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Prescribed fire and its impacts on ecosystem services in the UK

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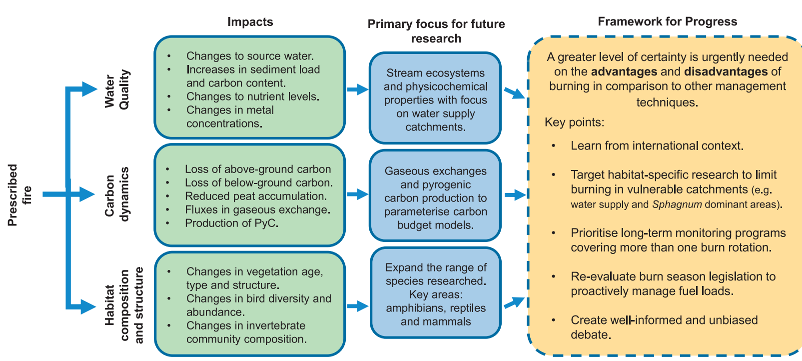
HIGHLIGHTS

- Prescribed fire is a controversial and highly debated topic in UK land management.
- Water supply catchments are at risk from water quality impacts of fire.
- Irresponsible burning in the UK uplands threatens to reduce vital carbon storage.
- Prescribed burning over inappropriate time-scales reduces faunal and floral diversity.
- More research is needed to reliably inform management practices in the UK.

GRAPHICAL ABSTRACT



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ABSTRACT

The impacts of vegetation fires on ecosystems are complex and varied affecting a range of important ecosystem services. Fire has the potential to affect the physicochemical and ecological status of water systems, alter several aspects of the carbon cycle (e.g. above- and below-ground carbon storage) and trigger changes in vegetation type and structure. Globally, fire is an essential part of land management in fire-prone regions in, e.g. Australia, the USA and some Mediterranean countries to mitigate the likelihood of catastrophic wildfires and sustain healthy ecosystems. In the less-fire prone UK, fire has a long history of usage in management for enhancing the productivity of heather, red grouse and sheep. This distinctly different socioeconomic tradition of burning underlies some of the controversy in recent decades in the UK around the use of fire. Negative public opinion and opposition from popular media have highlighted concerns around the detrimental impacts burning can have on the health and diversity of upland habitats. It is evident there are many gaps in the current knowledge around the environmental impacts of prescribed burning in less fire-prone regions (e.g. UK). Land owners and managers require a greater level of certainty on the advantages and disadvantages of prescribed burning in comparison to other techniques to better inform management practices. This paper addresses this gap by providing a critical review of published work and future research directions related to the impacts of prescribed fire on three key aspects of ecosystem services: (i) water quality, (ii) carbon dynamics and (iii) habitat composition and structure (biodiversity). Its overall aims are to provide guidance based on the current state-of-the-art for researchers, land owners, managers and policy makers on the potential effects of the use of burning and to inform the wider debate about the place of fire in modern conservation and land management in humid temperate ecosystems.

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1. Introduction

Fire is an important ecological process for many ecosystems and has played a complex role in shaping landscapes across the globe (Bixby et al., 2015). Throughout the last millennium, humans have used fire as a means of clearing land, facilitating hunting, and maintaining favourable grazing and leisure habitats (Goodfellow, 1998; Worrall et al., 2010a). During the last century, prescribed fire (i.e. controlled or management burning) has been used increasingly as a management tool across parts of the Mediterranean, and the seasonally dry regions of Australia and North America to control natural fire regimes and reduce the risk of severe wildfires spread by managing fuel loads (Burrows and McCaw, 2013; Fernandes et al., 2013; Ryan et al., 2013). The scientific literature on prescribed fire is dominated by research from these regions where fire is also part of the natural ecosystem cycles. Management burning, however, is also a common practice in non-fire prone ecosystems in the world's temperate zones (e.g. New Zealand, Tasmania, Northern Europe, South America and East Asia) (Holden et al., 2007), and the need to fully understand its impacts maybe even greater. The UK uplands have been burnt by humans for centuries (Worrall et al., 2010a). This paper aims, therefore, to provide (i) a comprehensive review of the existing knowledge on the impacts of this practice on key ecosystem services and (ii) to identify future research directions, with a focus on providing guidance to land managers and policy makers on the potential effects of the use of burning.

Early evidence of human management burning in the UK begins in the late Mesolithic/early Neolithic times (approx. 4000 years ago) as a hunting strategy and for clearing land (Fyfe et al., 2003; Tucker, 2003; Davies et al., 2008). By the late medieval period, burning was recorded as a common land management practice, notably in southern England and Scotland (1300s) (Rackham, 1986; Fyfe et al., 2003). It was not until the mid-19th century, however, that the use of burning for habitat management spread rapidly because of grouse moors (Worrall et al., 2010a). Over the last 150 years, this practice has taken the form of rotational prescribed burning (Davies and Legg, 2008). Rotational prescribed burning consists of using deliberately-ignited fires to create a mosaic of burnt patches of different ages. This produces a diverse vegetation structure, allowing the regeneration of younger, more palatable shoots (Worrall et al., 2010a). Burning occurs over a variety of patch sizes with individual patches being burnt on cycles of between 8 and 25 years. Some burning ideally takes place within a given area every year (Davies et al., 2008). This is deemed beneficial for the productivity of livestock-grazing pasture and increasing red grouse population for sports shooting where relevant (Worrall et al., 2010a).

Upland habitats form the primary focus of this review. In the UK, prescribed burning is conducted almost entirely in upland areas, focused on controlling the density, structure and age of *Calluna vulgaris* and *Molinia caerulea* dominated communities (Tucker, 2003). Upland areas are categorised as areas above the upper limits of agricultural enclosure, between 250 and 600 m altitude, depending on climatic conditions (Reed et al., 2009). Uplands cover approximately one-third of the land surface in the UK and support a diverse range of semi-natural habitats (Reed et al., 2009). These incorporate a range of ecosystem types from blanket bog, heathland and grassland assemblages, containing a variety of both vegetal and animal species (rare and priority conservation species; e.g. Hen harrier - *Circus cyancus*, Black grouse - *Lagopus lagopus scoticus* and *Sphagnum* sp.) and different operating land management practices (e.g. burning, grazing, cutting and predator control) (Natural England, 2001).

It is widely established that these upland regions provide a range of ‘ecosystem services’ (i.e. services the environment provides for the well-being of humans) benefiting multiple stakeholders (Provisioning services; food, fuel and freshwater. Regulating services; water regulation, climate regulation. Supporting services; nutrient cycling, primary production.) (MEA, 2005; Reed et al., 2009). As a result, a large portion of upland habitats fall within areas awarded with special conservation and research significance (e.g. National Parks; Sites of Special Scientific Interest (SSSI); Special Areas of Conservation (SAC)) (Tucker, 2003).

There are concerns around the application of prescribed burning in these important upland ecosystems in the UK. Burning has been implicated in several potentially negative impacts on the health and diversity of upland habitats (Ramchunder et al., 2009; Brown et al., 2015; Davies et al., 2016). In recent decades, the use of prescribed fire in the UK has become a source of heightened controversy with negative public opinion fueled by opposition from popular media (Davies et al., 2008; Brown et al., 2014; Allen et al., 2016; Monbiot, 2016). This highlighted several important limitations within the subject knowledge and resulted in land managers requesting further clarification on the impacts of prescribed burning. This, in addition to several other driving forces, has produced a substantial increase in research output with 77% of the literature captured for this review being published since 2000, 37% since 2010. It is, therefore, timely to review these areas of focus (water quality, carbon dynamics and habitat composition and structure) not least to also provide a synthesis for land managers in the UK and in regions with comparable ecosystems.

Three key aspects of ecosystem services form the focus of this review due to their vulnerability and the significance of potential impacts:

i) Water quality: A prominent concern for the management of upland catchments as they provide 70% of the UK's freshwater resource and are heavily regulated and monitored (Bonn et al., 2009).
ii) Carbon storage: Upland areas in the UK are vitally important for carbon storage with 3000 Mt carbon estimated to be stored in moorlands alone, equating to a globally significant carbon store over 6 times the gaseous carbon emitted by the UK in 2015 (SEERAD, 2007; Department for Business, Energy and Industrial Strategy (DBEIS), 2017).
iii) Habitat composition and structure (biodiversity)[1]: Globally rare fauna and flora are found in the UK uplands with a variety of UK BAP (Biodiversity Action Plan) Priority Habitats and 75% of the total area of the world's natural heather moorland (Tucker, 2003).[1]

These three aspects of ecosystem services have been consistently cited as important features needing to be closely monitored when implementing burn practices. All of which require further research to clarify possible impacts (Tucker, 2003; Ramchunder et al., 2009; Worrall et al., 2010a; Glaves et al., 2013; Brown et al., 2015; Davies et al., 2016).

