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University of Durham

Doctoral Thesis

The Effects of Upland Peatland Vegetation Management on Carbon Exports and Water Quality

Suzane Michelle Qassim

Thesis submitted in accordance with the regulations for the degree of Doctor of Philosophy in the University of Durham, Department of Earth Sciences

2015

Abstract

Peatlands are important carbon reservoirs both nationally and globally, because they have the potential to be both sources and sinks of carbon. Dissolved organic carbon (DOC) is carbon lost from peatlands via the fluvial pathway. UK upland peatlands have a history of atmospheric deposition, degradation, and erosion as well as being extensively managed. Management of the upland peatlands presents an opportunity to maximise carbon storage and water quality benefits.

The research aim was to contribute toward the understanding of vegetation management effects upon peatland carbon exports and water quality. In the context of two studies: 1) bare peat ecological restoration (Bleaklow); 2) heather management through cutting and burning (Goyt Valley). Multi factorial designed in-field experiments were set up. Between 2007 and 2013, sites were monitored monthly for CO_2 fluxes, water table (WT) depth and water samples were collected and analysed for DOC concentrations. The results were statistically analysed using general linear models and were critically discussed.

In both studies, water sample DOC was better explained through inter-annual monthly variation than variation between sites. Bleaklow bare peat restoration and Goyt Valley management did not significantly influence soil pore water DOC concentrations. However findings supported the use of gully blocking and stabilisation techniques to revegetate bare peat, raised WT, promoted CO_2 influx through gross photosynthesis and reduced site acidification. Goyt Valley heather management through cutting was a good alternative to burning in dry localities (to raise WT). Runoff water and peat through-flow (at 10 cm depth) DOC was influenced by managed cutting and burning. Water sample DOC significantly varied along a peat profile (horizontally) and catchment. Through-flow DOC concentrations were greater than soil pore water at the wet locality and lower at the dry locality. The findings emphasised the importance of temporal and spatial scale when considering vegetation management effects on peatland carbon exports.

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I confirm that no part of the material presented in this thesis had previously been submitted by me or any other person for a degree in this or any other university. Where relevant, materials form the work of others has been acknowledged.

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1 Introduction

1.1 Peatland

With the exception of the largest geological carbon (C) store within the Earth's crust, mantle and core, there are three global C reservoirs: the ocean, atmosphere, and terrestrial system (Eswaran *et al.* 1993). The biggest terrestrial C store is peaty soils, accounting for approximately 20-30% of the world's soil C reserve (Updegraff *et al.* 2001). Soils more than 30-40cm deep with greater than 65% organic matter (OM) are classified as peat (Charman 2002, Johnson *et al.* 1963). Globally peatlands account for only 2-3% of land cover (Clymo 1984, Gorham 1991, Updegraff *et al.* 2001). Peatlands store disproportionally large amounts of soil carbon compared with other ecosystems. They therefore impact on atmospheric carbon and the greenhouse gas (GHG) pool in addition to the Earth's radiative balance (Frolking *et al.* 2006) and net irradiance (Ramaswamy *et al.* 2001).

1.1.1 Peatland formation

Peat-forming plants thrive where high water tables are present in water logged, anoxic, acidic and nutrient poor conditions where aerobic decomposition processes are inhibited. Peatland systems form peat by adding material into their waterlogged lower layers and OM accumulates as a consequence of photosynthesis rates exceeding respiration (Shepherd *et al.* 2013). Peat growth is initiated by water retention and low evaporation and subsequently OM is buried and hummified leading to the formation of peat soil (Holden *et al.* 2007, Turunen *et al.* 2002). Over time this leads to a large accumulation of C (Holden *et al.* 2007, Turunen *et al.* 2002), usually in basins or depressions with impeded water flow (Moore and Bellamy 1974).

Peatlands geographical occurrence is related to peat formation conditions, dependent on topography and climate. Their greatest abundance is on flat land where climate conditions are cool (Kuhry and Turunen 2006), and in continuously moist and hyperoceanic climatic conditions (Taylor 1983).

Peatlands can generally be split into gradients of two types, ombrotrophic (precipitation fed) or minerotrophic (ground water fed) systems (Keller *et al.* 2006), related to the variation in morphology, hydrology and biology (Charman 2002, Moore and Bellamy 1974). Generally, peat accumulates deeper on flats and shallower on sloping ground (Lindsay 1995). Nevertheless, they usually form over impermeable substrate (Taylor 1983) which allows peat to form on slopes and summits (Lindsay 1995, Taylor 1983).

Peatlands are sensitive to environmental changes (Holden *et al.* 2007). As a carbon reservoir they can gain and incur losses of carbon via atmospheric pathways (Danevčič *et al.* 2010, Nykänen *et al.* 2003, Rowson *et al.* 2010, Worrall *et al.* 2003b) or fluvial outputs (Clutterbuck and Yallop 2010, Volk *et al.* 2002, Worrall *et al.* 2003a, Yallop *et al.* 2010). Budgets by Dinsmore *et al.* (2010) demonstrate the importance of aquatic fluxes, which can account for 30–50% of net ecosystem exchange (Nilsson *et al.* 2008, Roulet *et al.* 2007). Peatland degradation (main causes in the England described (Table 1.1) could significantly increase losses of terrestrial carbon and even convert what would naturally be a carbon sink into a carbon source (Joosten 2009, Joosten *et al.* 2012, Rowson *et al.* 2010).

1.1.2 Blanket bogs

Ombrotrophic bogs, blanket bogs or mires, receive all nutrients and water input through atmospheric processes as opposed to being ground water fed. Typically, they form in upland areas in regions of oceanic climate in the maritime fringes of the continental masses (Lindsay *et al.* 1988). Peatlands comprise of only 2% of the Earth's land area (Brooks and Stoneman 1997) in the northern hemisphere (Figure 1.1) occupying a relatively narrow band (Mighall *et al.* 2006, Moore and Bellamy 1974). Increased elevation correlates with decreased temperatures and increased precipitation, decreased rates of decomposition, and favourable conditions for the formation of peat and blanket bogs (Franzén *et al.* 2012).

Blanket bogs form on slopes and summits (Lindsay 1995, Taylor 1983) where other wetlands could not form (Charman 2002). Peat formation requires continuous moist conditions, low nutrient availability (limited phosphorus and nitrogen), with a pH of ~3.5-4 (Charman 2002), with high rainfall (>1000 mm p.a) (Bell and Walker 2005). These conditions combined with impermeable substrate such as acid rock deposits, surficial glacial or periglacial strata and stony deposits (Tallis 1983, Tallis 1985, Taylor 1983). During the Holocene period, since the last glacial period, between AD 270−455 Gt C approximately 4.5 Gt C has accumulated in northern peatlands alone (Gorham 1991, Turunen *et al.* 2002), at an average rate of 0.96 Mt C/yr (Gorham 1991).

A blanket bog in a good condition has a balanced range of *Sphagnum spp.* mosses, cotton grasses, dwarf shrubs, sedges and other typical wetland plants (Charman 2002). It is also characterised by a high water table that fluctuates in a surface zone. New rain falling travels mainly across the bog surface as surface runoff. The surface runoff flow is slowed by rough surface vegetation, bog mosses and cotton grasses, causing friction; the water accumulates within the peat soil due to the peat potential for high water retention, and the slow movement of the rainwater (Holden *et al.* 2008). The high water table results in slow rates of decomposition due to low aeration of the soil (Shepherd et al., 2013).

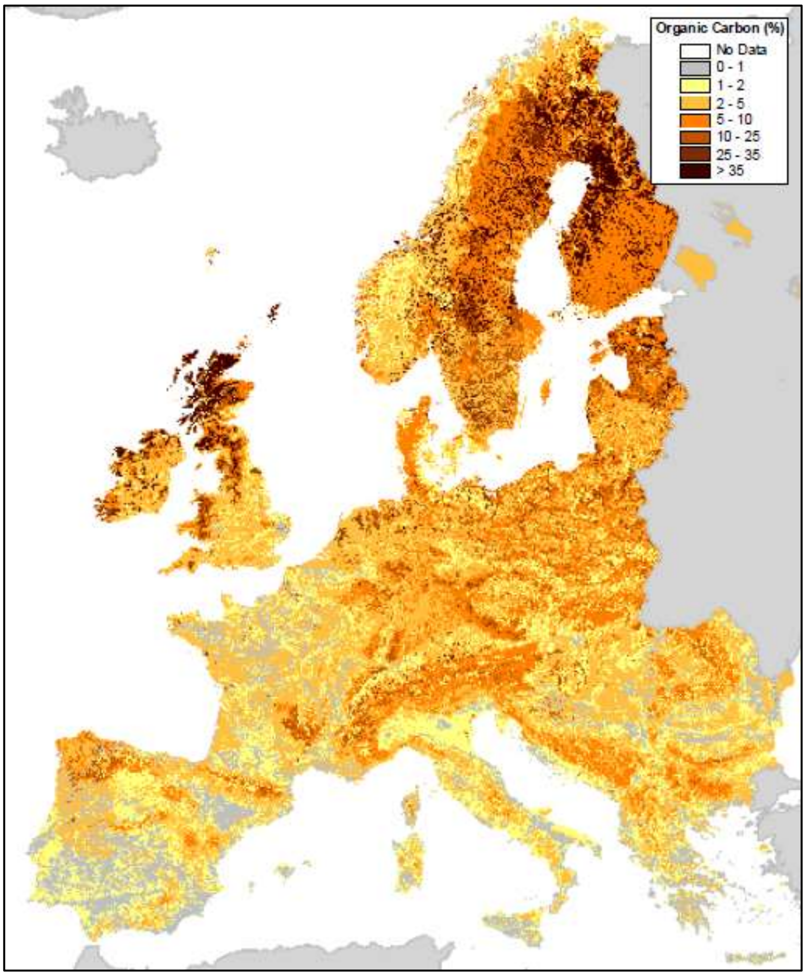


Figure 1.1: European topsoil organic carbon percentage. The ranges of organic carbon are greater at darker shades, which coincide with the presence of peat (Jones et al. 2005).

Factors affecting English blanket bogs	Percentage of English bogs affected
Vegetation management - Semi natural vegetation not suitable for carbon sequestration	51%
Overgrazing	9%
Rotational moorland burning	30%
Artificial drainage	21%
Drainage by gully incision	14%
Afforestation	7%
Bare peat - due to degradation	1%

Table 1.1: Natural England (2010) mapped factors affecting English blanket bogs. These values are not mutually exclusive, i.e. an individual peatland area may be affected by one or several of these factors and as such the percentages are not additive.

1.1.3 An ideal blanket bog

A peatlands status with regards to carbon can be divided into one of three groups: a) damaged, with poor vegetation cover, not depositing peat potentially eroding; b) transitionary sink, in which a peatland's vegetation is maturing or has nursery crops (vegetation will continue to change), peatland is not yet stable or not long term; c) long term/perpetual site, which is pristine peat in which peat is continually accumulating (Natural England 2010)

A blanket bog is typically dominated by small shrubs, including heather (*C.vulgaris)*, bilberry *(Vaccunium mytillus (L*)) and sedges such as cotton grass (*Eriophrorum spp*.), as well as the peat forming bog mosses (*Sphagnum spp. imbricatum*) (Coulson 1992, Holden *et al.* 2007). *Sphagnum spp.* promote peat deposition as they decay at a lower rate than vascular plant litter (Bragazza 2008). Bryophytes are poikilohydric plants and their water content is controlled by their environment, *Sphagnum spp.* therefore grow well in wet peatland habitats (Harris 2008). Some *Sphagnum spp.* species are better than others at surviving drought conditions (Bu *et al.* 2013). A blanket bog in a good condition is defined by the JNCC (2009) as including: a) no loss of extent of blanket bog habitat; b) at least 4 indicator species present within a 4 m^2 quadrant; c) low cover of non-native species such as trees and scrub (discounting dwarf species) and mesotrophic grasses/forbs/bracken; d) low grazing/browsing on dwarf shrubs (particularly juvenile plants); e) no burning on sensitive areas, into moss/lichen layers, or to expose the peat surface; f) less actively eroding peat than re-deposited peat (in the wider area); g) less than 10% disturbed bare ground or showing signs of drainage or track damage; h) and less than 10% of *Sphagnum spp.* should be damaged (crushed, disturbed).

1.2 Blanket bogs in the UK

United Kingdom (UK) peaty soils are largely located in upland regions, and represent a pool of 6–7 billion tonnes of carbon (C) (Emmett *et al.* 2010, Shepherd *et al.* 2013). The most extensive type of peatland soil in the British Isles, at an estimated 86.8% of peat area cover, are blanket bogs (Lindsay *et al.* 1988) (Figure 1.2) These peaty systems are estimated to be a net sink of 0.32 MtC/yr (Holden *et al.* 2007). Around 10% (an estimated 25,000 km^2) of global blanket bogs are located within the UK (Tallis 1997). According to the Natural England (2010) ~99% of UK bogs are classed in poor condition. Ameliorating UK bog condition is increasingly important. Only 1% of British peatland is classed as 'undamaged', equating to only 35 km^2 of the 3,553 km^2 of blanket bogs in England. Although blanket bogs are extensively developed in the UK, they also lie in a unique geographical location in that they are located at the southern climatic margin of blanket peatlands (Daniels *et al.* 2008, Tallis 1997). Blanket bogs are largely confined to upland regions and are located in Dartmoor, North Pennines, Cambrian mountains and The Peak District National Park (PDNP) in South Pennines (Mighall *et al.* 2006, PDNPA 2008).

Peatlands are sensitive to environmental changes (Holden *et al.* 2007). English blanket bogs emit 0.89 Mt CO_2-equivalents annually with rotational burning being the largest emitter at 0.26 Mt CO_2-e y^{-1}. These estimates are found in the Natural England (2010) report. In order to improve peatland conditions, it is important to identify the causes of their degradation (Table 1.1). The 2010 report indicated that although large areas of fen peatland remain, the majority has become wasted through drainage and cultivation.

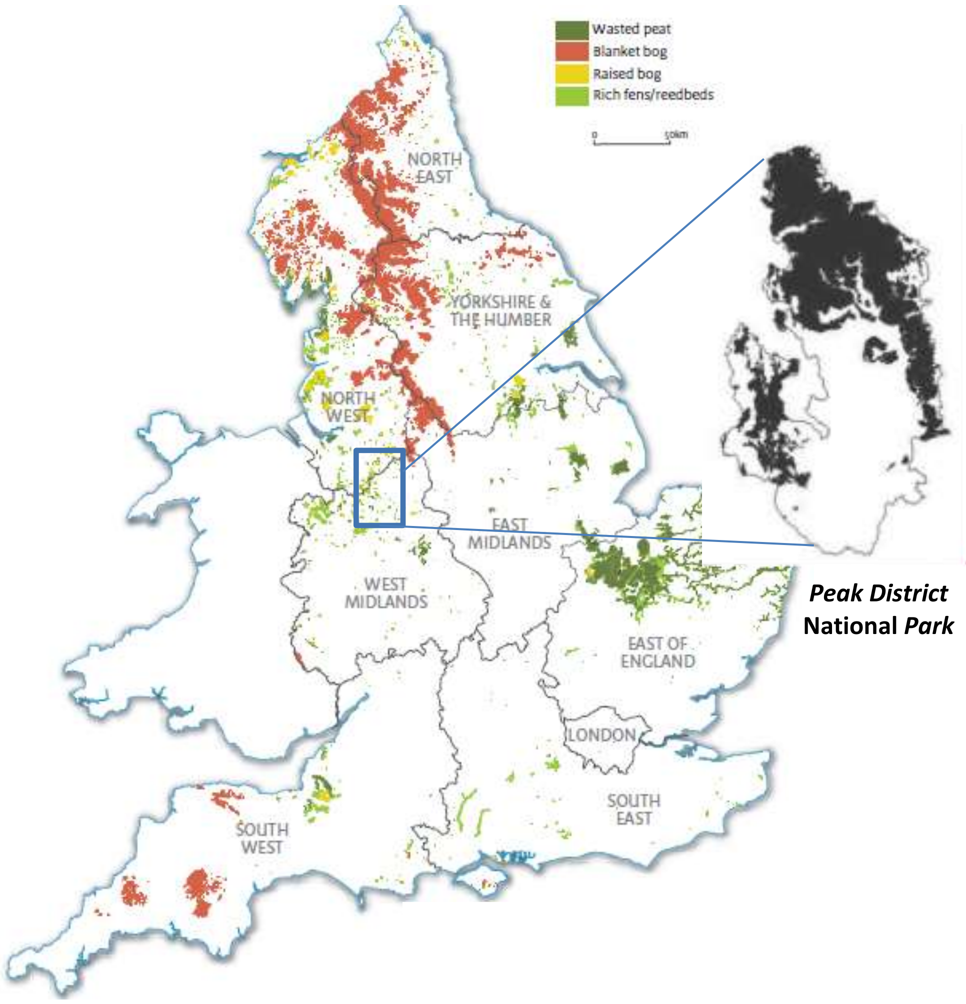


Figure 1.2: England peatland distribution, colour coded according to type of peatland: wasted, blanket peat, fens, raised bog (Natural England 2010), and the Peak District National Park (black shade indicated peat presence) (JNCC 2011).

1.3 The Peak District National Park Peatland

Up to three quarters of the PDNP is covered in peatland (Figure 1.2). The South Pennines are very unique as they lie at the southern climatic margin of blanket peatlands and receive lower rates of precipitation than any other British upland peats (Tallis 1994, 1997).

Up to three quarters of the PDNP peat soil (Figure 1.2) is degraded and/or eroded (Anderson and Tallis 1981). Studies in the PDNP by Phillips *et al.* (1981), Worrall *et al.* (2011), Warburton (2003) and Anderson *et al.* (1995b) observed peat surfaces recede by up to 62 mm annually. This recession could explain why in the South Pennines, raised blanket bog eroding catchments experience 80% of fluvial C loss in the form of particulate organic carbon (POC) (Pawson *et al.* 2008). Similarly, the largest single carbon loss from the North Pennines system according to Evans *et al.* (2006), is also demonstrated to be POC losses associated with the fluvial suspended sediment load. While monitoring other regions on a national scale however, a survey of gully erosion over 2-5 years found no detectable changes in erosion features (McHugh et al., 2000, Wishart and Warburton, 2001). This suggests that the erosion is site specific associated with regional pressures.

The Moors for the Future Partnership (2010) identified the main causes of peat degradation in the Peak District and South Pennines as: air pollution (e.g. sulphur dioxide) pollution, tourism, managed burning, overgrazing, weather, drainage, non-native species, over grazing, in addition to natural causes in particular climate change (Tallis 1997). Acid deposition associated with oxidised nitrates and sulphur compounds, derived from fossil fuel combustion, had contributed towards the acidification of the peatlands and surface waters (Clark *et al.* 2005, Curtis *et al.* 2000). Bleaklow's (a locality within the PDNP) proximity to industrial centres (e.g. Manchester) has resulted in a legacy of atmospheric deposition. Studies have found disappearance of *Sphagnum spp.*, an important peat forming species, during the 19th century.

The disappearance is suggested to be due to air pollution and/or climate, followed by severe fires during the period 1918–1930 which resulted in decreased shrub species such as heather (*C. vulgaris)* and the domination of graminoid species (Yeloff *et al.* 2006). In addition to the change in vegetation type, severe fires in the summer of 1959 led to a reduction in catchment vegetation cover and bare peat (Yeloff and Hunt 2005). Without vegetation to protect the exposed surface, overland flow over bare peat is faster than over vegetated peat (Holden et al., 2008). The reduction in water resistance and peat forming species increased peat vulnerability to desiccation, weathering and erosion, allowing erosion to exceed the rate of peat accumulation potential. Studies on pollen and spores by Yeloff and Hunt (2005) confirmed increased rates of erosion between 1976-1984 within the Peak District National Park.

1.4 Legislation to protect peatlands

The UK is part of the EU, it must therefore abide by international and EU environmental legislation. The preservation of peatlands is being supported by several pieces of legislation both directly and indirectly. Protection of peatland habitats is significant on a national scale in addition to international obligations. The interest in peatlands is due to their: support of species and habitats; carbon storage potential; relations to water quality and flood risk. Due to peat's physicochemical properties it has recorded the historic environment that it preserves, in its artefacts, stratigraphy and landforms in addition to the wild landscapes in which it forms. Management practices such as peat cutting, extraction or draining have consequently become more restricted within the UK. Under the 1971 Ramsar Convention, an intergovernmental treaty, guidelines were set out for Global Action on Peatlands (GAP), with specification around maintenance of global biodiversity, storage of water and carbon vital to the world climate system, and the promotion of wise use of peatland (Bell and McGillvary 2006). As a result of GAP, sites, such as those covered by peat, deemed of ecological significance are given site designations set up to provide them extra protection; land management/government bodies are then obliged to follow recommendation and guidance to a regional/national and/or international scale.

Within the UK, blanket bogs, raised bogs and fens form three of the national protected habitats under international Council Directive 92/43/EEC of 21 May 1992 Conservation of Natural Habitats and of Wild Fauna and Flora(CEC 1992). The UK as a member state of the EU, follows its obligation through the Directive on a national scale through the Natural Environment and Rural Communities (NERC) Act 2006 Section 41, (known as the Habitat Directive) (Bell and McGillvary 2006) and Biodiversity Action Plan (BAP). Specific species of flora and fauna are listed and protected under the UK Biodiversity Action Plan (BAP). Site designations relevant to peatland ecosystems include: Ramsar, Sites of Special Scientific

Interest, Natura 2000 – Special Area of Conservation (SAC) and Special Protection Area (SPA). Upland peatlands are a UK Priority Habitat as they support a range of plant and invertebrate communities and unique bird assemblage, detailed in a report by the Biodiversity Reporting and Information Group (2007). The Countryside and Rights of Way Act 2000 (CRoW), provides guidance on land management, and strengthening the legal protection for threatened species, with specific attention to protecting birds.

A partnership was set up in 2007 between Natural England, Defra, the Environment Agency, Forestry Commission, the Welsh Assembly Government, Countryside Council for Wales and the Northern Ireland Environment Agency to protect and enhance peat soils (known as 'The Peat Project'). Other government bodies and statutory agencies may also become involved in this project in the future. The Peat Project aims to protect peatlands and promote their importance to a range of policy areas, climate change, biodiversity, water quality and flood risk. One of the ways it does this is through 'Good Practice', research on restoration and management in order to develop advice, products and guidance. The work in this thesis is aimed at contributing towards future guidance on peatland restoration and management practices.

1.5 Peatland Management

Management can be both a threat and an opportunity to control the magnitude at which a peatland is a sink or source of C (Worrall and Clay 2012b). Some benefits and adverse consequences of peatland management can be measured in terms of: atmospheric carbon fluxes, water quality, in addition to biodiversity and ecology. The effect of peatland vegetation management on peatland atmospheric and fluvial carbon cycling is the main focus of this thesis.

Poor management has been identified as a significant driver of upland peatland degradation, linked to carbon release (Clutterbuck and Yallop 2010, Joosten *et al.* 2012, Schumann and Joosten 2008, Worrall and Clay 2012b). The main contemporary land management practices include: drain blocking; cattle and sheep grazing; prescribed, managed burning; vegetation cutting (Holden *et al.* 2007, Mitchell *et al.* 2008). According to Nartural England almost 84% of the original English area of peatland in the Fens has been lost, mostly due to cultivation and drainage. Only 4% the original area of raised bog remains and much of our blanket bog has been eroded into haggs, drained by grips, or is rotationally burned for grouse rearing.

Studies have found evidence of management impacting upland peatland C storage as far back as Mesolithic times (Reed *et al.* 2009, Sleutel *et al.* 2003). Radio carbon dating of pollen in South Wales blanket bogs by Smith and Cloutman (1988) found peat accumulation began ~8000 year BP when heather presence combined with abundant charcoal is evident, supporting evidence of heathland management through burning. Sleutel *et al.* (2003) postulates there is evidence of a decline in organic carbon (OC) contents in many soils as a consequence of agricultural expansion during the 20th century. Between 1960-1970s, moorlands were drained to improve grazing pastures; this drastically altered the peatland ecology and hydrology (Wallage *et al.* 2006). A common practice now used to restore peatland hydrological regimes (as a result of drainage in particular) has been physical alteration to block gullies and ditches. Holden (2005) identified land management as an important factor in altering peat structure. These alterations have impacts on above and below ground water flow paths (Holden and Burt 2002), water retention and change in soil structure following draining causing shrinkage, cracking or decomposition, in addition to influencing change in water quality and ecology (Holden et al., 2007).

Action is usually required to preserve and restore a peatland ecosystem. The ecological aim of interventionist management is primarily to promote further deposition of peat and increase the longevity of the existing peat environment. In order to restore a site, the optimum condition is often stablished based on prior research. A peatlands chemical, physical and biological condition may therefore be altered/ enhanced to reach an ideal environmental status or desired habitat. Peat forming species are greatly desired, particularly when restoring peat, as they lock carbon into the terrestrial reservoir, offering greater longevity for the habitats. The hydrology of the peatland must be restored (Price *et al.* 1998, Quin *et al.* 2014), in order to promote the re-colonization of peat forming vegetation such as *Sphagnum spp.* moss, which obtains their water through capillary action (Price *et al.* 1998).

1.5.1 Peatland ecosystem services

Peatlands are socio-economically important as they provide many ecosystem services. Some of the economically viable uses of a peatland are: influencing water quality, as peatlands covers catchments which feed into large reservoirs; flood defence, as upland peat stores water and the vegetation slows downhill water flow; agriculture, for example sheep grazing (Clay *et al.* 2009, Rawlins and Morris 2010, Worrall and Adamson 2008); for ecotourism, ramblers and walkers; game hunting (e.g. grouse or deer); as a fuel/agricultural fertiliser resource, for which peat extraction is conducted.

Peatlands provide valuable ecosystem services which are becoming increasingly realised. The physical and chemical properties of peatlands support rare flora species. These biotas are adapted to surviving within the harsh weather conditions on a peatland, and to the acidic, nutrient poor environment. These flora also support wildlife with varying home ranges, some threatened or vulnerable reptiles, small mammals and bird species (particularly birds of prey) (Tharme *et al.* 2001).

Peatlands systems are carbon reservoirs; they can range from being a source of carbon and GHG to a sink. This is determined by a peatlands status as either: an eroding peatland or an accreting/actively depositing peat forming ecosystem. Peatlands can therefore play an important role as a carbon stores to influence: atmospheric GHG concentrations, the greenhouse effect, global warming and the rate of climate change. Land management and land use change have the potential to significantly alter C cycling and provide important mitigation against increasing greenhouse gas emissions (Parry *et al.* 2014, Rowson *et al.* 2013, Worrall *et al.* 2011).

1.6 Vegetation management in the Peak District National Park

The PDNP is reported by Moors for the future (2007) as being the world's second most visited national park (22 million visitors per day). The PDNP also has large areas of degraded and/or eroded peatland due to a combination of mismanagement, significant inputs of flux of both regional and local atmospheric pollution, and environmental weathering (Andersen *et al.* 2010, Hutchinson and Armitage 2009, Tallis 1985). Large efforts have been invested into managing the PDNP through vegetation management. The overall aim is to maximise the benefits of the peatland ecosystem. The PDNP was the settings selected for the purpose of research, due to the wide variety of management (restorative, interventionist and maintenance) types used within the PDNPs. Dixon (2011) found that: a) altitude significantly influences CO_2 fluxes, b) there is a relationship between photosynthesis and respiration, and c) respiration is temporally lagged by an estimated 3 hours. Dixon (2011) did not consider the effects of vegetation management at an extended multiannual scale.

1.6.1 Bare peat revegetation – ecological restoration (Bleaklow Plateau)

Damage or degradation to peatlands due to mismanagement, can more readily be restored (Worrall and Clay 2012b, Worrall *et al.* 2011) than that caused by external drivers such as increased air temperature (Holden et al., 2007). Restored sites show improved C budgets after restoration, with the benefit of avoiding C losses (Rowson *et al.* 2010). Land management can therefore represent an opportunity to reduce atmospheric and fluvial C outputs and improve water quality in the runoff from peat-covered (Parry *et al.* 2014, Quin *et al.* 2014).

A multiannual study by Dixon *et al.* (2014) comparing bare sites to revegetation sites found that, depending on the revegetation methods used, the site can be two to eight times more likely to become a net sink for CO_2. Revegetation of bare peat can therefore reduce atmospheric C losses in the long term. Generally there is a lack of long term monitoring and measurements of peatland C inputs and outputs (van den Berg *et al.* 2012). Hence, little is known about the consequences of long-term disturbances or management on the individual components of the carbon cycle (Blodau 2002).

Eroded sites, such as those found in the PDNP, benefit most from restoration management (Shepherd *et al.* 2013). Rates of erosion are higher on bare peat (Daniels *et al.* 2008). Bare and eroded peat is less resilient to stochastic weather events such as drought which are linked to long term effects (of up to 25 year) on carbon-out fluxes from a peatland ecosystem (Worrall *et al.* 2006). The aim of vegetation restoration is dependent on the target habitat. However in general bare peatlands are being restored to form a functional peat accumulating ecosystem (Sottocornola *et al.* 2007). The methods used to restore Bleaklow are linked to land-uses which include water supply, agriculture, sport and leisure tourism, and game shooting (Bonn *et al.* 2009, Reed *et al.* 2009).

Changes in hydrology and ecology can lead to physical degradation. Such changes can be a result of external or internal pressures (Parry *et al.* 2014). Methods used to restore vegetation of bare peat must be designed to address the degradation pressures and their specific effects on the bog. Several interventions were used as part of the Bleaklow bare peat restoration. These methods were conducted in several stages. Firstly causes of external pressures and damage were assessed, measures were then put in place to reduce and prevent continued damage, such as erosion of bare soil. Internal pressures were then addressed through manipulating soil chemistry and stabilising the soil, particularly on steep gullies, after which the vegetation was introduced. Finally, water tables were manipulated to reduce desiccation and decomposition in addition to enhancing surface re-vegetation (Mitchell *et al.* 2008). The majority of flow in areas of intact blanket peat occurs within the upper 50mm (Daniels *et al.* 2008).

Moors for the Future have implemented peatland restoration in six phases: 1) identify causes and preventions, 2) managing sheep, 3) stabilising bare peat, 4) liming, seeding and fertilisin, 5) increasing diversity, 6) gully blocking. In this section the phases are discussed as two main stages giving details of the challenges to restoration, the methods used to overcome them, and research related to the techniques used. The first stage is on the prevention and reduction of damage caused by external pressures; the second is on the intervention for manipulation of bare peat chemistry, hydrology and biology.

1.6.1.1 Stage 1 - Prevention and reduction of damage

Models by Gallego-Sala *et al.* (2010) of variables influencing bog distribution in the British Isles found temperature changes to be the most important driver. Degraded, bare peat is more vulnerable to stochastic weather events, such as drought, associated with climate change (Clark *et al.* 2010, Worrall *et al.* 2010). Climate is an external driver which is not easily controlled and can increase peatland degradation rates. Climate change and the global rise in atmospheric temperatures are a threat to peatlands, particularly ombrotophic bogs as they depend on atmospheric inputs. Drought events have been linked to long term effects (up to 25 years) on of carbon out fluxes from a peatland ecosystem (Worrall *et al.* 2006). Rates of erosion are higher on bare peat (Daniels *et al.* 2008), bare and eroded peat is less resilient to stochastic weather events. Furthermore increased temperatures can elevate decomposition rates, while increased precipitation can increase the rate of erosion (Heathwaite 1993) and carbon exports (Dinsmore *et al.* 2013).

Peatland water table is influenced by precipitation, and although changes in precipitation cannot be easily influenced, the peatland's ability to store the water can be influenced. *Sphagnum spp.* species are sensitive to changes in the water table (Rochefort

2000). The promotion of important bog forming species, such as *Sphagnum spp.,* is one of the aims of revegetation as *Sphagnum spp.* form part of an ideal bog ecosystem.

