Reviews and Syntheses: Understanding the impacts of peatland catchment management on DOM concentration and treatability

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Abstract

In the United Kingdom, most large reservoirs constructed for public water supply are in upland areas. Many are ~~and~~ situated in catchments characterised by organic-rich soils, including peatlands~~, that are often in poor condition~~. Although these soils naturally ~~Such catchments~~ leach large amounts of dissolved organic matter (DOM) to water, the widespread degradation of upland peat in the UK is believed to have exacerbated rates of DOM loss.~~, with water draining peatlands tending to release the most~~. High and rising DOM concentrations in these regions raise treatment challenges for the water industry.

In the UK, water companies are increasingly considering whether upland catchment peat restoration measures can slow down or even reverse rising source water DOM concentrations and thus reduce the need for more costly and complex engineering solutions. There remains considerable uncertainty around the ~~efficacy~~ effectiveness of such measures, and a comprehensive overview of the research in this area remains lacking. Here we review the peer-reviewed evidence for the effectiveness of four catchment management options in controlling DOM release from peat soils: ditch blocking, revegetation, reducing forest cover, and cessation of managed burning.

Results of plot scale investigations into effects of ditch blocking on DOM leaching are currently largely equivocal, while there is a paucity of information regarding impacts at spatial scales of more direct relevance to water managers. There is some, although limited, evidence that terrestrial vegetation type may influence DOM concentrations and treatability. The presence of plantation forestry on peat soils is generally associated with elevated DOM concentrations, although reducing forest cover ~~has~~ appears to have little short-term benefit, and associated disturbance ~~can~~ may even ~~exacerbate~~ increase concentrations further.

Catchment management measures have rarely been monitored with downstream water quality as the focus. To mitigate the uncertainty surrounding restoration effects on DOM, measures should be undertaken on a site-specific basis, where the scale, effect size and duration of the intervention are considered in relation to subsequent biogeochemical processing that occurs in the reservoir, the treatment capacity of the water treatment works, and future projected DOM trends.

Introduction

Dissolved organic matter (DOM) is ubiquitous across surface waters, with particularly high concentrations occurring in waters draining catchments with peat soils (e.g. Williamson et al., 2021). DOM originates from the decomposition of plant material and soil, and from plant and algal production and microbial transformation within the water column (Tranvik et al., 2009). DOM in rivers and lakes is subject to both biotic and abiotic processing, which change both concentrations and chemical structure (e.g. Algesten et al., 2004; Tranvik et al., 2009) so that DOM concentrations at the point of abstraction from reservoirs represent the sum of these removal and generation processes (Figure 1).

~~Peatland restoration has become an integral part of the UK's environment strategy, particularly in the drive toward Net Zero (Hm Government, 2021). It is founded on the potential to achieve multiple benefits that include improving biodiversity, protecting existing carbon stocks and enhancing carbon sequestration, and controlling water runoff and quality, in catchments that are deemed to have been degraded by anthropogenic stressors. Nearly three quarters of the storage capacity of drinking water reservoirs in the UK is sourced from peatland areas (Xu et al., 2018).~~ Peatlands release particularly high amounts of organic matter as dissolved organic matter (DOM) into drainage waters, and DOM concentrations have been rising since the 1980s (e.g. Naden and Mcdonald, 1989; Robson and Neal, 1996; Harriman et al., 2001; Worrall et al., 2004; Evans et al., 2005). Mean DOM concentrations in UK Upland Waters Monitoring Network (UWMN) surface waters, most of which are dominated by organic-rich soils, have approximately doubled over the last three decades (Figure ~~1~~2). At the sub-catchment scale, Chapman et al. (2010) found that water colour increased by between 22 and 155 percent over a 20 year period between 1986 and 2006. This phenomenon has now been observed across much of industrialised North America and ~~north-west~~Northern Europe, and appears to result largely from a~~n~~ long-term increase in the solubility of terrestrial organic matter as soils recover from the effects of acid rain (Monteith et al., 2007; De Wit et al., 2021; Monteith et al., 2023). One consequence of these changes is that water treatment works in some regions are having to adjust to much higher source water DOM concentrations than they were originally designed to cope with, since most were built at a time of much higher atmospheric deposition, and hence lower DOM. Atmospheric deposition of pollutants across the UK uplands has now declined to a very low level, and it is expected that in future, changes in DOM export will be increasingly affected by other factors including temperature, changes in precipitation seasonality and intensity and marine ion deposition (Monteith et al., 2023). Rising levels of DOM in waters draining many of these peatland catchments pose considerable water treatment challenges, with respect to increasing treatment costs and risks of regulatory failure (see Figure ~~1~~1). It has been proposed that peatland restoration measures might help slow or even reverse these DOM trends, along with other important benefits including increased terrestrial carbon storage, water retention and improvements in upland biodiversity (e.g. Glenk and Martin-Ortega, 2018).

Although consumption of DOM in drinking water is not directly harmful to people, coloured water reduces customer satisfaction (Ritson et al., 2014) and can be indicative of further problems. Indirectly, elevated DOM concentrations have implications for human health due to their potential influence on treatment processes and the production of carcinogenic disinfectant by-products (DBPs) such as trihalomethanes (THMs) during chemical disinfection, which are regulated by the Drinking Water Inspectorate due to their potential carcinogenic properties (Ding and Chu, 2017). DOM also may hamper the efficacy of chlorine as a disinfectant while simultaneously acting as a substrate for bacterial regrowth (Prest et al., 2016), thus increasing the risk of regulatory failure from bacterial contamination and the subsequent loss of customer trust.

The composition of DOM can have a large influence on the performance of the water treatment processes and the formation of DBPs upon chlorination (Matilainen et al., 2010). DOM in water draining peatland areas tends to be predominantly hydrophobic, and relatively photoreactive and biologically recalcitrant (Anderson et al., 2019) and~~. It~~ is relatively easily removed by conventional coagulation and filtration during drinking water treatment due to the presence of charged functional groups (Matilainen et al., 2010). Hydrophilic DOM, on the other hand, is mostly produced within the waterbodies by phytoplankton (Imai et al., 2002), and is biologically labile but less easily degraded by sunlight (Berggren and Del Giorgio, 2015; Berggren et al., 2018). The relative balance of hydrophobic to hydrophilic DOM in water is referred to as hydrophobicity, and is conventionally assessed in the water treatment system using Specific UV Absorbance measurements at 254 nm ($SUVA_{254}$), i.e. absorbance at 254 nm per unit dissolved organic carbon concentration (Weishaar et al., 2003). Values greater than 4 L mg^{-1} m^{-1} indicate hydrophobic dominance, while values less than 2 L mg^{-1} m^{-1} ~~show~~ indicate that the DOM is primarily hydrophilic and will not be effectively removed using conventional coagulation and filtration alone (Matilainen et al., 2010).