To collect the relevant literature used in this review searches in scientific journals were conducted using several online databases, assessing articles at title and abstract level (Scopus, Web of Science and Google Scholar). Extensive searches for non-peer reviewed work through key UK agency sites were also undertaken resulting in an overall bibliography of 95 publications (Natural England, Natural Resources Wales, Scottish Environmental Protection Agency, Forestry Commission, Yorkshire Water, Severn Trent Water, and Welsh Water).

Of the 95 publications identified, 64 were peer-reviewed research papers, 10 peer-reviewed review papers and 21 agency reports. A systematic outline of the captured publications is given in the supplementary information (Table S1). The following section provides a brief overview of the current use of fire in the UK, which is followed by sections on the three key ecosystem services. The final two sections highlight the major research gaps and suggest future directions followed by the overall conclusions and a framework for progress. An executive summary is also provided in the Supplementary Information to make

[1] In this review the broad term of habitat composition and structure has been used instead of biodiversity as the relevant literature discussed includes the assessment of widely different levels of biodiversity.

the key findings and suggestions outlined by this review as accessible to land managers and policy makers as possible.

2. Current use of prescribed fire

In modern terms a prescribed fire is any supervised burn conducted to meet specific land management objectives (Santín and Doerr, 2016). The current practice of prescribed burning (also referred to as management or controlled burning, swaling or muirburning) has been created using a combination of traditional knowledge and practical experience combined with modern technology and scientific research (Fernandes and Loureiro, 2010). The global distribution of the use of prescribed burning is highly variable and the development of effective practices has been quicker in more fire-prone and fire adapted regions (Australia, USA and Canada) due to the greater need to control fuel loads (Goldammer and Bruce, 2004). In highly fire-prone regions prescribed burning is integral to sustaining healthy ecosystems (Burrows and McCaw, 2013; Gharun et al., 2017).

In areas where the use of prescribed fire is well established often proactive burn strategies are employed to protect communities and the environment. For example, in Victoria (southeastern Australia) the location and burn rotational length are predetermined depending on the main management objective (wildfire risk reduction, vegetation management and/or biodiversity) (Burrows and McCaw, 2013). Short rotations (<4 years) are used in areas within a close (~5 km) radius of human settlements to protect life and assets. Longer rotations are employed in lower risk areas further away from human settlements. The current management policy aims to burn approximately 2000 km^2 per year (8%) of the forested area to effectively manage fuel loads (Burrows and McCaw, 2013). In highly fire-prone regions such as, Australian Eucalyptus forests and North American tallgrass prairies burn rotations can be as short at 2–3 years, resembling return periods close to a natural wildfire regime (Burrows and McCaw, 2013; Valkó et al., 2014).

In contrast, the use of prescribed burning is prohibited in most or all circumstances in several non-fire prone European countries despite burning being a traditional part of their land management (Valkó et al., 2014). These countries include, Estonia (Liira et al., 2009), Germany (Kahmen et al., 2002), Sweden (Antonsen and Olsson, 2005), Switzerland (Köhler et al., 2005), the Czech Republic, Poland, the Netherlands, Romania, Ukraine, Austria and Greece (Valkó et al., 2014). This has likely resulted from the unpredictable and often negative impacts of uncontrolled burning in unregulated land clearance and vandalism (e.g. Romania, Ukraine, Austria and Greece) in addition to a lack of relevant research and operational expertise.

Prescribed burning is tightly regulated in the UK and it is estimated on average 114 km^2 (0.4–0.6% of total cover) of upland areas are being burnt each year (Yallop et al., 2006). Evidence suggests the area of land being managed by prescribed burning has increased since the 1940s, however, trends are localised with large regional variations (Yallop et al., 2005; Yallop et al., 2006). The typical length of burn rotations is also variable. It is recommended that burn rotations should not be shorter than a 15–20 year reoccurrence on UK moorland, however, local conditions and vegetation types inevitably alter the appropriate return period (Defra, 2007a).

The timing of prescribed burning is set by principal legislation and burning practice codes, which differ across the UK (Table 1). Despite previous research not recommending any alterations to the existing burn restriction dates (Glaves et al., 2005), opposition is growing within the operational community. In practice, it is argued that these dates do not enable fuel loads to be effectively managed due to the highly variable weather experienced in the UK (Allen et al., 2016; Hope, 2017). Prolonged periods of wet conditions often make it difficult to meet fuel reduction objectives during the current legal burn seasons. The statutory agencies presiding over Howden Moor (Peak District), for example, had to raise the target annual burn area from 7% to 10% of the moorland area per year to compensate for a build-up of burnable heather following several years of bad weather during the prescribed burn season (2003–2005) (Allen et al., 2016).

Nationally, there is limited data on the extent, distribution and frequency of burning in the uplands. A large proportion of the available mapping studies are locally based using different methodologies making the collation of these resources difficult.

3. Water quality

Water quality is one of the critical issues relating to the impacts of burning on ecosystem services in the UK. It is estimated 70% of the UK's freshwater is sourced from upland catchments (Bonn et al., 2009) and this is closely regulated by the European Water Framework Directive (Kallis and Butler, 2002). Outside the UK, many areas also heavily rely on the water provided from upland, particularly forested, catchments including one-third of the world's largest cities (e.g. Los-Angeles, Rio de Janeiro, Tokyo and Sydney) and two-thirds of municipalities in the USA (CHIFM, 2008). For this review, we identified 23 relevant peer-reviewed papers, five agency reports and six relevant peer-reviewed review papers addressing impacts of prescribed fires on water (Table S1). Here the assessment of water quality has been divided into two distinct categories (i) research focusing on dissolved organic carbon (DOC) and water colouration, two closely related and carbon-focused characteristics and (ii) research focusing on other aspects of water chemistry.

Throughout the water quality literature, there are three different types of water that are commonly sampled; runoff water, soil water and stream water. Runoff is the body of water that flows across the land surface in times of reduced infiltration rate or capacity, often sampled using crest-fall traps (Worrall and Adamson, 2008; Clay et al., 2010a). Soil water refers to the water contained, flowing or percolating through the upper layers of soil. Although there is no standardised definition or methodology, soil water is commonly sampled using dipwells up to 1 m below the surface (Clay et al., 2010a). Sampling is also conducted on water from directly within a stream system, often 2nd order streams, providing an indicator of the catchment scale impact of burning on water quality (Brown et al., 2013; Ramchunder et al., 2013).

Table 1
Legal prescribed burn seasons with relevant legislation.
Adapted from Worrall et al. (2010a).

	Uplands	Legislation	Code
England	1st October–15th April	The Heather and Grass Burning Code Regulations (England) 2007	The Heather and Grass Burning Code (Defra, 2007)
Wales	1st October–31st March	The Heather and Grass Burning Code Regulations (Wales) 2008	The Heather and Grass Burning Code for Wales (Welsh Assembly Government, 2008)
Scotland	1st October–30th April	Hill Farming Act 1946	Muirburn Code (SEERAD, 2001)
Northern Ireland	1st September–14th April	Game Preservation Act 1928.	The Heather and Grass Burning Code (Defra, 2007)

3.1. DOC and water colouration

DOC is a broad classification for the quantity of organic matter at varying stages of decomposition that is dissolved in aquatic systems, often considered to be organic compounds of <0.45 μm in size (Clay et al., 2009a). Water colour is a measure of the absorbance of water at a given wavelength and refers to the humic component of DOC within a water body (Holden et al., 2012). Elevated DOC and water colouration in water supply catchments can lead to breaches in drinking water standards and have health implications as the chlorination of such waters can lead to the production of carcinogenic by-products (Chow et al., 2003). This can have substantial costs implications for water companies and is likely responsible for these parameters becoming dominant foci in the recent literature. DOC is also a significant part of the carbon lost through freshwater systems, particularly in peatland catchments (Yallop et al., 2010). It is estimated aquatic carbon losses in peatland ecosystems account for 30–50% of the net exchange of carbon (Armstrong et al., 2012).