Historical atmospheric pollution deposition is a challenge to peatland management (Parry *et al.* 2014). Alteration of the geochemical conditions, such as nutrient availability, pH, conductivity, and water table level can alter the competitive advantage of the niche peat forming species such as *Sphagnum spp.* (Dube *et al.* 2011). Prior to restoration the soil pore water pH on Bleaklow was between 2.8 and 3.5 (Rothwell *et al.* 2006). A study by Wind-Mulder *et al.* (1996) also found peat soil water and peat soil chemistry were altered post removal of surface peat in a Canadian bog.

A pH of 3.5-4 inhibits growth of soil microbial communities advantageous for root establishment and nutrient uptake (Smith and Read 1997), and does not allow the formation of a beneficial bacterial and fungal community which in turn supports vegetation (Caporn *et al.* 2007). Liming of Bleaklow sites (Parry *et al.* 2014) and raising pH enabled plants to make better use of the nutrients and was thus essential for the promotion of growth on bare peats (Caporn *et al.* 2007), as was fertilisation (with a nitrogen, phosphorus and potassium based fertiliser). According to Caporn *et al.* (2007) liming had a greater importance in influencing soil pH than fertiliser. However, fertilisation is required in bare peat restoration as peatbogs are phosphorous limited, and although N deposition would ordinarily inhibit *Sphagnum spp.* growth, the lack of on-site vegetation or a litter layer resulted in a depletion of nutrients which are required for vegetation growth (Tomassen *et al.* 2003).

Wildfires such as the April 2003 Bleaklow fire have occurred frequently in the PDNP, and according to a Moors for the Future report (2009) there have been over 400 wildfires in the PDNP since 1976. Wildfires can cover very large areas and in some cases are more intense and severe than controlled burns (Davies *et al.* 2008), and can result in peat ignition and the exposure of large areas to erosion (Albertson *et al.* 2010). If >10cm of peat is ignited all of the viable seed bank will be destroyed (Legg *et al.* 1992). Thus on Bleaklow the loss of the surface vegetation and peat required the active reintroduction of flora.

1.6.1.2 Stage 2 – Intervention

Blanket bogs (e.g. on Bleaklow), are susceptible to damage from visitor pressure and animal trampling. Low levels of sheep grazing can initiate or increase erosion of a peatland (Ellis and Tallis 2001) consequently influencing vegetation species dominance and succession (Hope *et al.* 1996, Ward *et al.* 2007). A study by Worrall and Clay (2012a) found that grazing could lead to peatland environments being net sources of GHG, as well as enhancing the effect of burning on dissolved organic carbon (DOC) exports and decreasing water table depth (WTD) (Worrall *et al.* 2007a).

On the Kinder plateau (a locality adjacent to Bleaklow) Anderson and Radford (1994) reported on benefit to reducing sheep grazing. The benefits included the reduction of vegetation fragmentation and soil erosion, encouragement of revegetation, and prevention of fresh young growth being eaten on restoration sites. On Bleaklow, grazing was also prevented by the Moors for the Future partnership (2012a) by working with the local farmers to reduce grazing and install 31km of stock proof fencing (Anderson *et al.* 2011) around 25.5km^2 of Bleaklow, in order to excluding livestock (Caporn *et al.* 2007). Visitor pressure was managed through channelling access using fencing, paths, walkways and sign posts.

Vegetation takes time to establish. Natural revegetation on Bleaklow was a challenge due to the loss of the surface peat and its viable seed bank (Legg *et al.* 1992, Salonen 1994). A study by Lavoie and Rochefort (1996) of Canadian extracted peatlands sites found that without intervention an abandoned, bare, degraded peatland did not return to a functional peatland ecosystem even 30 years after abandonment.

Further investigation about the reintroduction of *Sphagnum spp. diaspores* was conducted by Rochefort (2000) as part of a Canadian revegetation program (investigating straw mulch and phosphorous fertilizer in addition to *Sphagnum*-moss transfer). After 10 years there was much improvement in the surface vegetation and below ground processes to a degree similar to that of natural neighbouring bogs (Andersen *et al.* 2013). Thus on Bleaklow, seeding (nurse crops) or use of plug plants and on site fertilization was conducted. *Sphagnum*-moss pellets have also been used in some areas (Parry *et al.* 2014).

On Bleaklow, the use of stabilisation techniques significantly increased vegetation cover (Anderson *et al.* 2011). To give vegetation the opportunity to grow on Bleaklow, altering the pH and the creation of germination sites was conducted alongside the addition of seeds/diaspores and physical alterations (installation of gully dams or geojute netting and spreading heather brash) (Mitchell *et al.* 2008). Similar techniques were also used on Bleaklow to restore the bare sites (species reintroduction, mulch spreading and drain blockage) as those also studied by Rochefort (2000) to restore excavated bare peat.

Locally-cut heather mulch (brash) was spread over gently sloping bare peat surfaces of up to 30^{o} (Parry *et al.* 2014) to act as a barrier to weathering and reduce erosion, provide a more suitable microclimate, and add seeds and fungi that will support ecosystem development. The addition of litter was studied by Waddington *et al.* (2003) who found that the use of mulch increases soil moisture and decreases the WTD, as well as reducing surface albedo in comparison to the dark bare peat, thereby moderating peat temperatures. It was found that mulching increases CO_2 flux, however the magnitude of this effect decreased as the mulch aged. A study by Strack and Zuback (2013) replicated the methods used by Waddington *et al.* (2003) in North Alberta. They found that restoration greatly reduced DOC concentration carbon losses relative to the unrestored extraction site; the losses were even lower than estimated at natural pristine peatland.

As part of the soil stabilisation conducted by the Moors for the Future partnership (2012b) rolls of biodegradable textile mesh (known as ‘geojute’) was pegged down on areas too steep for mulch to remain in place, such as steeper peat slopes and sides of haggs (Anderson *et al.* 2011). The geojute reduces erosion and traps sown seeds to aid germination rates on these steep areas. Geotextiles on soil surfaces have been shown to be an effective soil conservation practice reducing both runoff and water erosion (Bhattacharyya *et al.* 2010). Meyer *et al.* (1970) also found the use mulch on bare peat extracted sites reduced velocity of runoff, resulting in decreased soil erosion compared to a site without mulch (Meyer *et al.* 1970). Price (1997) found mulch also increased surface soil moisture.

Peat surface stabilisation should reduce the ongoing loss of carbon through erosion, while revegetation should increase sequestration of CO_2 through enhanced primary productivity (photosynthetic activity). McHugh et al. (2000), and Wishart and Warburton (2001) found that in long established gullies erosion was low, indicating not all gullies may need to be blocked to ensure the stability of the peatland. Drain blocking (analogous to gully blocking) has been shown to decrease the depth of a water table (WT) (Wind-Mulder *et al.* 1996) in addition to reducing carbon losses and water colour (Turner *et al.* 2013).

1.6.2 Rotational peatland vegetation management

Grouse (*Lagopus lagopus scotia*) shooting is an important economic use of the uplands and one of the few that continues, largely without direct government subsidy. For a profitable game grouse population, management is used to create a heather stand age mosaic composed of a mixture of habitats for grouse, the fist for grazing composed of juvenile heather and the second for nesting composed of mature heather are required (Holden *et al.* 2012). Traditionally heather burning has been preferred, as burning of vegetation allows the removal of undesired or aged species, reducing their competitive dominance (Mitchell *et al.* 2008). A recent study suggests that an estimated 114 km^2 area of peatland is managed through controlled burns annually (Yallop *et al.* 2006).

The frequency and temperature of a fire are important factors in a burning regime. If burned too infrequently, degenerate heather (~20 yr old) becomes dry and woody and can result in hot fires/wildfires (Hobbs and Gimingham 1987). Hot fires are discouraged by the Defra Burning Code (Defra 2007). Alternative methods to burning, such as cutting, are being pursued (Calvo *et al.* 2002, Cotton and Hale 1994). Both burning and cutting of vegetation allows for the removal of undesired or aged species (Mitchell *et al.* 2008).

1.6.3 Effects of heather burning *vs.* cutting on a peatland

There is an ongoing debate on the benefits and disadvantages of managed vegetation burning to upland water quality. However there is agreement that managed burns should follow the Defra heather and grass burning code (2007); burns should be carried out at a temperatures below 200°C at which the heather seeds would be killed (Whittaker and Gimingham 1962). There is evidence to support that vegetation burning has a negative impact on peatland carbon balance (Brown *et al.* 2014, Farage *et al.* 2009, Holden *et al.* 2012, Imeson 1971, Yallop and Clutterbuck 2009). Clutterbuck and Yallop (2010) confirmed that when comparing two non-burnt controls to four newly burnt catchments on blanket peat, concentrations of DOC in drainage waters from the four burnt catchments increased, relative to the unburnt controls. Furthermore, Yallop and Clutterbuck (2009) found that across 50 British catchments (during

2005), areas where burns exposed bare peat surface resulted in the alteration of the hydrological status of the underlying peat allowing enhanced aerobic decomposition, DOC productivity and release in upland environments. These increases in DOC flux and concentration were not explained by regional or global phenomena (such as changes in air temperature), which could explain only 20-30% of the increase in DOC concentrations. The debate surrounding the impacts of burning on peatland carbon cycling is active with some suggesting that burning which does not expose bare peat, does not significantly increase DOC concentration in the long term (Clay *et al.* 2009).

Alternative methods to burning such as vegetation cutting are being pursued (Cotton and Hale 1994), as both burning and cutting of vegetation allows for the removal of undesired or aged species (Mitchell *et al.* 2008). Research into the comparative effects of upland vegetation cutting and burning has mainly focused upon the ecology and succession of vegetation (Calvo *et al.* 2002, Calvo *et al.* 2005). Cotton and Hale (1994) found that re-growth of heather on cut sites is lagged, by approximately one year, when compared to sites managed by burning. Calvo *et al.* (2002) support the finding that cutting heather *does* not encourage its preservation, particularly as aged heather regenerate poorly and slowly. Cutting can therefore promote replacement of ericaceous shrubs species (such as heather) by other faster growing plants (Mitchell *et al.* 2008), such as herbaceous species (Calvo *et al.* 2005) and grasses (such as *Molinia)* (Ross *et al.* 2003). In the interest of heather regeneration, Calvo *et al.* (2002) recommended burn treatment practiced with cycles of 10-15 years. On the other hand to increase diversity and to create a mosaic of vegetation with different succession stages and structures, Muñoz *et al.* (2012) recommended cutting as a useful vegetation management tool.

Burning and/or cutting of vegetation on peat soils have been shown to raise water tables at the plot scale but reports on the effects upon DOC concentration vary (Worrall *et al.*

2012) Ward *et al.* (2007) and Clay *et al.* (2009) found no significant difference in DOC concentrations in soil waters between burnt and unburnt sites. While Worrall *et al.* (2007b) and Helliwell *et al.* (2010) showed a significant decrease in DOC concentration in soil water on burnt sites relative to unburnt sites. Worrall *et al.* (2013) found that in a short term study soil pore water DOC concentrations significant decline post both vegetation cutting and burning. Clay *et al.* (2009) was the only study to consider concentrations in surface runoff of burnt and unburnt plot scale. Clay *et al.* (2012) found no significant change in DOC concentration for burnt plots up to 10 years after burning, however they did find that there was a significant increase in water colour for up to 4 years after a plot was burnt indicating burning did negatively influence water quality during that 4 year post burn period. The impacts of burning at a catchment-scale have been variable and contradicted findings of Clay *et al.* (2012) at plot scale (Brown *et al.* 2014, Clutterbuck and Yallop 2010, Yallop and Clutterbuck 2009).

1.7 Peatland hydrology and cycling

In order to investigate the effects of peatland vegetation management on peatland C losses, the peatland carbon cycle must be considered. Scientists Ingram (1967) and Ivanov (1981) coined the terms 'acrotelm' and 'catotelm', which are associated with the Ingram and Braggs (1984) diplotelmic peat functional systems model; a concept used to understand the general hydrology of the peat system (Moore 1995). The 'acrotelm', is located 10−50 cm below the surface (Franzén 2006). It is defined as the upper partly living soil, frequently aerated layer (Figure 1.3), in which water moves freely and usually laterally. The 'catotelm' is the permanently waterlogged, anaerobic peat mass, through which water movements are usually slow. Water filters down from the peat surface rapidly, but is retained at depth in the catotelm (Farrick and Price, 2009). This system as discussed by Moore (1995) and Holden and Burt (2003), is a model describing a natural intact 'active' bog system (Rochefort 2000). However it is not comprehensive enough to describe all peat systems (Lindsay 2010).

As peatland vegetation photosynthesise (Equation 1.1) and grow, they act as C stores, taking in atmospheric carbon through the leaf stomata. During daytime peatland vegetation has an increased rate of photosynthesis, while at night rates of photosynthesis are decreased and rates of the vegetation and peat soil microbes respiration become dominant (Equation 1.1 in reverse). Deposition of OM occurs on peat when the rates of photosynthesis exceed the gaseous production of respiration, and there is increased vegetation growth. The combination of OM deposition and anaerobic conditions (and shallow water tables) leads to the formation of peat soil at the surface of the acrotelm (Holden *et al.* 2007, Turunen *et al.* 2002). C peatland storage is consequently controlled by the balance between vegetation productivity, decay and respiration (Franzén *et al.* 2012).

$$6CO_2 + 6H_2O\ (+\ light\ Photon) \rightharpoondown C_6H_{12}O_6 + 6O_2$$

Equation 1.1

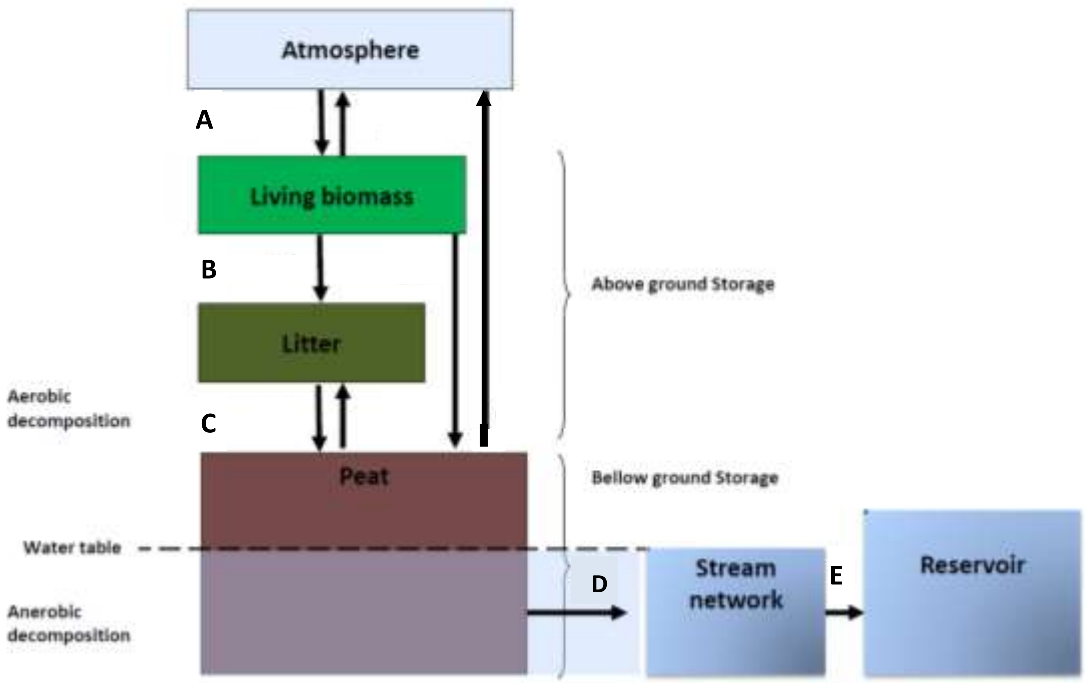


Figure 1.3: Peatland carbon cycle. The Black arrows represent carbon flow. A) Gaseous carbon flow between the atmosphere and living biomass. B) Organic carbon flow from vegetation to soil as dead biomass (litter). C) Incorporation of litter into the peat and the release of root exudates into peat soil. D) Carbon being dissolved into the soil water as DOC and POC and exported into the stream networks. E) The eroded and dissolved carbon reaches the reservoirs.

Carbon waste that is not incorporated into cells is transferred from the plant into peat via the plant roots. The plant roots continuously produce and secrete compounds into the rhizosphere (Flores *et al.* 1999). The rhizosphere is the zone of soil occupied by plant roots, the zone biology and chemistry of the soil are influenced by the root presence. Root exudation includes the secretion of ions, free oxygen, water, enzymes (complex protein molecules), mucilage, and a diverse array of carbon-containing primary and secondary metabolites (Mitchell *et al.* 2008, Wallage and Holden 2010). These organic molecules have a varying degree of solubility. Hydrophobic DOC compounds are adsorbed onto the soil particles/aggregates and are leached slowly into the soil pore water (Scott *et al.* 1998). Phosphate limitation may induce the release of organic exudates. Organic exudate in turn can affect soil microbial populations indirectly through their effects on soil pH (Flores *et al.* 1999).

Vertical peatland growth is controlled by the balance between primary production and decay (Franzén *et al.* 2012). Only ~5−10% of the biomass produced at the peat surface is incorporated into the catotelm as peat (Clymo 1984, Gorham 1991, Warner et al., 1993). The rate of decomposition of dead plant (leaf litter) material depends on temperature, aeration and supply of nutrients to micro-organisms (Franzén *et al.* 2012). The biomass is either broken down and decomposed releasing carbon back into the atmosphere as CO_2, or dissolved or eroded and washed away as POC (Pawson *et al.* 2008). The loss of C as POC is important as it can then be converted to DOC in stream processes, which is then oxidised and released into the atmosphere gaseous CO_2 (Moody *et al.* 2013).

Within the peat soil (Figure 1.3), DOC production is driven through oxidative (Mcknight et al., 1985) and microbial processes (Scott *et al.* 1998). Furthermore factors such as air and soil temperatures, frequency and intensity of rainfall, pH and sulphate concentrations, create a complex variation of DOC (Clark *et al.* 2005, Clark *et al.* 2008, Scott *et al.* 1998). Hydrological process control the export of DOC (McDowell and Likens 1988) along the fluvial pathway. Studies by (Clark *et al.* 2008, McDowell and Likens 1988) documented an increase of DOC concentrations within surface waters as a result of the passage of the waters through peat.

The Ingram and Braggs (1984) diplotelmic peat model describes a natural intact bog system. The term ‘haplotelmic’ describing a one layer bog system is a more representative model of a bare eroded peat system which has lost its surface vegetation and much of its viable seed bank (Salonen 1994), and its hydrological regime (Holden *et al.* 2011). According to the diplotelmic peat model, the fluctuating water table at the surface of the peat increases oxygen availability in the acrotelm (Moore 1995). Lavoie and Rochefort (1996) found that bare extracted sites had deeper water tables than revegetated extracted sites. In the absence of vegetation, the C path from the atmosphere to the peat through photosynthesis (1.3.B) and litter deposition is missing from the C cycle. Furthermore, the hydrology of bare peat created aerobic conditions at greater depths within the peat profile. This aeration results in shrinkage, oxidation and compression of peat (Price 1996). The habitat is thus altered and promotes the replacement of *Bryophyte* peat forming plants (such as *Sphagnum spp.)* with non-peat forming vegetation (including non-*Sphagnum spp.* mosses or tussock-forming species). Thus haplotelmic peatlands are non-carbon sequestering environments (Lindsay 2010). The continued erosion of the peat acrotelm surface can cause the loss of surface layer peat and disruption to a complicated hydrological regime (Ingram 1967), in which case studies have found these difficult to restore (Holden *et al.* 2011).

The C cycling within peat is complex, therefore when creating a carbon budget for a peatland many carbon species have to be accounted for. According to Worrall at al. (2003b), rainfall DIC and DOC, CO_2 exchange, CH_4 emissions, DOC export and POC export are required to make a full, comprehensive study of carbon balance within a peat system. However, with limited resources, one must focus the research effort on measuring specific C species, understanding their role within the system and the processes which lead to their production.

1.8 Carbon cycle – atmospheric pathway

1.8.1 Carbon Dioxide (CO_2)

It is important to monitor CO_2 atmospheric concentration, as CO_2 is one of the most abundant greenhouse gases. It is therefore an important component of a peatland terrestrial–atmospheric carbon cycle (Figure 1.3). The rise in CO_2 emissions since the industrial revolution has been linked to an increased rate of climate change. The Stern report (2006) proposed that in order to reduce the future negative impacts of climate change on economically valuable ecosystem services, a global coordinated effort must be made to reduce GHG atmospheric concentrations such as CO_2.

Carbon dioxide is measured as a gas concentration in the atmosphere and is discussed in terms of increasing or decreasing fluxes over time (positive or negative). There are three important fluxes: 1) Net Ecosystem Respiration (R_{eco}, also referred to as NER), the movement of C from peat (including: plants and superficial microbes) to the atmosphere. 2) Net Ecosystem Exchange (NEE) is the movement of C to and from the peat and atmosphere, and is used to explain if a peatland is a sink or source of C. Measurements of NEE are taken during light conditions, and can be used to calculate the third flux which is not easy to directly measure. 3) Gross photosynthesis ($P_{g.}$), is an influx associated with the vegetation component of the C cycle. P_g is derived by the subtraction of R_{eco} from NEE fluxes. Important drivers for CO_2 exchange are soil moisture, WTD, soil temperature and vegetation cover (Glatzel *et al.* 2006, Lloyd and Taylor 1994, Updegraff *et al.* 2001). Further details on the measurement and calculation of CO_2 fluxes are available it the Methodology section (2.1.2.2).

1.8.2 Methane (CH_4)

Methane is a prolific greenhouse gas, more effective than CO_2 at trapping heat in the atmosphere (Lashof and Ahuja 1990). Studies by Hargreaves *et al.* (2001) demonstrate peat bogs' contribution of CH_4 into the atmosphere. For example, the largest fluxes of methane have been measured from cotton grass (in addition to non-Sphagnum mosses). The structure of cotton grass' lacunal (air chamber) systems provides a conduit for the passage of CH_4 emissions from the peat soils into the atmosphere (Thomas et al., 1996). Studies have found that releases of atmospheric C as methane from blanket bogs is higher than fluvial losses of DOC. However that is not the case in an eroded or damaged bog due to the deeper water table and increased aeration of the peat surface (Shepherd et al*.,* 2013. Change in water table depth is therefore a major driver of CH_4 fluxes in peatlands (Danevčič *et al.* 2010, Nykänen *et al.* 2003). Methane was not directly measured as part of the research presented in this thesis due to time limitations.

1.9 Carbon cycle – fluvial pathway

1.9.1 Particulate Organic Carbon (POC)

Export pathways of POC are predominantly within the fluvial environment and according to studies, the amount of POC exported varies according to the state of the catchment. Some studies have measured POC to represent 10-15% of total organic carbon flux. Dinsmore *et al.* (2010) found that POC represents a small portion of peatland carbon flux; POC was the smallest portion of C export within the fluvial output (after, DOC, DIC), also smaller than CO_2 atmospheric losses. Peatland soil erosion is an important driver for the production of POC; the correlation between soil erosion and POC is evident in bare and eroded catchments, in which POC represented up to 80% of fluvial exports according to studies by Evans *et al.* (2006), Pawson *et al*. (2008), and Worrall *et al*. (2011). Peatlands revegetation can effectively control erosion and sediment flux (Evans *et al.* 2006). According to Moors for the Future intervention and ecological restoration efforts can reduce erosion and cut POC losses by up 95% within two years.

1.9.2 Dissolved Organic Carbon (DOC)

DOC is a general term describing a wide range of molecules from simple acids and sugars to complex humic substances with large molecular weights (Moore 1998). DOC is identified as carbon that is capable of passing through a 0.45µm syringe filter (Tranvik 1998). Compounds of DOC are formed in various stages of decomposition ranging from acids to complex humic substances (Wallage *et al.* 2006). Studies report DOC plays a major role in fluvial carbon export (Dawson et al., 2002). Studies are biased towards aquatic fluxes of DOC (Clark *et al.* 2008, Hope *et al.* 1996, McDowell and Likens 1988).

Surface waters DOM concentration and speciation is dependent on import, washout, indigenous primary production and processes. Internal system loss can be incurred due to abiotic mineralization (particularly photo-oxidation), microbial mineralization and flocculation followed by sedimentation (Tranvik 1998). Broad climatic and site factors have been identified as key factors influencing DOC concentrations (van den Berg *et al.* 2012). For example drought years were linked to observations of lower DOC concentration by Clark *et al.* (2005). There are several mechanisms which increase DOC production at varying time scales. These mechanisms can be divided into abiotic and biotic factors related to the following: a) abiotic such as increased air temperatures (Freeman *et al.* 2001) and severe drought events (Clark *et al.* 2009, Neff and Hooper 2002, Worrall and Burt 2004), potentially associated with climate change (Dinsmore *et al.* 2013, Frolking *et al.* 2006, Worrall *et al.* 2003b); changes in soil pH (Clark *et al.* 2008, Scott *et al.* 1998); changes in water flow volume and nature; increases in atmospheric concentrations of CO_2, and changes in atmospheric deposition and eutrophication (Freeman *et al.* 2004). b) Biotic, historic vegetation type, which controls the physical and geochemical characteristics of the peat mass (Brown *et al.* 2014); vegetation cover and composition (Armstrong *et al.* 2012, Neff and Hooper 2002), land management (Clutterbuck and Yallop 2010, Yallop and Clutterbuck 2009). All these mentioned factors are enhanced by anthropogenic activities and local land management (Freeman *et al.* 2004, Mitchell *et al.* 2008, Wallage and Holden 2010).

The impact of land management upon DOC concentrations in soil pore water has been investigated for a number of land management types on peatlands, including: prescribed burning (Clay et al., 2009a), drainage (Gibson et al., 2009), deforestation (Glatzel et al., 2003), afforestation (Jandl et al., 2007), and grazing (Ward et al., 2007). There is little research on the long term effects of bare peat restoration or comparative heather management methods on DOC.

1.9.2.1 DOC and water quality

Dissolved organic carbon is mobile, non-fixed OM (Whitbread 1994). At high concentrations DOC is considered a pollutant which can lead to biological contamination and is linked to water taste, odour (Volk *et al.* 2002) and discoloration (Butcher *et al.* 1995). DOC influences river water quality through the transport of complex metals and nutrients and its effect on acidity and pH (Driscoll *et al.* 1989, Driscoll *et al.* 1994). Microbial energy and nutrient supply, light absorbance and photochemistry in surface waters can be influenced by DOC (Evans *et al.* 2005). DOC represents a significant challenge to water supply companies who may have to remove DOC to meet drinking water quality standards (Dinsmore *et al.* 2013). DOC is costly to remove. In some cases incomplete removal can result in the formation of disinfectant by-products (DBP's) such as carcinogenic trihalomethane (Condie *et al.* 1983, Gough *et al.* 2014, Volk *et al.* 2002). Water colour is therefore a primary precursor for trihalomethanes and other DBPs (Reckhow *et al.* 1990). Trihalomethane concentrations in drinking water are limited by law in the UK under the Water Supply (Water Quality) Regulations (2010). The regulation specifies a maximum total trihalomethanes concentration of 100 μg L-1 (Gough *et al.* 2014, Hsu *et al.* 2001).

Studies over the past five decades on terrestrial surface waters report a significant rise in DOC concentrations (Evans *et al.* 2005, Monteith *et al.* 2007, Worrall and Burt 2007) in the UK (Freeman *et al.* 2001), central Europe (Hejzlar et al 2003), and the USA (Skjelvale et al., 2001, Driscoll et al., 2003). Trends have been attributed to increasing temperatures due to climate change. However, other studies have not been able to sufficiently explain DOC from temperature variation alone. The observed trends have given rise to the interest to conduct further research into water quality and carbon flux within a peatland system. This is of interest to water companies which own land (e.g. in the Goyt Valley) which constituted the catchment feeding into reservoirs for the following reasons: The UK is an estimated annual net sink of

0.32 MtC/yr (Holden *et al.* 2007), these upland regions source 70% of the UKs drinking water (Burt *et al.* 1997). The distribution of organic-rich peat, present in these upland regions, is closely associated with increased DOC, POC (Hope *et al.* 1997) and water colour (Butcher et al., 1995).

1.9.2.2 DOC components

As disused earlier, DOC variation is influenced by microbes, frequency and intensity of rainfall, pH and sulphate (Scott *et al.* 1998). Composition of DOC can vary from fulvic, lignin derived, relatively lower molecular weight molecules to the humic relatively higher molecular weight associated with darker water colours (Carlsen *et al.* 2000). The degree of peat humification is related to initial peat breakdown and decomposition (Whitbread 1994). Specific UV absorbance (SUVA), which is the UV absorbance of a water sample at a given wavelength , is related to DOC (Weishaar *et al.* 2003). A proxy for the degree of sample humification is E_4/E_6 (discussed in General methodology section).

Humic	Fulvic
Present in deeper peat Lower E_4/E_6 Macro Molecular/ heterogeneous	Lignin derived Higher E_4/E_6 – above 11 Lower molecular weight (MW)
Chemical structure: – More aromatic – More acid groups – Polyelectrolytes – Carbolic and phenolic function groups – Higher C:N ratio due to protein/peptides	 – Phenolic units – Carboxylic structures – Less aromatics and O-alkyl C – More carboxyl carbon – Low aromatics – Large aliphatic structures
Environmental Consequence: – Can effect water sorption capacity towards pollutants – Increase water colour	 Better for bioactivity (Chen *et al.* 1977)(above E_4/E_6 12)

Table 1.2: Characteristics of DOC Humic and fulvic components (Carlsen *et al.* 2000, Chen *et al.* 1977, Thomsen *et al.* 2002).

1.10 Peatland water quality (Nutrients/ acidity and ionic strength)

Nutrient concentrations have important roles in atmospheric carbon cycling as they can impact the rate of photosynthesis and plan growth (Shaver *et al.* 2000). Peatland plant productivity is often limited by phosphorus and/or nitrogen (Bridgham et al., 1996, Bedford et al., 1999). Nutrient changes can therefore mediate shifts in vegetation communities (Nykänen *et al.* 2003). As mentioned previously blanket bogs receive all nutrients from the atmosphere (Shepherd et al., 2013). Meteorological factors that affect Atlantic storminess, such as the North Atlantic Oscillation influence sea salt deposition variation. Climatic factors can therefore result in fluctuation in nutrients such as nitrogen in the form of nitrates (Evans and Monteith 2001).

Anthropogenic activities, such as agriculture and industrialisation, are likely to increase nutrient availability (Keller *et al.* 2006) This can be done directly at source through leaching or indirectly through altering atmospheric chemistry. Industrialisation and increasing dependence on fossil fuels resulted in sulphur (S) emission peaks in 1970s. The increased S and nitrogen (N) compounds in precipitation, contributed to the acidification of soils and waters as they are two of the most significant acid species. Oxidation of reduced sulphur and mineralisation of organic sulphur result in the disassociation of sulphuric acid and increase of protons H^+ ions in peat (Equation 1.2 and Equation 1.3), thereby decreasing pH. The Gothenburg protocol (1999) on Multi-Pollutant Multi–Effect, resulted in a significant reduction of S emissions as well as nitrogen oxide (NO_x) (NEGTAP 2001).

$$H_2SO_4 \rightarrow 2H^+ + SO_4^{2-}$$

Equation 1.2

$$HNO_3 \rightarrow H^+ + NO_3^-$$

Equation 1.3

Atmospheric deposition can affect soil OM solubility by influencing the acidity of soils and/or the ionic strength of soil solutions, and the ionic strength on the coagulation of DOC. Decreasing inputs of acidic deposition or sea salt should lead to lower concentrations of a suite of multivalent ions found in soil solution—including SO_4^{2-}, Ca^{2+}, Mg^{2+} (Monteith *et al.* 2007). A Study by Keller *et al.* (2006) found that increased phosphorus availability within peat inhibited CH_4 production in bogs, and high nitrogen concentrations had an inhibitory effect on CH_4 oxidation. Studies at 10cm soil pore water depth have also found a strong relationship between sulphate and DOC (Clark *et al.* 2005). Ionic strength and pH control the solubility of DOC (Freeman et al., 1993, Scott et al 1998, Adam et al., 2001). It is therefore important to monitor nutrients, H+ ions and ionic strength through measuring anions, pH and conductivity.

