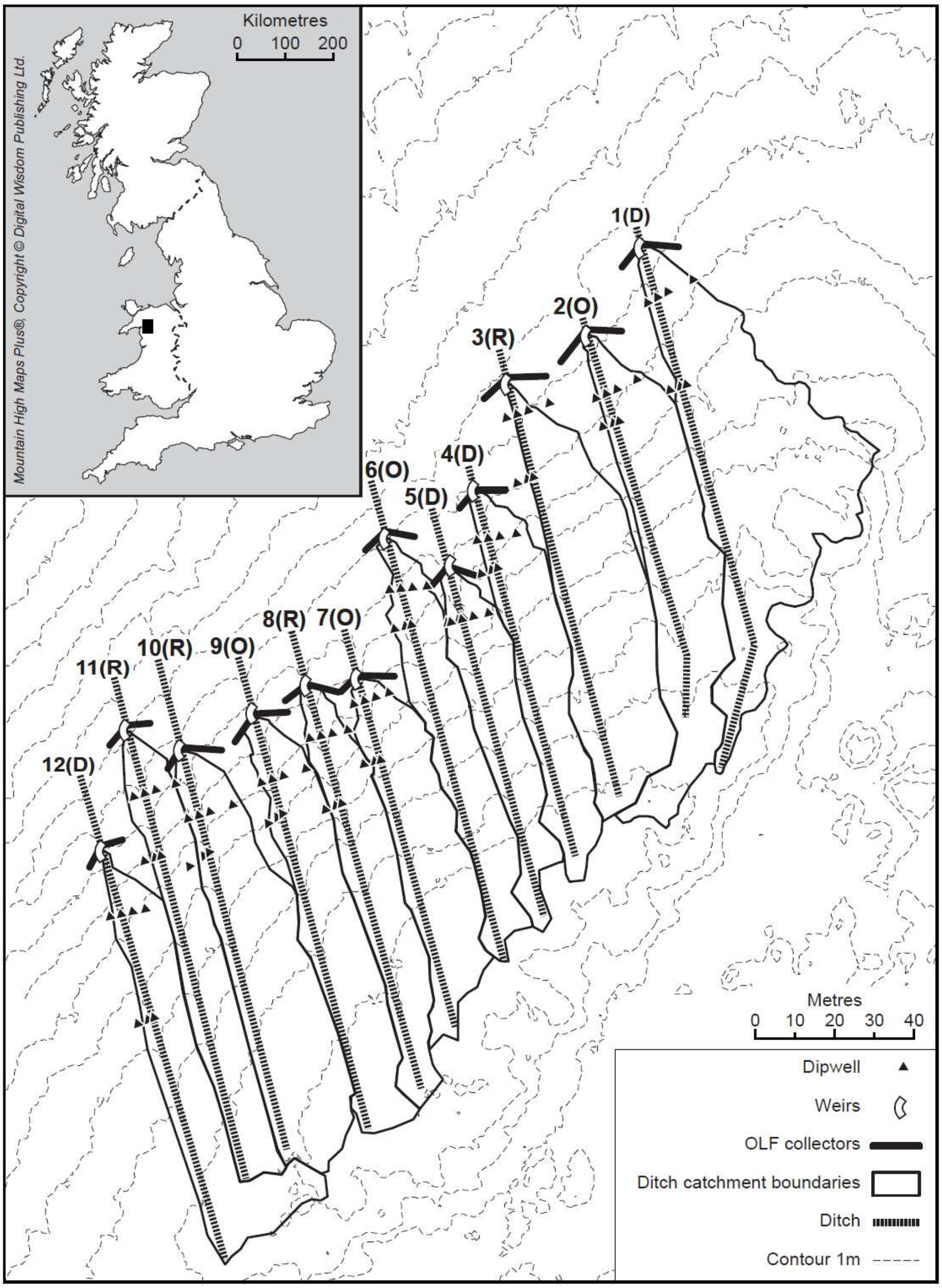
Figure 3. Mean and standard error of a) porewater (piezometer) and b) overland flow (crest stage tube) DOC concentrations by treatment on each sampling occasion, from 2011 to 2015.