Higher concentrations of DOM in raw water necessitate a greater amount of treatment to provide potable water to customers (Monteith et al., 2021). This may include larger coagulant dosages, shorter filter run times, and longer and more frequent cleaning of filtration units, and result in higher energy costs, higher sludge removal costs and an increase in direct and indirect (energy-related) greenhouse gas (GHG) emissions from the treatment process (Jones et al., 2016). Major additional costs are incurred where capital investment is needed to upgrade treatment infrastructure designed for lower concentration ranges experienced in the past (Monteith et al., 2021).

Peatland restoration (physical interventions to return them to a more natural state i.e. high water table and active peat-forming vegetation) has been suggested as a catchment scale method for reducing DOM concentrations in water draining peatlands (IUCN Peatland Programme). The primary restoration methods undertaken to date in the UK uplands are: blocking of peatland drainage to raise the water table, revegetation of bare peat with peatland species, removal of plantation forestry to allow peatland species to recolonise and water tables to rise, and cessation of managed burning to encourage growth of peatland plant species (Figure 2) (IUCN Peatland Programme). It is important, therefore, for water industry decision makers to understand the extent to which peatland restoration could make a positive contribution to reducing DOM concentrations of raw water and thus relieve stresses on the treatment system and potentially remove the need for major additional capital investment in treatment plant. ~~This~~ In this study, we ~~work~~ review~~s~~ the available peer-reviewed literature relating to ~~and provides an assessment of~~ the impacts of~~UK~~ peatland restoration on DOM concentrations and treatability of raw drinking water. Finally, we consider the possible influence of catchment land-use on in-reservoir DOM cycling, and what impact this may have had on drinking water treatability. We focus on the UK as a well-studied area in which peatlands make an important contribution to drinking water supplies (Xu et al., 2018), and where rising DO~~C~~M concentrations are having a significant impact on water treatment processes and costs, but the conclusions of the work are likely be relevant to other areas with peat-derived water supplies.

2. Methods

To answer the question “will peatland restoration reduce DOM concentrations in raw water” we explored the evidence within the peer-reviewed scientific literature for catchment management approaches within peatland dominated drinking water catchments to influence DOM concentrations in the soils and waters of peatland catchments. This was achieved by applying a standard set of Boolean search terms within Web of Science and Google Scholar. The terms were: (“dissolved organic matter” OR “dissolved organic carbon” OR “DOM” OR “DOC” OR “colour”) AND (“peatland” OR “bog” OR “fen” OR “moor”) AND (“ditch blocking” OR “forest” OR “plantation” OR “managed burning”). Initial results, including titles and abstracts, were rapidly reviewed to determine whether the information within the papers was relevant, ~~both in terms of subject matter and in region (limited to temperate peatlands with a primary focus on UK and Irish peatlands where evidence was available)~~, then relevant papers were read in full and included in the review. Given the geographic focus of the project, we prioritised papers from the UK and Ireland where available, but also drew on data from other temperate peatland regions where required. From the original searches, 272 papers were considered relevant enough for further reading and 104 were included in the review.

~~2~~3. Evidence for the efficacy of catchment management approaches in the reduction of DOM

~~To answer the question “will peatland catchment management reduce DOM concentrations in raw water” we explored the evidence within the peer-reviewed scientific literature for catchment management approaches within peatland dominated drinking water catchments to influence DOM concentrations in the soils and waters of peatland catchments. This was achieved by applying a standard set of Boolean search terms within Web of Science and Google Scholar. The terms were: (“dissolved organic matter” OR “dissolved organic carbon” OR “DOM” OR “DOC” OR “colour”) AND (“peatland” OR “bog” OR “fen” OR “moor”) AND (“ditch blocking” OR “forest” OR “plantation” OR “managed burning”). Initial results, including titles and abstracts, were rapidly reviewed to determine whether the information within the papers was relevant, both in terms of subject matter and in region (limited to temperate peatlands), then relevant papers were read in full and included in the review.~~

3~~2~~.1. Ditch blocking

Extensive areas of upland peatlands across the UK uplands were drained in the mid-20th century in an attempt to increase agricultural productivity. Peatland drainage reduces water tables (Holden et al., 2011), resulting in a loss of peat forming plant species. The consequent drying and cracking of peat surfaces exposes previously permanently saturated organic matter to oxidative processes, making them more vulnerable to erosion and dissolution into DOM (e.g. Clark et al., 2009). Extensive efforts have been made by the water industry and organisations concerned with peatland ~~conservation~~ restoration to block ditches in an attempt to restore the hydrological, biogeochemical and ecological functions of these landscapes (IUCN Peatland Programme 2023) (Figures ~~2~~ 1 & 3).

Search results of the scientific literature showed that the impact of ditch blocking on DOM concentrations had been assessed in pore waters, ~~and~~ in ditches ~~a~~and in~~t~~ streams at the sites being restored. Of the five plot-scale studies of ~~peat soil~~pore water identified during this review, four (Table 1) reported significant changes in DOM concentrations~~-~~. The studies investigated effects between five and twenty years following ditch blocking, and reported a cross-study ~~average~~ mean 34% reduction in DOC concentration (range 0 to 69%) (Wallage et al., 2006; Holl et al., 2009; Haapalehto et al., 2014; Strack et al., 2015; Menberu et al., 2017). While therefore suggesting a general tendency for ditch blocking to reduce pore water DOM concentrations, these studies do not necessarily imply that effects will be translated through to surface waters and ultimately to the point of abstraction.

Changes observed in DOM concentrations at a drainage ditch scale, are more variable than those for pore waters (Table 1, Figure 3). The eleven studies reviewed showed a mean 8% increase in DOM concentrations following ditch blocking, although this figure is skewed by the large increases reported by Worrall et al. (2007b) and Haapalehto et al. (2014) (100% increase immediately following ditch blocking and 50-75% increases after ten and five years respectively); the median change is zero. Importantly, no significant change in DOM concentration was reported in over half of these studies (O'brien et al., 2008; Gibson et al., 2009; Armstrong et al., 2010; Wilson et al., 2011; Evans et al., 2018; Pickard et al., 2022). Likewise, a recent study found no reduction in DOM concentrations in ~~the~~ a restored site compared to the ditched site six years after ditch blocking, while both drained and restored site DOM concentrations remained elevated compared to the non-drained control (Pickard et al., 2022). Differences between studies in apparent effect size may in part be related to experimental design, including whether the work included a simultaneous control, and the time period over which post-restoration monitoring was carried out.

Studies of DOM flux changes following ditch blocking report ~~an average~~ a mean24% reduction (range 0 – 88% reduction) in DOM flux, primarily attributed to decreased water fluxes from the restoration site. However, the measurement and reporting of water fluxes (and hence DOM fluxes) at a site- or catchment-scale requires careful consideration of the potential for dominant water flow pathways to be altered following ditch blocking. For example, Holden et al. (2017) showed that damming of drainage ditches in North Wales reduced discharge along the original ditch lines, but that most, or all, of the displaced flow instead left the peatland via overland flow or near-surface through-flow. Subsequent reporting from the same experiment demonstrated that DOM concentrations in water displaced along these surficial pathways were approximately the same as those in water travelling along the ditches, with the result that ditch-blocking was not found to have any clear effect on either DOM concentrations or fluxes at the catchment scale (Evans et al., 2018).

