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Biogeosciences Discussions

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

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Abstract

In the UK most large reservoirs constructed for public water supply are in upland areas and situated in catchments that contain at least some organic-rich soils. Dissolved organic matter (DOM) leaching from these soils imparts a brownish colour to water and raises treatment challenges for the water industry since excessive post-treatment concentrations result in the generation of potentially harmful disinfection by-products in drinking water. The primary method for maintaining sufficiently low pre-disinfection DOM concentrations is chemical coagulation, but in the past 15 years water companies have increasingly considered the capacity for catchment interventions to improve raw water quality at source, reducing the need for costly and complex engineering solutions in treatment works. There remains considerable uncertainty around the effectiveness of these catchment engineering-based 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 management options for upland organic soil-dominated catchments that are being considered by the water industry as options for controlling DOM releases. These are ditch blocking, revegetation, reducing forest cover, and cessation of managed burning. Results of plot scale investigations into effects of ditch blocking on ditch-blocking are available but largely equivocal, while there is a paucity of information regarding impacts at spatial scales of more direct relevance to water managers. 'The presence of plantation forestry on peat soils is generally associated with increasing DOM concentrations, although canopy removal has little short-term benefit and can even further increase concentrations. Although not widely studied, the available evidence suggests that *Sphagnum* mosses produce DOM that is more easily removed via conventional treatment processes compared to vascular plants such as heather and grass species. We found surprisingly little published research around the extent to which manipulation of in-reservoir processes might be used to mitigate or exacerbate changes in inflowing DOM as part of a catchment management approach.

This review concluded that catchment management measures have rarely been monitored with downstream water quality as the focus, and that restoration impacts vary across sites. 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.

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Introduction

Peatland restoration has become an integral part of the UK environment strategy, particularly in the drive toward Net Zero. It is founded on the potential to achieve multiple benefits that include improving biodiversity, 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). The dissolved organic matter DOM concentrations of these water tend to be relatively high, and have been rising since the 1980s (e.g. Naden and Mcdonald, 1989; Robson and Neal, 1996; Harriman et al., 2001; Freeman et al., 2001; Worrall et al., 2004). 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 being approximately double those seen in the late 1980s (Figure 1). 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 Europe, and appears to largely result from an 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). Rising levels of DOM in waters draining many of these catchments pose considerable water treatment challenges, with respect to increasing treatment costs and risks of regulatory failure (see Figure 1). It has been proposed that peatland restoration measures might help slow or even reverse these DOM trends, but while some of the benefits of peatland restoration are now becoming clear (e.g. Glenk and Martin-Ortega, 2018), evidence for impacts on water quality have been more difficult to glean.

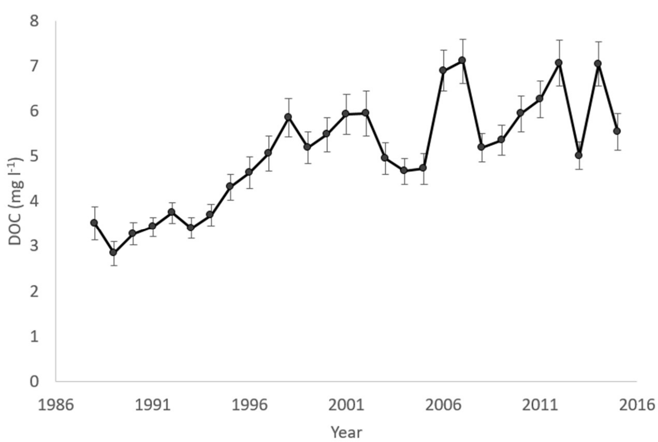

Figure 1: Mean (+/- Standard error) annual dissolved organic carbon (DOC) concentrations from UWMN sites. These sites are predominately situated in the north and west of the UK – see www.uwmn.uk for more details.