Water colouration and DOC concentration research has been conducted using a range of methodologies including laboratory studies (McDonald et al., 1991), plot-scale studies (Clay et al., 2012; Worrall et al., 2013a), catchment-scale studies (Yallop et al., 2008; O'Brien, 2009; Chapman et al., 2010) and modelling studies (Grayson et al., 2008, 2012). The evidence presented from this combination of different approaches allows the overall conclusion that burning on moorlands is correlated with an increase in DOC and water colour (Yallop and Clutterbuck, 2009; Yallop et al., 2010; Grayson et al., 2012). Yallop et al. (2010) estimated prescribed burning on upland peat habitats resulted in between a 5 to 15-fold increase in carbon exports as DOC as compared to equivalent unmanaged areas (Yallop et al., 2010). Ramchunder et al. (2013) in a comparison of streams in unburnt (3) and burnt (3) catchments over one year recorded a mean increase in DOC from 14.67 mg l^{-1} (unburnt) to 29.93 mg l^{-1} (burnt). It has also been observed that catchments producing the highest colour concentration had been subjected to >40% of the land surface area being burnt, with less colour production experienced in catchments with lower burn area cover (Grayson et al., 2012).

There are, however, a few studies that report rather different outcomes (Table 2) (O'Brien et al., 2005; Chapman et al., 2010; Clay et al., 2012). Several small-scale plot studies conducted on both soil water and runoff water produced inconsistent evidence with, at most, short-term (<1 year) elevations in water colouration and DOC (Ward et al., 2007; Clay et al., 2009a; Clay et al., 2012). A plot-scale study by Clay et al. (2012) found burning on moorland did not significantly correlate with changes in the concentrations of DOC or water colour in runoff. They did, however, witness a significant correlation (increase) in water colour concentration of soil-water after a burn, only returning to normal conditions after approx. 4–5 years. The existing evidence suggests plot-scale and catchment-scale studies are likely to exhibit different responses to burning, with the distinction between the compositions of samples collected using different methods (runoff, soil water and stream water). Runoff and soil water measurements at the plot-scale both only represent individual expressions of catchment water discolouration at a given location. Catchment-scale studies addressing the extent to which changes propagate downstream are highly beneficial to assess the broader impacts of burning on water colour and DOC. A summary of the published outcomes are presented in Table 2.

Vegetation type present in a burn area is a good example of one of the factors identified as crucial in the production of DOC and the significance of which may have been underestimated or even overlooked by earlier studies. Research into the effects of different vegetation types on DOC concentration in soil and surface water after a burn have shown statistically significant correlations (Beharry-Borg et al., 2009; Armstrong et al., 2012). Semi-natural ecosystems dominated by *Calluna vulgaris* (Heather) are associated with the highest levels of DOC in comparison to sedge-dominated and mixed vegetation assemblages (Armstrong et al., 2012). This is suggested to be because of their ability to suppress the water table through evapotranspiration. Increased water demand by shrubby vegetation (e.g. *Calluna vulgaris*) for transpiration leads to greater depth of root systems and declines in the water table (Worrall et al., 2007). This suppression of the water table in addition to *Calluna*-dominance promoting the creation of peat pipes (natural tunnels/macropores created by root penetration which transport water through the soil/peat) alters the hydrological transport in *Calluna*-dominated peatlands by reducing the interaction between ground water, soil water and the water table (Holden, 2005; Miller, 2008; Clutterbuck and Yallop, 2010; Smart et al., 2013).

Table 2
Representation of the varying results presented within the UK literature on the impacts of prescribed burning on DOC and water colouration. Study details in Table SI.

Parameter	Increase	No impact	Decrease
DOC			
Runoff	Yallop et al., 2010	Clay et al., 2009a Clay et al., 2010a Clay et al., 2012 Worrall et al., 2013a	
Soil water		Clay et al., 2009a Clay et al., 2012 Ward et al., 2007	Worrall et al., 2007 Worrall et al., 2013a
Stream water	Clutterbuck and Yallop, 2010 Yallop and Clutterbuck, 2009 Ramchunder et al., 2013	Brown et al., 2013	
Water colour			
Runoff	Yallop et al., 2010	Clay et al., 2012	Clay et al., 2009a
Soil water	Clay et al., 2012 McDonald et al., 1991	Ward et al., 2007	Clay et al., 2009a Worrall et al., 2007
Stream water	Beharry-Borg et al., 2009 Clutterbuck and Yallop, 2010 Yallop and Clutterbuck, 2009 Grayson et al., 2008 Grayson et al., 2012	Chapman et al., 2010 O'Brien et al., 2005	

In addition to investigating vegetation type, cover and structure, an important aim for future research would be to quantify the impacts of a range of other influencing parameters, which are seldom considered, on levels of DOC production in different ecosystems (e.g. Burn severity, extent and properties of surface soils/peats).

It is estimated the mean DOC concentration of freshwater drainage from upland peat catchments has increased by 91% in the UK between 1988 and 2003 (Evans et al., 2005). Typically, 60–70% of the variance in DOC concentrations in burnt peatlands over the past two decades are driven by changes in burn activity (Glaves et al., 2013). The combustion of vegetation and subsequent changes in erosional and fluvial processes, particularly on blanket bog and peatlands, are likely significant in altering DOC and water colour production but the exact causality is still not fully understood (Chapman et al., 2010). More research is needed to investigate the mechanisms controlling the dynamics of DOC and water colouration. It is important to differentiate the changes in DOC and water colour because of burning from the background of increasing DOC concentrations in freshwater drainage evident across the whole of Northern Europe (resulting from several different climatic and atmospheric drivers, notably changes in temperature and sulphur emissions) (Freeman et al., 2001).

Despite the importance of this area of study, there are still ongoing debates on the impacts of prescribed fires on the range of key physical and chemical parameters for water quality. The current literature contains many contradictory research findings, likely an indicator not only of the complexity of this area of research but also of the limited number of studies (e.g. Table 2). This presents opportunities for further study in the UK and is also cited as an area requiring global attention, with additional studies required in a range of geographic areas and biomes (e.g. tropical South America, Africa, Asia, Australia, boreal regions, temperate rainforests, grasslands and semi-arid savannas) (Bixby et al., 2015).

3.2. Water chemistry

Water chemistry is another important aspect when considering the impacts of burning on freshwater systems. The removal of vegetation and litter cover by burning, the creation of ash and the increased vulnerability of soils to erosional processes often results in elevated deposits of nutrients and metals into stream and ground-water systems (Tucker, 2003; Abraham et al., 2017). Elevations in the concentration of nutrients and metals can have significant effects on water quality with the potential to breach legal regulations and pose a health risk in water supply catchments. Over the last two decades, fires have affected the water supply in major world cities such as, Denver (USA), Sydney, Adelaide and Melbourne (Australia), increasing metal and sediment concentrations to substantially above World Health Organisation drinking water guidelines (Abraham et al., 2017).

Water chemistry studies in the UK are conducted using a range of methods. Most studies follow either plot-scale or catchment-scale sampling methodologies (Worrall et al., 2010a). At the plot-scale, experimental blocks are used with a combination of dipwells for soil water sampling and crest-fall traps for runoff sampling (Worrall and Adamson, 2008; Clay et al., 2010a). Catchment-scale studies have often employed direct river sampling techniques using continuous sensors with dataloggers and physical water sample extraction for laboratory analysis (Brown et al., 2013; Ramchunder et al., 2013). Studies have also monitored indicator species such as macroinvertebrates as an indirect assessment of stream chemical properties (Brown et al., 2013; Ramchunder et al., 2013).

The current knowledge suggests after a burn soil water experiences an increase in aluminium (Al), iron (Fe) and sodium (Na) and a decrease in calcium (Ca) and chlorine (Cl) (Worrall and Adamson, 2008; Clay et al., 2010a) (Table 3). Similarly, runoff water has been observed to experience an increase in Al and Fe following a burn (Brown et al., 2013; Ramchunder et al., 2013). A comparison of the chemical composition of water from 5 streams in unburnt and 5 streams in burnt peatland catchments in northern England by Ramchunder et al. (2013) detected an average increase in Al from 0.10 mg l^{-1} (unburnt) to 0.30 mg l^{-1} (burnt) and increase in Fe from 0.39 mg l^{-1} (unburnt) to 26.13 mg l^{-1} (burnt) over 1 year. The overall findings on water chemistry are, however, based on a limited number of studies, some of which do not entirely agree (Table 3). The evidence for the response of other metals (magnesium, potassium, manganese) and nutrients (chloride, phosphates and nitrates) are insufficient to form any reasonable conclusions (Table 3).