We identified nine studies that have assessed the potential impact of ditch blocking on DOM treatability and hence the ease of treatability within a conventional water treatment works. The majority of studies at UK and continental European ditch blocking locations, along with results from their experimental work, showed little effect of ditch blocking on DOM treatability as measured by commonly reported metrics such as SUVA, E2:E3 ratios (ratio of light absorbance at 250 and 365 nm) and E4:E6 ratios (ratio of light absorbance at 465 and 665 nm) (Glatzel et al., 2003; Strack et al., 2015; Gough et al., 2016; Lundin et al., 2017; Peacock et al., 2018). While none of the studies included direct measures of DOM hydrophobic and hydrophilic fractions, one measured THM formation potential and found no change between water samples taken from drained and rewetted blanket bog mesocosms (Gough et al., 2016), suggesting that in the short term ditch blocking may not reduce THM formation following water treatment.

More broadly, therefore, while the evidence suggests that ditch blocking may reduce DOM concentrations within pore waters (Table 3, Figure 3), there is no published evidence for such activities to have successfully influenced DOM concentrations in runoff at a catchment scale, and thus at a level of potential relevance to raw water supply to treatment works. It is important to note, however, that catchment-scale studies are hugely challenging logistically and financially to design and maintain.

3~~2~~.2. Revegetation of bare peat

Exposure of bare peat following anthropogenic disturbance has been an extensive problem in a number of UK peatland regions, most notably in the Peak District of Northern England (Pilkington et al., 2015). The subsequent erosion of the peat has caused significant problems for the water industry because of the high particulate loads from the catchment to the downstream reservoirs. There have been significant efforts in recent years to revegetate some of the most degraded upland peatland areas in order to stabilise these systems (Pilkington et al., 2015).

Published research on the impacts of revegetation of peatland areas on DOM is limited, but Qassim et al. (2014) found that pore water DOM concentrations were higher in revegetated sites compared to bare peat areas and vegetated controls over a five-year period. The initial revegetation mix in this work was a nurse crop of *Agrostis* sp., *Deschampsia flexuosa* and *Festuca* sp. applied in combination with additions of lime and fertiliser to ensure grass growth. Heather brash was also applied to stabilise the peat surface and provide a seed source of peatland species. The use of lime is likely to have increased DOM solubility through a reduction in acidity of the peat (Evans et al., 2012), and the re-establishment of vegetation may have increased the production of 'new' DOM via root leachate and fresh litter decomposition. Particulate losses from peatland systems decreased following stabilisation of the peat surface through revegetation irrespective of gully blocking activities (Pilkington et al., 2015), as overland flow velocities are lower on vegetated peat than bare peat (Holden et al., 2008). However, the same study (Pilkington et al., 2015), and more recent assessments of the effects of revegetation on DOM concentrations (Stimson et al., 2017; Alderson et al., 2019), found no long-term changes in DOM concentrations following revegetation at the headwater catchment scale.

Radiocarbon (^{14}C) measurements of DOM in UK upland waters indicate that the principal source of DOM in waters draining relatively undisturbed soils is recent primary production, probably formed within the last few years (Evans et al., 2014). It follows, therefore, that plant productivity, and plant tissue composition and degradability, which depend both on ambient environmental conditions and species composition, may be important factors, both for DOM concentrations and the treatability of the DOM produced. In a laboratory-based extraction experiment, DOM leached from *Sphagnum* was more easily removed by a conventional coagulation process and decomposed more rapidly than DOM leached from *Molinia caerulea* or *Calluna vulgaris* litter. In addition, *M. caerulea* and *C. vulgaris* litter released more DOM per unit dry weight compared to Sphagnum litter (Ritson et al., 2016). At the field scale, published results are less clear cut: one study found that DOM concentrations in pore waters were higher in areas of blanket bog dominated by *C. vulgaris* compared to areas dominated by sedges or *Sphagnum* species (Armstrong et al., 2012). In contrast, Parry et al. (2015) found no correlation between dominant vegetation type (differentiated into ericoids, grasses, sedges and bare peat) and stream water DOM concentrations in headwater catchments. This may reflect the greater biotic (as well as soil) heterogeneity of peatland environments at the catchment scale in comparison to single species plot experiments.

The evidence available to date suggests that while revegetation of peatland sites has stabilised bare peat surfaces (e.g. Pilkington et al., 2015), and is likely to have reduced particulate organic matter loss, it ~~has~~ may not have significantly changed DOM export from peat headwater catchments. Laboratory based work has shown that the species present could impact DOM treatability, with *Sphagnum* derived DOM being more easily treatable that *M. caerulea* or *C. vulgaris* litter (Ritson et al., 2016). This suggests that catchment management via revegetation should aim to achieve high cover of *Sphagnum* species compared to vascular plants to maximise DOM treatability (Table 3).

3~~2~~.3. Plantation forestry / deforestation

It has long been recognised that plantation forestry activities can have detrimental impacts on reservoir water quality and treatability. For example, in 1984 it was shown that drainage and deforestation resulted in large sedimentation issues at Crai Reservoir in south Wales (Stretton, 1984 cited in: Hudson et al. 1997), while large pulses of nutrients (N and P) to upland streams were observed after forest-felling (Neal, 2002). This review covers the impact of ground preparation and forest planting, in-situ forest growth, and forest removal (including forest to bog restoration) on peat on DOM concentration and quality. Note that UK blanket bogs do not naturally support trees, and that virtually all forestry activities on peat in the UK involve drainage and planting with non-native conifers.

To reduce the impacts of forest operations on sediment and nutrient loss and consequent raw water quality in the UK, the Forest and Water Guidelines now state that no more than 20% of a drinking water catchment should be felled in any 3 year period (Forestry Commission, 2017). In addition to this, although primarily to conserve soil carbon stocks rather than for improved water quality, the 2000 Forestry Commission guidance note on forest and peatland habitats (Patterson and Anderson, 2000) states that approval will no longer be given for forestry planting or regeneration on active raised bog or inactive raised bogs that could be restored to active bog, and areas of active blanket bog greater than 25 ha area and peat > 45 – 50 cm depth.

A recent review for Yorkshire Water (Chapman et al., 2017) noted that conventional conifer site preparation on peat, peaty gley and peaty podzol soils would be expected to increase DOM concentrations. This would be largely due to the implemented drainage ~~reducing the height~~increasing the depth of the water table and consequently ~~increasing~~ the production of DOM via increased aeration of the peat surface (Clark et al., 2009). In the absence of extensive primary data on the effects of forest establishment from the UK, research from Fennoscandia supports this conclusion; Jandl et al. (2007), in their review of studies of the effect of forest management on soil carbon sequestration, highlighted two Finnish studies where DOM concentrations increased following drainage ditch installation but returned to pre-drainage levels later in the forest cycle, while Schelker et al. (2012) observed increased colour in sites being prepared for forestry in northern Sweden. Furthermore, Rask et al. (1998) reported an increase in colour in streams draining peat dominated catchments following afforestation in Finland, while in Sweden afforestation has also been linked to long-term increases in water colour (Skerlep et al., 2019). However, it should be noted that forest management in Fennoscandia often involves relatively limited levels of disturbance (e.g. ditching to accelerate growth of existing mixed native trees species) whereas in the UK it typically involves ditching, ploughing and active planting with non-native monocultures.