1.11 Structure of Thesis

This thesis is composed of:

- Thesis introduction (Chapter 1), with the background to peatlands, carbon cycles and peatland management relevant to all the experimental chapters (Chapters 3-6). Each of the experimental chapters will have a brief introduction explaining their content and context within this thesis; however the thesis introduction (Chapter 1) will need to be referred to for greater detail.
- General methodology (Chapter 2), on field experimental and laboratory and statistical analytical techniques. The general methodology section is provided to avoid the repetition of information in the experimental research chapters. This section should therefore be referred to as a baseline for all the experimental chapters (3-6).
- Four experimental chapters (Chapter 3–6). The first two chapters (3-4) were based on a study on bare peat restoration intervention and management (over a 5 year period): Chapter 3 is focused on the effects of the management on water table depth and water quality; Chapter 4 is focused on carbon dioxide fluxes. Chapters (5-6) are based on research conducted in the Goyt Valley on the subject of managed vegetation cutting and burning in wet and dry localities: Chapter 5 is focused on the comparative differences between heather management types on water table depth and water quality over a (5 year period); Chapter 6 is focused on the effects of heather management at multiple spatial scales and the differences in DOC from head to reservoir.
- Conclusions (Chapter 7), outlines the main finding (using peatland cross section schematics)and limitation, provides a recommendation for future peatland management and highlights further areas of research (Chapter 7).
- Reference.
- Appendices, includes images of the study sites and useful abbreviations and acronyms.

1.12 Aims and objectives

The studies in this thesis will focus on the DOC component of the carbon cycle within the aquatic pathway. The broad aims of this thesis are to contribute toward ongoing debate on peatland vegetation management research to:

- Assess the multiannual effects of bare peat revegetation on soil and surface water DOC concentrations and CO_2 fluxes.
- Assess the multiannual effects of bare peat revegetation on CO_2 fluxes.
- Assess the multiannual effects of cutting verse burning of heather *on* soil and surface water DOC concentrations.

1.13 Co-author contributions

The research in this thesis formed part of two papers that have been published:

- Qassim S. M., Dixon S. D., Rowson J. G.,Worrall F., Evans M., Bonn A. (2014), A 5 year study of the impact of peatland revegetation upon DOC concentrations. Biogeochemistry.
- Dixon S. D., Qassim S. M., Rowson J. G., Worrall F., Evans M. G., Allott T. E. H., Boothroyd I. M. (2014). The impact of peatland restoration on CO2 fluxes and water table depths from a climatically marginal upland blanket bog. Biogeochemistry.

Other contributors towards the research in thesis (by chapter) are:

- Natural England and United Utilities: funded the research in this thesis.
- Prof Martin G. Evans: provided Bleaklow weather monitoring station and erosion pin data. The data on temperature were used (as covariates) in statistical analysis (Chapters 3 and 4).
- Defra - Acid Deposition (UKEAP): open access data on rainfall volumes and water chemistry were used during statistical analysis in Chapter 3.
- United Utilities and the Sustainable Catchment Management Programme: supplied data on Goyt valley reservoir water quality and environmental variables used in Chapter 6.

- Prof Fred Worrall: supervised of this doctoral research. Worrall led the project development and site selection in 2006. Worrall provided supervision, field training and helpful feedback on the chapter write ups, in addition to collecting data during the summer months of 2010.
- Dr Simon D. Dixon: the multiannual element of this thesis research was possible due to the initial data conducted between 2006 and early 2010 in Dixon (2011) (Chapters 3-6). Dixon provided training on the use of Geographical information Systems (GIS), guidance on statistical analytical methods and proof read the experimental chapters in this thesis. Some analysis on water table depth (chapter 3) and a figure on CO_2 fluxes (Chapter 4) were adapted from Dixon *et al.* (2014).
- Dr James G. Rowson: aided with the data collection between 2006 and early 2010 and lead installed the initial sites in the Goyt Valley (Chapters 5 and 6).
- Dr Ian M. Boothroyd: provided lab and field training and proof read several chapters.
- Dr Zhang Zhuoli: assisted with the field data and sample collection during several autum and winter month in 2012 (Chapters 6).

2 General Methodology

This chapter will give details of the methodologies employed to conduct the research presented in this thesis. This General methodology chapter includes details on: a) field site selection, with background on study localities and instrumentation; b) field monitoring regime and sample collection; and c) laboratory and statistical analytical methods. Any modifications to the methods will be highlighted where relevant within each experimental chapter (3-6).

The study sites were selected within the uplands of the Peak District National Park (PDNP), in Northern England. As areas are topographically high, the blanket bog in this region receives the majority of its nutrient input through atmospheric processes. The sites could therefore be considered ombrotrophic. The geographical localities used in this study are: the Bleaklow Plateau and the Goyt Valley (Big Moss and Ravenslow). The experiments within each locality involved the installation of set of monitoring and sampling sites. These are sites are considered as factors within an overall factorial experimental design (i.e. site with restoration or management, type of dominant vegetation cover, year etc). The sites used are the same as those used by Dixon (2011) (PhD thesis). Dixon (2011) focused his research efforts on considering the importance of altitude, vegetation and short temporal scales on peatland carbon fluxes. Some of the Dixon (2011) key findings are: altitude significantly influences CO_2 fluxes; there was a significant relationship between photosynthesis and respiration, and that respiration is temporally lagged by an estimated 3 hours. The focus of the research presented in this thesis is on the effects of different management types on water table depth, water quality and fluvial carbon (DOC concentrations) and CO_2 (fluxes) on a multiannual scale. Data from Dixon 2011 was used to allow the consideration of a multiannual record (5 years/ 5 project years).

2.1 Bleaklow Plateau – (Bare peat re-vegetation study)

The Bleaklow Plateau is situated on a raised topographic blanket bog to the west of Glossop (at 53˚ 46’ N 1˚ 84’), in the region of Dark Peak of the PDNP. In 2011 the total annual rainfall was 1152mm in 2011, and the mean monthly air temperature ranged from 0.57^{o}C to 12.39^{o}C between 2007 and 2011 (According to data obtained from Prof M. G. Evans, Manchester University). The plateau is 468 – 630 m above sea level and is composed of an extensive layer of peat, 2-3 m deep. This was formed on periglacial surface stratum and coarse grained Milstone Grit (Fearnsides *et al.* 1932, Rowson *et al.* 2013).

Bleaklow was selected because of its history of degradation which goes back to 1200-1000 years AD (Moors for the Future, 2010) and is association with historic climate change events such as the Medieval Warm Period and the Little Ice Age (Tallis 1997). These climatic events are associated with periods of increased peat wastage which lead to exposed bedrock rain channels and vast eroding and intersecting dendritic gullies (Bromehead *et al.* 1933, Daniels *et al.* 2008, Tallis 1983, Tallis 1994). In addition to changes in land management practices, the industrialisation of the nearby cities of Manchester and Sheffield subjected the region to pressures from atmospheric pollution deposition for 200 years (Tallis 1997). Acid deposition associated with oxidised nitrates and sulphur compounds, derived from fossil fuel combustion, had contributed towards the acidification of the peatlands and surface waters (Clark *et al.* 2005, Curtis *et al.* 2000). More recently the area has been subject to heavy grazing and visitor pressure. The region’s combined factors of historic weathering and erosion, heavy grazing, visitor pressure, wildfire and legacy of atmospheric deposition of metals and acids have led to the extensive gully erosion and dissection of the Bleaklow peat plateau (Tallis 1997), leaving ‘haggs’ in places where gullies met with isolated blocks of peat from the peat mass (Natural England 2000).

In 2003 a wildfire de-vegetated approximately 844 ha of the Bleaklow Plateau (Bonn *et al.* 2009, Worrall *et al.* 2011), adding to the erosion on Bleaklow and further exacerbating its poor ecological state to a condition similar to that displayed in figure 2.1 (Daniels *et al.* 2008, Evans and Lindsay 2010, Rothwell *et al.* 2007, Rowson *et al.* 2013). Consequently, the Bleaklow Plateau blanket bog was amongst the 75% of degraded bogs within the PDNP (Anderson and Tallis 1981). Intervention and treatment was needed to restore the site to a functional peat accumulating system (Sottocornola *et al.* 2007).

As a result of the damage caused by the 2003 wildfire, a group of partners (Peak District National Park Authority, National Trust, Natural England, United Utilities, Severn Trent Water, Environment Agency, Yorkshire Water, Derbyshire County Council and RSPB) joined together to form the Moors for the Future. Consequently the Moors for the Future have been the driving force for restoration on the Bleaklow Plateau and in order to achieve this, they have employed a range of interventionist management methods since 2003. The benefits of these management methods to a peatland system are discussed in detail in the Introduction chapter (section 1.6).

Figure 2.1: Bare eroded peat - Bleaklow Summit (2011).

2.1.1 Bleaklow site selection and monitoring regime

Bleaklow (Figure 2.2) was selected as the research locality in 2006, for a longitudinal study with the objective of monitoring the effects of bare peat re-vegetation on water quality and fluvial and atmospheric carbon fluxes. The Carbon Waste and Water research group at the University of Durham initiated the study installation four years after the restoration work began in 2003.

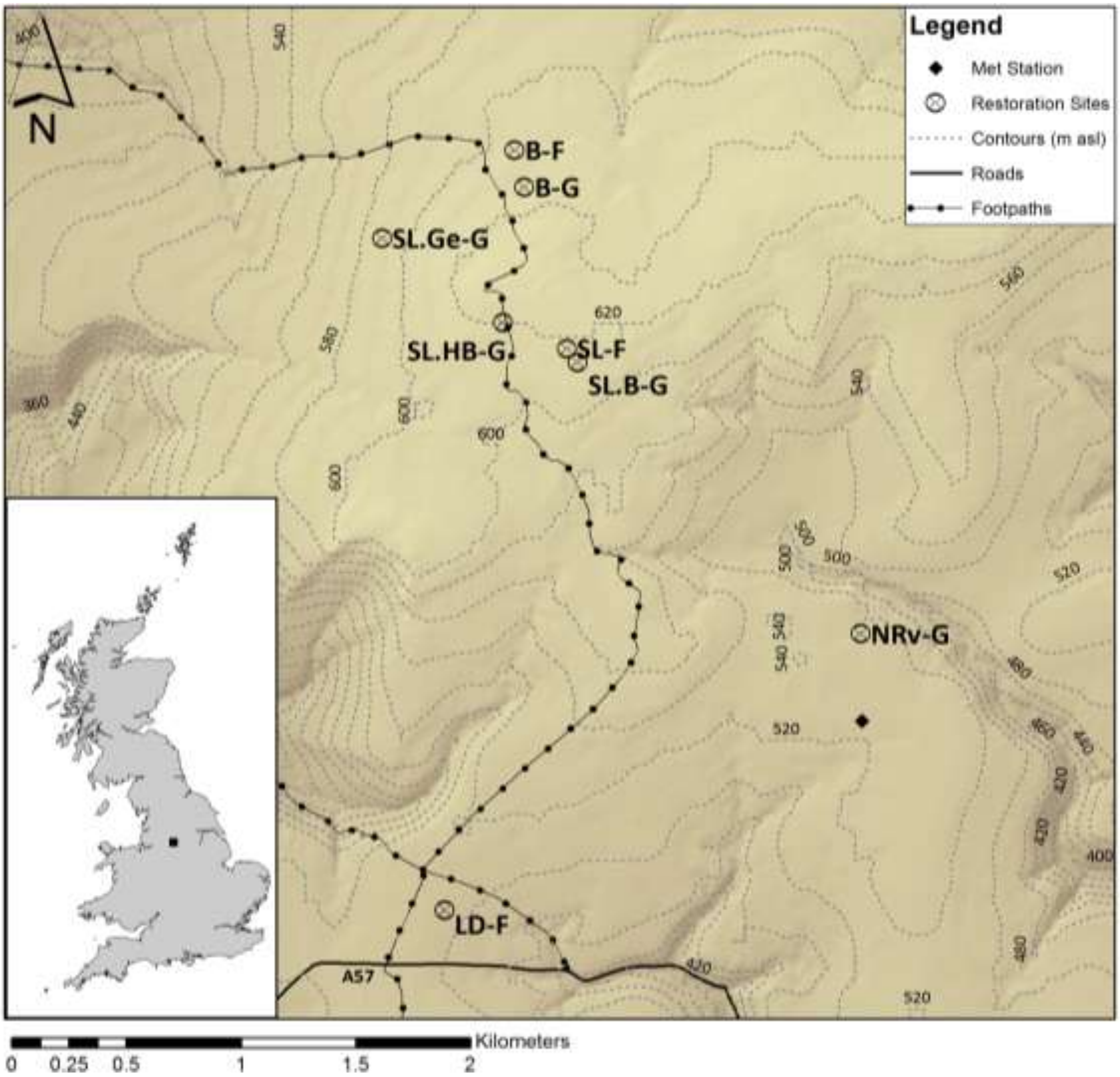


Figure 2.2: Location of study sites on the Bleaklow Plateau. For explanation of site codes see (Table 2.1). Bleaklow Plateau with field sites marked: Least disturbed flat (LD-F), naturally re-vegetated gully (NRv-G), seeded limed flat (SL-F), seeded limed gully blocked (SL.B-G), seeded limed heather brash gully (SL.HB-G), seeded limed geojute gully (SL.Ge-G), bare gully (B-G), bare flat (B-F) and met station. (Figure adapted from Dixon et al. 2014).

There were four control sites used as comparators to four sites with treatments. Two bare untreated (bare flat, B-F and bare gully, B-G) sites were used as bare controls, and the least disturbed vegetated site was used as a vegetated control (least disturbed vegetated control, LD-F). The naturally revegetated site (NRv-G) was the fourth control used during the eight sites (2008 - one year) comparison. The four restoration sites were monitored during this study (one was monitoring for a fewer number of years), all of which were seeded with a lawn grass mix (*Deschampsia spp.* and *Festuca spp.*), limed and fertilised (nitrogen, phosphorus, and potassium, NPK). One of the sites, received only an NPK intervention, and is named SL-F. Two of the remaining sites had their surface soil stabilised with different techniques to prevent erosion and protect the lawn grass seeds, promoting vegetation establishment. Biodegradable jute netting 'geojute' (Ge) was anchored to the soil using biodegradable pegs on SL.Ge-G. Mulch of *C. vulgaris* commonly referred to as heather brash (HB) was scattered on SL.HB-G. The final restoration site, SL.B-G, had a plastic dam gully block (B) reducing water flow from the site, in order to restore the water table to a shallower depth.

Eight study sites (Table 2.1) (were visited and sampled on a monthly basis between November 2006 and January 2012 except for periods when heavy snowfall prevented access (January 2008, February 2010, December 2012) and when collection and analysis of samples was not possible (July 2009, September 2009, April 2010, May 2010). The final year of the project (November 2011 to January 2012) included only three months. The final months of the project were included so that adequate winter coverage could be included in the analysis.

Weather variables were measured onsite during the monthly observation (air temperature, AT and photosynthetically active radiation, PAR). Additionally, hourly environmental variables were measured (precipitation and air temperature) from monitoring localities on Bleaklow from a Manchester University (MU) infield automated weather station

(AWS). Gaps in MU AWS rainfall data were filled using the Defra UK Air Information Service UK-AIR (Defra 2013) from a site called River Etherow (E 412410; N 398884) at ~440m altitude and ~4370m north west of Bleaklow summit. The Defra database included rainfall, monthly volumes and rainwater pH, conductivity, and concentration of sulphate and nitrates.

Original site names	Treatment	Site abbreviation	Eastings	Northings	Time since treatment, prior installation	Equipment installed (Nov 2006)	Equipment installed (Dec 2007)
Penguins drift	Control - least disturbed, Vegetated	LD-F	409054	393154	N/A	6 Gas collars & open dip wells & 1 wier plate	
Trenches South	Control - Bare, gully	B-G	409402	396384	N/A		
Trenches North	Control - Bare, Flat	B-F	409359	396549	N/A		
Josephs Patch Gully	Seeding, liming, Heather brash & Geojute	SL.Ge-G	408781	396156	1 yr		
Tubby west	Seeding, Liming & Heather brash, Gully	SL.HB-G	409309	395778	1 yr	6 Gas collars & open dip wells	
Tubby East	Seeding & Liming	SL-F	409588	395663	1 yr		
Oriental	Blocked Gully	SL.B-G	409635	395601	1 yr		6 Gas collars & open dip wells & 1 wier plate
Baskerville drift	Naturally re-vegetated, gully	NRv-G	410864	393922	N/A		6 Gas collars & open dip wells

Table 2.1: Bleaklow locality site names, geographical location and time scale of monitoring and installation.

2.2 Goyt valley – Big Moss and Ravenslow (heather management burning and cutting study/ burning in wet versus dry locality studies)

The Goyt Valley, 4 km west of Buxton, in the South West Peak District National Park, is where the headwaters of the river Goyt are sourced, flowing north through steep cloughs where the waters are dammed into the Erwood and Fernilee Reservoirs, after which the waters continue north into the River Goyt and then meander North West to later join the River Etherow. The syncline Goyt valley geology is composed of alternating shale-gritstone layers with bedded coal measures. Successive ice ages shaped and exposed the shales and gritstone. The valley was later carved by the Goyt River (Rice 1957).

At present day, part of the upper Goyt catchment is covered by blanket bog, specifically on the eastern flank of the north-south trending valley; including a summit at ~500 m asl to ~350 m asl (Barnatt and Smith 2004). United Utilities (UU) own and manage the dry areas of Big Moor (referred to as Big Moss in this thesis) and the wet locality (Ravenslow) through an onsite game keeper (Figure 2.3). The Goyt valley received a mean 1026 mm of annual rainfall during the study period, based on data from the Sustainable Catchment Monitoring Program (SCaMP) (2008-2012) collected within the Goyt Valley at locations adjacent to those used in this study (Anderson 2010). Measurements at Big Moss and Ravenslow give a mean annual rainfall of 1042.22mm (measured 2012-2011).

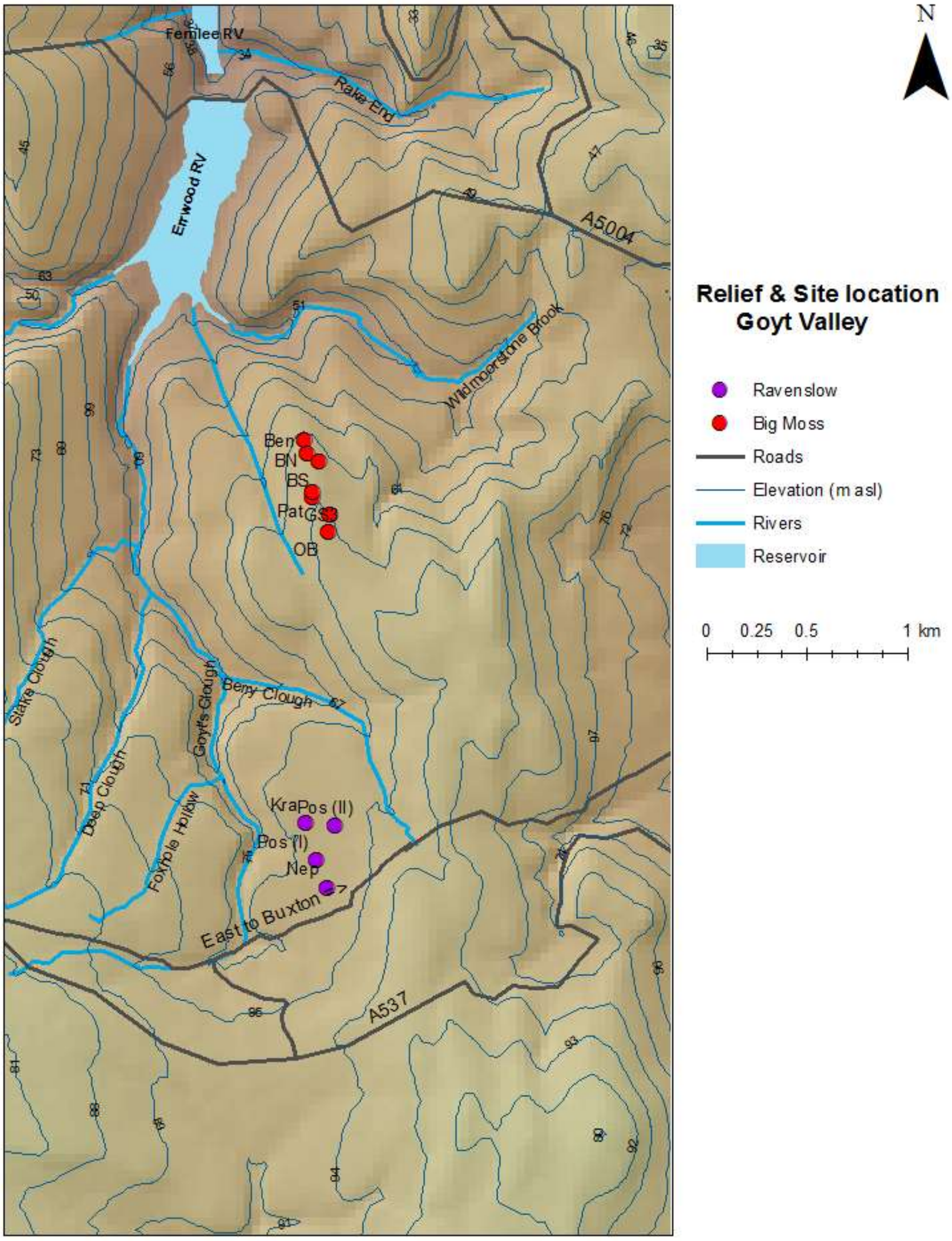


Figure 2.3: Location of study sites on the Goyt Valley. Ravenslow (in purple) is the wet site; Big Moss (in red) is the dry site. The site names were abbreviated: C. vulgaris controls (Kra and Pat) old burn (Nep, BS and BN), new burns (Pos and OB), cut and leave old (GS1), cut and scatter new (GS3 and Ben). Four rainfall gauges were installed and monitored between April 2012 and June 2013 (present at: Kra, Nep, OB and Ben (For explanation of site codes see (Table 2.2)).

Rotational controlled burning of *C. Vulgaris* was used to create a mosaic of multi aged *C. vulgaris* stands suitable for red grouse habitat (Figure 2.4). Cutting and flailing is however becoming the more common practice; as the site is designated a Site of Special Scientific interest (SSSI) onsite burns are done with the approval of Natural England (Barnatt and Leach 1997).

Figure 2.4: Mosaic of managed heather - Goyt Valley 2013.

The sites were chosen for their particular management; each treatment site had duplicate sets of triplicate plots. Within in each plot there was equipment to sample soil pore water and surface runoff water. The treatments were at the least duplicated between the two localities (Dry locality: 2 cuts, 2 burns, 1 control; Wet locality: 2 burns, 1 control) within the Goyt Valley (Table 2.2). The first locality was on Big Moss, relatively dry heath. The sites were selected on a broad fat interfluve as a priority. The second locality was chosen on relatively wet heath, in a topographic depression which could even be considered a blanket mire, due to the presence of *Sphagnum spp.* (Clymo 1987). The inclusion of wet and dry heath localities allowed the study to not only replicate treatments, but also consider the effect of a greater range of water table depths on water quality. Both sites and the treatment plots were selected on areas of deep peat, i.e. peat of greater than 500 mm deep according to Avery (1980).

The sites burnt at the outset of the experiment were burnt in April 2008, although this was after the season permitted in the DEFRA Burning Code (2007). The burns were permitted by license of Natural England for our research. Two fresh or new burns were conducted upon the study site. The burning was conducted by local estate staff trained and experienced in conducting managed burns of *C. vulgaris*. The treatments were then instrumented immediately after the burns, and allowed to settle such that sampling could begin in the following month (May 2008). Subsequently sampling took place every month until June 2013. There had been no managed burning within the catchment for at least 5 years prior to the start of the study; although an accidental burn occurred in the valley in April 2007. The size of all sites, including those subjected to vegetation cutting but excluding those designated as controls, was consistent with the typical size of prescribed burn plots as set out within the Defra burning code (2007), i.e. the burn area could not be more than 150m long by 30m wide monitoring regime.

Locality	Original site names	Treatment	Site abbreviation	Easting	Northing	Time since treatment, prior installation	2008 May equipment installed A	2008 June equipment installed A	2012 April equipment installed B	No. of 10DDs installed	2012 April equipment installed C	2012 May equipment installed C
Goyt valley - Big Moss (dry)	Goodship 1	Cut & lift – New	C.L-New	402140	373707	< 1 yr	6 Gas collars, open dip wells and runoff traps		10DDs	4		
	Goodship 3	Cut & leave – Old	C.L-New	402052	373795	1 yr				2		
	Bendigo	Cut & leave – Old	C.L-New	402013	374076	1 yr		6 Gas collars, open dip wells and runoff traps		4	1 Rain water sampler	
	Bottle North	Burn – Old	B-Old	402027	374012	1 yr				2		
	Bottle South	Burn – Old	B-Old	402088	373969	1 yr				2		
	Patang	Heather Control	Cont	402052	373818	N/A	6 Gas collars, open dip wells and runoff traps			4		
	Otterbox	Burn – New	B-New	402164	372600	< 1 yr				4		1 Rain water sampler
Goyt valley Ravenslow (wet)	Poseiden	New burn	B-New	402075	371985	1 month	3 Gas collars, open dip wells and			2		
	Poseiden	New burn	B-New	402165	372155	1 month				2		
	Neptune	Old burn	B-Old	402125	371850	1 < age < 5 yrs	6 Gas collars, open dip wells and runoff traps			4	1 Rain water sampler	
	Kraken	Heather Control	Cont	402020	372170	N/A				4		

Table 2.2: Goyt Valley locality site names, geographical location and time scale of monitoring and installation.

2.3 Monitoring and sampling regime

Monthly monitoring and sample collection began on Bleaklow in November 2006 and terminated in February 2011, and the Goyt monitoring extended from May 2008 until June 2013. The original installation was conducted by Rowson (2007) and Dixon (2011) (Refer to table 2.1 for the dates of installation). There are monitoring and data gaps as a result of heavy snowfall preventing access, changeover of site monitoring responsibilities in the late spring and early summer months of 2010, and equipment failures.

2.3.1 Instrumentation and sample collection

Both the Goyt and Bleaklow localities included a range of sites. Each site consisted of a collection of replicate monitoring apparatus (plots); with six plots per site, divided equality into a duplicate set (three nested plots) (30 secs - minutes walking distance). The plots within a set were positioned ~2-3 meters apart (Figure 2.5). Plots were composed of: a gas collar and dip well (at minimum). Additional water sampling traps were installed in the Goyt study.

2.3.1.1 Water table depth and water sampling

Each plot was instrumented with a dipwell (Figure 2.6.a) made of 1m long, 5cm diameter PVC pipes. The pipes had ~0.5 cm diameter holes drilled at ~10cm intervals running down the length of the pipes. The top and base of the pipes were left open. The dipwells were inserted perpendicularly to the ground, leaving ~20cm of pipe above ground. The holes allowed soil pore water passage along a pressure gradient from the saturated peat vertically and horizontally into the dipwell. The dipwells enabled the measurement of the water table depth (WTD) and the collection of a soil pore water sample at WTD. The WTD is the level between the inside of the dipwell and peat soil once the pressure is at equilibrium. A tape meter was used to measure depth the WTD. When the water table was deep a conductivity probe was used as the visibility in the dipwell was poor.

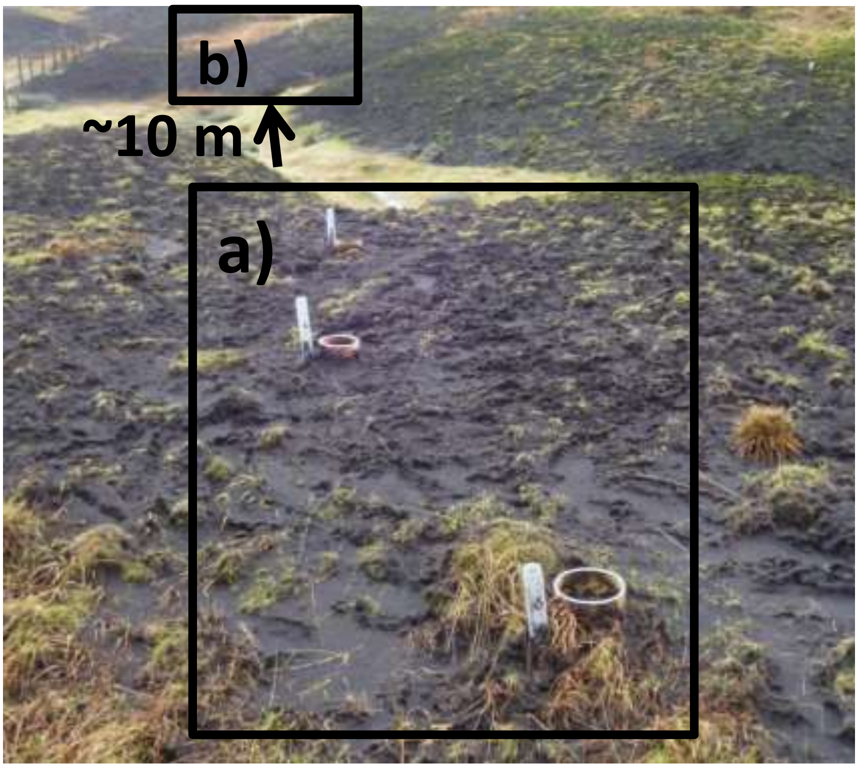


Figure 2.5: Site monitoring plots layout: a) a set/ nest of triplicate plots (half of a study sites set of plots); b) second set (Bleaklow 2011).

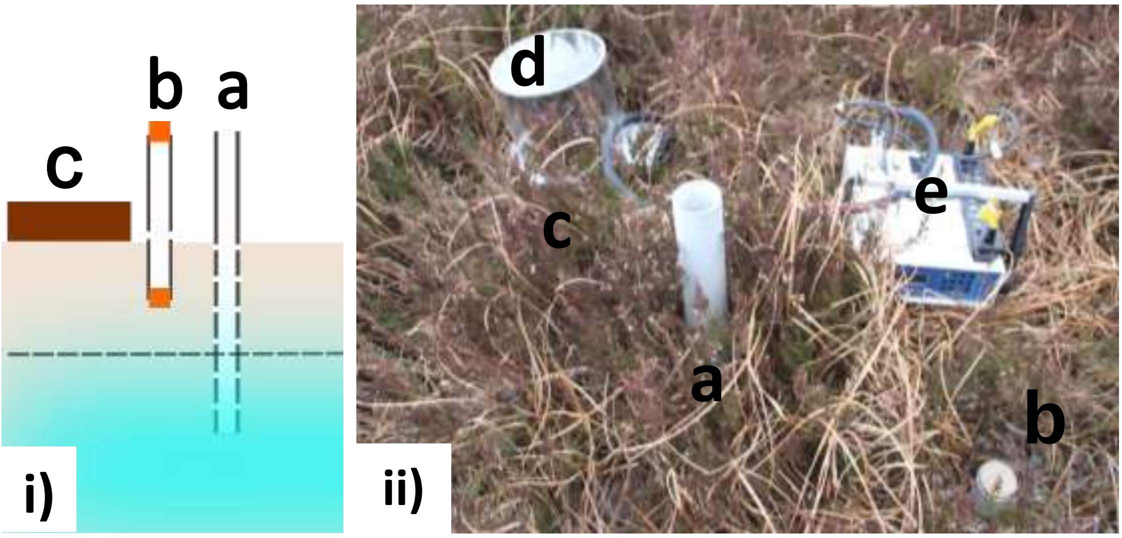


Figure 2.6: i) cross section diagram of sets of equipment diplaying instumentaion relative to the peat surface and the water table. ii) Example an experimental plot. The Lettering on both i) and ii) represent the following: a) dipwell b) runoff trap (present at the Goyt Valley study sites) c) gas collar d) gas chamber e) Infra-red gas analyser (IRGA).