The UK does not currently experience catastrophic wildfires (i.e. extreme intensity, large extent) with severe fires only causing an intermittent problem. Despite this, climate change is expected to increase the vulnerability of UK ecosystems to wildfires, and examples of high-impact fires have occurred over the last decade (e.g. Swinley Forest in 2011) (Gazzard et al., 2016). In regions where they do occur increases in nutrients and metals into stream systems can be substantial, particularly in forested areas (Tecle and Neary, 2015; Burton et al., 2016; Nunes et al., 2017). Global incidences of these types of fires demonstrate the potential severity of the impacts of fire on water quality providing useful context for the future of fire impacts in the UK. The Rodeo-Chediski fire in Arizona (USA) 2002, for example, burnt 189,648 ha of forested land in an upland area, resulting in a 4000% rise in stream peak flow (Gill, 2004; Tecle and Neary, 2015). The fire also produced a 2-month rise in stream phosphorus levels (reaching 39 ml l^{-1} - Environmental Protection Agency (EPA) drinking water guideline - 0.1 mg l^{-1}) and a 5-month rise in iron levels (3000% above the EPA drinking water guideline) (Tecle and Neary, 2015). These levels are dangerously high and present a major problem for drinking water supplies in the surrounding area.

There is a lack of data on the impacts of burning on stream chemical properties within the UK and further research is required (Ramchunder et al., 2009; Worrall et al., 2010a). The response of both nutrients and metals in runoff, soil water and stream water following a burn are not well understood and results appear to be highly species specific (Clay et al., 2010a). Worrall and Adamson (2008) suggest the hydrological impacts experienced by burnt catchments dictate the changes in these parameters.

A notable hydrological consequence of burning highlighted in the literature is changes to the source waters affecting the composition of runoff and soil water (Worrall et al., 2010a). Decreases in the depth of the water table by up to 26% on 10-year burn cycles reduces the importance of base-rich ground waters (e.g. Ca, Mg) in soil water, due to changes in evapotranspiration (influenced by vegetation cover and type – see Section 3.1) (Worrall et al., 2007; Clay et al., 2009b). Changes at the soil surface from the generation of hydrophobic compounds, intensification of soil crusting and/or the quality and quantity of organic

Table 3
Representation of the varying results presented within the UK literature on the impacts of prescribed burning on water chemistry. (SSC = suspended sediment concentration, Total P = total phosphorus). For study details see Table 1 SI.

	Increase	No impact	Decrease
Stream water/runoff			
pH	Ramchunder et al., 2013		Brown et al., 2013
			Clay et al., 2010a
SSC	Ramchunder et al., 2013		
Nitrates	Ramchunder et al., 2013	Brown et al., 2013	Clay et al., 2010a
Calcium	Clay et al., 2010a		Brown et al., 2013
Chloride	Ramchunder et al., 2013	Brown et al., 2013	
Magnesium		Brown et al., 2013	Clay et al., 2010a
Sodium		Brown et al., 2013	Clay et al., 2010a
Sulfate		Brown et al., 2013	Ramchunder et al., 2013
Aluminium	Brown et al., 2013		Clay et al., 2010a, 2010b
	Ramchunder et al., 2013		
Manganese	Brown et al., 2013		
Iron	Brown et al., 2013		
Silicon	Brown et al., 2013		
Soil water			
pH	Clay et al., 2010a		
Nitrates		Worrall and Adamson, 2008	
Total P			Worrall and Adamson, 2008
Potassium	Clay et al., 2010a		
Calcium			Clay et al., 2010a
			Worrall and Adamson, 2008
Chloride			Clay et al., 2010a
Magnesium			Worrall and Adamson, 2008
Sodium	Clay et al., 2010a		Worrall and Adamson, 2008
Sulfate		Worrall and Adamson, 2008	
Aluminium	Clay et al., 2010a		
	Worrall and Adamson, 2008		
Iron	Clay et al., 2010a	Worrall and Adamson, 2008	

matter available following a burn also leads to a divergence in runoff and soil water composition (Clay et al., 2010a). Increased separation of these two water-transport pathways allows runoff water at the surface to become more dilute with rainwater and its components, whilst soil water under the surface becomes less dilute (Worrall et al., 2010a). Further research is needed to fully understand these underlying mechanisms.

Several key limitations have been identified in the previous two subsections from the lack of mechanistic understanding to the limited number of studies currently available on the impacts of prescribed fire on water quality. The factors influencing the creation and transport of water pollutants because of burn management, for example, appear to be complex and varied. The influence of in-situ vegetation cover (Armstrong et al., 2012), weather, burn regime and hydrological dynamics (Worrall and Adamson, 2008) are also important factors but often not fully considered. The wide range of sampling techniques, time and geographic scales used in the research published thus far is not helpful for directly comparing results. In many cases this limits the ability to draw firm conclusions, highlighting the need for greater standardisation of methodologies across water quality research.

In addition, the geographic distribution of the research collected on the impacts on water quality is significantly restricted, with over 80% originating from northern England. Although this region contains a large proportion of the UKs upland moor and peatland areas, it is vital the spatially sensitive nature of these parameters be reflected in the distribution of research. It is widely suggested that all forms of research need to be expanded across the full range of UK ecosystems with interest on water supply catchments (Holden et al., 2012; Glaves et al., 2013; Brown et al., 2015; Davies et al., 2016).

4. Carbon dynamics

Prescribed burning affects several aspects of the carbon cycle and the literature provides strong evidence for this (Imeson, 1971; Kinako and Gimingham, 1980; Ward et al., 2007; Allen et al., 2013). Most of the carbon-focused literature has investigated the impacts of burning in peatlands on DOC concentrations, as discussed in Section 3. Other key themes identified include carbon sequestration (Garnett et al., 2000; Ward et al., 2007) and carbon storage (Farage et al., 2009; Allen et al., 2013) as well as some more recent attempts to model full carbon budgets (Worrall et al., 2010b). We identified 37 relevant publications on the impacts of burning on carbon dynamics (27 peer-reviewed papers, 4 peer-reviewed review papers, and 6 agency reports).

4.1. Carbon storage

Burning on peatlands reduces above-ground carbon stocks through the combustion of vegetation and has the potential to reduce the carbon storage in surface peats (Ward et al., 2007; Glaves et al., 2013). Several studies have attempted to quantify the above-ground carbon loss in peatland ecosystems in the north of England. These studies estimate carbon losses by sampling biomass in selected plots/quadrats and calculating the difference between areas burnt and equivalent unburnt areas in similar *Calluna vulgaris* dominated habitats (Ward et al., 2007; Farage et al., 2009; Allen et al., 2013). Ward et al. (2007) estimated a 56% (88 g C m^{-2}) reduction in above-ground carbon on a 10-year burn cycle for their peatland study area at Moor House (Pennines, North England. Farage et al., 2009 investigated two burn events both on *Calluna*-dominated moorland areas experiencing ~15 years burn cycles in Mossdale Moor (Yorkshire, North England) and found a 16 ± 4% (103 ± 22 g C m^2) and 24 ± 5% (201 ± 62 g C m^2) carbon loss. These estimates are relatively similar considering the uncertainties involved in estimating losses. Due to the differences in burn cycle lengths and characteristics of individual burns, variations in loss estimates are expected. It is, however, important to recognise that even if the amount of above-ground carbon reduction can be substantial, this reduction may not be significant in relation to total ecosystem carbon storage, as the majority of carbon in peatland ecosystems is stored below-ground (Ward et al., 2007).