At a regional to national scale in the UK, recent work suggests that the presence of plantation forestry on peat soils is associated with higher DOM concentrations in streams and rivers compared to peat soils supporting semi-natural vegetation (Williamson et al., 2021).

The presence of conifers on peat soils in a UK and Irish context is associated with higher pore water DOM concentrations across the four studies covered in this review (Table 2), with a mean difference of approximately 130%. The exception to this pattern was found in spruce plantations in north Wales where DOM concentrations in pore waters were 19% lower than in adjacent blanket bog, though this pattern was not seen in pore water samples from under other plantation species (Gough et al., 2012). We found only one study (Gaffney et al., 2018) that compared DOM concentrations in drainage ditches between forested and ~~intact~~ blanket bog areas, with DOM concentrations approximately 100% higher in the former. The presence of forestry on peat had less clear cut impacts on streamwater DOM concentrations, with two out of three studies reporting no significant difference between streams draining catchments with forestry and intact blanket bogs (Shah et al., 2021; Flynn et al., 2022), and the third showing ~~an~~ DOM concentrations approximately 25% higher in a stream draining a forested catchment compared to a blanket bog catchment (Cummins and Farrell, 2003).

Clear~~Tree~~ felling tends to cause an increase in DOM, though the effects are not universal across studies and locations. Three of five studies of streamwater DOM concentrations reported increases following felling (Cummins and Farrell, 2003; Zheng et al., 2018; Shah and Nisbet, 2019), with a mean increase of approximately 43%, although the two studies in the Flow Country showed no change (Muller et al., 2015) and ~~a~~ 6% lower~~decrease in~~ concentrations compared to the control site (Muller and Tankere-Muller, 2012), which was attributed to the success of buffer strips between the plantation and the monitored stream. The mean increase in DOM concentrations in ditches was nearly 200% (ranging from a 50% increase to a 500% increase, see Table 2) (Cummins and Farrell, 2003; Muller and Tankere-Muller, 2012; Muller et al., 2015; Gaffney et al., 2018). Most studies measuring DOM concentrations from forestry on peat were relatively short-term in timeframe, lasting two years or shorter. Only two studies monitored DOM concentrations for five years or longer.

There has been comparatively little research on the effects of forest presence on the treatability of DOM, although Gough et al. (2012) evaluated DOM concentrations and $SUVA_{254}$ values in waters draining catchments forested with different tree species. They found that pore water leachates from pine and larch plantation yielded particularly high DOM concentrations relative to a blanket bog control (19 and 13 mg L^{-1}, respectively, compared to 9 mg L^{-1}). Leachates also had lower $SUVA_{254}$ values (1.2 and 2.4 respectively, compared to 3.3 L mg^{-1} m^{-1}). This would suggest that DOM leaching from plantations dominated by these tree types may be less easily treatable than DOM from blanket bogs. Similarly, samples taken from Scottish blanket and raised bog sites (Howson et al., 2021) found that $SUVA_{254}$ values were lower from forested sites, again suggesting that forestry on peat results in less aromatic, hydrophobic DOM that may be less easily removed via conventional coagulation, possibly because of additional DOM inputs from litter.

Recently there have been attempts to restore previously afforested fen and bog peatlands in parts of Europe and North America under what is often referred to as 'forest-to-bog' restoration (Chimner et al., 2017; Andersen et al., 2017), ~~. Although still a relatively new practice within the UK, this type of restoration has been carried out for 18 years in the Flow Country in northern Scotland,~~ and national policies on peat restoration may lead to its expansion in the future. Some of the studies listed in Table 2 (Muller and Tankere-Muller, 2012; Muller et al., 2015; Gaffney et al., 2018; Shah and Nisbet, 2019; Gaffney et al., 2020; Howson et al., 2021; Shah et al., 2021) monitored the impacts of felling as part of ongoing forest-to-bog restoration monitoring, with the main differences in management being that the trees were felled to waste (the practice of leaving felled trees *in-situ* to rot) and there was less ground disturbance at the site compared with the use of machinery to extract felled timber (Gaffney, 2017). However, the practice of felling trees to waste has been suggested to provide a potential additional DOM source as the trees slowly decompose (Muller et al., 2015), with mulched fallen trees providing a major source of water soluble DOM (Howson et al., 2021).

As bog vegetation regenerated after ~~such~~ forest to bog restoration in the Flow Country, DOM concentrations reduced from elevated levels towards those seen in forest control areas. The time frame for complete recovery to pre-intervention levels is to date inconsistent, with some areas still showing elevated DOM in the restoration sites relative to the control sites after 17 years (Gaffney et al., 2018). In others, DOM concentrations had returned to those seen in intact blanket bog within the same time frame (Howson et al., 2021), or were showing inconsistent effects across sub-catchments ~~, with the most upstream catchments showing increased DOM concentrations compared to bog controls, an effect not seen further downstream~~ (Pickard et al., 2022). Other studies have reported shorter-term perturbations in DOM (~4-5 years) following forest-to-bog restoration, including within a Scottish lowland raised bog area, Flanders Moss, where stream water baseline DOM levels were reached within two years at one site (Shah, 2018). In a Finnish study of the impacts of forest to mire restoration, a short-term peak in pore water DOM concentration following initial restoration activity was followed by a return to reference concentrations within six years (Menberu et al., 2017).

In summary, coniferous afforestation of peatlands increases DOM concentrations in pore waters and streams, both during site establishment, potentially during the forest growth, and again as the trees are felled ~~(by up to 500%)~~ (summarised in Table 3). Forest-to-bog restoration as a method of land management produces short-term increases in DOM concentrations while trees are felled and brash remaining on site decomposes. However, given a long enough timeframe, DOM concentrations appear to reduce back towards levels seen from comparable control locations. From a water company perspective it is important to note that this time frame can be up to 20 years in blanket bogs, i.e. considerably longer than the standard funding cycle. Removing felled timber and brash from the site, rather than felling to waste, would be expected to greatly reduce the magnitude and duration of any DOM peak.