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), which are regulated by the Drinking Water Inspectorate (DWI) due to their potential carcinogenic properties. Chlorination, a standard disinfection process in most UK WTWs, leaves free chlorine in the water supply as a residual disinfectant. Free chlorine reacts with DOM remaining in the water supply following coagulation and filtration to form DBPs, including THMs. Chloramination, the treatment of drinking water with chlorine and ammonia to form chloramine, has been used as a method of reducing THM formation. However, it has been found that chloramination promotes the formation of nitrogenous DBPs (e.g. Bond et al., 2011; Lavonen et al., 2013), which are more carcinogenic than THMs (Ding and Chu, 2017) and are likely to be regulated in the future. 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. 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 activity (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 indicate hydrophobic dominance, while values less than 2 show 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). Overall, the cost of DOM removal in UK water supplies is estimated to be hundreds of millions of pounds, and has risen sharply in recent years as a direct consequence of rising DOM concentrations. Major additional costs are incurred where capital investment is needed to upgrade treatment infrastructure designed for lower concentration ranges experienced in the past. 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 work reviews the available peer-reviewed literature and provides a qualitative assessment of the impacts of peatland restoration on DOM concentrations and treatability.

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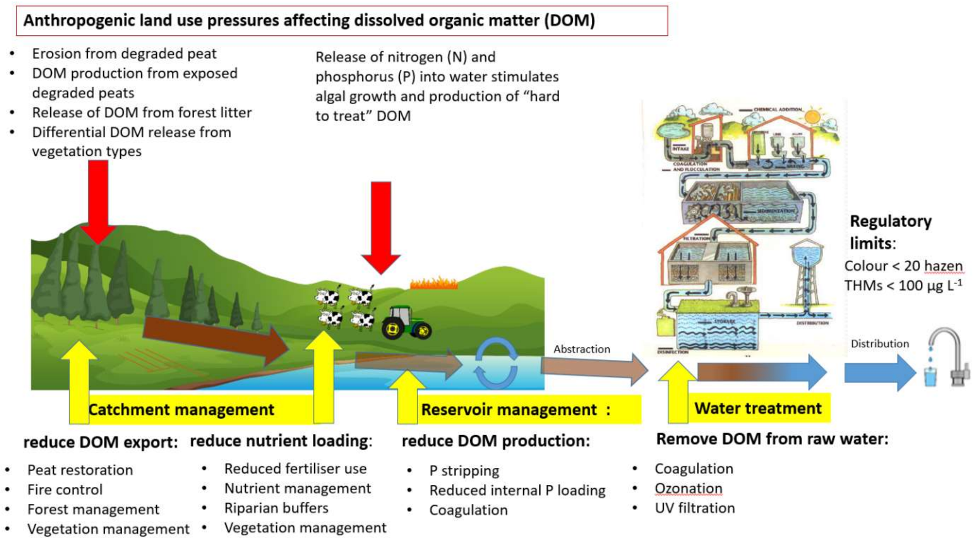

Figure 2: Schematic showing anthropogenic pressures on peatland catchments, and the potential peatland management processes covered in this review.

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

2.1. Ditch blocking

Following peatland drainage, the resulting reductions in water tables, loss of peat forming plant species, and consequent drying and cracking of peat surfaces, exposed previously permanently saturated organic matter to oxidative processes, making it 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 to block ditches in an attempt to restore the hydrological, biogeochemical and ecological functions of these landscapes (Figure 3).

Of the studies relevant to UK peatlands found during this review, four out of five (Table 1) reported significant changes in DOM concentrations within peat soil pore water (i.e. plot scale) following ditch blocking, with a cross-study average 34% reduction (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.

At the ditch scale, results are more variable than those for pore waters (Table 1). The ten studies reviewed showed a mean 10% 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% and 50-75% increases respectively); the median change is zero. Importantly, no significant change in DOM concentration was reported in 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). Likewise, a recent study monitoring DOM concentrations six years after ditch blocking on a blanket bog found no reduction in DOM concentrations in the restored site compared to the ditched site (and both drained and restored site DOM concentrations remained elevated compared to the non-drained control (Pickard et al.,

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2022). Differences in apparent effect size may 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.

Measuring and reporting 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 did reduce discharge along the original ditch lines following blanket bog re-wetting, 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). Studies of DOM flux changes following ditch blocking report an average 24% reduction (range 0 – 88% reduction) in DOM flux, primarily attributed to decreased water fluxes from the restoration site.

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	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)

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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.