The conductivity probe was comprised of a hollow 1.5 m pole (< 5 cm diameter), with a 1 m tape measure fixed lengthways and a simple open electrical circuit. At the top of the pole is a compartment containing batteries connected to resistors and an LED. The connecting wires are extended through the pole to create anodes at its base. When the conductivity pole is placed into the dipwell the anodes make contact with the soil pore water, the conductive soil pore water allows the completion of the circuit, thus lighting the LED and allowing the detection of the water table, and a reading is noted. The offset (height of dipwell from the soil surface) is deducted from the measurement to give the WTD (Equation 2.1).

$$WTD = Inside\ dipwell\ WT - Dipwell\ offset$$

Equation 2.1

A dip-probe was used to collect soil pore water samples from the dipwell after WTD was measured. The same depth is therefore variable as it is determined by the WTD. The sampling probe is composed of a bamboo cane and a 30ml steralin (sealable, polycarbonate tubes) attached at one end. Once collected, samples were poured into, and stored in, 30ml steralin. The samples were labelled with the date of collection, site name, plot number and sample type.

2.3.1.2 Gaseous carbon fluxes: (CO_2)

To measure the effects of treatment on CO_2 gas flux, monitoring must be conducted infield. Measurements can be made in either a steady state (open) or non-steady state (closed) chamber (Kutzbach *et al.* 2007). The closed chamber method is frequently used to measure the net CO_2 exchange between the atmosphere and low saturated canopies typical of peatlands (Nykänen *et al.* 2003, Rowson *et al.* 2013, Sottocornola *et al.* 2007, Worrall *et al.* 2011). This method was chosen for the research as it allows the assessment to be conducted over short time intervals (minutes long) (Kim and Henry 2013), and so it is both time and cost effective. Importantly, the method is simple to operate in remote, difficult to reach areas such as upland peatlands (Kutzbach *et al.* 2007), and enables measurement relative to a range of covariates (e.g. Water table depth and photosynthetically active radiation (PAR).

At each of the plots (six per site) plastic collars were installed (Figure 2.6.C). At the collars, measurements were taken of CO_2 gas concentrations in order to calculate the flux of C to and from peat surface. Measurements were made using a dynamic, closed chamber method, with an infrared gas analyser (IRGA) (EGM-4, PP-Systems, Hitchin, UK) connected to a 20 cm tall by 15 cm diameter acrylic closed chamber (CPY-2, PP-Systems, Hitchin, UK) as per Rowson *et al*. (2010) and Dixon *et al.* (2013). The chamber was placed onto the collar where the IRGA measures the concentrations of CO_2 (in ppm) within the chamber. Over a period of two minutes the IRGA took measurements at intervals of 4 seconds over a period of 2 minutes. A gas flux was then calculated using a linear regression of CO_2 concentration over time (g CO_2 m^{-2} h^{-1}) (Rowson 2007). Gas observations were made prior to any sample collection to minimise the impact the observers' presence on the flux.

Separate readings were taken from each collar, per site, per monthly visit. These include: ecosystem respiration (R_{eco}) (measured in the dark), and a net ecosystem exchange (NEE) (measured in light). Absence of light is simulated by using a close-fitting u-PVC sleeve, which prevents the passage of all PAR into the chamber. The R_{eco} was measured first as it reduced the greenhouse warming effect on the ecosystem within the collar, minimising the impact of the presence of the chamber upon the following reading taken on that collar for NEE. The difference between these two readings (Equation 2.2) was used to derive gross photosynthesis (P_g) as it is not directly measured. By convention a flux into the peat is given a negative sign, i.e. a negative NEE is indicative of a net sink of CO_2.

$$P_g = NEE - R_{eco}$$

Equation 2.2

Using sensor probes inside the chamber, environmental variables of air temperature (K) and PAR (µmol m^{-2} s^{-1}) were recorded concurrently with the measurement of CO_2.

2.3.2 Laboratory Analysis

2.3.2.1 Water quality - Dissolved organic carbon (DOC)

Using DOC as a measure of fluvial carbon losses can be an advantage as it is comparable to many other studies. Using disposable syringe membrane filters, water samples were filtered to 0.45 µm to remove particulates (Evans *et al.* 2005, Roulet and Moore 2006). Filtrate conductivity and pH were measured using electrode methods, and light absorbance at 400, 465 and 665 µm was subsequently measured using a spectrophotometer calibrated with deionised water (DI) blank. DOC concentration was measured colourmetrically following the method of Bartlett and Ross (1988). An oxalic acid ($H_2C_2O_4$) standard was used in a series dilution (at 60, 30, 15, 7.5 ppm), in duplicate, used to create a calibration curve. The standards are run along with blanks and the samples (oxalic acid at 0 ppm). Samples with more than 60 mg C/L of DOC were dilution, as the test is most sensitive within the range of the standards.

2.3.2.2 Water quality - Conductivity and pH

The conductivity and pH of the sample 0.45 µm filtrate were measured using electrode probe methods. Conductivity is a measure of the total concentration of ions in solution measured in S/m (Mastrocicco *et al.* 2011) and it is linked directly to the total dissolved solids (TDS). The probes were calibrated using: The conductivity probe with a 12880 µS/cm HI 70030 solution; the pH probe using a pH 7 and 4 stock solutions. The probes were rinsed in DI between samples. Deionised water has a conductivity of ~5.5 µS/m and pH of ~6. The pH probe measures the potential difference of hydrogen ions in the probe (at a known concentration and pH) against the solution outside the electrode (samples unknown pH).

2.3.2.3 Water quality - UV-Vis absorbance

Specific ultraviolet absorbance (SUVA), UV absorbance of a water sample at a given wavelength, is related to DOC concentration and character (Weishaar *et al.* 2003). Sample filtrate absorbance was measured at three wavelengths: 400, 465 and 665 nm. In water samples with low iron (Fe) concentrations such as those associated with peaty environments, the absorbance of samples at different wavelengths can provide strong correlation to different DOM composition. Absorbance is a useful measure of specific functionalities such as aromatic carbon at 280nm (Chin *et al.* 1997) and 254nm (Weishaar *et al.* 2003). Absorbance at wavelengths 280nm and 254nm were not measured as the data would not be comparable to the existing data in Dixon (2011), which are important for the multiannual analysis on treatment type. Absorbance at 465 nm and 655 nm were used to calculate the E_4:E_6 ratio (absorbance at 465 nm divided by that at 655 nm). The E_4:E_6 ratio was used as an additional measure of DOC quality, as it has a good capability of characterising DOM ratio, and is used as a proxy for the degree of sample humification (Artinger *et al.* 1999, Chen *et al.* 1978, Lassen *et al.* 1994, Thomsen *et al.* 2002). This provides an indication of molecular weight, hydrophobicity (Edzwald *et al.* 1985), the degree of humification (Lassen *et al.* 1994), and aromaticity of DOM (Figure 2.) (Weishaar *et al.* 2003). Generally a higher E_4/E_6 is related to organic molecules with relatively lower molecular weight (MW) fulvic components (Figure 2.), while lower E_4/E_6 is associated with higher MW humic components (Carlsen *et al.* 2000).

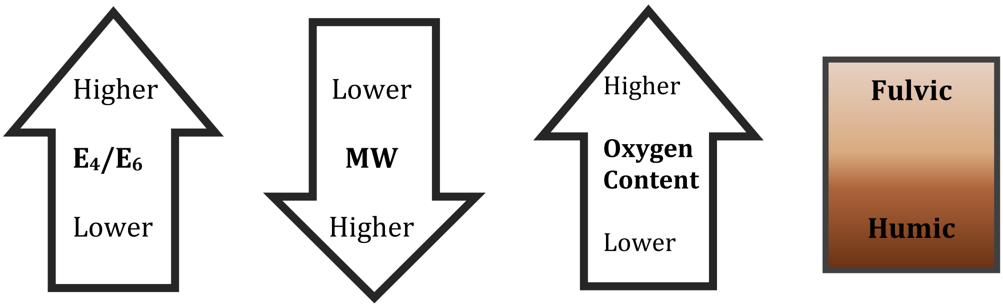


Figure 2.7: How to interpret E_4/E_6 ratios.

Using a UV-Vis spectrometer (Jenway 6505) was used to measures water sample UV-vis absorbance; filtered water samples were placed in clear plastic curvette (Kartel 4.5 ml Micro-Curvette) from which readings were taken. The same cuvette was used for each set of samples in order to minimise measurement errors. To correct for drift on the machine and to reduce the possibility of sample residue affecting subsequent sample measurements, a blank of DI was used to calibrate the spectrophotometer. The blank was run at the start of the series of measurements and subsequently after every 10 samples. The cuvette was rinsed with DI (three times) prior to calibration using the blank.

2.4 Data analysis

Qualitative analysis was conducted using plots and graphs of raw and relative data (relative to the controls). Although this is a valid method of data analysis this thesis sets a higher reliance on statistical analysis, as such methods are more robust and rely on testing the data for the probability that observations are random.

Each of the experimental chapters (Chapters 3 - 6) has a detailed section on the statistical analysis and experimental design. In this section the general methods of statistical analysis used in this thesis are outlined. The data analysis was conducted in the follow way: the data were compiled into an Excel spreadsheet database; it was then processed and quality checked. All statistical analysis was conducted using Microsoft Excel and Minitab (A statistical software package).

2.4.1 Common statistical methods

There are several types of error in statistics. These errors must be addressed in order to produce valid results and accept or reject the H_0. A type I error occurs if the H_0 is rejected when it is true. A type II error occurs when the H_0 is not rejected when in fact the alternate hypothesis is true. Both type I and II errors can occur due to systematic errors, inherent in invalid procedures (Hurlbert 1984). This potential error was addressed by using experiments and observations with factorial design. The factorial experimental design was employed in this research, whereby the levels of one factor (e.g. site or treatment) were sampled within all the levels of the others factor(s) present (e.g. month). The factors are said to be cross classified when the sites are fully factorial. Limitations can disallow for a fully factorial design, this is described in each experimental chapter.

Prior to analysis, all data were checked for normality using the Anderson Darling Test, to minimise bias, as a result of outliers and to allow the use of parametric statistical tests (which assume normal data distribution). Analysis of variance (ANOVA) is robust against the assumption of normality with large data sets (>100 samples). Normalisation, if required, was achieved through the removal of outliers (usually outside of 3 or 4 sigma variation) and data transformation. In the case of WTD data as >90 mm or >80 mm depth represent cases in which the WTDs were too deep to detect. These data points were removed during the quality check phase of analysis of statistical analysis. In cases where soil pore water samples are collected at great depth, there is higher potential of sampling sludge in place of soil pore water. Such samples could result in a greater number of DOC data outliers. The removal of such outlier data points from a dataset is conducted during analysis as they are not representative of soil pore water DOC.

Data were logarithmically transformed when variance of a sample (of count data) was larger than the mean (i.e. DOC and WTD). Square root transformation was used where the variance of a sample was more or less equal to the mean. This was done for DOC data, but not with the data which had both negative and positive values (e.g. WTD and NEE), or with pH which is already a logged value. Logging of data also removed zero values. In cases where the data was still not normal post removal of outliers and dataset transformation, the transformed dataset with the lowest Anderson-Darling normality statistic was chosen. This was done for datasets as a whole in addition to individual sites.

Parametric methods make stricter assumptions and are considered more robust for larger data sets than non-parametric tests which may not detect differences as they compare medians (Fowler *et al.* 1998). They avoid those assumptions of the classical model that are overly inconsistent with the nature of the dependent variable (Akritas and Brunner 2003).

Thus, analysis of variation of means was conducted using ANOVA and analysis of covariance (ANCOVA). The homogeneity of variance (how evenly distributed the error is between factor levels) was tested using Levene's test. Outlier removal and transformation procedures were conducted if the Levene test failed.

The data were multivariate. As such they were investigated using analysis of variance (ANOVA) and General Linear Model Analysis of Covariance (GLM-ANCOVA) to determine the statistical significance of independent factors. Individual factors (e.g. site, month, year), factor interactions (treatment with month and year) and covariates (WTD, air temperature, PAR, P_g, R_{eco}, NEE) were used to create and test models for the dependent variables (i.e. P_g, R_{eco}, NEE and DOC). The models were built through forward selection and bidirectional elimination of variables, one at a time, to find the best combination. To be accepted within the model, the variables required a minimum benchmark for statistical significance (i.e. $p \leq 0.05$). The best fit model was determined by finding the combination of significant predictors that yielded the greatest coefficient of R^2 (determining the predictable portion of variation). Adjusted R^2 (adj R^2) accounted for addition of independent variables. The GLM tests the relationship between one dependent variable and several independent variables. The GLM explains data model variation as: data = model + error. The model is the dataset variation, attributable to the factors (input being analysed; error is dataset variation unaccounted for by the model, otherwise known as residual error). The most physically interpretable, parsimonious models with the highest R^2 were used. Residual analysis was carried out on the dataset residuals post ANOVA and ANCOVA. The magnitude of the effects of each significant factor and interaction, were calculated using ω^2, Equation 2.3 (Olejnik and Algina 2003). Here Seq SS_a = the sequential sum of squares for a given factor or covariate; df_a = the degrees of freedom from the given factor or covariate; $AdjMS_{error}$ = the mean square error; and Seq SS_{tot} = the sequential sum of squares total.

$$\omega^2 = \frac{(Seq\,SS_a \; - \; df_a \times \; AdjMS_{error})}{(Seq\,SS_{tot} + AdjMS_{error})}$$

Equation 2.3

The locations of the significant differences between factor levels were investigated with *post hoc* Tukey's pairwise comparisons. The Tukey test is more sensitive at pinpointing the significant differences between means than other available tests such as Dunnett's test. The results were hence used to establish where the significant differences lay between factor levels (e.g. site/year/month). Dunnett's test operates in a similar way to Tukey's, although it only compares the data against the control group. To compare the magnitude of the main effects within a model, a main effects plot is constructed. It reports the least squares means which are the means of factor levels (i.e. site, month, year), adjusted for the role of all other factors and covariates included in the model.

Analysis using ANOVA/ ANCOVA was primarily conducted using absolute data. A common method also used in some of the thesis chapters was, to analyse the data of sites with intervention/treatment relative to their locality control (i.e. Bare and vegetated controls on Bleaklow, unmanaged heather control on Ravenslow and Big Moss). These relative data sets were analysed using ANOVA/ ANCOVA, in order to identify which factor influenced the data variation in sites with intervention in relation to the control. Relative data were also plotted by site over time in order to investigate how if site predictors (e.g. WTD, DOC, CO_2) were becoming more or less similar to the locality controls over. Summaries of chapter ANOVA/ ANCOVA outputs are available within table at the end of each chapter analysis section.

3 Bare peat restoration effects on: water table and soil pore water DOC

3.1 Rationale

The most important cause of peatland degradation identified by Natural England (2010) was vegetation management leading to bare peat and its associated erosion and gully incision. Bare peat is void of vegetation and prone to desiccation, aeration, aerobic decomposition and increased rates of erosion. Erosion of blanket peat will tend to localise in areas of bare peat (Evans and Lindsay 2010), thus they are a particularly large source of carbon from peatlands to the environment.

Peatland vegetation cover provides ecosystem services. It stabilises the surface (Rochefort 2000), influences microclimatic conditions and protects against frost heaving (Groeneveld and Rochefort 2005), and increases soil moisture and albedo and reduces rates of soil erosion (Mackay and Tallis 1996). Anthropogenic interventions are used as they can strongly influence upland vegetation dynamics (Reed *et al.* 2009). Mismanagement can impact carbon sequestration. Therefore revegetation strategies must be investigated (Natural England 2010). The research presented in this chapter formed part of two published papers[1,2].

[1] Qassim S. M., Dixon S. D., Rowson J. G., Worrall F., Evans M., Bonn A. 2014. A 5-year study of the impact of peatland revegetation upon DOC concentrations. Journal of Hydrology 519: 3578-3590.

[2] Dixon S. D., Qassim S. M., Rowson J. G., Worrall F., Evans M. G., Allott T. E. H., Boothroyd I. M. 2014. The impact of peatland restoration on CO_2 fluxes and water table depths from a climatically marginal upland blanket bog. Biogeochemistry 118: 159-176.

Management practices are being continuously developed and implemented to manipulate ecosystem function, carbon fluxes and water quality. Studies on the loss of peat vegetation and revegetation have been conducted in relation to: water chemistry (Caporn *et al.* 2007), hydrology (Daniels *et al.* 2008, Holden *et al.* 2011, Price *et al.* 1998), ecology (Lindsay 2010, Rochefort 2000), carbon atmospheric fluxes as CO_2 (Waddington *et al.* 2003), and fluvial flux as POC (Anderson *et al.* 2011, Pawson *et al.* 2008). Studies by Worrall *et al.* (2003a), Evans *et al.* (2005) and Evans and Monteith (2001) have documented an increase in water colour and DOC concentration, within and from blanket peat, over time scales of over two decades. Plausible mechanisms to explain this increasing DOC trend include increased air temperature, changes in soil pH, the amount and nature of water flow, and atmospheric CO_2 (Freeman *et al.* 2004, Worrall *et al.* 2003a, Worrall *et al.* 2003b). All of these factors are affected by land management (Mitchell *et al.* 2008, Wallage and Holden 2010, Worrall *et al.* 2011b). The multiannual, comparative effect of different bare site ecological restoration methods on soil pore water DOC is poorly understood and there is a paucity of research in relation to bare peat revegetation and DOC particularly in the form of multiannual studies, thereby warranting further research.

More recently in April 2003, a wildfire spread across 7 ha (5.5 m^2), burning away the vegetation, litter, and in some places the upper soil surface layers (Moors for the Future, 2010). This event added to the carbon losses associated with the erosion and poor ecological state of Bleaklow (Daniels *et al.* 2008, Evans and Lindsay 2010, Rothwell *et al.* 2007). Peatland vegetation management practices and policy were required to revegetate the bare surface to prevent peat loss through erosion and allow for the potential of a peat forming system. This would also reduce carbon losses from the peatland associated with climate change (Gallego-Sala *et al.* 2010).

Loss of surface peat can occur as a consequence of natural erosion and degradation in addition to peat extraction. The research on peat revegetation has largely focused on the restoration of bare extracted bogs, specifically in Canada (e.g. Lavoie *et al.* 1996 and Rochefort 2000 in southern Quebec, and Strack and Zuback 2013 in Northern Alberta). Due to paucity in research with regards to restoration effect on carbon fluxes, Bleaklow restoration can be compared to the Canadian extracted peat revegetation studies, particularly to the methods used and their success on bare peat restoration of vegetation, hydrology and reduction of carbon losses. Although little is known about the effect on fluvial carbon losses, prior to restoration POC was found by Pawson *et al.* (2008) to be 80% of fluvial exports. Worrall *et al.* (2011c) found losses of POC were reduced on revegetated plots relative to bare soil control plots. Revegetation studies on bare peat with nurse and moorland grasses, and *C. vulgaris* were found not to reduce DOC loss in a two year time scale (Anderson *et al.* 2011, Worrall *et al.* 2011a). The need to investigate the longer term (>2yrs) effects of different revegetation methods on water quality forms the basis of the research in this chapter.

3.2 Aim

The aim of this study was to investigate the effects of re-vegetation on soil pore water DOC concentration and composition. The null hypothesis of this study is that there is no significant difference in soil water DOC concentration and composition between sites with a revegetation treatment and control sites. The hypothesis will be tested through analysis of a multiannual data set of soil water table, soil pore water sample DOC concentration, UV-Vis absorbance, pH, and conductivity. Analysis of seasonal and annual data trends will be conducted using graphs, comparative analysis of the data, ANCOVA GLM Models and post hoc tests to find where differences lie.

3.3 Methodology

3.3.1 Study site and experimental design

Bleaklow is a locality situated within the region of Dark Peak of the Peak District National Park. On Bleaklow, eight research sites (plots per site, in sets of three) were installed in geographical proximity, allowing for similar histories of environmental conditions and pollution deposition. The Eight sites were monitored between November 2006 and January 2012. All plots were composed of a dipwell and gas collar and were visited, monitored (for CO_2 gas fluxes and environmental variable) and sampled (for soil pore water) on a monthly basis between November 2006 and January 2012. The sites used (Table 3.1) in this study are the same as those used in Worrall *et al.* (2011a), Clay *et al.* (2012), Rowson *et al.* (2013), and Dixon *et al.* (2014). In this chapter the sites will be referred to by their names given in Dixon *et al.* (2014). Details on Bleaklows' history, geology and research sites are found in the general methodology (Section 2.1).

Type of site	Site management and vegetation	Gully or Flat?	Site abbreviations (Worrall *et al.* 2011)	Site abbreviations (Dixon *et al.* 2014)	Coordinates	
					Eastings	Northings
Control	Least disturbed, Vegetated	Flat	Ne	LD-F	409054	393154
Control	None - Bare	Gully	Ug	B-G	409402	396384
Control	None - Bare	Flat	Uf	B-F	409359	396549
Treatment	Seeding, liming, heather brash & geojute	Gully	R1	SL.Ge-G	408781	396156
Treatment	Seeding, Liming & heather brash	Gully	R2	SL.HB-G	409309	395778
Treatment	Seeding & liming	Flat	R1	SL-F	409588	395663
Treatment	Seeding, Liming, and blocked Gully	Gully	-	SL.B-G	409635	395601
Control	None - Naturally re-vegetated	Gully	Nv	NRv-G	410864	393922

Table 3.1: Restoration study: revegetation of bare peat, site details. This is detailed in Worrall et al. (2011) and Dixon et al. (2014).

3.3.2 Water sample analysis

As described in the general methodology (Chapter 2), water samples were analysed for DOC, pH, conductivity and absorbance at 400, 465 and 665 nm (abs_{400}, abs_{465} and abs_{665}). Concentrations of DOC were measured colourimetrically using the method of Bartlett and Ross (1988). In addition, the E_4/E_6 ratio (ratio of abs_{465} to abs_{665}) was calculated for all samples. Chen *et al.* (1978) have shown that the E_4/E_6 ratio is mainly governed by the particle size or molecular weight and can be used to measure the relative proportions of fulvic acid to humic acid in the coloured component of the DOC, and also to measure the degree of humification (Thurman 1985). Environmental variables data were obtained from three sources: a) in field measurements during site sampling and monitoring (for air temperature and photosynthetically active radiation (PAR)); b) From MU AWS (for air temperature, irradiance and rainfall volumes); and c) the UK Eutrophying and Acidifying Atmospheric Pollutants (UKEAP) (for monthly rainfall volume and chemistry).

3.3.3 Vegetation quadrat surveys

In addition to DOC and water quality data collected, vegetation quadrat surveys were conducted in July 2009 and August 2011. The survey in 2009 was three years post site installation and seven years post treatment. Seven sites (45 plots) were surveyed for the most dominant species by plot area. A plot area is defined as a 0.5 m^2 area surrounding the dipwell and collar (Figure 3.1.a). The collars were surveyed by visually dividing the collar into four sections (Figure 3.1.b) and recording the dominant cover (plant covering largest surface area) in each section. In 2011 the survey was conducted on the six sites monitored (36 plots) for the five year study (six sites: vegetated control, bare soil gully, bare soil flat, seeded and limed, heather brash, and geojute site). The 2011 survey was performed for every collar and plot area. However, it measured the vegetation cover in terms of percentage area covered in the over and understory.

The vegetation was grouped into functional groups instead of species type, as there was a limitation for two reasons. Firstly the poor numbers of replicate measures, as there were only 6 plots per site and only one site representing each treatment. Secondly the surveys were conducted at different times of year and so there may be seasonal variation between those conducted in 2009 and in 2011. The groups included were: shrub, the woody shrubs species such as *C. vulgaris*, bilberry (*Vaccinium myrtillus*) and crowberry (*Empetrum nigrum)*; sedge, such as cotton grass (*Eriophorum vaginatum* and *Eriophorum angustifolium*); grass, purple moor grass (*Molinia caerulea*); litter, which is fallen dead vegetation; *Sphagnum* Bryophyta such as *Sphagnum palustre*; Moss, that are non-*Sphagnum* Bryophyta; and finally Lichen and liverworts, grouped as 'other'.

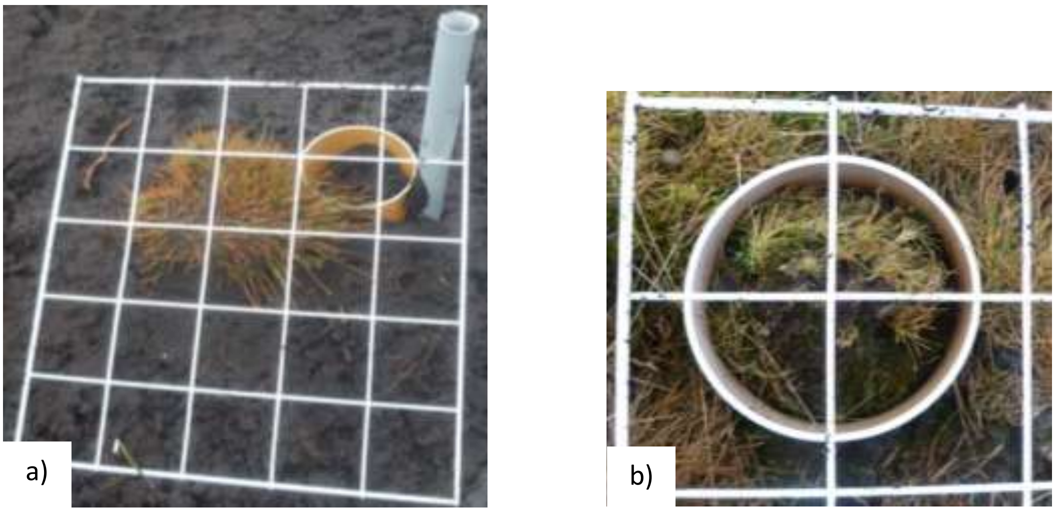


Figure 3.1: Vegetation quadrat survey method: a) Vegetation survey of plot area (0.5m^2) including area closest to collar and dipwell, b) Vegetation survey of collar.

3.3.4 Statistical analytical methods

Statistical analytical methods are presented for weather variables (temperature and rainfall), vegetation cover, and WTD as they are used as covariates in the analysis of DOC and E_4/E_6 ratio. It was conducted in this order to better understand the variables used during the analysis of the covariates of DOC concentrations and $E_4/E_{6.}$ Refer to the General methodology for detail on data analysis (Section 2.4).

3.3.4.1 Weather variables

Using The MU AWS (hourly data), installed adjacent to the naturally revegetated site, gaps in temperature and PAR were filled for periods when it was not possible to measure them at the time of sampling (Section 3.7.1). This was achieved by fitting the measured temperatures against the AWS temperature data. The line of best fit was then used to predict what the unknown temperature at the Bleaklow study sites were, using the known temperatures measured by MU AWS.

3.3.4.2 Vegetation analysis

An Investigation of vegetation plot cover was conducted according to vegetation functional cover type. Some species *bryophytes and species* within the *cladoniaceae* family (lichen) or *marchantiophyta* (liverwort) were not recorded in the 2009 survey and so it was not possible to consider the significance of lichens or liverwort interactions between year and cover type, or site and cover during the multivariate analysis.

In 2009 the study included 46 plots across 7 sites, whilst in 2011 there was an evenly distributed 36 plots across 6 sites. An investigation of vegetation cover was made by sites (9 sites) and survey years (2009 and 2011). The vegetation survey conducted in 2009 recorded data on the vegetation cover in relative (order of dominance) not quantitative (percentage) values. The survey conducted in 2011 reported the data as vegetation functional group percentage cover. The use of the term vegetation level of dominance is used to indicate the relative importance of vegetation cover on a site. The most dominant cover (D1) refers to the functional group that covers ≥ 50% of a plot area. A second functional group occupying <50% plot cover is referred to as D2. As one of the sites (naturally revegetated) had a greater number of plots (9 in place of 6) the vegetation cover was compared between sites as a percentage instead of number of plots. There are four levels of dominance used in the analysis (Table 3.2): first most important (D1); second most important (D2); third most important (D3); and fourth most important (D4). For example a site B-F has 6 plots; of the plots those in plot 5 were completely bare and one plot was 25% covered by vegetation in the sedge functional group, 25% was covered in litter and the remaining 50% was bare.

Site	Year	Family	D1	D2	D3	D4
B-F	2009	Bare	100.0	0.0	0.0	0.0
		Sedge	0.0	0.0	0.0	0.0
		Shrub	0.0	0.0	0.0	0.0
		Grass	0.0	0.0	0.0	0.0
		Other	0.0	0.0	0.0	0.0
		Litter	0.0	0.0	0.0	0.0
		Moss	0.0	0.0	0.0	0.0
	2011	Bare	100.0	0.0	0.0	0.0
		Sedge	0.0	0.0	0.0	0.0
		Shrub	0.0	16.7	0.0	0.0
		Grass	0.0	0.0	0.0	0.0
		Other	0.0	0.0	0.0	0.0
		Litter	0.0	16.7	0.0	0.0
		Moss	0.0	0.0	0.0	0.0

Table 3.2: Percentage of site plots dominated by each cover type (in 2009 and 2011). The percentages refer to 4 dominance levels (D1-D4). The percentage between columns cannot be added together.

The most dominate cover on the site (across all 6 plots) was bare and was therefore reported as D1 = 100% bare. The second most dominant covers on the site (1/ 5 plots) were equally the sedge and the litter functional group, it was therefore reported as D2 = 16.7% sedge and 16.7% sedge.

The data for the 2011 survey was collected qualitatively as percentage cover type. In order to compare the vegetation cover between the years 2009 and 2011, the 2011 data were converted into an order of level of cover dominance by functional group (D1-D4). Using levels of vegetation cover dominances allows easy comparison of the data from both survey years. The base vegetation was not compared between 2009 and 2011 as the base vegetation data were insufficient for analysis.

Changes in vegetation between 2009 and 2011 are discussed in terms of four levels of cover dominance with emphasis on plot cover as opposed to collar cover. The percentage of plots per site was calculated by cover type for each dominance level (D1-D4), as shown in the example (Table 3.2). A one way ANOVA was used to analyse the difference in vegetation dominance between 2009 and 2011. There was a small sample size and some cover types were replicated as they were not represented in every site. Thus, it was not possible to use an ANOVA to analyse the differences between sites and year. Instead direct relationships were drawn between year and bare cover percentage, site and cover type, and dominance of a single cover on Bleaklow. Cover between collars and of plots was also compared using an ANOVA in order to asses if there was a significant difference between plot and collar and assess if the effect of treatment was equal between sites and plots. The end point of the study was in 2011 and vegetation was measured in terms of percentage. The year 2011 was the end point of the study. It is assumed the treatment sites were bare 3 years prior installation (at SL-F, SL.Ge-G and SL.HB-G). Hence, in order to evaluate the progress of the restoration efforts, the difference between the three controls and three treatment sites at the endpoint of the five years study were compared using the percentages cover of the different functional types. The use of an ANOVA to analyse the 2011 bare cover (percentage) quantitative (using six plots as a replicate within site). Post hoc analyses were conducted to find where the differences in bare cover lay on the Bleaklow sites.

3.3.4.3 Water table depth, DOC and E_4/E_6 statistical methods

The sampling survey design implemented in this study represents a factorial approach to the problem of understanding the impact of revegetation on the soil pore water DOC concentrations in peatlands. This study could not directly consider restoration treatment as a factor because it was never repeated between sites, i.e. no combinations of site and treatment were available on the Bleaklow Plateau. Equally, no measurements were made prior to the fire because it could not be foreseen when or where the wildfire would have occurred nor were any measurements made prior to restoration treatment; i.e. any differences identified between sites could be ascribed to pre-existing differences between sites rather than to restoration. The experiment is multifactorial with respect to: year (five levels; 2007-2011), month (twelve levels: January to December), site (eight levels based on the restoration methods used at the site: bare flat, bare gully, naturally revegetated, vegetated control, seeded and limed, geojute, heather brash and gully blocked), and vegetation dominance (seven levels: bare, shrub, sedge, grass, moss, *Sphagnum spp.*, other).