3~~2~~.4. Managed burning

Managed burning of peatland vegetation (Figures ~~2~~ 1 & 5) (primarily the burning of *Calluna* sp. as part of grouse moor management) is a contentious issue within peatland conservation and management (e.g. Davies et al., 2016) and has been extensively reviewed and debated over the past decade, particularly in relation to the impacts on DOM (Worrall et al., 2010; Holden et al., 2012; e.g. Brown et al., 2015; Harper et al., 2018)~~, and most recently by Harper et al. (2018)~~. There is little evidence within these reviews to suggest that DOM concentrations or colour increase within peat pore waters following managed burns. A recent study showed no change in DOM concentrations following low and high intensity burning (Grau-Andres et al., 2019), and in previous studies pore water DOM concentrations were unchanged (Clay et al., 2009; Clay et al., 2012; Worrall et al., 2013) or decreased (Worrall et al., 2007a). At the catchment scale, positive correlations between the extent of burning and DOM concentrations and water colour have been interpreted as causal (Clutterbuck and Yallop, 2010; Yallop et al., 2010; Ramchunder et al., 2013) ~~although~~ but this has been questioned in the literature (Holden et al., 2012)~~,~~. Burning as a management practice is designed to ensure that there is a mosaic of variously aged heather habitat, so it seems plausible that these effects are more linked to changes in vegetation cover. As previously discussed *C. vulgaris* produced higher amounts of DOM than *Sphagnum* in the laboratory (Ritson et al., 2016) and at plot scale (Armstrong et al., 2012). It is also worth noting that Evans et al. (2017b) found that a wildfire in Northern Ireland resulted in a temporary reduction of DOM concentrations in a downstream monitoring lake, which was attributed to re-acidification of catchment soils following the fire, as well as the loss of DO~~C~~M-producing vegetation cover.

4~~3~~: Discussion~~ and conclusion~~

4.1: Role of peatland catchment management

Table 3 summarises the range and extent of the current peer-reviewed evidence for the impacts of peatland restoration on DOM concentrations in raw water and the treatability of the DOM present. However, considerable knowledge gaps remain regarding the effects of peatland restoration on raw water DOM concentrations and treatability. Our thorough screening of the literature revealed remarkably few published primary studies in this area, despite a widespread belief among UK conservation, policy and water industry groups that peatland degradation has driven increased DOM concentrations in upland water supplies (Anderson, 2012). This lack of evidence, and the mixed findings of those studies that have been undertaken, suggest ~~to the extent~~ that generalisations of the effects of most of the interventions examined must be taken with considerable caution.

The available literature does indicate that both revegetation of bare peat (particularly to *Sphagnum* dominated bog) and ditch blocking is associated with decreased DOM concentrations within pore waters and ditches, at the location where restoration occurs. However, and in contrast to ~~much more widely~~ reported positive impacts of these restoration actions with respect to carbon sequestration, soil particulate losses, flood management and upland biodiversity (Loisel and Gallego-Sala, 2022), evidence that such impacts may translate to ~~comparable~~quantitatively significant- changes within the ~~wider~~ larger and more heterogeneous catchments that provi~~v~~de ~~of more relevance to~~ drinking water resources is generally lacking.

There is ~~arguably much~~ stronger evidence pointing to the risks posed by the afforestation of (naturally unforested) peatlands, and the subsequent management of such plantations. ~~, with plantations~~F~~f~~elling operations, including those associated with ongoing forest management and those associated with forest-to-bog restoration, tend~~ing~~ to lead to increasing DOM concentrations and potentially reduced treatability of exported DOM. In the published literature we have been unable to find experimental evidence incorporating local changes in water chemistry ~~in the vicinity of~~following interventions with changes in downstream DOM processing, to show whether water quality effects are detectable at the point of abstraction for water treatment works. This extension beyond the plot and hillslope scale represents a significant gap in current understanding, as DOM processing continues within the aquatic environment downstream of peatlands~~ restoration sites~~, and may be affected by upstream management.

Robust quantification of the impacts of catchment management on DOM concentration and treatability at the point of abstraction clearly represents a major current evidence gap. The ~~size of the research challenge with respect to the necessary~~ spatial and temporal scale required to understand these impacts, as well as the ~~and~~ need for robust Before-After-Control Impact (BACI) ~~of any field~~ experiments, entail significant cost, which ~~cannot be underestimated, and~~ perhaps explains ~~in part~~ the current dearth of reliable information. This is particularly pertinent when changes in water chemistry may take a number of years to be seen, depending on catchment dynamics ~~and~~ as well as possible ~~within~~ in-reservoir processes. Our review has highlighted that studies of different ~~catchment~~ land management approaches have not been followed downstream to monitor their impacts to the wider catchment.

4.2: DOM processing in drinking water catchments

The general paucity of evidence to support widespread (terrestrial) -catchment-focussed interventions specifically to manage source water DOM concentrations and treatability leads then to the question as to whether there are other water quality management options that could be applied within reservoirs and whether these have been comparatively overlooked for DOM. DOM in rivers and lakes is subject to both biotic and abiotic processing, which ~~change~~ affect both concentrations and chemical structure (e.g. Tranvik et al., 2009) and hence ~~affect~~ treatability. For example DOM is lost to respiration (Koehler et al., 2012; Stets et al., 2010), sedimentation (Einola et al., 2011; Von Wachenfeldt and Tranvik, 2008), photo-oxidisation (via UV radiation) (Moody et al., 2013; Koehler et al., 2014) and flocculation with naturally-occurring aluminium and iron (Mcknight et al., 1992; Koehler et al., 2014).

More importantly for treatability, however, DOM is generated within lakes and reservoirs via photosynthesis (production of algal exudates and release via cell lysis) and through processing of particulate matter (Tranvik et al., 2009) so that DOM concentrations at the point of abstraction from reservoirs represent the sum of these removal and generation processes. Consequently, the resulting DOM tends to be relatively transparent and hydrophilic in comparison with DOM generated by organic rich soils, and thus presents different challenges for treatment, particularly as the hydrophilic DOM is not easily removed through coagulation (Matilainen et al., 2010) and may lead to the need for additional capital investment in order to effectively reduce residual DOM in drinking water.

Algal production, and hence within-reservoir generation of DOM, is often limited by the availability of phosphorus, nitrogen, or both. Hence, waterbodies with high concentrations of inorganic nutrients~~, either delivered externally from their catchments or re-released internally from sediments,~~ are likely to generate additional DOM within the water column (Feuchtmayr et al., 2019; Evans et al., 2017a). Further, evidence is growing on the importance of lake and reservoir bed sediments as a direct source of DOM to the water column, with reducing conditions occurring during stratification of lakes and reservoirs causing redissolution of previously sedimented organic matter (Peter et al., 2017).