Evidence also suggests prescribed burning reduces the rate of peat accumulation and below-ground carbon storage in comparison to non-burning management (Garnett et al., 2000; Ward et al., 2007). Ward et al. (2007) estimated a loss of 167 g C m^{-2} from the peat surface under regular burning regimes (10-year rotations) in a peatland ecosystem (Moor House, North England). Garnett et al. (2000) also found there was significantly less carbon in the peat formed under a regime containing burning (10-year rotations) in comparison to one that did not include burning (Burnt = 3.1 ± 0.4 kg C m^{-2} and not burnt = 5.4 ± 0.6 kg C m^{-2}). The weight of evidence for this conclusion is, however, limited in the UK. Notably, Garnett et al. (2000) acknowledged their study was unable to determine the main influencing processes as the data used was unable to establish the burn history of the peat formed before the study began. This result is also at odds with the finding of Clay et al. (2010a) who found burning to significantly decrease the carbon loss from a catchment comparing burnt (117.8 g C m^{-2} yr^{-1}) and unburnt (156.7 g C m^{-2} yr^{-1}) catchments. They suggest that burning reduces carbon loss by increasing primary productivity and reducing net respiration of ecosystems. Research from fire-prone ecosystems in the USA have also shown wildfires that lead to organic matter loss in mineral soils (via soil heating) can reduce soil aggregate stability enhancing the vulnerability of soils to post-fire runoff and erosion (Neary et al., 2005); however, direct effects of prescribed fires on organic matter of mineral soils are usually very minor (Santín and Doerr, 2016).

Some studies have attempted to combine these aspects of carbon storage and estimate the total loss of carbon from the peat surface and above-ground vegetation resulting from prescribed burning (Garnett et al., 2001; Ward et al., 2007). Over a 10-year burn rotation in peatland, these studies produced estimates of carbon loss of 25.5 g C m^{-2} yr^{-1} by Ward et al. (2007) and 73 g C m^{-2} yr^{-1} by Garnett et al. (2001), both from study areas at Moor House National Nature Reserve, North Pennines. Clearly further studies are required to assess the longer-term impacts of burning on carbon over several burn cycles and in other areas.

4.2. Gaseous exchange

Research investigating the fluxes of gaseous carbon are vital to parameterise climate change models as well as, to understand the effects of prescribed burning on carbon cycling (Grace, 2004); however, the exchange of gaseous carbon (fluxes of CO_2 and CH_4) at sites managed by prescribed burning has received limited attention within the UK with few studies adding to the body of knowledge (Allen, 1964; Grace, 2004; Ward et al., 2007). Fluxes in CO_2 have been shown to be significantly affected by the prolonged use of prescribed burning on peatland ecosystems. Early laboratory experiments simulating heather burning estimate that 61–68% of original vegetation carbon is released to the atmosphere during combustion (Allen, 1964). Over relatively short burn rotation (10-years) the gross CO_2 fluxes of both photosynthesis and respiration have been shown to increase, relative to unburnt treatments. This represents an acceleration of carbon processing rates (Ward et al., 2007). Fluxes of this kind are responsive to the vegetation changes attributed to burn management, but are also strongly correlated with seasonal changes in climate, particularly temperature (Ward et al., 2007).

4.3. Carbon budgets

In recent years, considerable attention has turned to estimating complete carbon budgets, mostly using modelling approaches (Clay et al., 2010b; Worrall et al., 2010b; Farage et al., 2009). Clay et al. (2010b) used both fluvial (DOC, particulate organic carbon - POC, excess dissolved CO_2) and gaseous parameters (Methane, net ecosystem respiration of CO_2 and uptake of CO_2 by primary productivity) to estimate a carbon budget for the widely-researched Trout Beck catchment in the North Pennines. This study found the catchment to be a net source of carbon under all management techniques. 10-year burn rotation plots equated to an average source of 109.6 g C m^{-2} yr^{-1} and 20-year plots 125.9 g C m^{-2} yr^{-1} (Clay et al., 2010b). However, burning did significantly decrease the magnitude of the carbon source in comparison to the unburnt plots (average of 156.7 g C m^{-2} yr^{-1}). The influence of burn rotational length is evidently a key determinant on the carbon budget.

Worrall et al. (2010b) derived from a meta-analysis of carbon research that prescribed burning is likely not to benefit the carbon budget. Based on the current literature, however, estimating total ecosystem carbon budgets is difficult as the limited number of UK studies leads to large uncertainties and subsequent models will be substantially affected by the differences in the estimates of parameters between studies and sites (Glaves et al., 2013). It is clear that burning affects the processes controlling carbon budgets in peatlands. Moorland management including the use of prescribed burning, however, may not have a substantially detrimental effect on the carbon balance of upland areas if burning is conducted using appropriate fire regimes tailored for the chosen catchment (Farage et al., 2009). It is also important to consider the relatively small loss of carbon from prescribed fires as a necessary and beneficial reduction in fuel load, reducing the probability of a wildfire which would have a more detrimental effect on the carbon budget (Davies et al., 2006; McMorrow et al., 2009). A schematic diagram of the factors identified within the literature as being important in the carbon budget of burnt catchments is provided in Fig. 1.

There are several additional limitations upon the current state of knowledge relating to the carbon dynamics of prescribed burn management. The continued theme of the limited geographic distribution of research across the UK is also an important issue for the assessment of carbon dynamics. Much of the carbon-focused research again originates from the Pennines (North England) which is not directly applicable to UK upland ecosystems as a whole. More research is therefore, required to extend the geographic distribution of research across a more representative sample of upland ecosystems. Few studies have directly related differences in burn characteristics (e.g. severity) to the effects on carbon dynamics within the UK context (Glaves et al., 2013). Potentially important aspects of the full carbon budget are also rarely considered in the UK literature, such as the production of pyrogenic carbon (PyC; charcoal) (Worrall et al., 2013b), which is more resistant to degradation than original biomass and lead to long-term carbon sequestration (Santín et al., 2016). Actually, a substantial portion, around 13.5%, of the organic carbon accumulated in northern peatlands during the Holocene may have originated from PyC (Leifeld et al., 2017). Research conducted in Australian *Eucalyptus* sp. forests has demonstrated a significant increase in PyC in the surface soil (0.4 Mg ha^{-1}) and litter (0.3 Mg ha^{-1}) following prescribed burning (Krishnaraj et al., 2016). To improve the accuracy of carbon budget estimates in UK ecosystems, additional factors such as PyC production must be fully considered in future research.

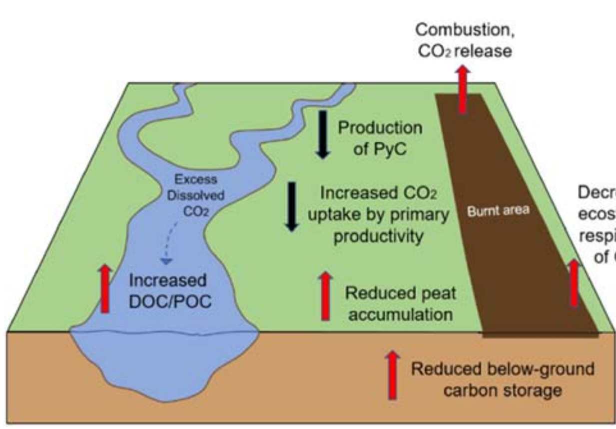

Fig. 1. Schematic diagram of the potential impacts of burning on catchment carbon budgets. Arrows represent the direction of flux resulting from burning (up (red) indicates increased loss of C from a catchment and down (black) indicates relative gain in catchment C); arrow size does not display magnitude of effect. (PyC = pyrogenic carbon, DOC = dissolved organic carbon, POC = particulate organic carbon). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

5. Habitat composition and structure (Biodiversity)

There are many ways of defining and approaching the topic of biodiversity across the relevant literature but in this review biodiversity is considered through the broader term of 'habitat composition and structure'. In this review habitat composition and structure (biodiversity) is defined both as a regulator underpinning ecosystem processes and directly as an ecosystem service itself (sensu. Mace et al., 2012). In much the same way as biodiversity, habitat composition and structure can also be considered an indication of the health of a system.

In this section, the impacts of prescribed burning on habitat composition and structure have been split into two distinct categories; the effects on flora (vegetation) and fauna (animals). We identified 33 peer-reviewed papers, 2 peer-reviewed review papers, and 14 agency reports providing insights into these parameters. The UK uplands contain many vital habitats (EC Annex 1 priority habitats and UK BAP habitats) for both flora and fauna of national and global conservation significance, such as Heather moorland, active Blanket bog, European dry heaths (Tucker, 2003). The impacts of burning on these habitats is an important discussion but crucially one that must assess the trade-offs of the use of fire in management.