Figure 4. Mean concentrations of ditch, porewater and overland flow concentration on each sampling occasion, for all treatments combined. Ditch water sampling started in October 2010 (4 months before ditch-blocking), pore water sampling in January 2011 (1 month before ditch-blocking), and sampling in July 2011 (5 months after ditch-blocking). Error bars indicate standard error of the mean for each measurement occasion.

Figure 5. Time series of measured DOC concentrations from the Afon Ddu (ditch-blocked catchment) and the Nant y Brwyn (unblocked catchment) from January 2008 (3 years prior to ditch-blocking) to March 2017 (6 years post-blocking).

Figure 6. Paired DOC samples from the Afon Ddu (ditch-blocked catchment) versus the Nant y Brwyn (unblocked catchment). 'Pre-blocking' samples are from January 2008 to February 2011 (n = 42), 'post-blocking' samples are from March 2011 to March 2017 (n = 79). The solid line shows the linear least square regression for the pre-blocking period, and the dashed line shows the regression for the post-blocking period.

Figure 1



Kilometres
0
100
200
Mountain High Maps Plus®, Copyright © Digital Wisdom Publishing Ltd.
1(D)
2(O)
3(R)
4(D)
5(D)
6(O)
7(O)
8(R)
9(O)
10(R)
11(R)
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20
30
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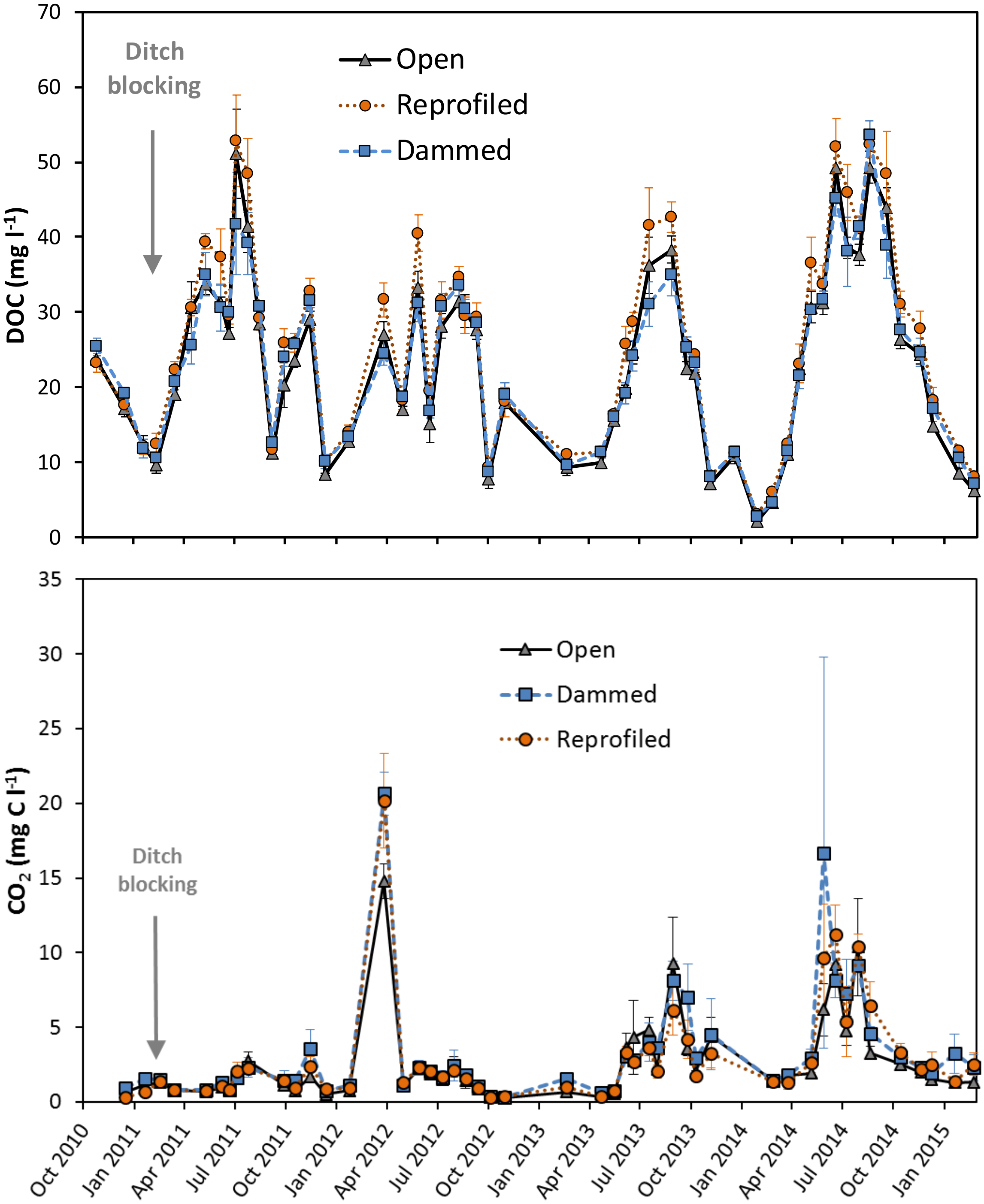
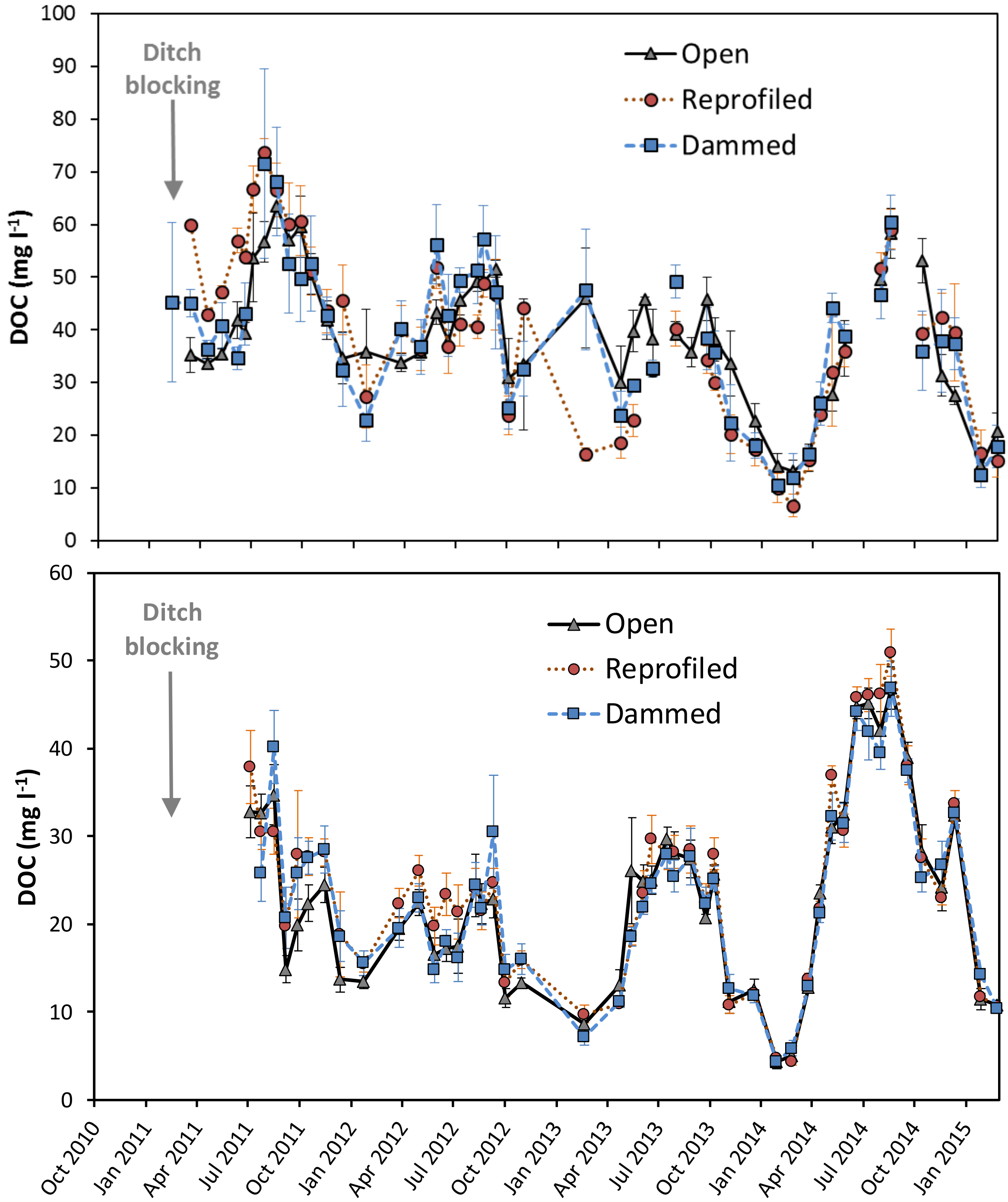
Dipwell
Weirs
OLF collectors
Ditch catchment boundaries
Ditch
Contour 1m