All data were quality checked for normal distribution and outliers (within 3 sigma error) as per the methodology section. Analysis of the data was performed using the four untreated sites (bare controls: bare flat and bare gully; vegetated controls: vegetated control and naturally revegetated) as comparators to the four treated sites. Two of the sites (naturally revegetated and SL.B-G) have a shorter data record. Analysis on all eight sites was therefore only conducted for 2008, when all eight sites had data. A full multiannual analysis was made for the remaining six sites. There are data gaps for two main reasons. Firstly, the lack of monitoring explained earlier (Section 2.2). Secondly, the WTDs were too deep to detect and were therefore recorded as >90 mm or >80 mm depth (often in the case of the bare sites). For the purpose of statistical analysis using ANOVA, these points were removed during the quality check phase of analysis.

Similar processes were conducted for WTD, DOC and E_4/E_6 (Section 2). Data ranges were reported and coefficient of variation (CV) (Equation 3.1) was used to show the data fluctuation (of WTD, DOC and absorbance) by site, as it is the extent of the dispersion of data and variation in relation to the mean: where σ, is standard deviation divided by μ, the mean. The CV is used as an indicator of variables' (WTD, DOC and UV-Vis) fluctuation.

$$C_\upsilon = \frac{\sigma}{\mu} \text{ x } 100$$

Equation 3.1

The quality checked untransformed data were analysed first, after which logged and square root of the data were investigated. The data with the most normal distribution was then used. The data was primarily investigated using General Linear Model Analysis of variance (GLM-ANOVA). As part of ANOVA covariate pre-selection, stepwise regression was used to identify variables that improved the prediction of changes in the response variable (e.g. DOC, E_4/E_6). The variables are included when their p-values are greater than a specified Alpha-to-Enter (0.15), and excluded from the model when their p-values are less than or equal to the Alpha-to-Remove value (0.15). The relationship between a response (WTD/ DOC/ E_4/E_6) and environmental variables (temperature/rainfall) at different time steps were investigated using a stepwise regression. The best model output was then input onto GLM to develop the best models possible. The experiment is cross-classified with respect to month, site and year. The difference between plots within a site can also be considered as a nested factor within the site factor and allowed the study to directly assess whether the variation within site was greater than the variation between sites. The percentage cover measured in 2011 was used as covariates for WTD, DOC and E_4/E_6 as although the data does not vary with time it does vary with site and plot and provide greater plot scale detail into the models.

Sampling month was included as a factor in ANOVA. Month is related to seasonality and is either used as factor of month (in 2008 ANOVA) or nested factor within year (in five year analysis). The levels 1 and 12 of the factor month are not dissimilar as they are temporally consecutive to one another. The use of the factor month nested within year incorporated seasonal variation between years into the GLM.

It was possible to use multiple predictors to investigate the data using GLM: factors (i.e. site, month, year, plot most dominant cover), nested factors (i.e. month nested within year, plots within site), factor interactions (i.e. treatment, gully or flat, and plot cover dominance with month and year), and covariates (i.e. WTD; PAR and air temperature – inside chamber, temperature C° rainfall mm mean at - day of sampling/month/days prior to sampling; P_g; R_{eco}, NEE; rainfall chemistry) on the dependent variables (DOC, UV-Vis and WTD). The models produced were required to meet a minimum benchmark for statistical significance (i.e. p = 0.05). A best fit model was determined by finding the combination of significant predictors that yields the greatest coefficient of determination (R^2). Adjusted adjR^2 (adjR^2 will be referred to as R^2) was used as it is adjusted for the number of explanatory terms in a model. To avoid type I errors, models were required to meet a minimum benchmark for statistical significance (i.e. p = 0.05). Models are discussed and the best model output results are displayed in main effects plots. The magnitude of the effects of each significant factor and interaction were calculated using generalised ω^2 (Olejnik and Algina 2003). Difference between significant factor levels was assessed by post hoc Tukey analysis. Factors can be significant within an ANOVA and have insignificant differences between factor levels in the post hoc analysis. However, while this study does often have unequal sample sizes between factor levels; the Tukey's test is conservative when looking at larger numbers of factor levels. It is also conservative for three, unequal level factors when interpreting the results of ANOVA/ANCOVA models. It is therefore problematic with a greater number of factor levels. The factor levels compared the ANOVA least square mean in brackets along with the standard error of the mean (SE mean). They were presented as normalised least squared mean along with SE mean.

Residuals calculated for each model output were analysed for normality of residual frequency distribution, and the fit of residuals against plotted values. The distribution and variation are hence examined and models were rejected where residuals did not display a normal distribution, or the residual plotted against the fitted values give a pattern other than random. Normal residual distribution was prioritised over a higher R^2.

3.3.4.4 ANCOVA Covariates

To develop models which explain a greater portion of the WTD, DOC and E_4/E_6; covariates identified as significant in stepwise regression were investigated further in the ANCOVA. Regression analysis was used between a response variable (i.e. WTD, DOC and E_4/E_6) and an independent variable (i.e. rainfall, temperature). Water table depth was analysed as a response variable in its own ANCOVA and used as covariates in DOC and E_4/E_6 ANCOVA. The response variables DOC and E_4/E_6 were also used in each other's ANCOVAs. Several weather covariates were included in the WTD, DOC and E_4/E_6 ANCOVA. The covariates considered were measured inside the chamber during sampling, (air temperature and PAR), or obtained from a Manchester University (MU) an infield automated weather station (AWS). Hourly environmental variables have been measured; Defra rainfall data (including rainfall; monthly volumes and rainwater pH, conductivity sulphate and nitrate). Significant covariates were identified and investigated. Analysis of soil pore water pH differences between sites and year was conducted, in addition to the relative difference between soil pore water and rainfall pH by using site month and year plots. A brief summary table of the ANOVA/ANCOVA results for the WTD and soil pore water DOC and E4/E4 ratio are included at the end of the results section.

3.4 Results

3.4.1 Weather

A trend line was used to express the relationship (Equation 3.2) between temperature measured at monitoring sites and the temperatures obtained from the AWS data. The AWS measured solar radiation (as W/m^2) instead of PAR. Although PAR is only a fraction of solar radiation (400 to 700nm), it was possible to create a calibration. The best fit trend line equation for both the temperature (Figure 3.2.a) and PAR (Figure 3.2.b) was used to calculate the Y value with a known X (temperature/or solar radiation value).

$$Y = A + bX$$

Equation 3.2

Where Y = Response variable (i.e. Temperature, PAR), A = intercept and response variable coefficient, and X = independent variable predictor coefficient.

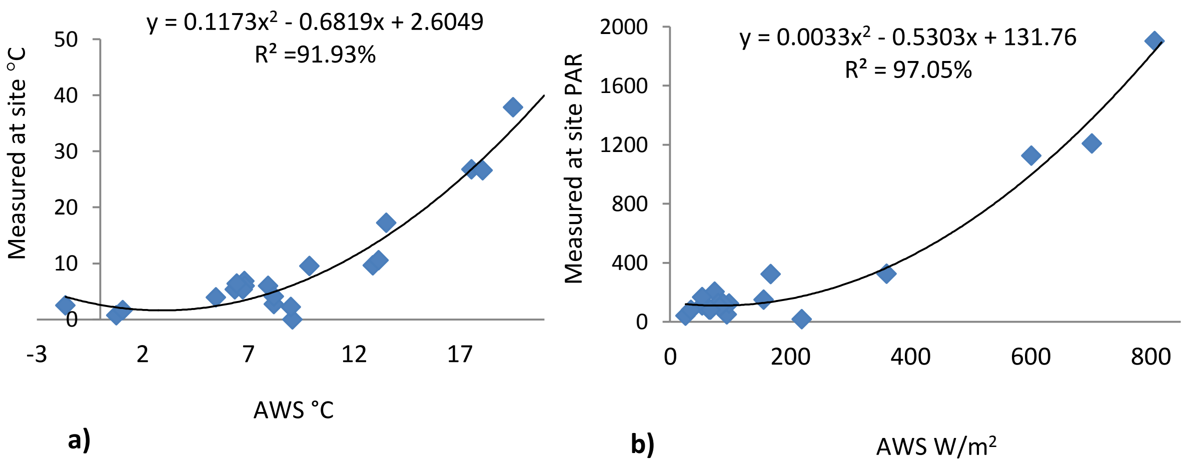


Figure 3.2: Data calibration plots: a) Temperature calibration b) Solar radiation-PAR calibration.

The annual cycles in mean monthly temperature and total monthly rainfall cycle are presented in Figure 3.3. Annual temperatures have a sinusoidal oscillation pattern of variation. The lowest daily mean temperature was -1.77 °C in December 2010 and the highest 13.30 °C in September 2006. Between 2006 and 2012 the annual temperature ranges increased from a variation of 9.89°C in 2007, to 12.39 and 10.52°C in 2009 and 2011. The largest monthly rainfall total was recorded at 230.6 mm in September 2008 and the lowest was 17.9 mm in April 2010. The measured monthly solar radiation maximum between July 2009 and April 2012 was 466.76 W/m^2. The mean monthly solar radiation received was 71.22 W/m^2, with a peak in June (Figure 3.4). The only full year of solar radiation data was available for 2011 with a total 408445 W/m^2.

Between July 2006 – April 2012, the mean annual precipitation on Bleaklow was between 1042 – 1237 mm/annum. Calibration between the Defra rainfall data and the MU AWS data was attempted (Figure 3.2). The calibration was used to calculate predicted rainfall values and compare them to known values. It was found that the model greatly overestimated the higher volumes of rainfall visible during autumn 2011 (Figure 3.5. b). The use of the raw rainfall data was therefore preferential to the calibration. Annual total rainfall (between 2006 and 2012) is presented in Table 3.3 . Over the study period, the greatest mean monthly total rainfall was found during autumn months specifically peaking during October (118.9 ± 18.6 mm) (Figure 3.4). The lowest rainfall was measured during late winter and early spring reaching its average lowest point during month February (57.1 ± 15.2 mm). Annual variation in rainfall occurred between 2006 and 2012. The total rainfall steadily rose from 2006 to 2008 then dropped to its lowest point in 2010 (913.6 ± 14.0 mm), it then sharply rose to its peak of annual rainfall in 2011 (1320.5 ± 17.4mm).

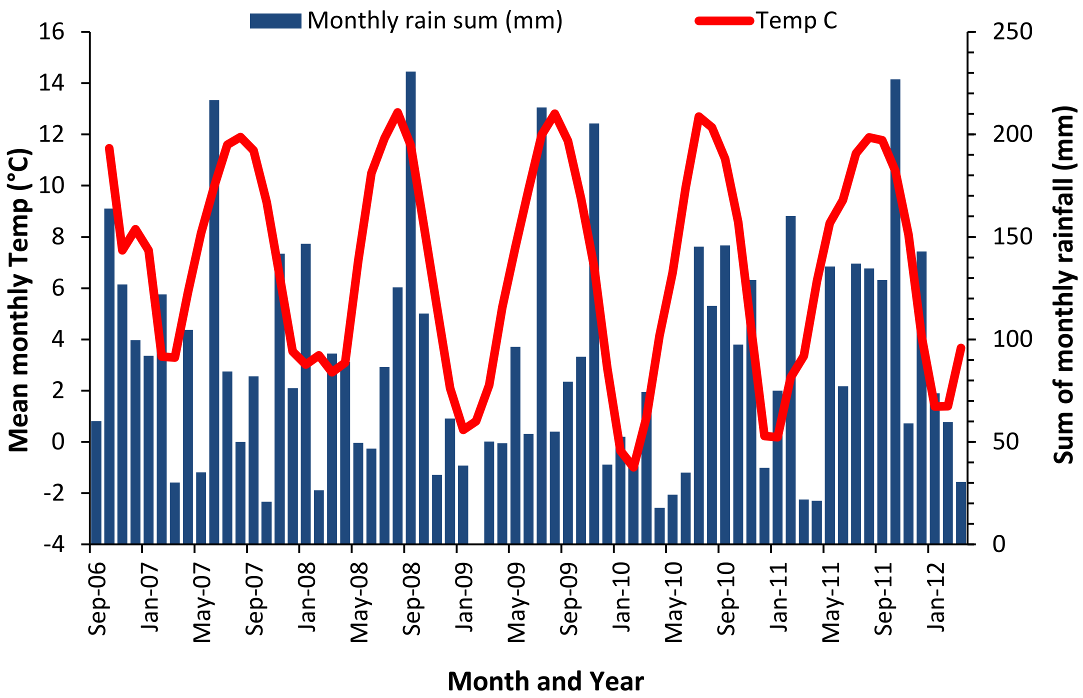


Figure 3.3: Monthly sum rainfall (mm) and mean daily temperature (°C) (2006 – 2012). Precipitation data are derived from both the MU AWS and the Defra UK-AIR data.

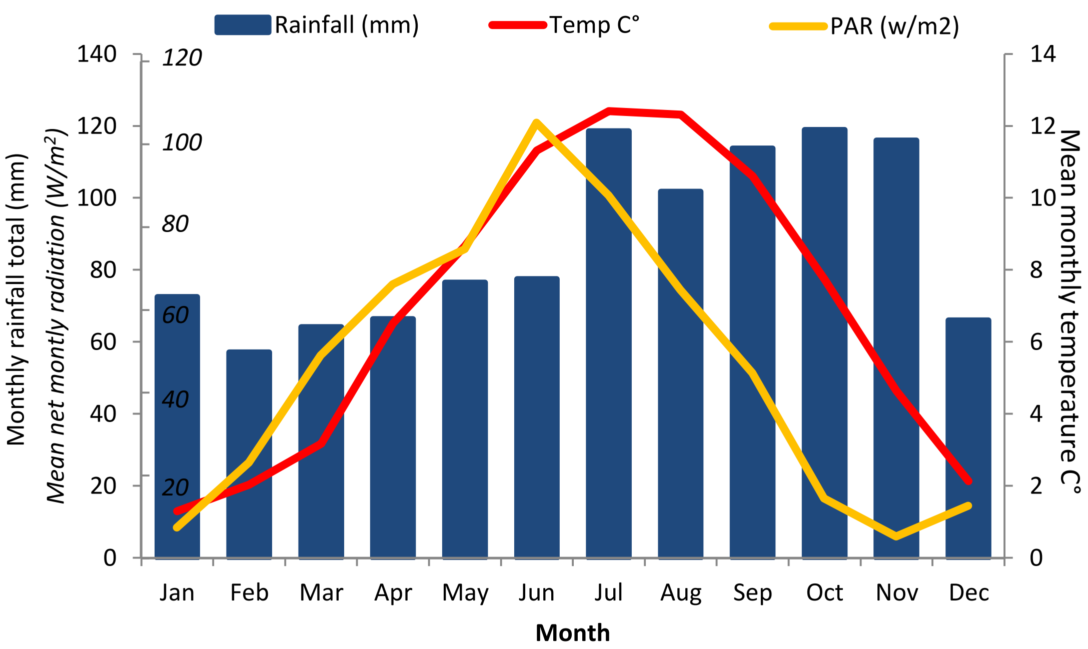
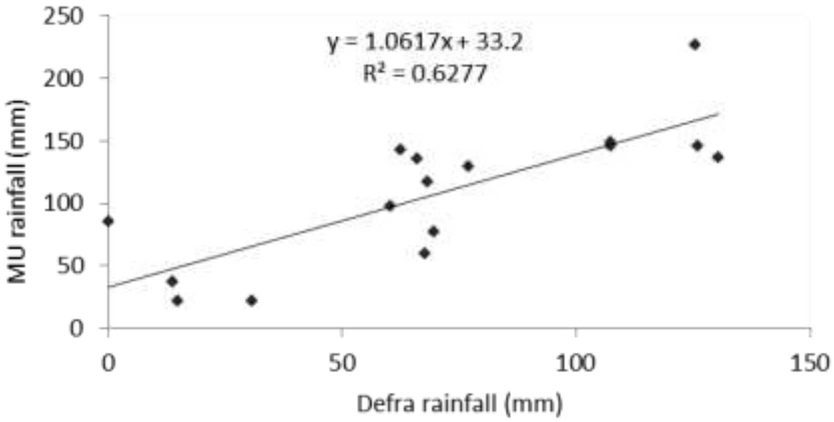


Figure 3.4: Mean monthly sum rainfall (mm) and mean temperature (°C) (2006-2011). Precipitation data is a compilation of both the MU AWS and the Defra UK-AIR data.

Year	Annual Rainfall (mm)
2006	1097.4
2007	1056.2
2008	1102.0
2009	971.7
2010	913.6
2011	1320.5

Table 3.3: Bleaklow annual rainfall (mm) (2006–2011). The final year 2012 sum is not included in the table as the data record was until April 2012.

a)



b)

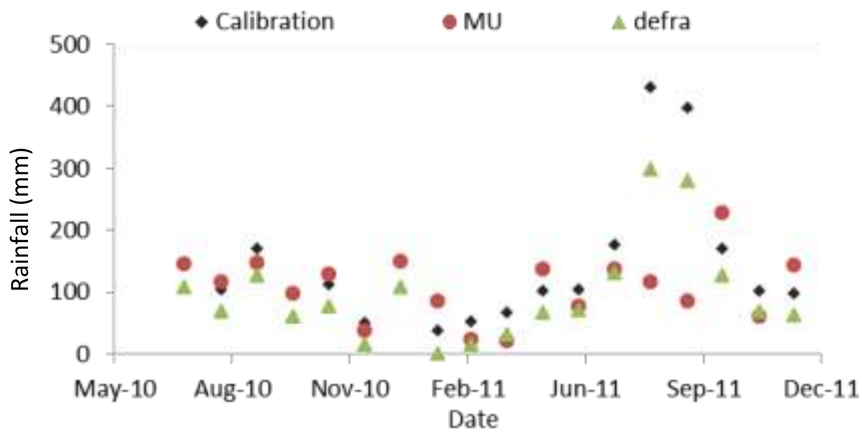


Figure 3.5: Rainfall data calibration a) Calibration of defra data to Bleaklow MU AWS. MU AWS Monthly rainfall is along the y-axis and Defra rainfall is along the x-axis.; b) monthly rainfall sum Defra data (green triangle), MU AWS data (red circle) (Jul 2010 – Dec 2011) and calibrated rainfall values using calibration in graph 3.5a (black diamonds).

3.4.2 Vegetation

In 2009 (Figure 3.6) sites where treatment was conducted appear to have multiple dominant cover types, with vegetation recorded in multiple dominance levels (D1 - D4), more than vegetation types and dominance levels than at the untreated bare soil controls (Bare flat and bare gully). There were also fewer dominant cover types in 2009 than 2011. This suggests a potential increase in biodiversity from year 2009 to 2011. No *Sphagnum spp.* was recorded at any of the six site collars or plots (treated or untreated) monitored for the period of 2006-2011. However, in 2009 the naturally revegetated site was the only site in which *Sphagnum spp.* was observed (Figure 3.6); the naturally revegetated site was not surveyed in 2011, hence no comment can be made on the progress of *Sphagnum spp.* on the vegetated control.

Analysis of vegetation cover by most dominant cover D1 on Bleaklow, using a one way ANOVA, found that year was not a significant factor ($p = 0.335$, $R^2 = 0.0001\%$). Therefore the dominance of any one cover type at a site did not significantly change between 2009 and 2011. The most dominant cover in 2009 remained most dominant in 2011.

The dominance of a cover type varied significantly between sites ($p = 0.041$, $R^2 = 40.59\%$), hence some sites could have more dominant vegetation types than others. Bleaklow sites as a whole had significantly different function group dominance ($p < 0.0001$, $R^2 = 27.83\%$). Post hoc analysis revealed bare soil cover was the most dominant cover type, whilst litter and moss and other cover types were less dominant. On Bleaklow in 2009 and 2011 respectively, bare soil was most dominant cover at 53% of site plots (36 total plots). The second most dominant cover was sedge functional group (25% and 31%), followed by the grass functional group (22% and 17%).

An analysis of percentage plot cover by functional group in 2011, using one way ANOVA, found significant differences between functional groups percentage cover on Bleaklow ($p < 0.0001$; $R^2 = 24.43\%$). Post hoc analysis revealed vegetation importance was divided into three groups, the first was bare cover (47.67% ± 7.79), which covered the greatest percentage of the sites on Bleaklow in 2011. Sedge (26.33% ± 6.47), grass (13.22% ± 4.71) and moss (9.67% ± 3.76) covered significantly lower surface area than bare, however the significance of grass and moss overlapped with the group with least cover. This group included shrub (3.11% ± 1.75), litter (1.33% ± 0.60) and other (0.22% ± 0.16).

Analysis of 2011 by bare soil plot coverage percentage (the highest cover on Bleaklow), using a one way ANOVA, showed site was a significant factor ($p < 0.0001$; $R^2 = 68.61\%$). Post Hoc analysis revealed that the differences between sites could be divided into three groups; according to their percentage of bare cover. The significantly highest bare soil cover was at the bare soil controls, bare gully (98.67% ± 1.33) and bare flat (96.67% ± 3.33), and the seeded and limed site (56.00% ± 18.4). There was no significant difference between the seeded and limed and the heather brash sites (26.00% ± 17.1). The heather brash site had lower bare cover than the bare controls and was not significantly different to the sites with significantly lowest bare soil cover which are the geojute (7.33% ± 6.57) and least disturbed vegetated control (1.33% ± 0.48).

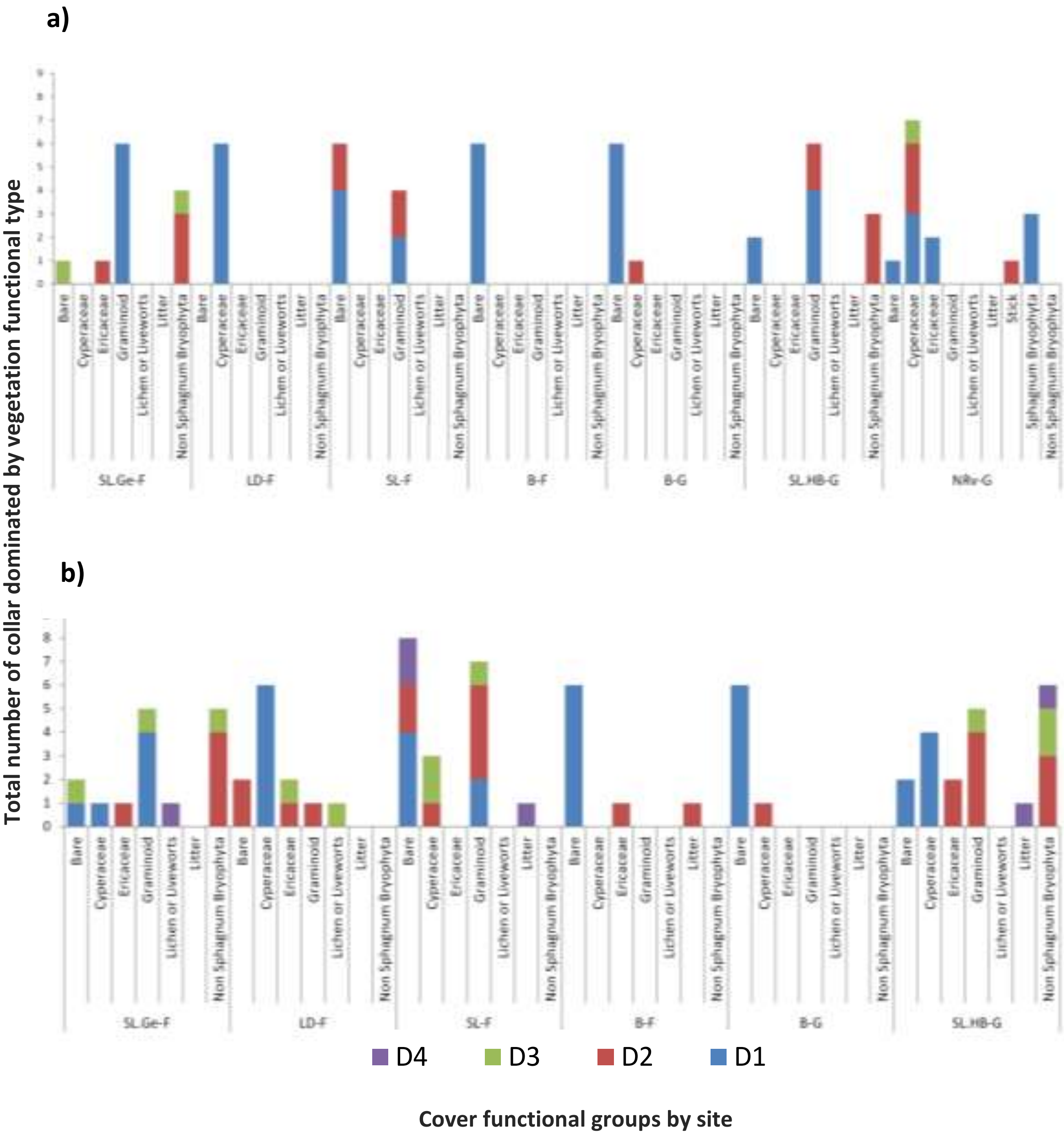


Figure 3.6: Bar chart of total plots vegetation dominance, by site and functional group (Bare, sedge, shrub, grass, other, Litter, Non Sphagnum moss and Sphagnum spp.) along the y-axis a) In 2009 and b) In 2011. The bars display present plot vegetation cover, in terms of 4 levels dominance levels, most dominant (D1) to least dominant (D4). The vegetation dominance at each site is not an accumulative value. Note: the only site with Sphagnum spp. was NRv-G. Refer to Table 3.1 for detail on site names.

3.4.3 Water table depth

3.4.3.1 Eight site comparison (2008) relationship between WTD and site

Statistical analysis of WTD in 2008 using ANOVA outputs indicate significant differences between sites ($p < 0.0001$; $\omega^2 = 47.6\%$) and months ($p < 0.0001$; $\omega^2 = 9.8\%$) and produced a model with $R^2 = 56.02\%$. Post hoc analysis revealed that the bare restored (without gully block) sites had much deeper water tables than the vegetated control, naturally revegetated and gully blocked sites. The deepest mean WTD position was observed in the bare flat site (-432.8 ± 18.4 mm) which was 236.2 mm greater than the blocked site (-196.6 ± 19.8 mm) and 371.7 mm greater than the vegetated control (-61.1 ± 9.2 mm).

The inclusion of covariates into the model initially showed that rainfall was a significant covariate within the ANCOVA, however when the other factors (month, site) and covariates (PAR and percentage of cover by one type) were included rainfall was relatively unimportant in explaining the WTD data variation and was dropped out of the model. The covariates slightly improved the R^2 from 56.02% to 56.14%. ANCOVA analysis found that site was the most important factor ($p < 0.0001$; $\omega^2 = 46.21\%$) but the importance of variation between the factor month was reduced ($p < 0.0001$; $\omega^2 = 5.59\%$). Variation in WTD was positively correlated to PAR ($p = 0.001$; $\omega^2 = 4.28\%$). Post hoc (Figure 3.7) analysis indicated the deepest WTD was at the bare flat control site (-434.6 ± 17.76 mm), which was ~9 times deeper than the vegetated control LD-F. The bare soil controls, bare flat and bare gully (-373.0 ± 17.77 mm) did not significantly differ. The heather brash site (-324.9.0 ± 19.49 mm) had shallower WTD than both the bar soil controls. The shallowest water table was at the least disturbed vegetated control (-48.6 ± 17.4 mm) followed by the naturally revegetated site (-156.2 ± 16.62 mm), which did not significantly differ to gully blocked site (-197.3 ± 17.86 mm).

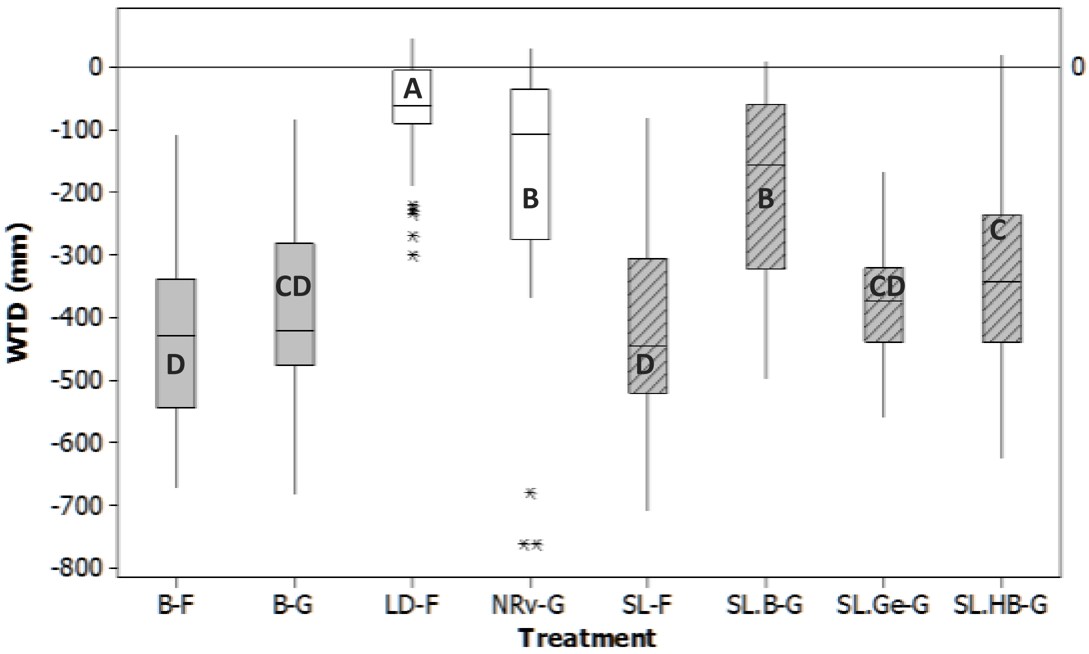


Figure 3.7: Box plot of WTD by site (2008). Letters on the plot are post hoc results of an ANCOVA (Factor inputs include: month, site and PAR). Shared letters indicate significant differences between site factor levels. The zero line indicates the peat soil surface. There are three box plot colours: grey, bare soil controls; white, vegetated controls; and grey striped, site with restoration treatment. Refer to Table 3.1 for detail on site names and treatment types.

3.4.3.2 Eight site comparison (2008) relationship between WTD and vegetation

In an ANOVA investigating the link between WTD and vegetation cover, the most dominant plot cover was the most important factor ($p < 0.0001$, $\omega^2 = 54.52\%$), followed by monthly variation ($p = 0.001$, $\omega^2 = 9.48\%$). Distribution of residuals was normal. Post hoc analysis revealed *Sphagnum spp.* dominated plots had the shallowest WTD (-51.8 ± 21.74 mm) which did not significantly differ to sedge plots (-79.5 ± 13.39 mm). Shrub dominated plots (-155.6 ± 26.37 mm) had significantly deeper water depth than sedge dominated plots. The deepest WTD was at bare soil plots (-415.4 ± 9.64 mm) and grass dominated plots (-330.1 ± 13.39 mm).

The addition of covariates in an ANCOVA did not greatly improve the R^2 value (from 64.06 to 64.59%). The most dominant plot cover remained the most important factor in explaining WTD variation ($p < 0.0001$, $\omega^2 = 55.29\%$), while monthly variation importance was reduced ($p < 0.0001$, $\omega^2 = 5.90\%$). The covariate PAR explained a small portion of the variation ($p = 0.006$, $\omega^2 = 3.33\%$). Residuals were normal. Post hoc analysis revealed (Figure 3.8) that, with the addition of PAR, water table depth at plots dominated by *Sphagnum spp.* (-55.3 ± 22.25 mm) did not significantly differ to sedge plots (-68.3 ± 14.26 mm); both cover types had significantly shallowest WTD. Shrub dominated plots (-155.2 ± 28.43 mm) had deeper WTD followed by grass dominated plots (-332.1 ± 12.49 mm). Bare plots had the significantly deepest WTD (-416.0 ± 9.90 mm) (Figure 3.10). WTD was ~2-2.5 times deeper at bare dominated plots than at *Sphagnum spp.* and sedge dominated plots.