In their assessment of DOM in lake and reservoir inflows and outflows~~, including those of several reservoirs~~, Evans et al. (2017a) concluded that any measures that can reduce N and P export from the catchment (e.g. Spears and May, 2015) or release from sediments, or which can strip nutrients from the water column (e.g. Spears et al., 2016), could provide effective mitigation for high DOM concentrations by reducing algal DOM production.~~. For example, measures for reducing nutrient loading to lakes from the catchment (Spears and May, 2015) and bed sediments (Spears et al., 2016) can be effective in reducing algal biomass in UK lakes - although the effects on algal DOM production in relation to drinking water treatment require further assessment. To date, this option has rarely been considered in relation to DOM-related treatment issues, although nutrient management is often considered in relation to other (taste and odour) related treatment issues. The available evidence therefore suggests that measures to reduce taste and odour problems could deliver co-benefits in relation to DOM levels.~~

It is pertinent, therefore, to consider whether measures which reduce in-reservoir DOM production, and/or favour in-reservoir DOM removal, may be as – or perhaps more – effective than measures aimed at reducing DOM export from the terrestrial catchment. For lakes acting as DOM sources, management regimes that reduce nutrient (primarily N and P) inputs from catchments and/or internal loading of nutrients and DOM from sediment to the water column may be more effective than those focussed on reducing inflowing DOM concentrations directly. Restricting nutrient inputs is also likely to reduce organic nitrogen concentrations relative to organic carbon concentrations, which has the added benefit of reducing the formation potential of nitrogenous DBPs. In addition, Birk et al. (2020) suggest that rising DOM loading from the catchment may act to dampen algal responses to nutrients through light limitation of primary production within some European lakes. If, by extension, this also limits in–reservoir DOM production then catchment interventions that relieve DOM load, but not nutrient load, may result in an increase in in-reservoir DOM production. Even in the case of less nutrient-rich water bodies, it appears that reducing N and P loadings would be beneficial for water treatment as this is likely to restrict additional DOM formation.

5. Conclusion

In summary, our review demonstrates that catchment management initiatives, while providing clear overall restoration benefits for peatlands, have yet to deliver a generalised solution to the challenge of stabilising or reversing DOM increases in drinking water sources.~~, although t~~There is some evidence that catchment interventions may provide benefits for DOM export in specific cases, but (with the possible exception of forest management activities) these have rarely been demonstrated consistently or at the whole-catchment scale. Furthermore, it now seems clear that the recent decadal-scale increase in surface water DOC concentrations was the result of an external driver (i.e. decreasing acid deposition), both in the UK and across large parts of Europe and North America, and cannot realistically be ‘managed away’. However, c~~C~~atchment management measures that reduce in-reservoir DOM production, or favour in-reservoir DOM removal, may be as or more effective, particularly with respect to more nutrient rich systems. More generally, it seems clear that catchment management should be considered part of the response strategy to rising DOM levels, and as part of a process to improve the resilience of source waters, not a panacea. It is therefore important that research science and the water industry work together to measure variables at the temporal and spatial scale required and to ~~also~~ develop~~s~~ effective tools to predict likely future DOM levels resulting from a combination of large-scale and catchment-scale drivers, to ensure that investments in both catchment management measures and DOM treatment infrastructure are correctly targeted, integrated, timely and cost-effective.

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Tables:

Table 1: Summary of the impacts of drainage ditch blocking on DOM concentrations and fluxes from peatlands, reported in increasing time since ditch blocking. BA = Before/After, CI = Control/Intervention. Reference to chronosequence in the survey design refers to a sampling strategy whereby sites that had had interventions at different times were used as a proxy for control sites, while survey refers to a short term one-off sampling of multiple locations.

Reference	Location	Sampling scale	Concentration or flux measured	Time since ditch blocking	Experimental Design	Change since ditch blocking
Worrall et al. (2007b)	UK, blanket bog	Ditches	DOM concentration	7 months	BACI	100% increase in DOM concentration.
Turner et al. (2013)	UK, blanket bog	0 and 1st order ditches	DOM concentration and flux	1 year	BACI	DOM concentration decreased by 2.5% compared to control, DOM flux decreased by 2.2 – 9.2% as a result of decreased water export.
Gibson et al. (2009)	UK, blanket bog	Ditches	DOM concentration and flux	1 year	CI	DOM concentrations unchanged, water flux decreased by 39% meaning DOM flux also declined by the same amount.
Wilson et al. (2011)	UK, blanket bog	Ditches and headwater streams	DOM concentration and flux	2 years	BACI	DOM concentrations unchanged, fluxes were 88% lower in streams draining ditch-blocked catchments due to much lower estimated water export.
O'brien et al. (2008)	UK, blanket bog	Headwater streams	DOM flux and water colour	2 years	BACI	Water colour was unchanged. Fluxes decreased by 24% in streams as a result of decreasing water export.
Menberu et al. (2017)	Finland fen, pine mire and spruce mire	Pore water	DOM concentration	3 years	BACI	41% reduction in DOM concentration.
Evans et al. (2018)	UK, blanket bog	Ditches	DOM concentration	4 years	BACI	No change in DOM concentration
Wallage et al. (2006)	UK, blanket bog	Pore water	DOM concentration	5 years	CI	DOM concentration lower in porewaters adjacent to blocked ditches (69% lower compared to open ditches)
Haapalehto et al. (2014)	Finland, raised bog	Pore water	DOM concentration	5 years and 10 years	Chronosequence	DOM concentration approx. 10% lower in sites 5 years post restoration and 25% lower in sites 10 years post restoration
Haapalehto et al. (2014)	Finland, raised bog	Ditches	DOM concentration	5 years and 10 years	Chronosequence	Concentrations approx. 75% higher in sites 5 years post restoration and 50% higher in sites 10 years post restoration
Armstrong et al. (2010)	UK, blanket bog	Ditches	DOM flux	7 years	CI	No change in DOM flux
Strack et al. (2015)	Canada, bog	Pore water and ditch water	DOM concentration	10 years	CI	No change in pore water DOM concentration. Ditch water DOM concentrations were

						similar in spring and summer and up to 30% lower in the restored site in autumn.
Armstrong et al. (2010)	UK, blanket bog	Ditches from a survey in Northern England and Northern Scotland	DOM concentration	6 months to 18 years	Survey	DOM concentrations 28% lower on average in blocked drains compared to unblocked drains.
Holl et al. (2009)	Germany, ex-fenland extraction site	Pore water	DOM concentration	20 years	CI	DOM concentrations 37% lower at restored site compared to drained site.
Urbanova et al. (2011)	Czech Republic, bog	Pore water	DOM concentration	NA comparison between drained and intact sites	CI	No difference in DOM concentration between intact and moderately degraded site, 50% higher DOM concentrations at highly degraded site.
Pickard et al. (2022)	UK, blanket bog	Headwater streams	DOM concentration	6-8 years	CI	No difference in DOM concentration between drained and restored sites. DOM concentrations significantly higher (50% increase) in drained and restored sites compared to non-drained controls.

Table 2: UK studies reporting DOM concentration monitoring of forestry activities on peat. Note that where percentage differences are preceded by ~ concentrations were not explicitly listed in text, figures and tables or supplementary information so are estimated from graphs.