Nine studies to date have assessed the potential impact of ditch blocking on DOM treatability and hence the ease of treatability within a conventional water treatment works. They found that 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

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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), 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 and are currently very rare over timescales suitable to detect land management effects on water quality.

Figure 3: 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).

2.2. Re-vegetation 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. 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.

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. in combination with additions of lime and fertiliser to ensure grass growth. Heather brash was 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 ericoid, grasses, sedges and bare peat) and stream water DOM concentrations in headwater catchments. This may reflect the greater heterogeneity of peatland environments at the catchment scale in comparison to single species 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 not 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). However, as with other restoration measures there is currently little in the peer-reviewed literature to demonstrate the effectiveness of this at a catchment scale.

2.3. Plantation forestry / deforestation

It has long been recognised that 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). Large pulses of nutrients (N and P) can also occur after forest-felling (Neal, 2002).

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 > 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 of the water table and consequently increasing the production of DOM via increased aeration of the peat

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surface (Clark et al., 2009). 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). At a regional to national scale in the UK recent work suggests that the presence of plantation forestry on peat soils increases DOM concentrations in streams and rivers compared to peat soils with semi-natural vegetation (Williamson et al., 2021).

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	% difference
Muller and Tankere-Muller (2012)	Flow Country	Felling compared to blanket bog	Stream (upstream and downstream)	-6%
Zheng et al. (2018)	Central Scotland	Felling compared to windfarm on blanket bog	Stream	~ 100%
Muller et al. (2015)	Flow Country	Felling compared to blanket bog	Stream	No difference
Shah and Nisbet (2019)	Central Scotland (raised bog)	Before / after felling	Stream	0%, 29% & 51% (mean 27%)
Cummins and Farrell (2003)	Ireland	Before / after felling	Stream	~0 – 100%
Gaffney et al. (2020)	Flow Country	Before / after felling and felling compared to blanket bog	Stream	No significant difference
Muller et al. (2015)	Flow Country	Before / after felling	Ditch	~ 75%
Gaffney et al. (2018)	Flow Country	Before / after felling	Ditch	~ 150%
Cummins and Farrell (2003)	Ireland	Before / after felling	Ditch	~50%
Gaffney et al. (2018)	Flow Country	Felling compared to blanket bog	Ditch	~500%
Muller and Tankere-Muller (2012)	Flow Country	Felling compared to blanket bog	Ditch	30-325% (overall average 159%)
Gough et al. (2012)	North Wales	Presence / absence of forestry	Pore waters	-19% - 111% (average 45%)
Howson et al. (2021)	Flow Country	Presence / absence of forestry	Pore waters	~ 66%

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Howson et al. (2021)	Central Scotland (raised bog)	Presence / absence of forestry	Pore waters	~14%
Flynn et al. (2022)	Ireland	Presence / absence of forestry	Pore waters	~400%
Gaffney et al. (2018)	Flow Country	Presence / absence of forestry	Ditch	~ 100%
Flynn et al. (2022)	Ireland	Presence / absence of forestry	Stream	No significant difference
Shah et al. (2021)	Flow Country	Presence / absence of forestry – time series	Stream	No significant difference
Cummins and Farrell (2003)	Ireland	Presence / absence of forestry	Stream	~25%

The presence of forestry 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 (Gough et al., 2012). We found only one study (Gaffney et al., 2018) comparing DOM concentrations at a ditch scale between forested and intact blanket bog areas, with DOM concentrations being approximately 100% higher in ditches draining the forested areas. At the stream scale the presence of forestry on peat had less clear cut impacts on 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).

Tree felling tends to produce larger increases in DOM, though the effects are not universal across studies and locations. At the stream scale three of five studies 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 Thurso catchment showed no change (Muller et al., 2015) and a 6% decrease in concentrations (Muller and Tankere-Muller, 2012), which was attributed to the success of buffer strips between the plantation and the monitored stream. At the ditch scale the mean increase in DOM concentrations 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).

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.

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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 in the Flow Country, DOM concentrations reduced from elevated levels towards those seen in forest control areas, although the time frame for complete recovery to pre-intervention levels is to date inconsistent, with some areas showing elevated DOM in the restoration sites compared to the control sites after 17 years (Gaffney et al., 2018). However, 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 increases in DOM (~4-5 years), including an assessment of forest -to-bog restoration of 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).