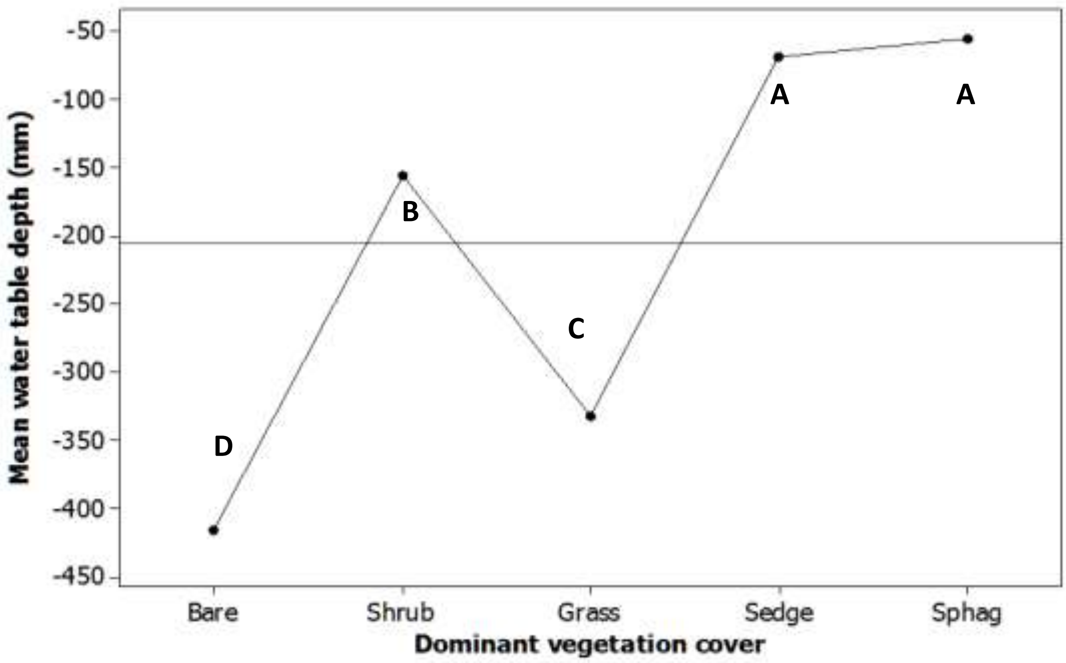


Figure 3.8: Main effects plot of WTD (2008) by primary plot cover dominance (2009). ANCOVA (Factor inputs include, month, most dominant site cover and PAR). Letters on the plot are tukey post hoc results. Shared letter indicate no significant different between levels.

3.4.3.3 Six site comparison (2007 – 2011) relationship between WTD and site

Between 2007 and 2011, the site with highest WTD fluctuation in terms of coefficient of variation was the vegetated control (140.28 mm), followed by SL.HB-G (47.70 mm); both had higher CV than the 4 other sites. The sites with the smallest CV were SL.Ge-G (30.56 mm) and B-F (38.97 mm). The sites dominated by bare soil cover (bare flat and bare gully) and seeded and limed site had an increasing trend in CV from year 2009 to 2011. This indicates that bare soil sites had the highest, annually increasing, water table fluctuation.

Analysis of the WTD over the five year study, using ANOVA, showed site was the most important factor ($p < 0.0001$, ω^2 = 44.48%). Months nested within years ($p < 0.0001$, ω^2 =

9.59%) was of similar importance in the model to plot nested within site ($p < 0.0001$, ω^2 = 9.28%). Inter annual variation of WTD ($p < 0.0001$, ω^2 = 0.26%) was significant but was of minor importance within the model. Residuals were normal. Post hoc analysis indicated that over the five year dataset, the bare flat (-311.4 ± 7.27 mm) and bare gully site (-310.3 ± 7.00 mm) had the deepest mean WTD, which were on average ~13 times deeper than the vegetated control (25.8 ± 6.0 mm), the site with the shallowest WTD. Water table depth at SL.Ge-G (-270.8 ± 11.58 mm) did not significantly differ to the bare soil controls nor the treated sites with significantly shallower WTD than the bare site, which were the seeded and limed (-277.3 ± 6.67 mm) and the heather brash site (-259.5 ± 6.69 mm).

A stepwise regression used to highlight the importance of temperature found no clear linear pattern of decreasing or increasing significance or with daily temperatures leading up to the day in which WTD was measured. The output with the greatest variation explained, included the monthly total rainfall ($p = 0.048$), total rainfall on day of WTD measurement ($p = 0.007$), total rainfall on day prior of WTD measurement ($p = 0.007$), total rainfall two days prior WTD measurement ($p = 0.002$), and temperature of day two prior WTD measurement ($p = 0.013$).

Regression analysis of water table to rainfall, between years 2007 and 2011, showed a significant positive correlation (coefficient value 0.46) with monthly rainfall ($p < 0.0001$) and the correlation varied by site ($p < 0.0001$). It was found that WTD was negatively correlated with rainfall (Pearson correlation -0.125; P-Value = 0.000), in which increase in rainfall results in decreased (shallower) water table. Residual distribution was normal. The regression plot in Figure 3.9 demonstrates clearly that LD-F has a shallower water table and different hydrology to the remaining five sites (two bare and three sites in restoration previously bare).

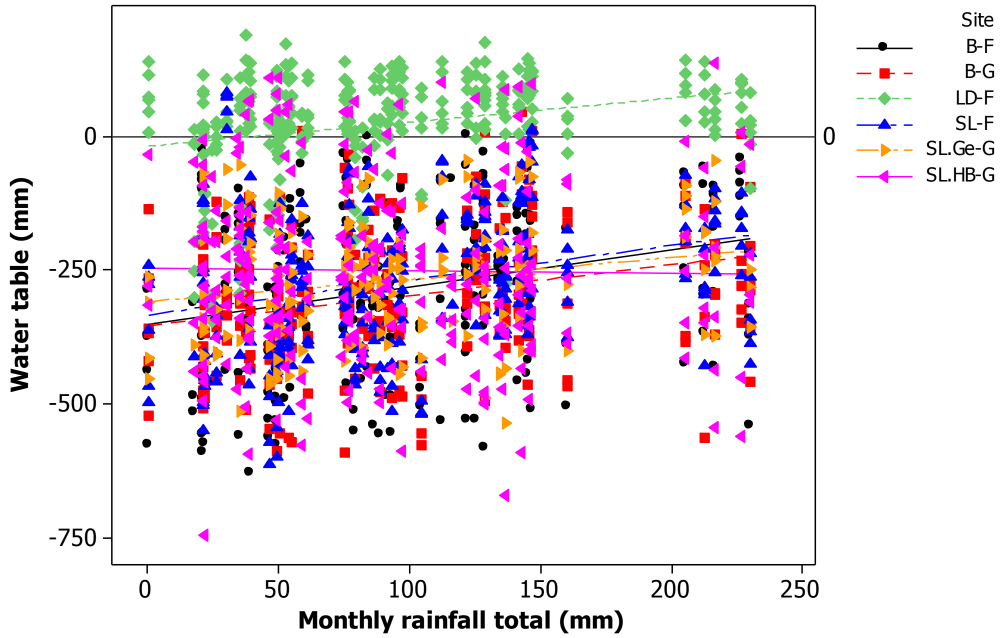


Figure 3.9: A regression scatter plot of monthly rainfall sum (mm) as a predictor of site water table.

Significant variables identified in the stepwise regression were included in an ANCOVA. Rainfall volume was not a significant covariate, as other covariates were more important and increased the R^2 from 63.64% slightly to 65.06%. The full five-year WTD ANCOVA found that inter site variation remained the most important factor ($p < 0.0001$; $\omega^2 = 42.89\%$), followed by variation of plot nesting within site ($p < 0.0001$; $\omega^2 = 9.86\%$). The importance of month nested within year was reduced ($p < 0.0001$; $\omega^2 = 7.42\%$). Inter-annual variation was the least important factor ($p < 0.0001$; $\omega^2 = 7.42\%$). Specifically WTD was positively correlated to PAR ($p < 0.0001$; $\omega^2 = 5.77\%$) and air temperature was a significant factor, although of little to no importance in the ANCOVA ($p < 0.0001$; $\omega^2 < 0.0001\%$). Post hoc analysis of the full five year analysis revealed that the bare gully (-310.8 ± 7.47 mm) and bare flat (-309.4 ± 7.63 mm) had the deepest WTD, with ~10 times deeper than vegetated control (35.9 ± 75.6 mm). The three restored sites: seeded and limed (-274.7 ± 7.66 mm), geojute (-270.8 ± 11.58 mm), heather brash (-264.9 ± 7.75 mm) had significantly shallower WTD than the bare sites. The restoration sites had ~8 - 9 times deeper WTD than the vegetated control. Water table depth in 2011 was significantly deeper than in 2007. Years 2008 to 2010 did not significantly differ (Figure 3.10).

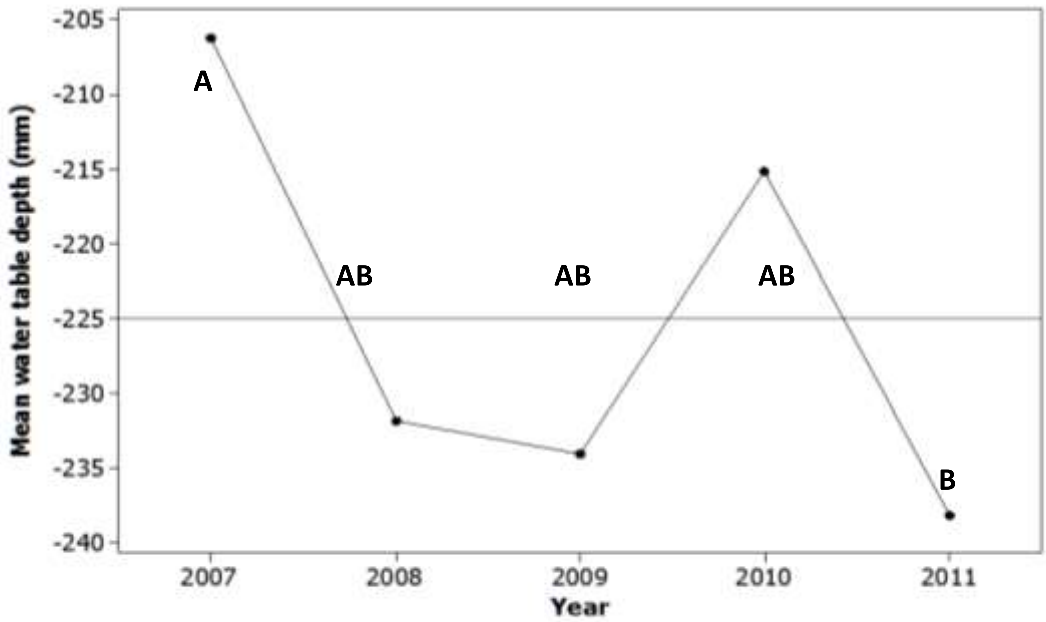


Figure 3.10: Water table depth ANCOVA main effects plot (2007 – 2011). Model inputs include: sites, plot nested within site, year, month nested within year. Covariates include: AT and PAR.

3.4.3.4 Analysis of WTD relationship to vegetation (2007 – 2011)

An ANOVA of the five year study found that the dominant vegetation was an important predictor for water table ($p < 0.0001$; $\omega^2 = 30.45\%$), followed by month nested within year variation ($p < 0.0001$; $\omega^2 = 8.44\%$). Annual variation was relatively unimportant ($p < 0.0001$; $\omega^2 = 0.20\%$), although there was a small interaction between dominant vegetation WTD and year ($p < 0.0001$; $\omega^2 = 0.99\%$). Post hoc analysis indicated that sedge dominated plots had the shallowest WTD (-83.6 ± 6.21), moss dominated plots (-236.9 ± 15.59) had deeper water tables than sedge. Grass (-294.9 ± 16.49) and bare soil plots (-300.7 ± 5.33) had significantly deepest water tables.

The addition of covariates in the ANCOVA slightly increased the R^2 from 40.02% to 40.86%. Plot dominance remained the most important factor ($p < 0.0001$; $\omega^2 = 27.5\%$). The importance of month nested within year was reduced ($p < 0.0001$; $\omega^2 = 5.70\%$). A small interaction was present between dominant vegetation and year ($p < 0.0001$; $\omega^2 = 1.71\%$). Inter-annual variation was the least important variation ($p < 0.0001$; $\omega^2 = 0.16\%$). The covariate PAR was important ($p < 0.0001$; $\omega^2 = 5.67\%$). Post hoc analysis (Figure 3.11) revealed that sedge dominated plots had the significantly shallowest WTD (-97.2 ± 7.37 mm). Moss (-243.1 ± 17.59 mm) and grass (-291.5 ± 19.01 mm) dominated plots were not significantly different and both had deeper WTD than sedge dominated plots. Grass plots were not significantly different to bare soil dominated plots which had the deepest WTD (-299.3 ± 5.77 mm).

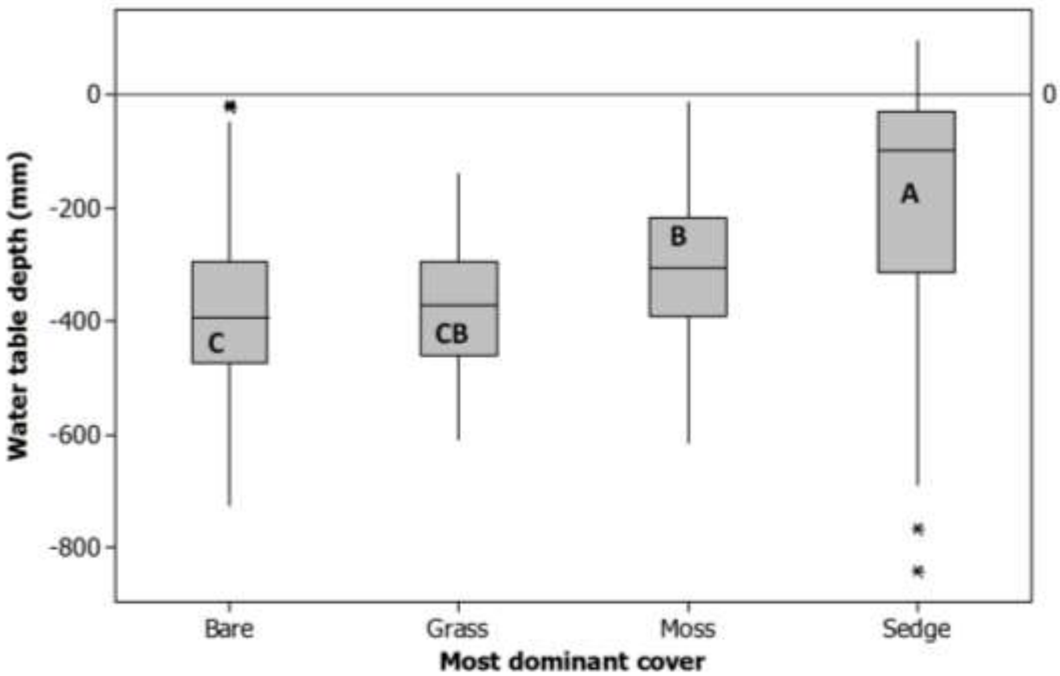


Figure 3.11: Box plot (five year study 2007-2011) of WTD plotted by 2011 primary site dominance. Letters on the plot are post hoc results of ANCOVA (Variables include: month and most dominant site cover, PAR, air temperature and total monthly rainfall).

3.4.4 Dissolved organic carbon concentration

3.4.4.1 Eight site comparison (2008) relationship between DOC and site

In 2008, DOC (N = 285) (Table 3.4) concentrations ranged between 0.83 mg C/ L to 308.07 mg C/L. Details (min, max and N number) on DOC and other measured variables (WTD, pH, conductivity and sample absorbance at 400, 465 and 665) are displayed in. The site data DOC distribution during 2008 period is represented in the box plots in (Figure 3.12).

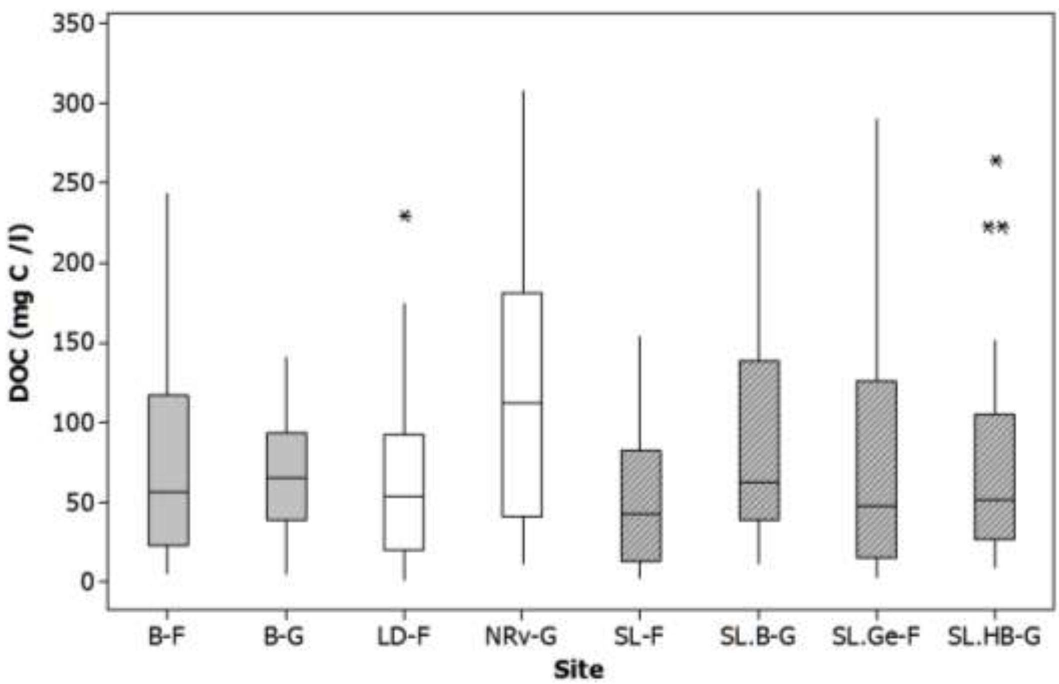


Figure 3.12: Box plot of site DOC (2008). The box plot colours represent the following: grey, bare soil controls; white, vegetated controls and grey striped, with restoration treatment.

Some of the sites have greater ranges in DOC, evident in (Figure 3.13) and their CV variation. The stabilised sites, geojute (100.88) and heather brash sites (83.19), had the greatest CV which indicates that DOC variation at stabilised sites is greatest relative to their mean. Three of the sites had very similar CV: bare flat (82.92), vegetated control (82.22) and seeded and limed (82.23) site. The sites with lowest CV were the naturally vegetated (74.83), followed by the gully blocked site (68.65) and finally the bare gully sites.

	DOC (mg C/L)	WTD (mm)	pH	Cond (S/m)	Abs 400 (nm)	Abs 465 (nm)	Abs 665 (nm)
N	285	489	321	320	323	322	319
Mean	77.34	28.4	4.53	54.46	0.064	0.149	0.008
SE Mean	3.78	0.9	0.03	1.42	0.003	0.006	0.001
Max	308.07	76.2	6.96	141.2	0.25	0.55	0.079
Min	0.83	-4.6	3.67	5.78	0.002	0.005	-0.007
% Removed	3.7	0.6	0.6	0.9	0.0	0.3	0.9

Table 3.4: Bleaklow study, descriptive statistics (2008) on DOC, WTD, pH, conductivity and abs 400, 456 and 665 nm.

In a stepwise regression, the relationship between DOC and temperature was investigated at several time steps (mean temperature on sampling day, on 1 – 7 days prior sampling, monthly mean and pre month mean). Temperature was significantly correlated to DOC in 2008. The inclusion of temperature at different time steps was used as covariates (mean temperature of the sampling day and that of six days prior sampling). It was revealed that there were significant differences between monthly soil pore water DOC concentrations. Differences in DOC were present between late autumn and winter to late spring and summer months. The mean temperature of the month preceding sampling was found to be most useful period to use in explaining DOC variation ω^2 = 8.6% (R^2 = 43.25%). Note that approximately 66% of sampling was conducted in the first 2.5 weeks of every month in 2008.

Using ANOVA, square root of DOC was found to significantly vary between months ($p < 0.0001$; ω^2 = 31.79%) followed by smaller, inter-site, plot nested within site variation which appeared significant ($p = 0.026$; ω^2 = 5.65%). Further analysis using a post hoc test did not identify significant differences between site DOC concentrations in 2008. In ANCOVA, temperature of the month preceding sampling month was significantly related to DOC concentration ($p = 0.010$; $\omega^2 = 10.52\%$). The addition of the covariate reduced the relative importance of month by 8.81% ($p < 0.0001$; $\omega^2 = 22.98\%$). Site remained the smallest contributing factor in the ANCOVA ($p = 0.002$; $\omega^2 = 5.20\%$). The addition of the covariate only slightly increased the R^2 from 37.43% to 38.97%. The residuals were normal. Post hoc analysis revealed a significant difference between the vegetated control sites' soil pore water DOC concentration (Figure 3.13.a). DOC at the naturally revegetated site (86.87 ± 0.21 mg C/L) was higher than the vegetated control (51.25 ± 0.19 mg C/L). DOC was highest during November (191.80 ± 0.41 mg C/L), followed by July - December (Between 62.96 ± 0.83 and 72.47 ± 0.51 mg C/L), and lowest in spring months, January - April (Between 49.80 ± 0.51 and 59.01 ± 0.67 mg C/L).

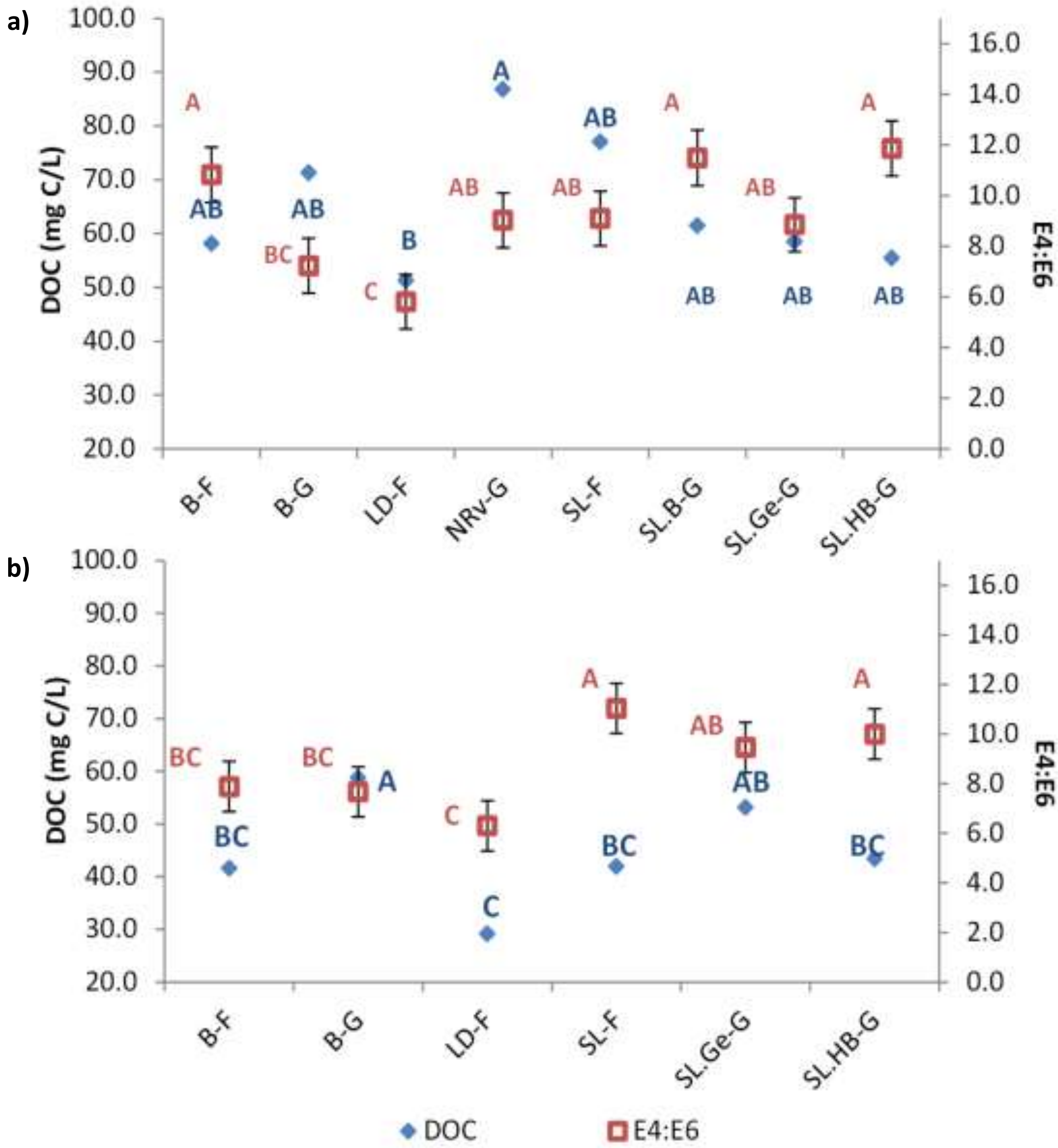


Figure 3.13: Main effects plot by site for DOC (normalised square root) in blue and E_4/E_6 (natural log) in red: a) 2008 data ANCOVA; b) 2007 – 2011 data ANCOVA. The ANCOVAs included the factors site and month. The Letters signify post hoc tukey test results whereby the means that do not share a letter are significantly different.

3.4.4.2 Eight site comparison (2008) relationship between DOC and vegetation

Analysis found soil pore water DOC concentration and the vegetation functional group cover were significantly related ($p = 0.05$, $\omega^2 = 4.09\%$), although month was the most important factor ($p < 0.0001$, $\omega^2 = 41.70\%$). Addition of the covariates temperature mean of the preceding month ($p < 0.0001$, $\omega^2 = 8.55\%$) improved the R^2 from 45.90% to 46.7%. The importance of the factors dropped. Monthly variation importance was reduced by 7.56% ($p < 0.0001$; $\omega^2 = 34.14\%$). Residuals were found to be normally distributed. Post hoc analysis (Figure 3.14) revealed two groups with overlapping significance.

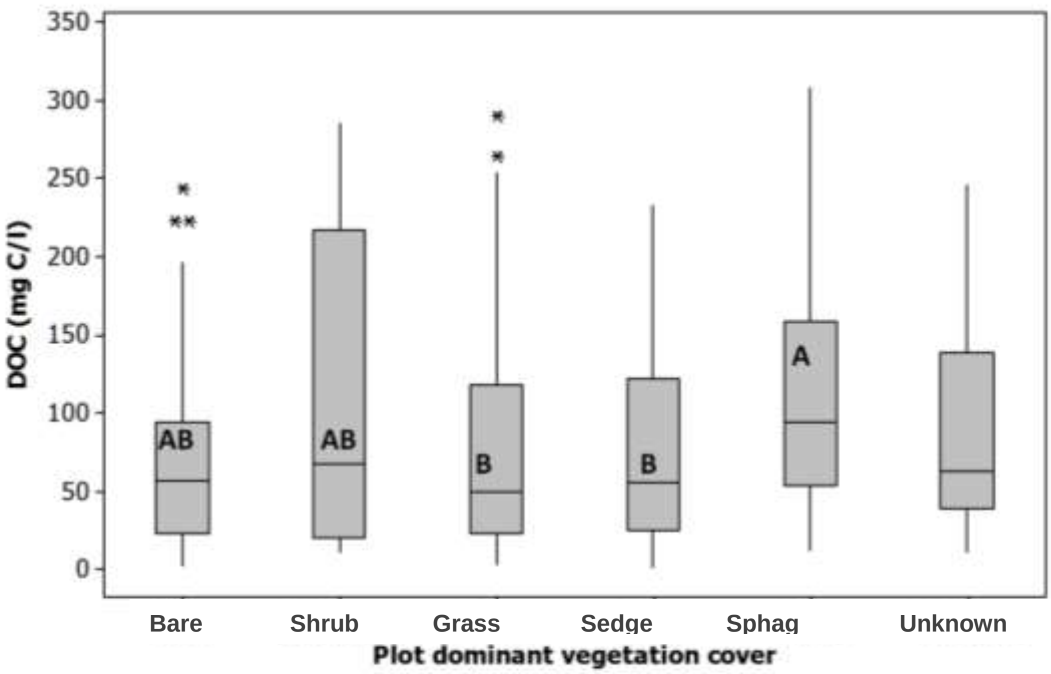


Figure 3.14: Box plot of DOC (2008) by most dominant vegetation cover. The ANCOVA includes: factors (site and month) and covariate air mean temperature of month prior to sampling. The letters signify post hoc tukey test results, whereby the means that do not share a letter are significantly different.

Mean DOC at *Sphagnum spp.* dominated plots (104.48 ± 10.33 mg C/L) was higher than both sedge (68.42 ± 6.62 mg C/L) and grass dominated plots (69.89 ± 6.19 mg C/L). Sites dominated by shrub (100.56 ± 16.06 mg C/L) and bare soil cover (72.64 ± 5.57 mg C/L) were not found to differ to the others, or the other functional groups. Plots with unknown cover were excluded from the model. The model predicted plots dominated by the sedge functional group would have the significantly lowest soil pore water DOC concentration.

3.4.4.3 Six site comparison (2007-2011) relationship between DOC and site

Post quality check, the six sites over the five year DOC dataset was composed of 1040 data points, ranging between 295.25 mg C /L to 0.02 mg C /L (Table 3.5).The monthly mean DOC is displayed in Figure 3.15. It shows three periods during the year at which the ranges of DOC concertation are greater. Specifically during late winter/ early spring, summer and later autumn/early winter; and two small peaks during spring and autumn (Figure 3.15).

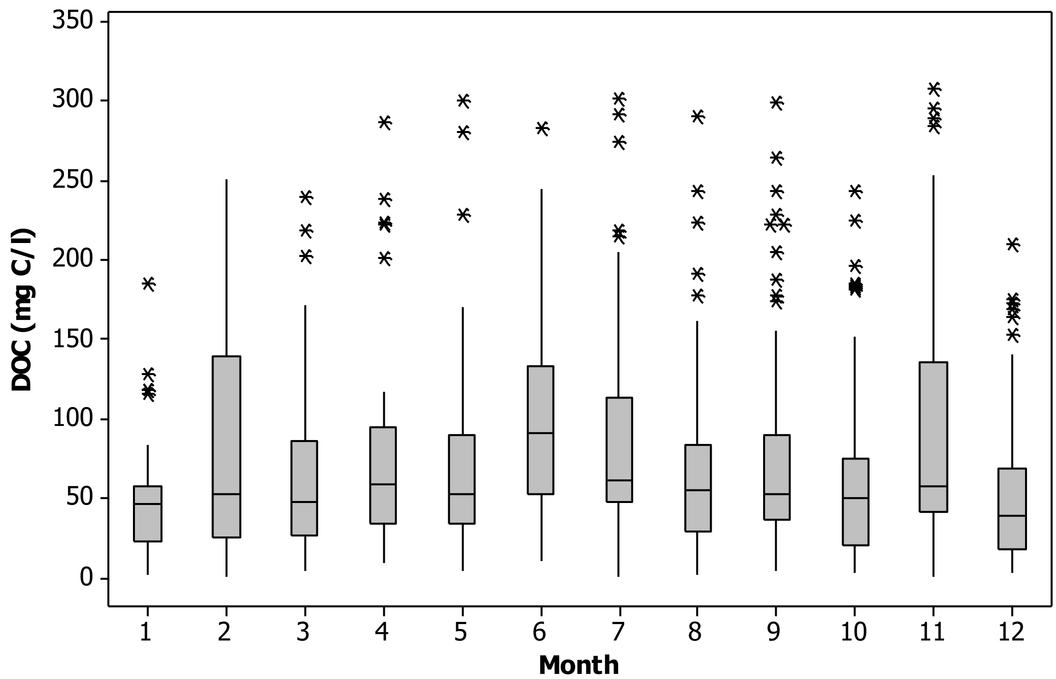


Figure 3.15: Boxplot of Bleaklow soil pore water DOC by month (2007 – 2011).