Paper	Location	Forestry activity monitored	Scale	Timescale of monitoring	% difference
Muller and Tankere-Muller (2012)	Flow Country	Felling compared to blanket bog	Stream (upstream and downstream)	1 year post felling	-6%
Zheng et al. (2018)	Central Scotland	Felling compared to windfarm on blanket bog	Stream	1 year ~ 8 years after felling	~ 100%
Muller et al. (2015)	Flow Country	Felling compared to blanket bog	Stream	3 months before ~ 1 year after	No difference
Shah and Nisbet (2019)	Central Scotland (raised bog)	Before / after felling	Stream	1 year before and up to 8 years after	0%, 29% & 51% (mean 27%)
Cummins and Farrell (2003)	Ireland	Before / after felling	Stream	5 years	~0 – 100%

Gaffney et al. (2020)	Flow Country	Before / after felling and felling compared to blanket bog	Stream	2 years	No significant difference
Muller et al. (2015)	Flow Country	Before / after felling	Ditch	3 months before ~ 1 year after	~ 75%
Gaffney et al. (2018)	Flow Country	Before / after felling	Ditch	1 year post felling	~ 150%
Cummins and Farrell (2003)	Ireland	Before / after felling	Ditch	5 years	~50%
Gaffney et al. (2018)	Flow Country	Felling compared to blanket bog	Ditch	0 – 17 years post felling. 1 year of measurement	~500%
Muller and Tankere-Muller (2012)	Flow Country	Felling compared to blanket bog	Ditch	1 year post felling	30-325% (overall average 159%)
Gough et al. (2012)	North Wales	Presence / absence of forestry	Pore waters	1 off sampling	-19% - 111% (average 45%)
Howson et al. (2021)	Flow Country	Presence / absence of forestry	Pore waters	~ 20 months	~ 66%
Howson et al. (2021)	Central Scotland (raised bog)	Presence / absence of forestry	Pore waters	~ 20 months	~14%
Flynn et al. (2022)	Ireland	Presence / absence of forestry	Pore waters	~ 2 years	~400%
Gaffney et al. (2018)	Flow Country	Presence / absence of forestry	Ditch	0 – 17 years post felling 1 year of measurement	~ 100%
Flynn et al. (2022)	Ireland	Presence / absence of forestry	Stream	~ 2 years	No significant difference
Shah et al. (2021)	Flow Country	Presence / absence of forestry – time series	Stream	25 years	No significant difference
Cummins and Farrell (2003)	Ireland	Presence / absence of forestry	Stream	5 years	~25%

Table 3: summary of the published impacts of catchment management activities on DOM concentrations and treatability, focussing on those studies relevant in a UK and Irish context. Numbers in brackets refer to the number of studies showing that effect in each case, while the overall impacts on DOM concentration and treatability for water treatment are shown as +/=/- (positive/neutral/negative) for concentrations and treatability respectively.

Catchment intervention	Impact on DOM concentration	Impact on DOM treatability
Ditch blocking (=/=)	**Increase (2)** (Worrall et al., 2007b; Haapalehto et al., 2014) **No change (8)** (O'brien et al., 2008; Gibson et al., 2009; Armstrong et al., 2010; Wilson et al., 2011; Urbanova et al., 2011; Turner et al., 2013; Strack et al., 2015; Evans et al., 2018) **Decrease (5)** (Wallage et al., 2006; Holl et al., 2009; Armstrong et al., 2010; Haapalehto et al., 2014; Menberu et al., 2017)	**No change (5)** (Glatzel et al., 2003; Strack et al., 2015; Gough et al., 2016; Lundin et al., 2017; Peacock et al., 2018)
Revegetation (to grass species) (=/-)	**Increase (2)** (Qassim et al., 2014; Ritson et al., 2016) **No change (4)** (Parry et al., 2015; Pilkington et al., 2015; Stimson et al., 2017; Alderson et al., 2019)	**Decrease (1)** (Ritson et al., 2016)
Revegetation (to heather) (-/-)	**Increase (2)** (Armstrong et al., 2012; Ritson et al., 2016) **No change (1)** (Parry et al., 2015)	**Decrease (1)** (Ritson et al., 2016)
Revegetation (to *Sphagnum*) (+/+)	**Decrease (1)** (Armstrong et al., 2012)	**Improve (1)** (Ritson et al., 2016)
Forest presence (-/-)	**Increase (5)** (Cummins and Farrell, 2003; Gough et al., 2012; Gaffney et al., 2018; Howson et al., 2021; Flynn et al., 2022) **No change (2)** (Shah et al., 2021; Flynn et al., 2022)	**Decrease (2)** (Gough et al., 2012; Howson et al., 2021)
Clearfell and forest-to-bog conversion (-/-)	**Increase (6)** (Cummins and Farrell, 2003; Muller and Tankere-Muller, 2012; Muller et al., 2015; Gaffney et al., 2018; Zheng et al., 2018; Shah and Nisbet, 2019) **No change (3)** (Muller and Tankere-Muller, 2012; Muller et al., 2015; Gaffney et al., 2020)	**Decrease (1)** (Zheng et al., 2018)
Managed burning (-/no evidence)	**Increase (3)** (Clutterbuck and Yallop, 2010, Yallop et al., 2010, Ramchunder et al., 2013) **No change (4)** (Clay et al., 2009; Clay et al., 2012; Worrall et al., 2013; Grau-Andres et al., 2019) **Decrease (1)** (Worrall et al., 2007a)	

Figure legends:

~~Figure 1: Mean (+/- Standard error) annual dissolved organic carbon (DOC) concentrations from the 23 UK Upland Water Monitoring Network sites. These sites are predominately situated in the north and west of the UK – see www.uwmn.uk for more details.~~

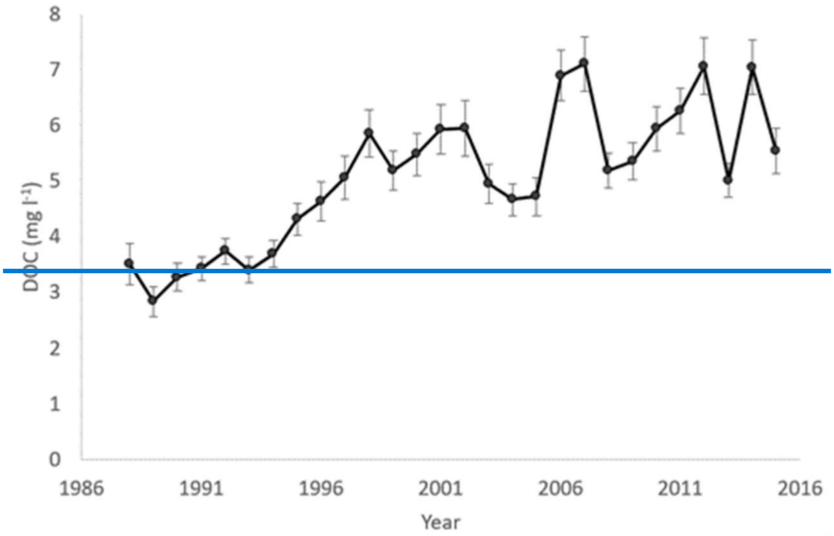

Figure ~~2~~1: Schematic showing anthropogenic pressures on peatland catchments, and the potential peatland management processes covered in this review.