Management of peatland for conifer plantation 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. Water companies should note that this time frame can be up to 20 years in blanket bogs, a time frame considerably longer than the standard funding cycle.

2.4. Managed burning

Managed burning of peatland vegetation (Figure 4) (primarily burning heather for grouse moor management) is a contentious issue within peatland conservation and management (e.g. Davies et al., 2016) and has been extensively reviewed 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), and most recently by Harper et al. (2018). There is little evidence within these reviews to suggest that DOM concentrations or colour increase within pore water at the plot scale 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 plot scale 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 it has been suggested that managed burning contributes to increases in water colour and DOM concentrations (Clutterbuck and Yallop, 2010; Yallop et al., 2010; Ramchunder et al., 2013). Burning as a management

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practice is designed to ensure that there is a mosaic of different aged heather habitat so it seems plausible that these effects are 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.

Figure 4: Burning of vegetation on peat in North Wales (Photo: Chris Evans).

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. Colour coding shows whether the overall conclusion is that effects are positive (green), no / limited change (yellow), or negative (red).

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	**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)	

3: Catchment management impacts on downstream DOM processing

As indicated by Table 3, there remain considerable knowledge gaps in the area of effects of peatland restoration on raw water DOM concentrations and treatability. This review highlights that both revegetation of bare peat (particularly to *Sphagnum* dominated bog) and ditch blocking have been associated with decreased DOM concentrations within pore waters and ditches at the location restoration occurs. The available evidence also suggests, again at this local scale, that plantation forestry presence and felling tend to lead to increasing DOM concentrations and potentially reduced treatability of exported DOM. However, the evidence for impacts at the stream scale is more equivocal. In the published literature we have been unable to find experimental evidence incorporating local changes in water chemistry in the vicinity of interventions with 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 peatland restoration sites.

DOM is not conservatively mixed through rivers and lakes but is subject to both biotic and abiotic processing, which change both concentrations and chemical structure (e.g. Tranvik et al., 2009). Loss pathways for DOM include: 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

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al., 1992; Koehler et al., 2014). 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.

DOM produced via these processes is 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.

Importantly, in-reservoir 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 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 or release from sediments, or which can strip nutrients from the water column, 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.

A future research focus should therefore include answering the question of 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.

4. Conclusions

Increasing DOM concentrations in reservoirs draining catchments dominated by peat soils are a cause for concern for water companies, from both regulatory compliance and treatment cost perspectives. To a large extent this increase appears to be a long-term large-scale phenomenon, driven by improvements in air quality, and thus beyond the direct control of catchment managers. While it is likely that atmospheric deposition-driven changes in DOM are beginning to level off it is also feasible that future climate change could also contribute to further increases in concentrations. The production of DOM in peat soils, for example, is known to be highly sensitive to soil temperature (Clark et al., 2009) while long-term increases in precipitation have also been linked with DOM increases (De Wit et al., 2021).

To date, catchment management initiatives, while providing clear overall restoration benefits for peatlands, do not appear to have produced a generalised solution to the challenge of stabilising or reversing DOM increases in drinking water sources, although there is some evidence that catchment interventions may provide benefits for DOM export in specific cases. We have identified some areas where there is mounting evidence for the importance of certain catchment interventions. In particular, short-term effects of forest felling and harvesting activities have repeatedly shown to have detrimental effects on DOM concentrations. Catchment interventions may also provide co-benefits such as reductions in sediment and particulate organic carbon loadings to reservoirs, reductions in greenhouse gas emissions and enhancement of biodiversity, which may justify the implementation of measures when all benefits are combined, even if the direct benefits for DOM alone may not.

Our review of the published literature highlights a major current evidence gap of importance to the water industry: the quantification of the impacts of catchment management on DOM concentration and treatability at the point of abstraction. The size of the research challenge with respect to the necessary spatial and temporal scale and need for robust Before-After-Control Impact (BACI) of any field experiment 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 within reservoir processes. Our review has highlighted that in-reservoir biogeochemical processes should be considered alongside catchment land management approaches by the water industry to maximise the potential for upstream solutions to rising DOM concentrations in source waters.

Catchment 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 the water industry also develops 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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