	DOC (mg/l)	WTD (mm)	pH	Cond (S/m)	Abs 400 (nm)	Abs 465 (nm)	Abs 665 (nm)
N	1148	1859	1335	1326	1325	1325	1323
Mean	69.1	29.4	4.34	55.068	0.120	0.090	0.015
SE Mean	1.71	0.4	0.02	0.639	0.004	0.002	0.000
Max	308.07	84.0	6.96	150.1	0.92	0.422	0.101
Min	0.05	-9.5	3.01	3.98	0.005	0.002	-0.007
% Removed	2.4	2.4	0.4	0.9	0.7	0.7	0.8

Table 3.5: Bleaklow (5 year study 2007 – 2011) soil pore water descriptive statistics for DOC, WTD, pH, conductivity and absorption at 400, 456 and 665 nm.

An ANOVA found that the soil water square root DOC concentration between years 2007 and 2011 (Figure 3.16) was most related to the factor of month nested within year ($p < 0.0001$, $\omega^2 = 22.90\%$). Inter-annual variation was the second most important factor ($p < 0.0001$, $\omega^2 = 9.53\%$), followed by a small annual and site interaction ($p < 0.0001$; ω^2 3.47%). Variation of plots nested within sites ($p < 0.0001$; ω^2 3.16%) explained the smallest portion of DOC variation during that time period along with site ($p < 0.0001$, ω^2 2.56%). Residuals were normal. Post hoc analysis revealed three significantly different groups. The site with the lowest soil pore water DOC was the vegetated control (28.36 ± 0.22 mg C/L), and the site with the highest DOC was the bare gully (71.03 ± 0.07 mg C/L). The site with restoration and least bare soil cover dominance, geojute site (61.80 ± 0.16 mg C/L), did not significantly differ from the bare gully, nor the three sites: bare flat (49.25 ± 0.06 mg C/L), seeded and limed (49.89 ± 0.07 mg C/L) and heather brash, which had lower DOC than the bare gully but higher than the vegetated control (Figure 3.13.b).

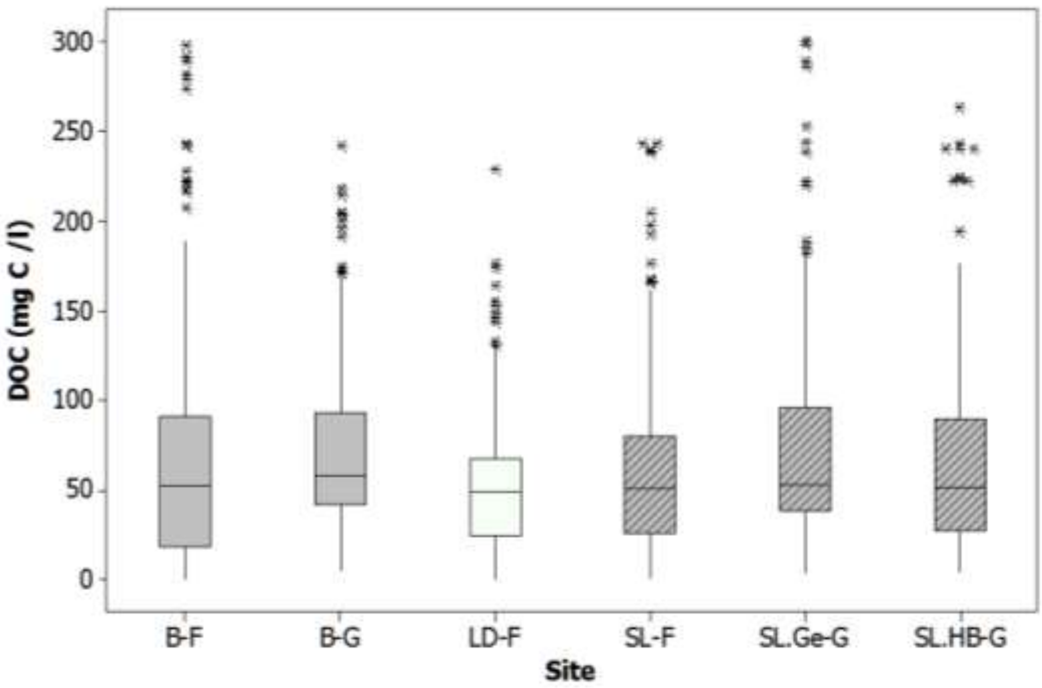


Figure 3.16: Box plots of soil pore water DOC concentrations (2007-2011) by site. The grey boxes are bare soil controls, white is least vegetated controls and grey striped are sites with restoration treatment.

The addition of covariates in the ANCOVA increased the R^2 from 41.64 to 45.18%. Month nested within year remained the most important explanatory variable ($p < 0.0001$; ω^2 23.24%), followed by inter-annual variation ($p < 0.0001$, ω^2 9.54%), an interaction between site and year ($p < 0.0001$; ω^2 4.85%), and plot nested within site ($p < 0.0001$, ω^2 3.80%). Water table depth was found to have a relatively small relationship to soil pore water DOC ($p = 0.005$, ω^2 2.23%). Monthly mean temperature ($p = 0.006$, ω^2 0.84%) and site variation ($p < 0.0001$, ω^2 0.65%) were found to be significant, although were relatively unimportant in the model. Residuals were normal. Post hoc analysis again revealed an overlapping significance between the levels (Figure 3.13). Soil pore water DOC was significantly lowest at the vegetated control (29.12 ± 0.26 mg C/L) and highest at the bare gully (58.70 ± 0.12 mg C/L). The geojute site had significantly higher DOC concentrations than the vegetated control (53.1 ± 0.20 mg C/L). The bare flat (41.60 ± 0.10 mg C/L) had significantly lower DOC than the bare soil gully site. The

DOC at the three treated sites did not significantly differ to each other. However, the heather brash (43.40 ± 0.07 mg C/L) and the seeded and limed site (41.60 ± 0.0.09 mg C/L) had significantly lower soil pore water DOC than the bare gully. Post hoc analysis also revealed that there were significant consecutive annual decreases in site DOC from 2008 to 2010, however, followed by a significantly increased in 2011 (Figure 3.17) the ANCOVA revealed a fluctuating trend in DOC. If internal and seasonal variation is not accounted for, an overall increasing trend is observed in DOC across the Bleaklow sites with the exception of the vegetated control, along with a peak in DOC concentrations observed year 2011 (Figure 3.18).

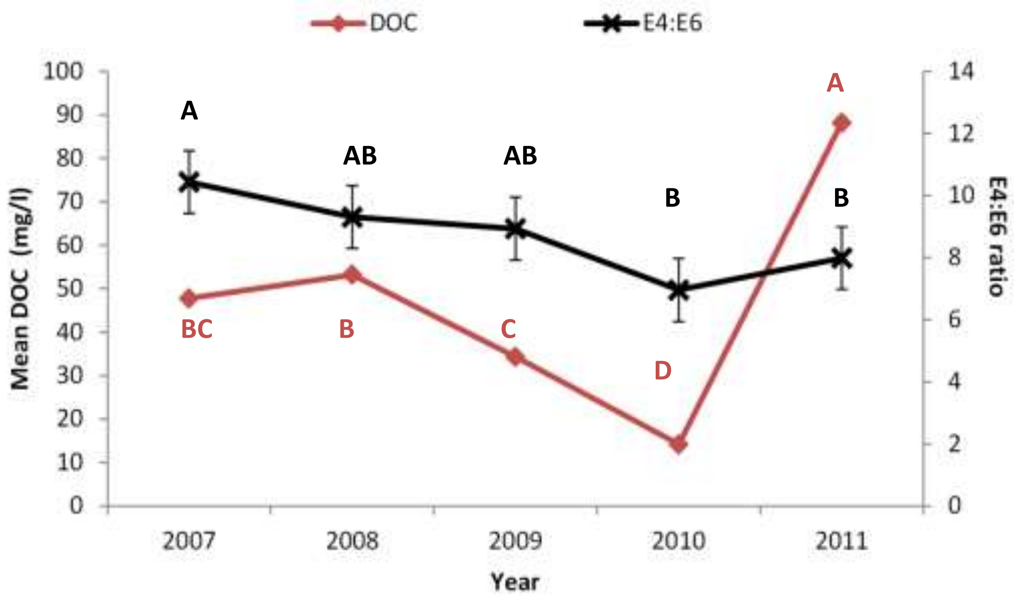


Figure 3.17: Main effects plot (2007 – 2011) of DOC (brown diamond) and E_4/E_6 (black cross), using natural log data valued in an ANCOVA. Model Inputs included site, year and month. Left axis is DOC (mg/L) and right axis is the E_4/E_6 ratio. The letters on the plot represent the tukey post hoc result. Shared letters represent no significant differences between year factor levels.

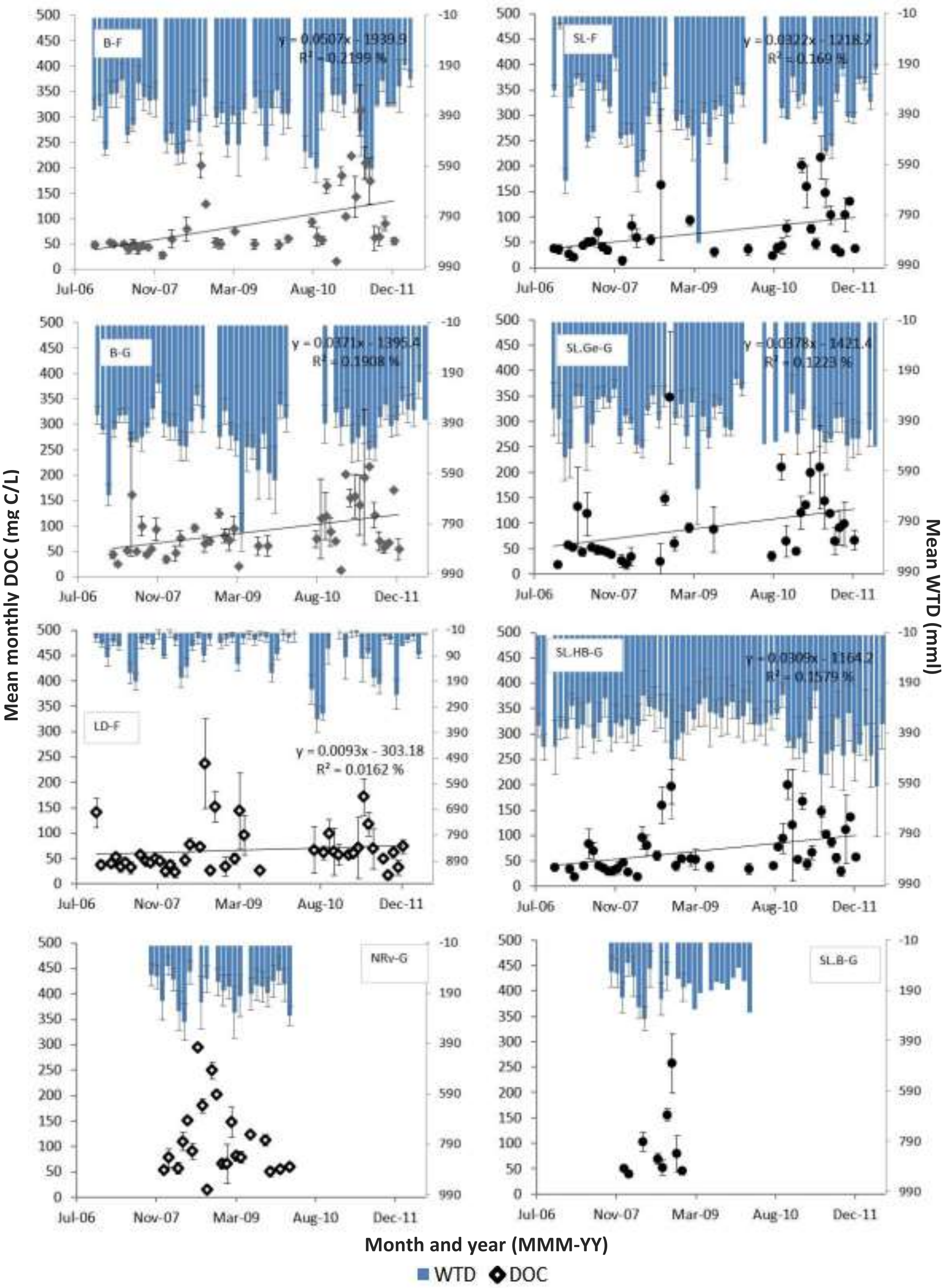


Figure 3.18: Bleaklow sites monthly mean soil pore water DOC conc. (left axes - mg C/ L – diamonds: grey, bare untreated; black hollow, vegetated controls; black diamond, treated) and WTD (right axes - mm - blue bars) (2007 – 2011). The error bars represent the SE mean. A fitted trend line illustrates site long term DOC concentrations general trends.

3.4.4.4 Six site comparison (2007-2011) relationship between DOC and site vegetation

Analysis of soil pore water DOC by the factor most dominant vegetation function group, using ANOVA produces a smaller R^2 = 32.54% than the ANOVA by site. Month nested within year remained the most important factor ($p < 0.0001$, ω^2 = 22.25%), followed by year ($p < 0.0001$, ω^2 = 8.78%). A small plot interaction with year was also present ($p < 0.0001$, ω^2 = 1.54%). However, site vegetation dominance alone was not a significant factor ($p = 0.313$, ω^2 = -0.06%). Residuals were normal. Post hoc analysis found no significant difference between vegetation types

Addition of covariates in ANCOVA increased the R^2 from 32.54% to 36.55%. Month nested within year remained the most important explanatory variable ($p < 0.0001$, ω^2 = 22.23%). Water table depth was significant, although of low importance in the model ($p < 0.0001$, ω^2 = 2.27%) as was monthly mean temperature ($p < 0.0001$, ω^2 = 0.78%). Post hoc analysis did not reveal any different results to the ANOVA (Figure 3.19).

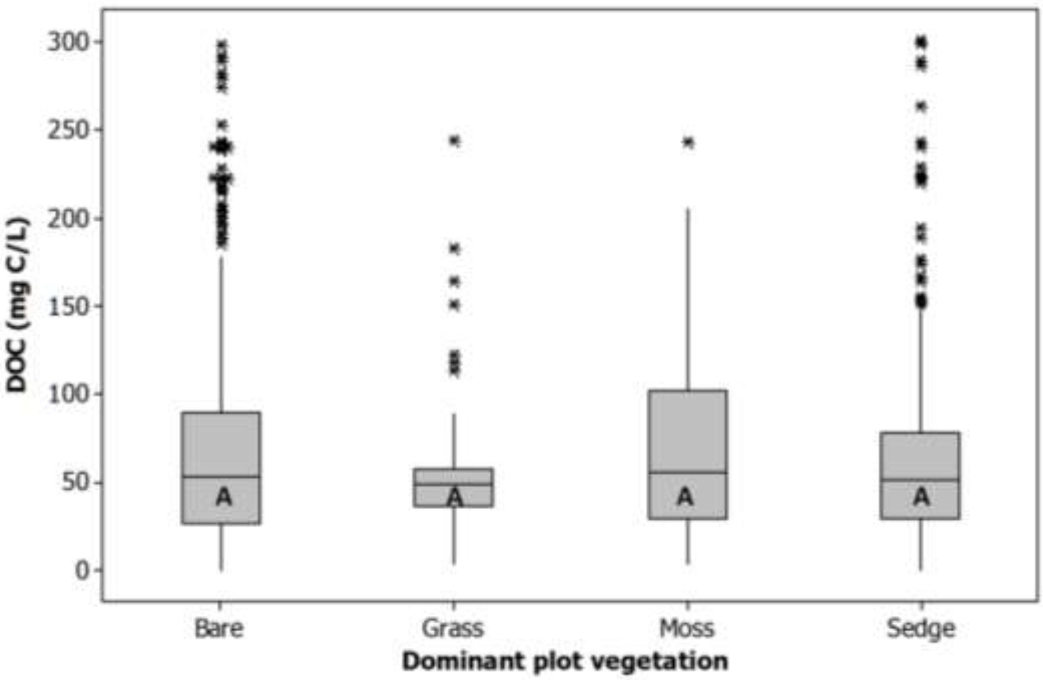


Figure 3.19: Box plot of vegetation cover DOC (2007 – 2011). ANCOVA post hoc results are displayed as letter on the plot.

3.4.5 Speciation UV-Vis and E_4/E_6

Soil pore water sample absorbance is highest at longer wavelengths (665 nm) and lowest at shorter wavelengths (400 nm) (Figure 3.20). The three absorbance wavelengths (400, 465 and 665 nm) were found to have measured a lower CV for three gully sites: the naturally revegetated gully, the bare gully, and gully blocked site. The geojute site had the greatest CV (*Table 3.6*) and widest range E_4/E_6 in 2008 (Figure 3.21).

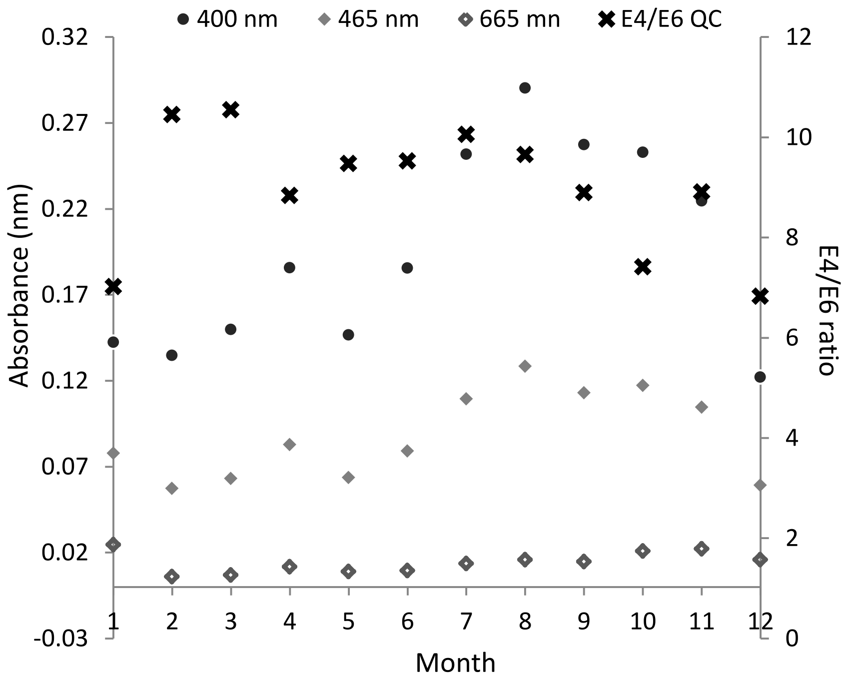


Figure 3.20: Mean Bleaklow soil pore water monthly absorption at 400, 465, 665nm and E_4/E_6 across (2007-2011).

	400nm	465nm	665mn	E_4/E_6
Low	NRv-G	NRv-G	NRv-G	B-G
↕	B-G	B-G	SL-F	SL-F
	SL.B-G	SL.B-G	SL.B-G	LD-F
	SL-F	B-F	B-F	SL.B-G
	LD-F	SL-F	B-G	SL.HB-G
	B-F	LD-F	LD-F	NRv-G
	SL. HB-G	SL.HB-G	SL.HB-G	B-F
high	SL.Ge-G	SL.Ge-G	SL.Ge-G	SL.Ge-G

Table 3.6 Bleaklow study sites soil pore water sample UV-vis absorbance CV. Sites are listed in order of low to high CV.

3.4.5.1 E_4/E_6 (2008)

3.4.5.1.1 *Eight site E_4/E_6 comparison (2008)*

The natural logged transformed E_4/E_6 (Figure 3.21) ANOVA indicated factors site ($p < 0.0001$, $\omega^2 = 15.28\%$) and monthly ($p < 0.0001$, $\omega^2 = 7.36\%$) were significant. Residuals were normal. Post hoc analysis revealed that the seeded and limed site (12.19 ± 1.09) had the highest E_4/E_6 and the vegetated control had the lowest (5.70 ± 1.10) and bare gully (7.40 ± 1.09). There was much overlap between site factor levels which made a pattern unclear.

Addition of covariates increased the R^2 from 22.64% to 24.53%. Inter-site variation was again the most important factor ($p < 0.0001$, $\omega^2 = 13.04\%$) followed by month ($p < 0.0001$, $\omega^2 = 6.46\%$), and pH as a covariate ($p < 0.017$, $\omega^2 = 3.48\%$). Natural log transformed E_4/E_6 was negatively correlated with pH (R2 = -0.194). Residuals were normal. Post hoc analysis of the site factor (Figure 3.13) found that the seeded and limed (11.87 ± 1.09), geojute (11.50 ± 1.10) and bare flat (10.83 ± 1.09) sites had significantly highest E_4/E_6, while the vegetated control (5.81 ± 1.07) and bare gully (10.83 ± 1.09) sites had significantly lower ratio. The three sites with the highest ratio did not differ to the gully block (9.10 ± 1.08), naturally revegetated and heather brash sites. These did not significantly differ to the bare gully either (7.24 ± 1.09).

There was no significant difference between the bare gully and vegetated control. Post hoc analysis revealed that E_4/E_6 was lowest during November (7.55 ± 1.10) and December (6.94 ± 1.69). It then significantly rose in January (10.03 ± 1.07) then dropped in March (7.13 ± 1.09) to a ratio insignificantly different to that from November and December. It rose in April (8.84 ± 1.11) and remained at a similar ratio over the summer months until November.

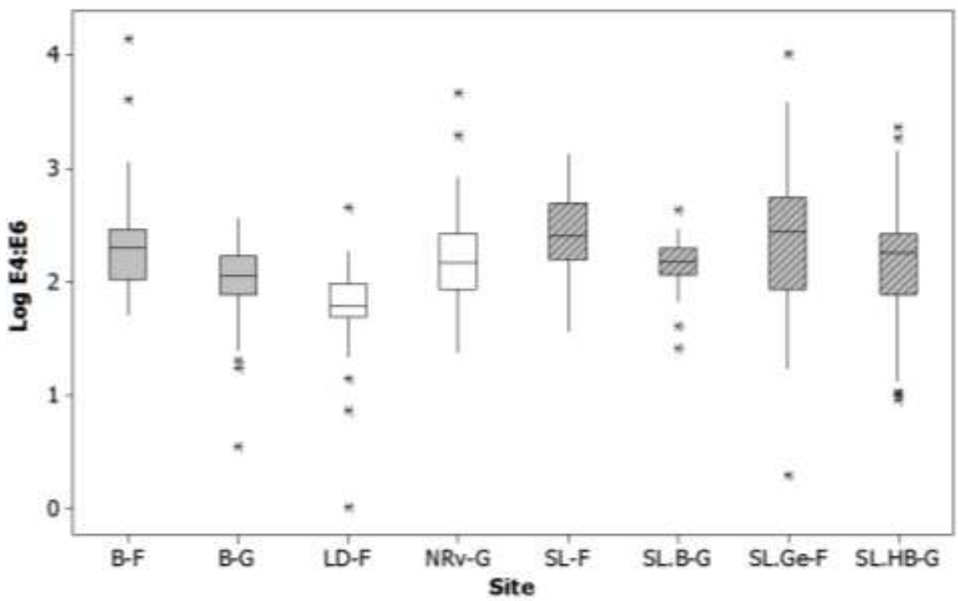


Figure 3.21: Box plot of Bleaklow sites soil pore water E_4/E_6 (logged) (2008). The box plot colours represent the following: grey are bare soil controls, white are vegetated controls and grey striped are sites with restoration treatments.

3.4.5.1.2 *Analysis of E_4/E_6 relationship to vegetation (2008)*

Analysis of variance of natural log transformed E_4/E_6 for 2008, found dominant vegetation type ($p < 0.0001$, $\omega^2 = 8.83\%$) was a more important explanatory variable than inter-month variation ($p = 0.004$ $\omega^2 = 5.74\%$). Post hoc analysis revealed the functional groups shrub (11.37 ± 1.17), grass (9.76 ± 1.06) and bare soil sites (9.76 ± 1.05) had the significantly highest soil pore water E_4/E_6 ratio. *Sphagnum spp.* dominated plot E_4/E_6 did not significantly differ to the other cover types (7.34 ± 1.12). Sedge plots had the lowest E_4/E_6 (6.26 ± 1.07).

The addition of soil pore water pH as a covariate ($p = 0.028$, $\omega^2 = 3.87\%$) only slightly improved the R^2 from 14.57 to 14.65%. Plot dominance remained the most important explanatory variable ($p < 0.0001$, $\omega^2 = 6.74\%$), followed by inter-monthly variation ($p = 0.022$ $\omega^2 = 4\%$). Natural log of E_4/E_6 was negatively correlated with pH; increased pH was thus correlated with decrease in E_4/E_6 ratio. Post hoc analysis revealed the same significant differences between the vegetation types with slightly different mean values. The soil pore water with significantly highest E_4/E_6 in 2008 was shrub (11.37 ± 1.17), grass (9.81 ± 1.06) and bare soil sites (9.57 ± 1.06). *Sphagnum spp.* dominated plot E_4/E_6 did not significantly differ to the other cover types (6.99 ± 1.13). Sedge plots had the lowest E_4/E_6 (6.51 ± 1.07).

3.4.5.2 E_4/E_6 (2006-2011)

3.4.5.2.1 *Analysis of E_4/E_6 relationship to site (2007 – 2011)*

There is an observed decreasing trend in E_4/E_6 over a project year. The annual mean ratio starts at 10.72 and by year 5 it drops to a mean of 8.86. The site with the highest CV was the geojute site followed by the bare flat site, and the sites with the lowest CV were the bare gully followed by the seeded and limed. The site with the smallest shift in E_4/E_6 from 2007 to 2011 was the vegetated control (*Figure 3.22*). The geojute site had greatest shift in $E_4/E_{6.}$ Analysis of the natural log of E_4/E_6 in ANOVA found month nested within year was the most important factor ($p < 0.0001$, $\omega^2 = 12.77\%$). Site was the second most important factor ($p < 0.0001$, $\omega^2 = 11.24\%$). Inter-annual variation was significantly related to E_4/E_6 ($p < 0.0001$, $\omega^2 = 4.13\%$). Post hoc analysis revealed the seeded and limed site (11.59 ± 1.01) had significantly highest soil pore water E_4/E_6. The vegetated control (5.90 ± 1.01) had the lowest ratio. The remaining four sites bare flat (9.04 ± 1.01), bare gully (8.30 ± 1.01), geojute (9.46 ± 1.01) and heather brash site (9.62 ± 1.01) did not significantly differ to one another, and had a low ratio than the seeded and limed but a higher ratio than the vegetated control. The only site with a mean E_4/E_6 ratio within the fulvic acid range (below 6) was the vegetated control.

The addition of covariates increased the R^2 from 22.64% to 24.53%. Month nested within year remained the most important factor ($p < 0.0001$, $\omega^2 = 9.99\%$), followed by variation between the factor site ($p < 0.0001$, $\omega^2 = 6.40\%$) and variation between year ($p < 0.0001$, $\omega^2 =$

3.16%). The most important covariate added was WTD (p = 0.027; 4.45%), followed by natural log transformed DOC (p = 0.019; 4.45%). Finally pH was significant although of low importance ($p < 0.012$, $\omega^2 = 1.30\%$). Post hoc analysis revealed (Figure 3.13.a) that the control sites, bare soil controls, bare gully (7.68 ± 1.01), bare flat (7.88 ± 1.01) and the vegetated control (6.30 ± 1.01) did not differ to each other and had lower soil pore water E_4/E_6 ratio than the heather brash and the seeded and limed sites. The seeded and limed (11.04 ± 1.01), heather brash (10.00 ± 1.01), and the geojute sites (9.46 ± 1.01) did not differ to each other. The bare soil controls did not significantly differ to the geojute site. The vegetated control was the only site to have an E_4/E_6 ratio within the humic range. Furthermore post hoc analysis revealed a similar pattern of E_4/E_6 variation from 2006 – 2011 (*Figure* 3.17), as found in the post hoc analysis of the DOC ANCOVA, the lowest E_4/E_6 ratio was in years 2010 (6.96 ± 1.02) and 2011 (7.99 ± 1.01). It was found that the highest mean ratio was during 2007 (10.43 ± 1.01).

The general trend would appear to be generally decreasing over time as observed in the raw data multiannual plots of E_4/E_6 (Figure 3.22). Using the Pearson's test, weak negative correlations were found between natural log transformed E_4/E_6 and water table depth (Pearson correlation coefficient; -0.078 P = 0.011). The only two sites to have an increasing trend in pH over the five years were the heather brash and geojute, which used stabilisation techniques (Figure 3.22). The pH at these sites had greatest fluctuation evident in their CV (heather brash = 12.62 and geojute = 10.71) compared to the bare controls and seeded and limed site which have decreasing pH and low fluctuation (bare flat = 10.04. bare gully = 8.78, and seeded and limed = 9.48). In 2011 both the bare sites had a pH below that of the vegetated control and three treatment sites.