5.1. Flora

The use of fire in land management has a range of complex impacts on the local flora from the initial combustion of vegetation during a fire to the redistribution of the balance of competitive advantage and changes in successional stages (Tucker, 2003; Ward et al., 2007). Much of the regeneration depends on the pre-fire conditions of vegetation, the length of burning rotation and the fire conditions on a given burn day (Tucker, 2003; Davies et al., 2010). As a result, the responses to fire in the UK uplands are highly variable.

The available literature suggests burning produces an initial period of dominance of graminoid species such as, *Molinia caerulea*, *Trichophorum cespitosum* or *Eriophorum vaginatum* and, at least an initial, decline in dwarf shrub cover and diversity on blanket bog and wet heathland (Marrs et al., 2004; Stewart et al., 2004; Ward et al., 2007). Ward et al. (2007) investigated the long-term impacts of repeat burning on peatland vegetation using the results of a 50-year old field experiment at Moor House (North Pennines). They found burning increased graminoid biomass by 88% and reduced the biomass of bryophytes by 91% and shrubs by 51% in the initial period of regrowth, typically 10–15 years. There is also strong evidence that *Calluna vulgaris* declines during this initial graminoid-dominant phase but typically then increases with time (Fig. 2) (McFerran et al., 1995; Stewart et al., 2004). Bracken fern is also one of the species that can increase in abundance in this early period following vegetation burning (Glaves et al., 2005). Caution must be taken, however, when forming generalisation as impacts will differ on individual circumstances (e.g. habitat type, weather, and burn dynamics). At some locations in the Peak District for example, under favourable conditions, *Calluna vulgaris* has been observed to return just one year after a fire event.

A prominent objective of prescribed burning in UK is to rejuvenate plant species such as *Calluna vulgaris* (heather). This improves the productivity of grazing pastures and creates a mosaic pattern of *C. vulgaris*

Fig. 2. Examples of the effect of different rotational lengths of prescribed burning on UK vegetation type and structure. A = recently burnt heather moorland, north western Scottish grouse moor (photo by Graham Lumsden). B = 0–10 year rotation – Graminoid-dominant ecosystem, northern Peak District. C = 15–20+ year rotation – *Calluna*-dominant ecosystem, North Pennines.

stands that increase the capacity of grouse on grouse moors (Tucker, 2003; Jones, 2005; Chen et al., 2008). As a result, *C. vulgaris* is perhaps the most commonly cited target species with regards to burn management. Some argue current burn practices reinforce the dominance of *C. vulgaris* creating habitats relatively low in species diversity (McVean and Ratcliffe, 1962; Lindsay, 2010). The presumption that current vegetation type and structure (i.e. at the time of burning) is what is ecologically best and should be maintained is also subject to continued debate (Worrall et al., 2010a). After burning and the initial decline in *C. vulgaris* (because of its relatively slow recovery rates), it often increases over a considerable period (15–20 years or longer), dominating particularly in drier sites where the rejuvenation of *Sphagnum* sp. is still inhibited (Ward et al., 2007; Harris et al., 2011b). This regeneration pattern highlights the importance of implementing site-appropriate burn rotational lengths to maintain the graminoid-*Calluna* balance and prevent loss of peat-forming *Sphagnum* sp. in areas managed for traditional purposes (Grazing pasture and grouse moors) (Harris et al., 2011a). In general terms, burning too frequently is thought to dramatically reduce *C. vulgaris* cover, damaging the ecosystem and likely leading to conversion to a grassland habitat (Tucker, 2003). If burn rotations are too long, *C. vulgaris* is likely to dominate at the expense of other species creating a monoculture (Tucker, 2003). In other ecosystems elsewhere, for example in North America, Australia and South Africa, overly long burn rotations resulting from fire suppression have been documented to allow the encroachment of woodland species into grassland and savanna habitats in addition to an increased risk of catastrophic wildfires from fuel buildup (Ratajczak et al., 2012; Valkó et al., 2014).

Sphagnum sp. is a group of Bryophyte species important to the creation and maintenance of bog and peatland habitats and they are shown to have variable responses to fire (Bain et al., 2011; Lee et al., 2013). The variable responses of *Sphagnum* sp. to burning likely reflects differences in individual burns and species of *Sphagnum* but it can be said with a reasonable degree of certainty that high *Sphagnum* cover and diversity is characteristic of less-modified and less-disturbed peatland ecosystems (Littlewood et al., 2010; Worrall et al., 2010a; Glaves et al., 2013). Burning has been stated to be particularly detrimental to peat-forming *Sphagnum* species (Grant et al., 2012), although results from a small number of experimental burns have contradicted this conclusion (Lee et al., 2013). Lee et al. (2013) suggested an open canopy and reduced cover of *Calluna vulgaris* aids the recolonization and growth of some *Sphagnum* species (e.g. *S. capillifolium*). There is also evidence from sites outside the UK suggesting that some species of *Sphagnum* (e.g. *S. balticum* and *S. magellanicum*) are able to recolonize effectively after burning in *Sphagnum*-dominated peatlands depending on site wetness (i.e. hydrogeological setting) (Sillasoo et al., 2011; Lukenbach et al., 2015). Lukenbach et al. (2015) in a study of post wildfire moss recovery in Alberts boreal peatlands (Canada) do, however, acknowledged that peatlands in late successional stages and those located in areas not well connected to groundwater flow are vulnerable to the detrimental effects of fire on *Sphagnum* sp. The processes dictating the changes in *Sphagnum* cover require further detailed study in the UK. Current burn policy airs on the side of caution and discourages burning on blanket bogs until there is greater clarity in the literature to recommend otherwise (Defra, 2007; Welsh Assembly Government, 2008; Scottish Government, 2011).

A notable proportion of the current literature associating the declines in particular species to the use of prescribed burning are not directly investigating the impacts of burning and rather making observational correlations in order to explain their results (Worrall et al., 2010a). These types of studies often acknowledge the potential influence of other factors (e.g. overgrazing, pollution and drainage) in these results and are only able to provide anecdotal rather than direct evidence of the impacts of fire. This evidence may still be valuable, but must be treated with caution.

To truly justify the use of fire for the purposes of vegetation management more studies need to be conducted directly addressing the benefits/drawbacks of burning in comparison to other techniques (e.g. cutting, layering or grazing) (Fig. 3) (Lunt et al., 2011). Few studies have focused on habitat composition or biodiversity as a whole and instead monitor the impacts of burning on one species or group of species.

5.2. Fauna

Fauna, in much the same way as flora, has a range of complex responses to the use of fire in UK upland habitats. The literature captured for this review contains several key areas of focus; birds, aquatic macroinvertebrates and terrestrial invertebrates.

In England, 35% of the moorland designated with site of special scientific interest (SSSI) status is based on ornithological features (Stroud et al., 2001). There are many breeding bird species in the UK that are globally significant (Hen Harrier - *Circus cyaneus*, Merlin - *Falco columbarius*, Golden Plover - *Pluvialis apricaria*, Lapwing - *Vanellus vanellus*, Curlew - *Numenius arpuata*, etc.) (Tucker, 2003). Breeding birds, therefore, constitute a prominent research focus into the effects of prescribed burning on fauna. Grant et al. (2012) noted burning affects birds primarily by the destruction of nests during burns and by changing the habitat (vegetation structure and condition as well as, availability of plant and invertebrate food sources). Most of these important species require short (mean height < 10 cm) open areas of vegetation for nesting, hunting and feeding. These behavioural preferences mean species such as Red grouse (*Lagopus lagopus scotica*), Golden plover, Curlew and Stonechat (*Saxicola torquata*) appear to benefit from prescribed burning as it provides an optimal habit of variable vegetation ages and heights (Thompson et al., 1997; Tharme et al., 2001; Daplyn and Ewald, 2006; Pearce-Higgins and Grant, 2006). Tharme et al. (2001) estimated densities of Curlew and Golden plover were two and five times higher, respectively, on grouse moors as on other moors.

Red grouse are a key species which rotational patch prescribed burning is specifically designed to benefit for the purpose of game shooting on grouse moors. As such, the rotational burning of vegetation covered a 34% larger area of moorland designated as grouse moors in comparison to other moorland areas (Tharme et al., 2001). Brown and

Fig. 3. Examples of the effects of different management techniques on catchment-scale vegetation structure. A = Mosaic pattern of prescribed burning created by Grouse moor management in northern Scotland (photo by Peter Cairns). B = Strip pattern of heather harvesting fuel management technique in the Brecon Beacons National Park.