Figure 2



70
60
50
40
30
20
10
0
DOC (mg l-1)
Ditch blocking
Open
Reprofiled
Dammed
35
30
25
20
15
10
5
0
CO2 (mg C l-1)
Ditch blocking
Open
Dammed
Reprofiled
Oct 2010
Jan 2011
Apr 2011
Jul 2011
Oct 2011
Jan 2012
Apr 2012
Jul 2012
Oct 2012
Jan 2013
Apr 2013
Jul 2013
Oct 2013
Jan 2014
Apr 2014
Jul 2014
Oct 2014
Jan 2015

Figure 3



Ditch blocking
Open
Reprofiled
Dammed
DOC (mg l-1)
100
90
80
70
60
50
40
30
20
10
0
Ditch blocking
Open
Reprofiled
Dammed
DOC (mg l-1)
60
50
40
30
20
10
0
Oct 2010
Jan 2011
Apr 2011
Jul 2011
Oct 2011
Jan 2012
Apr 2012
Jul 2012
Oct 2012
Jan 2013
Apr 2013
Jul 2013
Oct 2013
Jan 2014
Apr 2014
Jul 2014
Oct 2014
Jan 2015

Figure 4

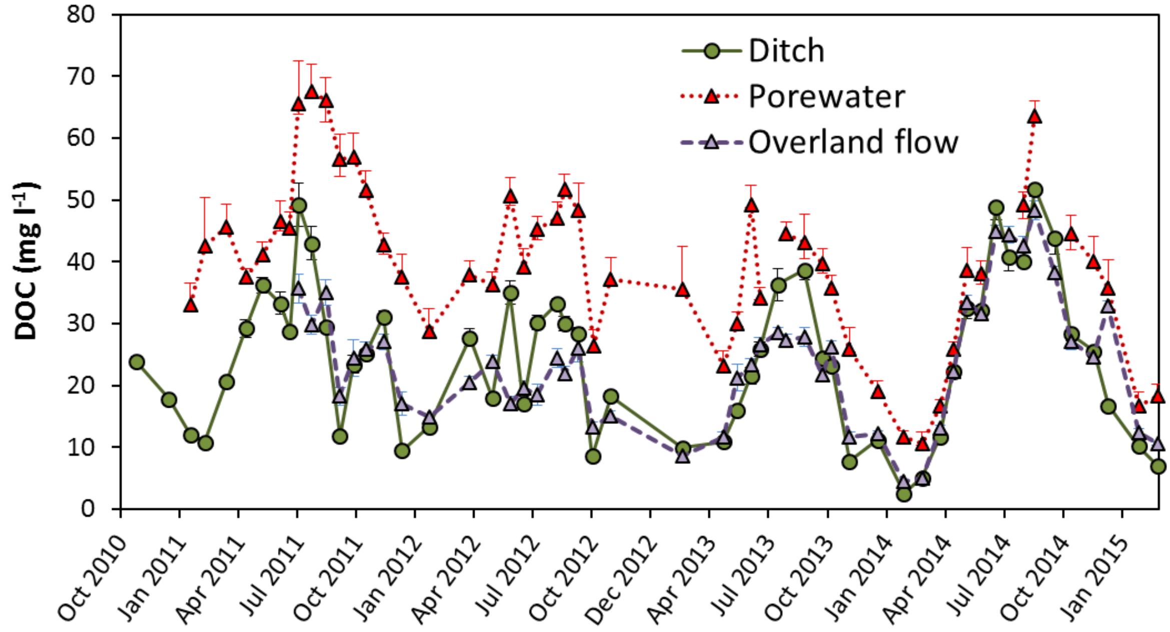


Figure 5

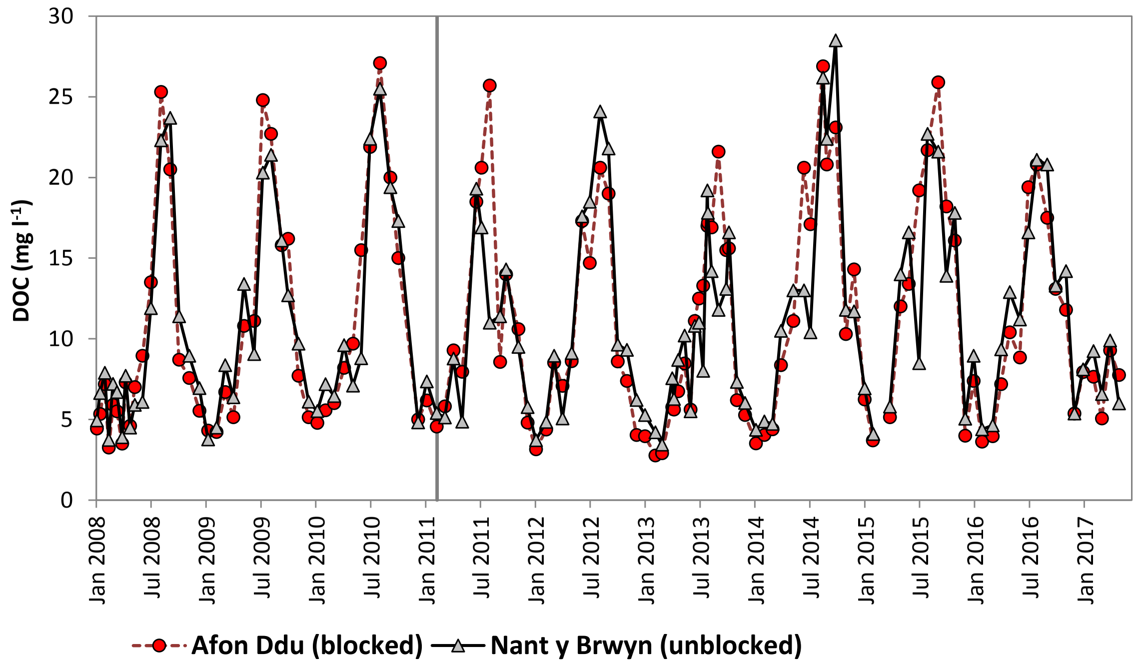
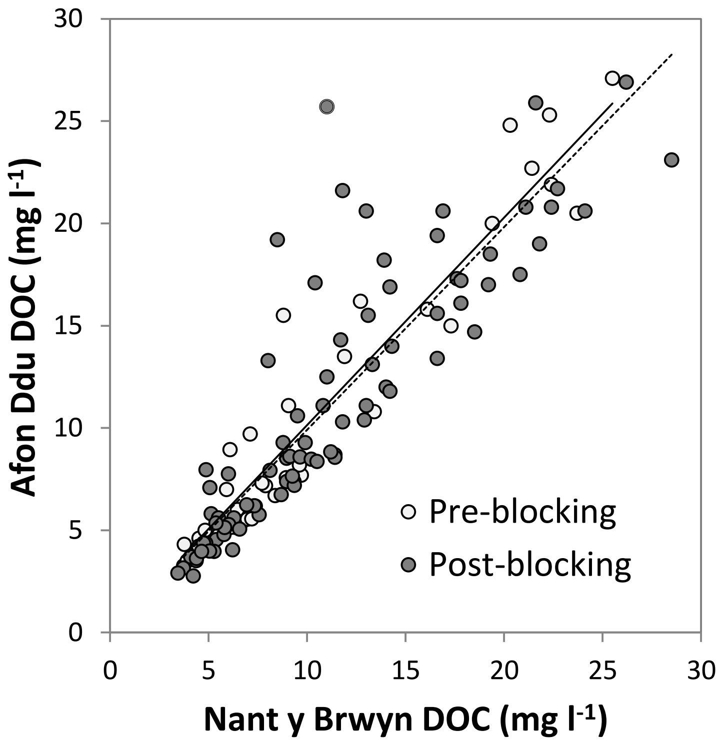


Figure 6



30
25
20
15
10
5
0
0
5
10
15
20
25
30
Afon Ddu DOC (mg l-1)
Nant y Brwyn DOC (mg l-1)
Pre-blocking
Post-blocking