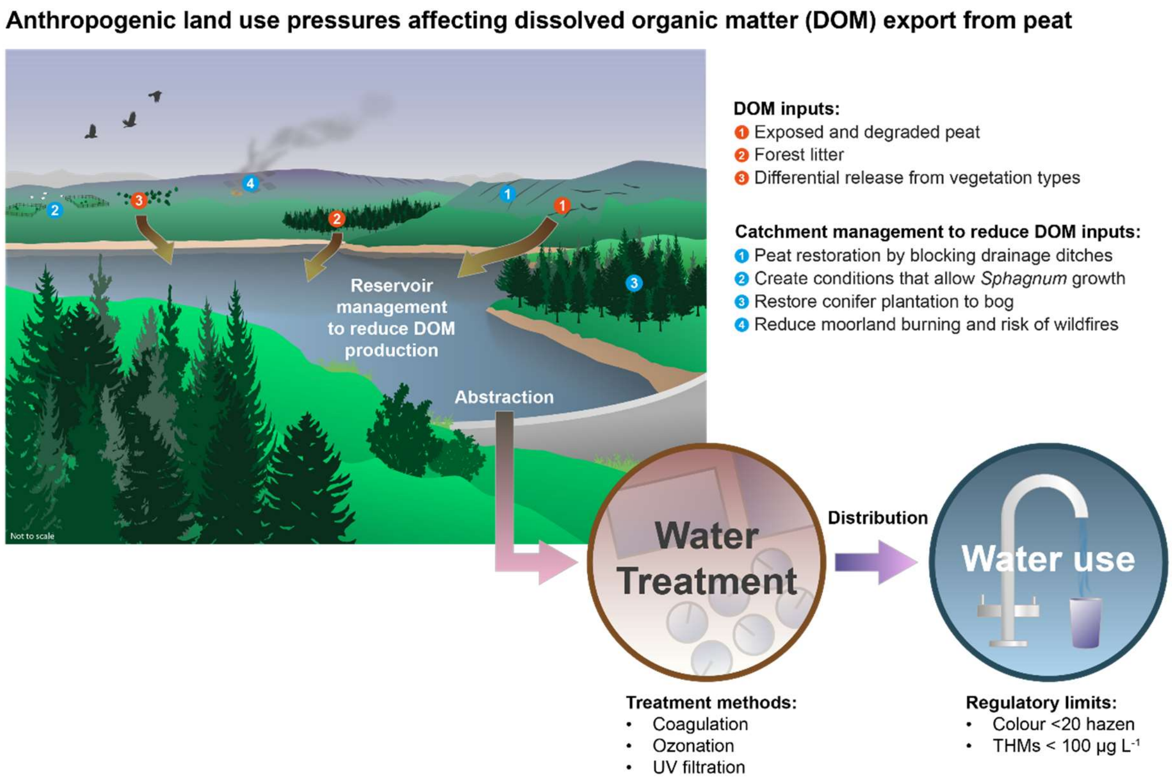

Figure 2: Mean (+/- Standard error) annual dissolved organic carbon (DOC) concentrations from the 23 UK Upland Water Monitoring Network sites. These sites are predominately situated in the north and west of the UK – see www.uwmn.uk for more details.

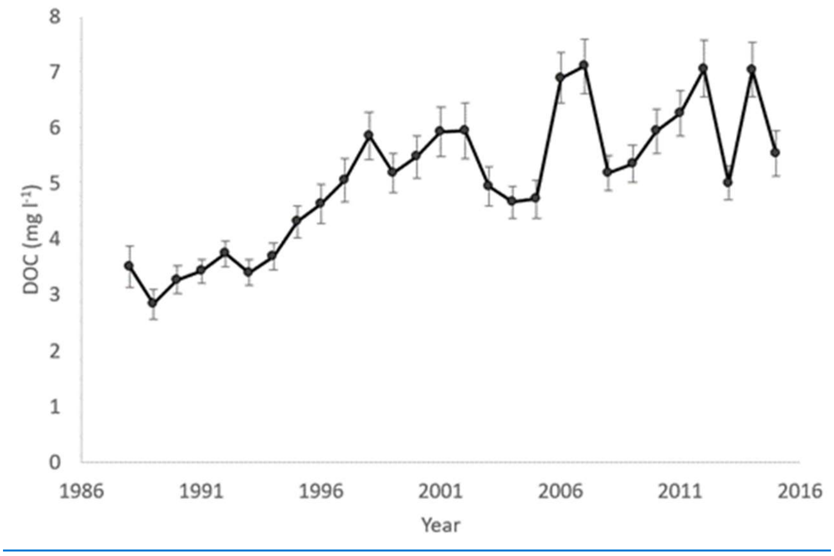

Figure 3: Percentage change in DOM concentration following ditch blocking. Grey circles show DOM percentage change in peatland pore waters, and black circles show DOM percentage change in ditches and streams.

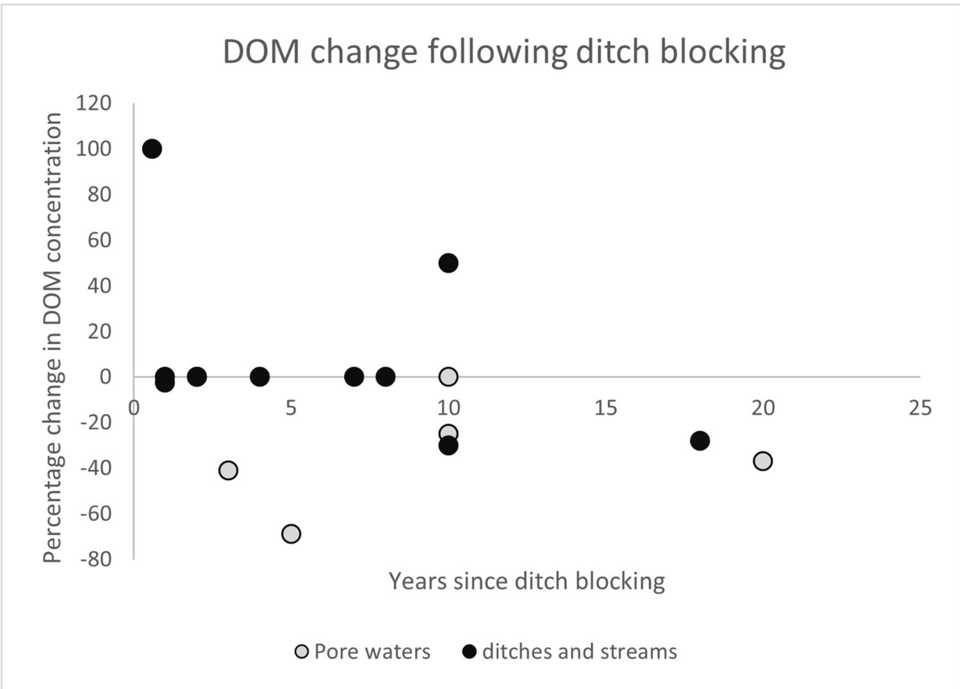

Figure 4: Drainage ditches before (left) and after (right) blocking on a blanket bog in North Wales, the ditches run down the slope and individual dams can be seen crossing the ditches (Photos: Chris Evans).

Figure 5: Burning of vegetation on peat in North Wales (Photo: Chris Evans).