Bainbridge (1990) estimate 5–15% (0.66 and 1.7 million ha) of the UK uplands are managed for grouse shooting. The creation of fresh palatable shoots of *Calluna vulgaris* for food and taller/older sections for nesting and shelter is highly beneficial to grouse (Glaves et al., 2013). As such, Red grouse numbers have been observed to be twice as high on grouse moors than other moorland areas (Tharme et al., 2001).

Other species of bird, however, do not appear to benefit from prescribed burning as they are commonly associated with different sets of vegetation characteristics (Tharme et al., 2001; Daplyn and Ewald, 2006). Whinchat (*Saxicola rubetra*) are associated with tall or dense vegetation types (Bracken - *Pteridium aquilinum*) and Skylarks (*Alauda arvensis*) with grassland vegetation with an open structure incorporating sedge and moss cover, both of which are not promoted by burn management (Pearce-Higgins and Grant, 2006). As a result, they are estimated to be 3.9 and 2.1 times less abundant on grouse moors, respectively (Tharme et al., 2001). Tucker (2003) also suggests burning is detrimental for Short-eared Owls (*Asio flammeus*), Hen Harriers and Merlin if patches of older heath are not retained for nesting purposes.

Many of the bird-focused studies used in the interpretation of the impacts of prescribed burning are not able to differentiate the impacts of burning on the densities of moorland birds from the impacts of other management practices often used simultaneously on grouse moors (e.g. predator control). There is strong evidence of a correlation between the intensity of burning and/or predator control on the densities of some species of moorland birds (Smith et al., 2001; Tharme et al., 2001; Pearce-Higgins and Grant, 2006). As it is unclear in these studies what proportion of changes to bird densities are caused by burning alone as opposed to the control/reduction in species that predate these smaller birds (Red Fox - *Vulpes vulpes*, Stoat - *Mustela ermine*, Cow - *Corvus corone*, Raptors) their results must be evaluated accordingly. Few studies directly focus on the impacts of burning on birds and rather focus on the impacts of grouse moor management or vegetation structure in general (Smith et al., 2001; Tharme et al., 2001).

The breeding behaviour of bird species has had a substantial effect on the timings of the legal burn seasons set across the UK (Section 2). Bird species that are ground or vegetation-nesting would be put under significant threat if prescribed burns occurred during nesting seasons. A review by Glaves et al. (2005) compiled a list of the date of first eggs laid by potentially vulnerable species and found that by the end of the current English burn season (15th April) 56% of Lapwing, 39% of Snipe (*Gallinago gallinago*), 26% Stonechat and 24% of Golden Plover had attempted their first nest. They go on to contextualise this as representing only a 1–2% chance of first nests being lost to burning and therefore, justifying the current regulatory dates. This potential vulnerability is an example of why prescribed burning needs to be tailored to specific locations, with burning avoiding nesting seasons and locations to reduce potential impacts.

Invertebrates, both aquatic and terrestrial, have been seen to be directly and indirectly influenced by prescribed burning (Usher, 1992; MacDonald and Haysom, 1997; Ramchunder et al., 2013). In addition to the direct combustion of terrestrial invertebrates during a fire, there is strong evidence to suggest that invertebrate density and community composition are significantly influenced by the changes in vegetation structure (McFerran et al., 1995; Eyre et al., 2003). Typically, species that prefer open ground environments such as, ground beetles and surface-active spiders tend to benefit from burning (Eyre et al., 2003). It is also proposed that species diversity and richness increase in habitats with a range of vegetation at different heights created by rotational burning practices (McFerran et al., 1995). Coulson (1988) suggested that under `good practice` burning terrestrial invertebrates are relatively effective at recolonizing areas as most are highly mobile. The timescale of recovery for various invertebrate species are seldom studied and have not been accurately quantified but are likely influenced by the burn rotational length, burn severity, vegetation dynamics and prevailing meteorology.

Studies investigating the impacts of burning on aquatic macroinvertebrates are useful for their insights into water quality. However, relatively little is known about the impacts on whole invertebrate assemblages in upland habitats (moorland/peatland) making this a key area for future research. There is currently evidence from a few studies that the use of prescribed burning in peatland catchments correlates with changes in aquatic macroinvertebrate assemblages (diversity and composition) (Ramchunder et al., 2013; Brown et al., 2013). Ramchunder et al. (2013) investigated the difference in macroinvertebrate assemblages over the course of one year between 2nd order streams in unburnt and burnt catchments in northern England and found no statistically significant difference in total abundance (unburnt = 2296 individuals per m^2, burnt = 2182 indiv. per m^2). A significant decrease in species richness (unburnt = 32 indiv. per m^2, burnt = 20 indiv. per m^2) was, however, found. Significant decreases were also reported in taxonomic richness by Brown et al. (2013). Changes in community composition often display a reduction in the density of pH and sediment-sensitive species such as, Ephemeroptera which presented a significant decrease in abundance (unburnt = 1061 indiv. per m^2, burnt = 271 indiv. per m^2) in Ramchunder et al., 2013). Along with an increase in more resilient species such as Chironomidae (unburnt = 568 indiv. per m^2, burnt = 1075 indiv. per m^2; Ramchunder et al., 2013) and Nemouridae (Brown et al., 2013).

In contrast to the assessment of terrestrial invertebrates by Coulson (1988), aquatic macroinvertebrates may experience longer recovery times in overall diversity if increased sedimentation disrupts feeding processes and fills interstitial spaces, potentially damaging filter feeding invertebrate taxa (Ramchunder et al., 2013).

We found no studies on the impacts of amphibians, reptiles or mammals within UK upland areas. Research from North America and Australia demonstrate prescribed burning has the potential to affect the overall abundance and diversity of species of amphibian (Schurbon and Fauth, 2003; Perry et al., 2012), reptiles (Gorissen et al., 2015; Harper et al., 2016) and mammals (Burrows and McCaw, 2013; Lashley et al., 2015; Harper et al., 2016) and this should constitute a key area for future study in the UK.

6. Research gaps and future directions

6.1. Spatial and temporal representativeness of prescribed fire research

The geographic distribution of research on prescribed fire in the UK is rather limited. Of the work examined here, 46% originates from northern England with 15% dedicated to, or including, data from one single catchment, Trout Beck at the Moor House Nature Reserve, in the North Pennines. Overall, England comprises 52%, Scotland 18%, Wales 3% and Ireland 1% of the captured literature. 26% of publications included multiple focus areas not confined to one specific area. This has likely resulted from the relatively small number of research papers in this field and because a large proportion of UK upland moors is in the north of England.

The research conducted at Moor House has substantially advanced our understanding of prescribed fire, however, it currently provides a bias towards a catchment that may not be representative of the broader context of the UK (Holden et al., 2012). Hence the need to expand all types of relevant research into a more representative distribution of locations and ecosystems remains (Brown et al., 2015; Davies et al., 2016). This should include efforts to quantify and monitor the distribution of the use of prescribed burning across the UK. In this context, the fact that 77% of relevant work in the UK has been published since 2000, with 37% since 2010 is encouraging. The relatively short time period over which an increased focus has been given to prescribed fire impact research, however, has meant that long-term assessments remain rare.

6.2. Ecosystem services

The body of knowledge on the impacts of burning on water quality has grown rapidly over the past few decades, with research expanding in numerous directions (Section 3). Despite this, further research is needed on all aspects of the impacts of prescribed burning on water quality, with a focus on water supply catchments (Brown et al., 2015). Water originating from within the Brecon Beacons National Park for example, supplies 90% of the drinking water to the wider urban area of Cardiff (Wales) (population of approx. 850,000) making water quality of substantial management importance (BBNPA, 2015). There is a lack of data on the impacts of burning on stream physicochemical properties (Ramchunder et al., 2009). This is perhaps a primary factor in the lack of consensus on the responses of soil and stream hydrology and nutrient cycling to burning. Future research should address this issue with a focus on understanding the underlying mechanisms and the provision of ecosystem services, particularly water quality from peatland systems (Worrall et al., 2010a). Expansion of studies investigating the extent to which water quality changes propagate downstream is also required to provide a wider environmental perspective of the impacts of burning (Brown et al., 2015). It is also important to further contextualise changes in water quality by extending studies investigating stream ecosystems. Relatively little is currently known about changes in the community composition of aquatic indicator species (e.g. macroinvertebrates, macrophytes) because of burning (Glaves et al., 2013; Ramchunder et al., 2013). Further investigation is also required on the effects of differences in the characteristics of burn patches such as size, shape, location, distribution, on water quality, chemistry and flow in peatland watercourses (Glaves et al., 2013).

Regarding impacts of burning on carbon dynamics, a prominent concern is the narrow geographic distribution of current research highlighted above, which limits the applicability of the findings given the diversity of terrain in the UK subjected to prescribed burning. In addition, few studies have directly related differences in burn characteristics, such as burn severity, to the effects on carbon dynamics within the UK (Glaves et al., 2013). This area of research would provide important context also globally to the effects of using prescribed fire in temperate climates. Furthermore, potentially important aspects of the full carbon budget are seldom considered in the UK literature. A lack of information exists on gaseous exchange and the production of char/PyC in the surface soil and litter following prescribed burning (Worrall et al., 2013b; Krishnaraj et al., 2016). Greater consistency in the methods used to monitor and estimate carbon balances would also enable more accurate comparative assessments.

Regarding habitat composition and structure, research is required across a more representative distribution of UK ecosystems directly focusing on the impacts of fire to separate them from other factors such as grazing, cutting, predator control, habitat type (Smith et al., 2001; Tharme et al., 2001). Including assessments of habitat biodiversity, as opposed to focusing on one or two individual species, are also important to further our understanding. To fully assess the value of fire for the purposes of vegetation management, more studies need to be conducted directly addressing the benefits of burning in comparison to other techniques such as cutting or layering (Lunt et al., 2011). There is also a notable lack of studies addressing the impacts on amphibians, reptiles or mammals within UK upland areas. Future research should progress in these new directions to provide a greater knowledge of ecosystem responses to fire. Research from outside of the UK can provide a useful context in these areas (Schurbon and Fauth, 2003; Perry et al., 2012; Gorissen et al., 2015; Lashley et al., 2015; Harper et al., 2016), but its applicability to the UK needs to be validated.

Cutting across the topic areas summarized above, a more detailed and consistent recording of site vegetation type, structure, composition and condition as well as surface topography and burn characteristic (type, intensity, severity) would substantially enhance the value and comparability of studies (Glaves et al., 2013). It is vital that future research strives to make the differences between the impacts of well managed and controlled uses of fire on ecosystem services in contrast to the impacts of more severe or poorly conducted prescribed burns.

7. Conclusions and framework for progress

Both prescribed and wildfires currently play a significant role in shaping ecosystems across the globe (Bixby et al., 2015). In many fire-prone regions, the use of prescribed burning is well established and integral to sustaining healthy ecosystems and protecting communities from catastrophic wildfires (Burrows and McCaw, 2013). In North America and Australia, for example, a large body of research underpins the effective use of prescribed fire to reduce accumulated biomass, support target species, manage open landscapes and control invasive species (Fuhlendorf and Engle, 2001; Cummings et al., 2007; Davies and Sheley, 2011; Ryan et al., 2013; Valkó et al., 2014). In the UK and large areas of Europe where fire does not serve such wide social and infrastructural needs, there is growing debate over the use of fire in land management. Opposition continues to question whether its role is an overall benefit. Current fire management practices in the UK uplands are closely regulated and it is estimated only 0.4–0.8% of upland areas are burnt each year (Yallop et al., 2006) mostly to maximise the productivity of *Calluna vulgaris*, red grouse and sheep (Worrall et al., 2010a; Allen et al., 2016).

The application of burning in the UK uplands has not dramatically changed throughout its modern usage (i.e. last 150 years) despite substantial changes in the environment and economy of the uplands (Davies et al., 2008). Taking inspiration from the effective use of prescribed fire in other regions (North America and Australia), the use of fire in the UK does not need to be limited to grouse moor management and agricultural land clearance if it is able to address any other desirable management objective. Habitat-specific research like that carried out in fire-prone regions needs to be conducted in areas relevant to Western Europe's humid temperate environment and different socioeconomic background to fire use to enable the appropriate use of prescribed fire in the UK. Land owners and managers need a greater level of certainty on the advantages and disadvantages of prescribed burning in different habitats, with historic perspectives of burning for biodiversity versus burning for productivity still prevalent.

The length and placement of prescribed burning seasons is an issue in the UK and one which could and should be subject to consideration in the devolved administrations as an easy step towards the better management of fuel loads. Dry springs coupled with an increase in illegal arson during the past decade (particularly in South Wales) have led to significant pressure being put on regional fire services (Jollands et al., 2011). This pressure has led to specialist task forces (specific wildfire response and prescribed burn management training for regional fire services) needing to be established, more equipment (assess to off-road response vehicles and helicopters) being deployed and closer cooperation between relevant agencies (Fire services, environment agencies, land managers and research institutes). More needs to be done to ease the financial and infrastructural strain on fire services resulting from accumulated fuel loads facilitating arson and wildfires (Section 2).

There is a potential blurring of the lines between controlled burns for land management purposes and for the purposes of mitigating the extent, likelihood and impact of illegal arson fires. If a consensus emerges where conservation burns in vulnerable ecosystems are more damaging than beneficial to biodiversity, water quality and/or carbon storage, there may remain a need to reduce the fuel load for wildfire prevention by other means until a beneficial cultural change leading to fewer fires by arson and accidental ignitions has been achieved.

Conservation land managers are caught in a paradigm. They maintain valued anthropogenic ecosystems that reflect past cultural and legislative requirements, whilst being uncertain of the long-term ecological resilience of ecosystems in a changing climate, culture and policy environment. Empirical, objective data is lacking in many areas and more needs to be understood about the long-term impacts of burning in addition to a range of other pressures facing upland systems (industrial pollution, acid rain deposition, historic overgrazing, human footfall and shifting weather patterns). When investigating the impacts of relatively long burn rotations (10–20 years) it is crucial that research includes data from the full post-burn recovery period and longer-term studies including more than one rotation are highly beneficial. Achieving this kind of long-term impact monitoring is, however, difficult and for the current knowledge to be significantly advanced more research areas need to be set aside (by National Parks or land owners) and partnerships with academic institutes set up specifically to target long-term research (e.g. Trout Beck, Moor House Nature Reserve).

When evaluating the current body of literature on the impacts of fire and the health and state of ecosystems in the UK it is important to acknowledge a key consideration to frame future progress. The biodiversity that remains at the present day has hung on in response to numerous pressures and it is often evaluated based on what is present now, after considerable human influence, rather than what was there before or what might come in the future. This feature of landscapes in the UK poses a range of open questions relating to what the 'natural' state of a given area is? What land-use type has/is an area intended for and what does a desirable or necessary future state of a chosen landscape look like (Worrall et al., 2010a)? There are perhaps no correct answers to these questions and differing opinions help to fuel the continued debate around the use of prescribed burning in land management (Davies et al., 2016). It is therefore, imperative that no one group, or opinion be allowed to drive the debate around what we want from our landscapes in the future. A collaborative effort incorporating the full range of stakeholders must be prioritied.

Prescribed burning, under a changing climate, could either be a useful land management tool or a highly damaging process if implemented without sufficient impact research. Based on the current knowledge it is still unclear which category prescribed burning falls into in the UK. This uncertainty around the impacts of prescribed burning has resulted in the national policy approach primarily focusing on suppressing the occurrence of fire in all forms (e.g. prescribed fire and wildfire) to avoid any potentially detrimental impacts (Gazzard et al., 2016). Although this may be appropriate in a densely populated country, there are considerable dangers around allowing fuel loads to build up. It is well documented in fire-prone regions that allowing fuel to accumulate under policies of suppression leads to increasing vulnerability to severe wildfires (Ryan et al., 2013). If the UK is not able to produce sufficient scientific evidence to inform management, climate predictions and international context suggests wildfires could become a major risk to water supplies, carbon storage and biodiversity. In addition to increasing the already high priority civil risk of wildfires and causing significant financial implications.

Creating and implementing progressive and adaptive management practices, including fire where sufficiently beneficial, supported by robust scientific evidence should be the primary focus of future research and policy (Allen et al., 2016; Davies et al., 2016).

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