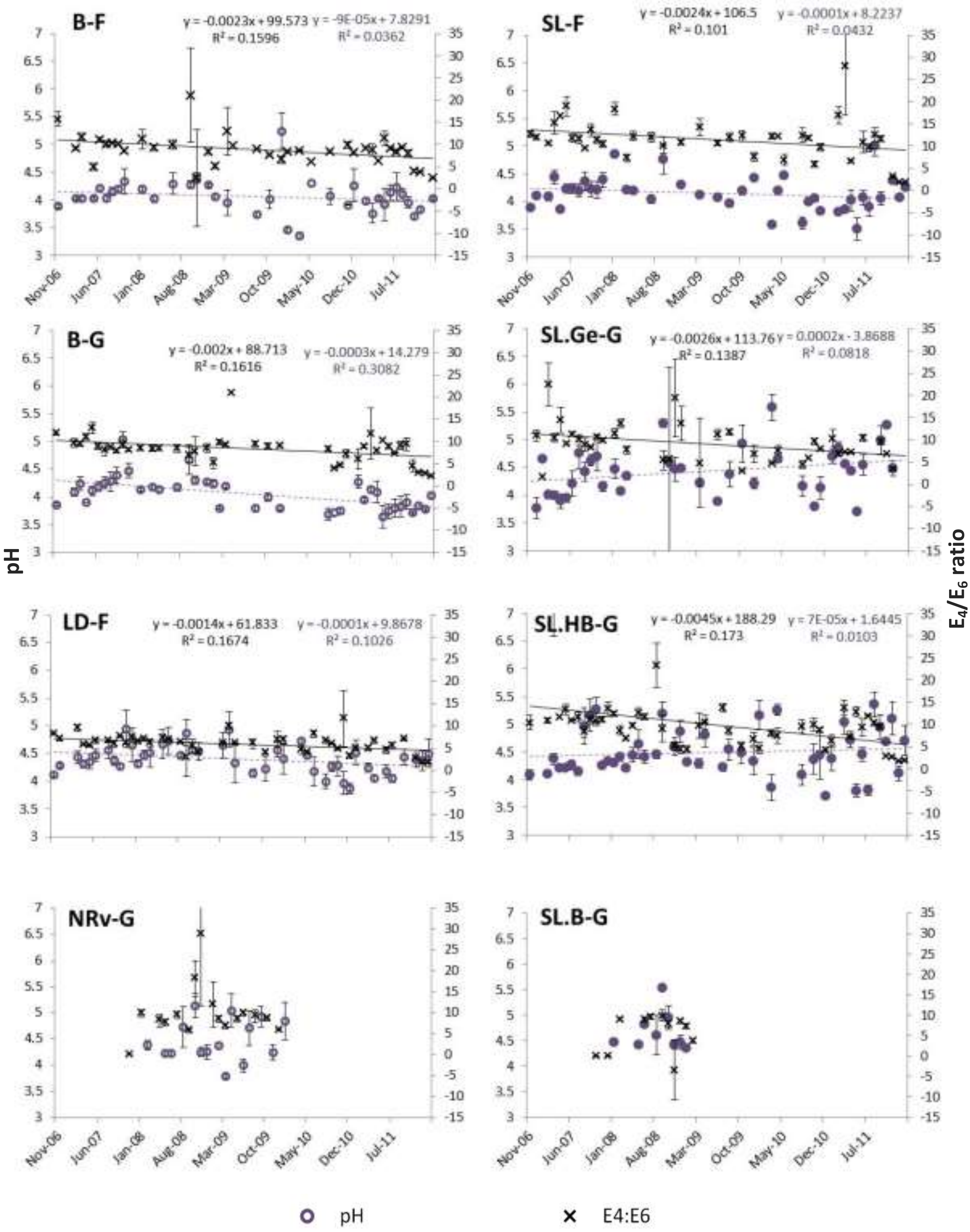


Figure 3.22: Bleaklow soil pore water monthly (x-axis, November 2006 – January 2012) mean pH (left y-axis) and E_4/E_6 *(right y-axis). All data-points are mean monthly values with error bars defining the SE mean. The purple circles are pH (hollow circles signify controls). The* E_4/E_6 *ratios are represented by black crosses. Multiannual linear trends are displayed for both the pH (dash purple) and* E_4/E_6 *(black line).*

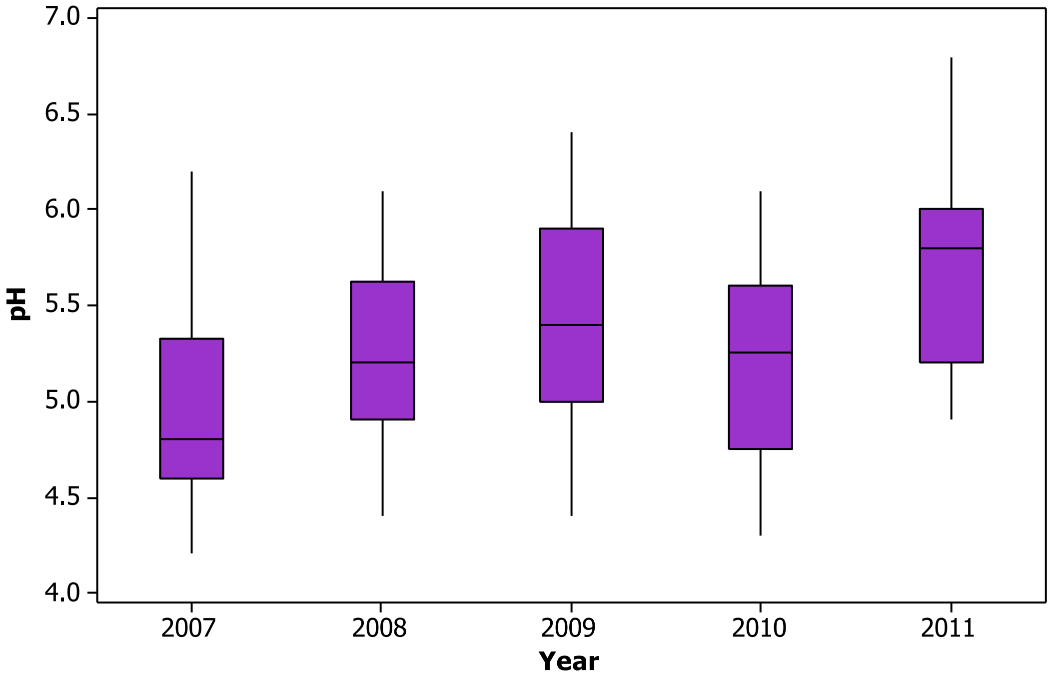


Figure 3.23: Box plot of rainfall water pH data distribution according (2007 and 2011). Data source Defra

Rainfall water pH had a general increasing trend, with a mean pH (5.34, ± 0.06) ~19% greater than the soil pore water pH (4.33 ± 0.02) throughout the study (2007 -2011). The soil pore water pH fluctuates seasonally and during the summer periods (Figure 3.22). The difference between the soil pore water and rainfall pH increased, with the soil pore water becoming more acidic (Figure 3.24). The difference between the rainfall pH at soil pore water was greater at the bare site controls and the Seeded and limed (dominantly bare sites) 30% lower than the rainfall water, while the pH at the vegetated control and sites with stabilisation techniques was 20% lower than the rainfall water pH (Table 3.7).

Site	Change in pH	Mean pH (2007)	Mean pH (2011)	Ratio of rainfall pH to soil pore water
B-F	-0.170	4.13	3.96	1.3
B-G	-0.435	4.31	3.87	1.3
LD-F	-0.220	4.50	4.28	1.2
SL-F	-0.174	4.24	4.07	1.3
SL.Ge-G	0.330	4.34	4.67	1.2
SL.HB-G	0.067	4.50	4.57	1.2

Table 3.7: Mean annual changes in pH on Bleaklow monitoring site, (2007-2011).

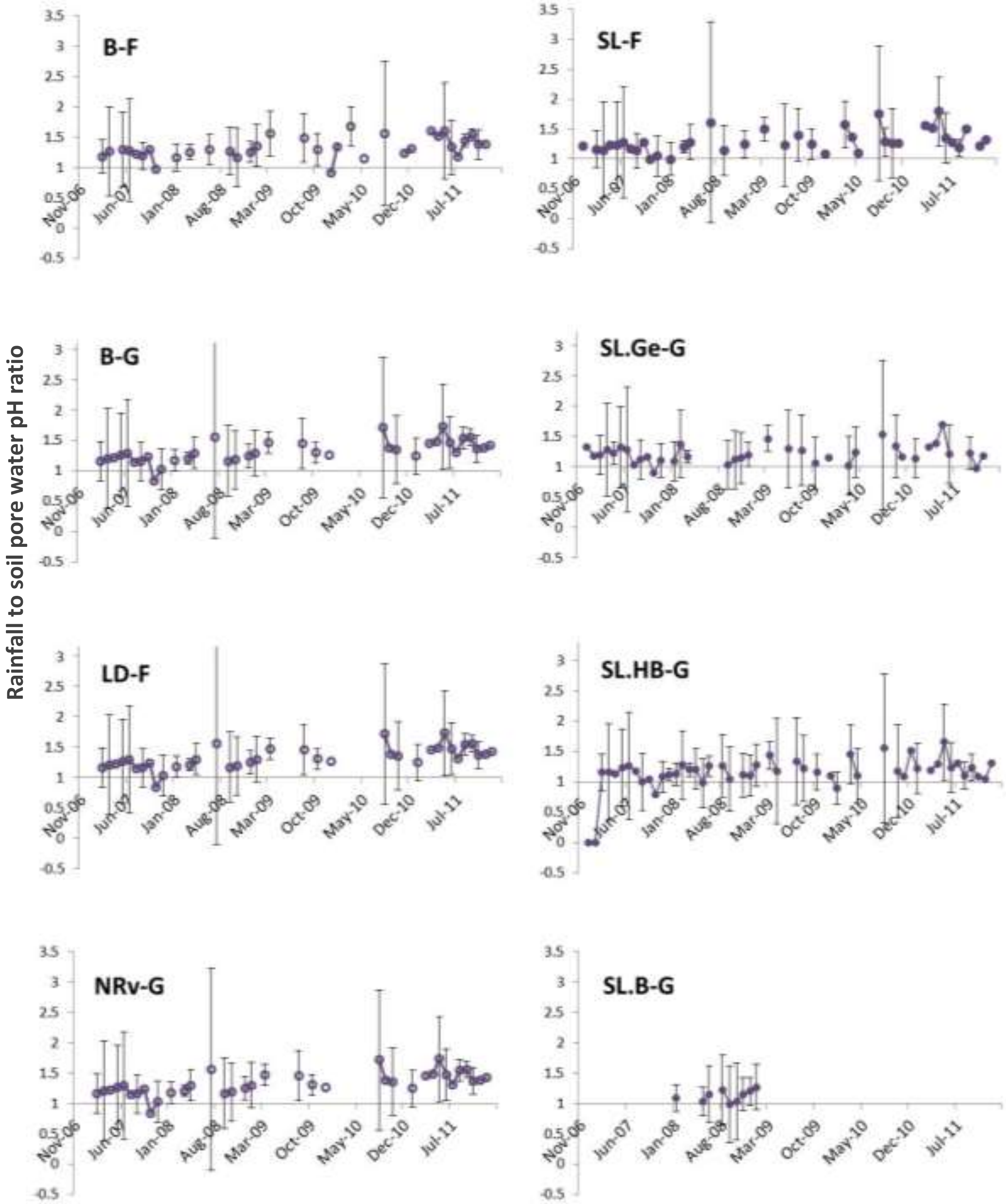


Figure 3.24: The ratio of rainfall to soil pore water pH (November 2006 – January 2012). Each data-points represents the monthly rainfall pH divided by site mean pH of the plot replicates on each site with error bars defining the standard error of the mean. The hollow purple circles signify site controls and the filled in purple circles represent sample pH measured at site with treatment. Point >1 signify that the mean rainfall pH was greater than the mean soil pore water during that month.

3.4.5.2.2 *Analysis of E_4/E_6 relationship to vegetation (2007 – 2011)*

From 2007 to 2011, E_4/E_6 was found to significantly relate to month nested within year ($p < 0.0001$, $\omega^2 = 28.58\%$), followed by annual variation ($p < 0.0001$, $\omega^2 = 8.64\%$). Dominant vegetation type was a significant factor ($p < 0.0001$, $\omega^2 = 5.77\%$). Post hoc analysis found that sedge dominated sites had the lowest E_4/E_6 ratio (6.63 ± 1.03). Post hoc analysis also found a generally decreasing trend in E_4/E_6 ratio from 2007 (10.22 ± 1.03) to 2011 (6.81 ± 1.03).

The addition of covariates in the ANCOVA improved the R^2 from 43.02% to 46.69%. It was found that month nested within year was still the most important factor ($p < 0.0001$, $\omega^2 = 25.77\%$), followed by annual variation ($p < 0.0001$, $\omega^2 = 9.64\%$). Water table depth was the most important covariate ($p < 0.0001$, $\omega^2 = 6.64\%$), followed by monthly mean temperature ($p < 0.0001$, $\omega^2 = 25.77\%$). The importance of the cover dominance was reduced ($p < 0.0001$, $\omega^2 = 0.95\%$), and although soil pore water pH was a significant covariate it was of low importance in the ANCOVA. Post hoc analysis revealed (*Figure* 3.25) again that sedge dominated plots had the significantly lowest E_4/E_6 ratio (7.30 ± 1.02) and it did not significantly differ to grass dominated plots (7.94 ± 1.06). Grass dominated plots did not differ to bare soil (8.33 ± 1.03) or moss dominated plots either (7.94 ± 1.03). It was also revealed that the first four years 2007 (9.36 ± 1.05), 2008 (9.26 ± 1.04), 2009 (8.26 ± 1.04) and 2010 (7.83 ± 1.09) did not have significantly different E_4/E_6. They were also significantly higher than the final year 2011 (6.21 ± 1.04). The ratio during year 2010 did not significantly differ to 2011 (*Figure* 3.17). There was a small significant relationship between DOC concentration and E_4/E_6 ratio. Although no significant correlation was found, they were both influenced by seasonal variation, which is evident in (Figure 3.26). Both DOC and E_4/E_6 where significantly related to temperature as a covariate. The relationship is visible in the seasonal variation, with peak concentrations of DOC during spring, summer and winter. E_4/E_6 also had peaks during those periods (Figure 3.26).

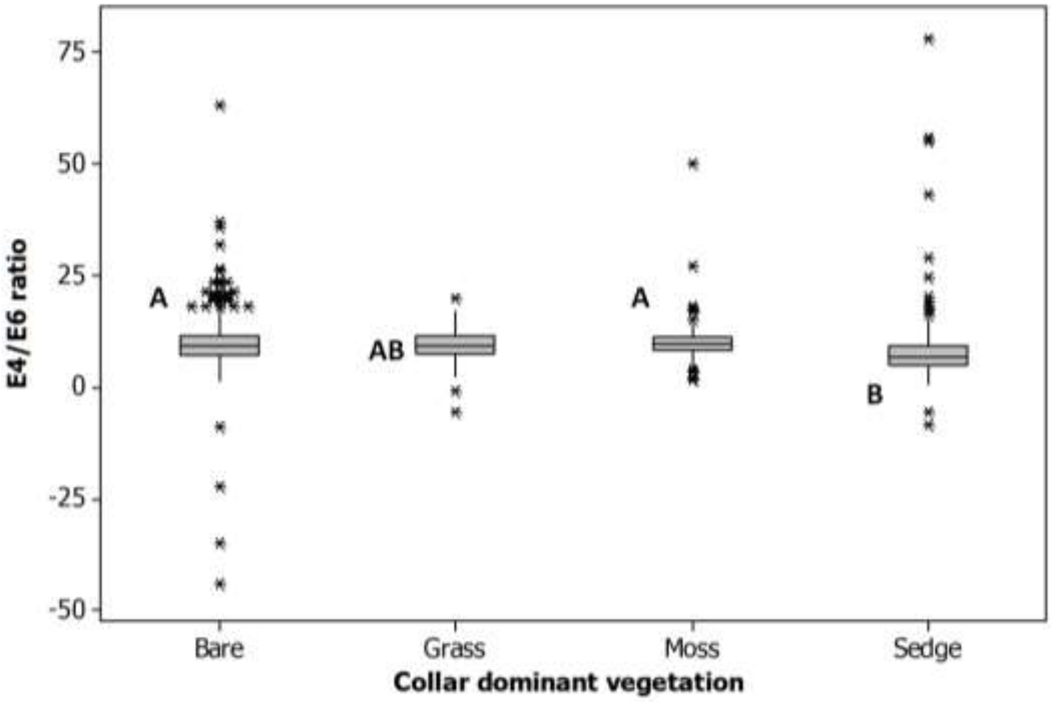


Figure 3.25: Dominance functional group soil pore water E_4/E_6 (2007 – 2011). Letters on the plot represent the ANCOVA post hoc results.

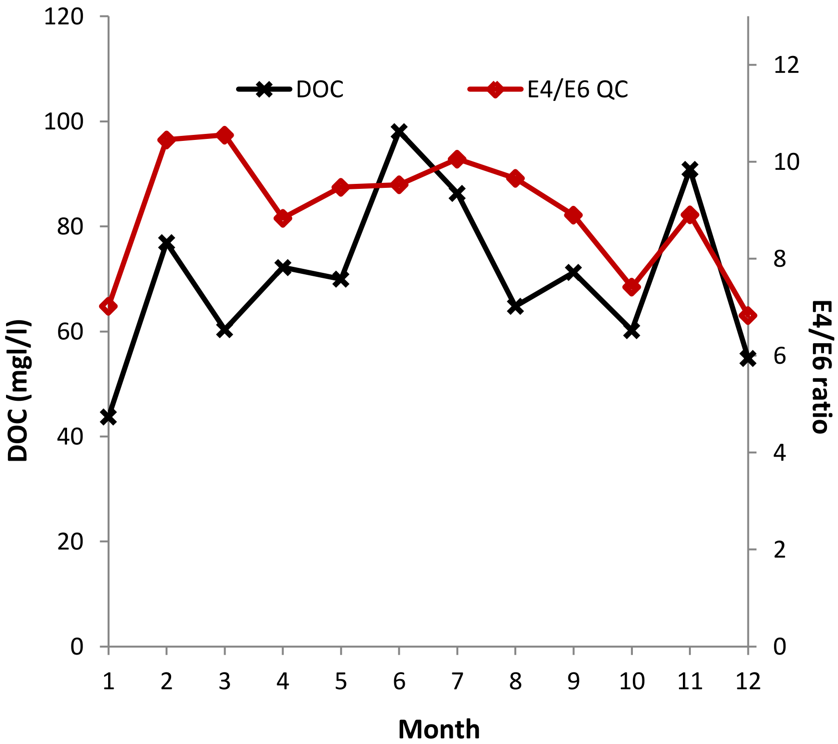


Figure 3.26: Mean monthly E_4/E_6 and DOC across all Bleaklow sites 2006-2011.

2008						
Response variable (n)	Test	Factor (F)/ covariate (C)/ interaction (I)/ nested factor (N)	Variable	P	ω^2	$_{adj}R^2$
WTD (490)	ANOVA	F	Site	< 0.0001	46.13	56.14%
		F	Month	< 0.0001	9.84	
		F	Dominant plot veg 09	< 0.0001	54.52	64.06%
		F	Month	< 0.0001	9.48	
	ANCOVA	F	Site	< 0.0001	46.21	56.14%
		F	Month	< 0.0001	5.59	
		C	PAR	0.017	4.28	
		F	Dominant plot veg 09	< 0.0001	55.29	64.59%
		F	Month	< 0.0001	5.90	
		C	PAR	0.006	3.33	
DOC (286)	ANOVA	F	Site	0.046	5.65	37.43%
		F	Month	< 0.0001	31.79	
		F	Dominant plot veg 09	0.050	4.09	45.90%
		F	Month	< 0.0001	41.70	
	ANCOVA	F	Site	0.009	10.52	38.79%
		F	Month	0.018	5.20	
		C	Pre Month mean C	< 0.0001	22.98	
		F	Dominant plot veg 09	0.055	2.76	37.85%
		F	Month	< 0.0001	24.49	
		C	Pre Month me	0.017	10.52	
E4/E4 (311)	ANOVA	F	Site	< 0.0001	15.28	22.64%
		F	Month	< 0.0001	7.36	
		F	Dominant plot veg 09	< 0.0001	8.83	14.57%
		F	Month	0.004	5.74	
	ANCOVA	F	Site	< 0.0001	13.04	23.40%
		F	Month	< 0.0001	6.46	
		C	pH	0.017	3.48	
		F	Dominant plot veg 09	< 0.0001	6.74	14.65%
		F	Month	0.022	4.00	
		C	pH	0.028	3.87	

Table 3.8: Analysis of variance and covariance output of WTD soil pore water sample DOC concentrations and E_4/E_6 data (2008). Outputs include p values, ω^2 and R^2.

			2007-2011			
Response variable (n)	Test	Factor (F)/ covariate (C)/ interaction (I)/ nested factor (N)	Variable	P	ω^2	$_{adj}R^2$
WTD (1104)	ANOVA	F	Year	< 0.0001	0.26	63.64%
		F	Site	< 0.0001	44.49	
		N	Plot (Site)	< 0.0001	9.28	
		N	Month (Year)	< 0.0001	9.59	
		F	Dominant plot veg 11	< 0.0001	30.45	40.02%
		F	Year	0.022	0.20	
		N	Month (Year)	< 0.0001	8.44	
		I	Dominant plot veg 11*Year	< 0.0001	0.91	
	ANCOVA	F	Year	< 0.0001	0.02	65.06%
		F	Site	< 0.0001	42.89	
		N	Plot (Site)	< 0.0001	9.86	
		N	Month (Year)	< 0.0001	6.51	
		C	PAR	< 0.0001	5.77	
		C	AT	< 0.0001	-0.01	
		F	Dominant plot veg 11	< 0.0001	27.59	40.86%
		F	Year	0.025	0.16	
		N	Month (Year)	< 0.0001	5.70	
		I	Dominant plot veg 11*Year	< 0.0001	1.71	
		C	PAR	< 0.0001	5.67	

Table 3.9: Analysis of variance and covariance output of WTD data (2007 2011). Outputs include p values, ω^2 and R^2

2007 - 2011						
Response variable (n)	Test	Factor (F)/ covariate (C)/ interaction (I)/ nested factor (N)	Variable	P	ω^2	$_{adj}R^2$
DOC (615)	ANOVA	F	Site	< 0.0001	2.56	41.64%
		F	Year	< 0.0001	9.53	
		F	Plot (Site)	< 0.0001	3.16	
		F	Month (Year)	< 0.0001	22.90	
		I	Site*Year	< 0.0001	3.47	
		F	Dominant plot veg 11	≤ 0.313	-0.06	32.54%
		F	Year	< 0.0001	8.78	
		F	Month (Year)	< 0.0001	22.25	
		I	Dominant plot veg 11*Year	≤ 0.001	1.54	
	ANCOVA	C	WTD	≤ 0.005	2.23	45.18%
		C	Month mean C	≤ 0.006	0.84	
		F	Site	< 0.0001	0.65	
		F	Year	< 0.0001	9.54	
		N	Plot (Site)	< 0.0001	3.80	
		N	Month(Year)	< 0.0001	23.24	
		I	Site*Year	< 0.0001	4.85	
		C	Month mean C	≤ 0.005	2.23	36.55%
		C	WTD	≤ 0.006	0.84	
		F	Dominant plot veg 11	< 0.0001	0.65	
		F	Year	< 0.0001	9.54	
		N	Month (Year)	< 0.0001	3.80	
		I	Dominant plot veg 11*Year	< 0.0001	23.24	

Table 3.10: Analysis of variance and covariance output of soil pore water sample DOC concentrations and E_4/E_6 data (2007 2011). Outputs include p values, ω^2 and R^2.

3.5 Discussion

3.5.1 Weather

The study began late in 2006. With eight sites in total, four site controls and four sites received combinations of treatment intervention of the following methods (seeded and limed, gully blocked, geojute and heather brash). The vegetated control LD-F was considered the ideal site scenario at Bleaklow and the bare controls B-G, B-F were what would occur without intervention. Ecological restoration of Bleaklow impacts on WTD, DOC and E_4/E_6 are discussed in terms of the eight sites (year 2008) and the six site (years 2007 to 2011) comparative studies, as per the results section layout. Studies have found significant relationship between DOC and temperature (e.g. Clark *et al.* (2005), Dinsmore *et al.* (2013), Pawson et al. (2008) as well as rainfall (e.g. Jager *et al.* (2009), Worrall *et al.* (2006) and Yallop *et al.* (2010)). Weather variables must therefore be considered in order to find what the potion of soil pore water DOC concentration data variation, is attributable to site effects (i.e. management type) verse natural seasonal.

3.5.2 Vegetation

3.5.2.1 Vegetation in 2008

The immediate target of the restoration was vegetation cover. According to the JNCC definition, none of the Bleaklow study sites fit the definition detailed in the introduction of a “good bog” neither in 2009 nor in 2011, with the exception of possibly the naturally revegetated site (in 2009). Analysis of the gully block site vegetation functional groups could not be made, as no quantitative data was recorded on the site. It can be said that the site was not bare based on anecdotal and photo evidence.

In 2009, three years into monitoring on Bleaklow, analysis of vegetation dominance in 2009 found that across the eight sites, bare soil was the most dominant cover. In 2009, it was evident that after three years of monitoring, the revegetation of untreated bare soil sites would be very difficult without intervention. As found at the bare flat and bare gully, which had all bare soil collars, their plots had little or no vegetation with the exception of one plot which was dominated by vegetation from the sedge functional group. Vegetation from the sedge function group was the second most dominant vegetation cover across all of the Bleaklow sites and found to be the most dominant cover type at the least disturbed vegetated site. The presence of sedge (*Eriophorum)* across Bleaklow could explain how sedge was able to naturally colonise the bare gully site without intervention or the introduction of seeds to the site.

3.5.2.2 Vegetation in 2011

In 2011, after a two year period since the last survey (six year period after monitoring began), the bare untreated sites remained dominantly bare at both bare gully and bare flat. This is likely due to bare sites losing much of their viable seed bank (Salonen 1994). The lack of revegetation on the bare control sites on Bleaklow supported the findings by Lavoie and Rochefort (1996), indicating that in some sites on Bleaklow, without restoration, the possibility that vegetation could re-establish is low even after six years.

The vegetative control had limited exposed peat and was dominated by sedge, although it was the least disturbed of the sites. Its ecological state was still not considered in a “good” condition due to the low biodiversity and absence of *Sphagnum spp.* Sliva and Pfadenhauer (1999) found that the benefit of seeding and liming, in vegetative restoration, was enhanced through soil stabilisation techniques. This is in agreement with the findings on Bleaklow. Specifically, the sites with seeding and liming had higher bare cover than the site with geojute, which had greatest success in revegetation and bare soil cover reduction. Thus, on an ombrotrophic dry peatland such as Bleaklow seeding and liming alone would not be sufficient to revegetate bare soil within six years. Sliva and Pfadenhauer (1999) found that heather brash had greater success in reducing bare cover than geojute, contradicting the findings on Bleaklow. This finding is supported by Price (1997), which suggests the use of mulch on an extracted bare peat surface could increase soil moisture and result in water table elevation and create conditions for the vegetation to establish more easily.

The dominance of bare soil at half the heather brash plots is potentially a site specific phenomenon difference, as expected. There is also a possibility the site location of heather brash may have resulted in greater erosion as a result of footfall from the adjacent Pennine way, whereas the geojute site was very isolated and would have been more difficult to access. Thus it would have incurred less erosion associated with walkers. The use of mulch assisted transitioning the vegetation from a non-natural lawn grass community to a sedge dominated community similar to that of the least disturbed vegetated control. Findings by Price *et al.* (1998) indicated mulch could increase *Sphagnum spp.* re-establishment, although none was found at the heather brash site. This is likely due the water table depth being too deep. Furthermore, with the exception of the naturally revegetated site, none of the sites had observed *Sphagnum spp*. The site had no restoration method used, however its site morphology (i.e. gully or interfluve) and hydrology allowed for the establishment of *Sphagnum spp.* that reduced the amount of bare peat. The presence of *Sphagnum spp.* would decrease bare peat erosion and encourage vegetation growth as found by Holden et al (2008) in Trout Beck in the North Pennies, where by the presence of *Sphagnum spp.* was associated with ~8% reduction of bare surface area over a twenty year period. The development of *Sphagnum spp.* is likely supported by the WTD which was found to be significantly shallower (nearer the surface) at the site. The WTD at the naturally revegetated site was relatively shallower than the least disturbed vegetated site, as analysis found that in 2008 the WTD at the naturally vegetated site did not significantly differ to the least disturbed vegetated control. Furthermore, the treatment site with shallower WTD than the bare site controls was the blocked gully site.

In both 2009 and 2011, plots had few levels of dominances (D1-D4), in which one cover type was most important. This could indicate a poor biodiversity across all sites. Important bog forming species such as *Sphagnum spp.* were poorly represented on Bleaklow in 2009 and in 2011. Vascular plants have higher decomposition rates than *Sphagnum spp.* (Moore *et al.* 2007). A vegetative shift could lead to more decay-resistant litter, in addition to increased carbon accumulation (Heinemeyer *et al.* 2010).

Litter was observed in 2011 where it was not in 2009. This could be due to the increase in litter forming vegetation onsite or due to the timing of the sampling. The 2009 survey was conducted in July, while the 2011 survey was in August. Litter is an important part of peat deposition (Clymo 1984), lower rates of litter decomposition are important for increased peat deposition which was observed at the heather brash and geojute sites (discussed further in chapter 4). Thus, restoration method which would encourage leaf litter deposition would be advantageous for the longevity of the peatland. Peat and litter decomposition are significantly linked to WTD; drier sites had higher decomposition rates (Moore *et al.* 2007), The investigation of water table in relation to site vegetation was important in order to consider the effect of restoration on DOC, as peatland decomposition and vegetative environmental stress is linked to DOC production (Crow and Wieder 2005, Mezbahuddin *et al.* 2014).

3.5.3 WTD

3.5.3.1 Water table depth (2008)

Restoration of the hydrological regime is vital in order to restore ombrotrophic peat bog to a peat forming system (Holden *et al.* 2004). Water saturation of peat and anoxic conditions retard the process of decomposition by heterotrophs (Hilasvuori *et al.* 2013). It is therefore important to determine the effect of restoration on water tables as WTD and soil moisture have been linked to soil pore water DOC concentrations (e.g. Clark *et al.* (2008) and Price *et al.* (1998)).

Re-establishment of vegetation and an intact-peatland hydrological regime was a principle aim of the restoration programme (Evans *et al.* 2005). Of the four sites treated, only one site (SL.B-G) received gully blocking, a specifically designed modification, to alter the site hydrology. In 2008, two years into the study treatment of sites influenced WTD, the shallowest WTD was found at the vegetated control, with ~ 9 times shallower WTD than the bare controls. At the gully blocked site the WTD was statistically indistinguishable to the naturally re-vegetated site, these two sites had the most similar WTD to the vegetated control. The naturally revegetated site had ~3 times shallower WTD than the bare controls. This may be a reflection of the vegetation cover dominance, as *Sphagnum* spp. dominated plot had a WTD 2.5 shallower than that of bare soil dominated plots. The differing site morphologies may also be responsible for differences in local hydrology and WTD, specifically the effect of natural gullies (Daniels *et al.* 2008) and a gully drawdown effect (Allott *et al.* 2009). The WTD at naturally revegetated site can also be explained by Clay *et al.* (2012), who found that the shallowest water tables were located on gully floors not interfluves (flat). Of the treated sites, only the gully blocked and heather brash site differed to the bare control sites. Monthly variation was important (specifically differences in monthly PAR) in explaining water table variation. Thus, it was evident that in 2008 at the two years stage of the study monitoring, although seasonality can influence WTD, the benefits in raising the WT nearer the surface can only be achieved through deliberate intervention.

Variation in WTD was further explained through analysis of the data using the factor plot cover dominance in place of site. It was found that PAR was again important. Furthermore, bare soil dominated plots had the deepest WTD, this further supports the need to employ revegetation efforts to reduce depth of WT, as it contributes to explaining the deeper WT found at the bare controls and the seeded and limed site. The functional groups with the shallowest WTD were sedge (-68.3 mm) and *Sphagnum spp.* (-55.3 mm); these were the functional groups dominant at the vegetated control and naturally revegetated site. The plots with the deepest WTD were dominated by shrubs followed by grass, this finding is supported by Urbanova *et al.* (2012) who linked vascular plants to deeper WT. Thus in order to raise the WTD toward the soil surface over a short time scale (up to 3 years post restoration): reduction in bare cover dominance is key as any vegetation cover is preferred to bare soil. Furthermore where gully blocking is not possible, the use of heather brash would be preferential to only seeding and liming.

3.5.3.2 Water table depth (2007 – 2011)

Over the five year study monitoring six Bleaklow sites, WTD variation was linked to differences between study sites. Furthermore, seasonal and annual variation was as important as variation between site plots. The factor month was related to seasonality (linked to environmental variables and vegetation). The importance of inter-annual seasonal variation explains the general increasing trend in water table depth over the five year study period. Environmental variables were investigates and it was found that rainfall was linked to WTD, at multiple temporal scales (monthly mean rainfall; and rainfall total at a short time period, 1 - 2 days prior WTD in-field measurement). Temperature was important in influencing WTD at a short temporal scale (1 - 2 days prior field monitoring). Increased rainfall volume was correlated to an in increase in WT. Hence increased rainfall would raise WTD closer to the soil surface. However, temperature was more important on a shorter time interval. Therefore rainfall and temperature events could influence WTD. However, the importance of rainfall was dropped out of the ANCOVA as the variables most important to WTD other than the factor site were the environmental covariates PAR and air temperature (measured during sampling period). The significant difference between bare control sites and the naturally revegetated site indicate that site morphology influences WTD, as neither of the sites received treatment and yet they significantly differ in both WTD and vegetation cover.

Analysis of WTD using dominant plot cover as a factor instead of site, found over the five year period, vegetation cover was important but explained less WTD variation than site could. In term of reduced depth of WT any vegetation cover was preferential to bare soil cover. The low of sensitivity of WTD to rainfall and the importance of site over vegetation cover in explaining WTD variation, support the finding by Allott *et al.* (2009) there is a distinct temporal-hydrology in Bleaklow blanket peats sites. The intact least disturbed sites such as that at the least distubed vegetated control, had near-surface (<10 cm) WTDs most of the time.

Water table fluctuation according Breeuwer *et al.* (2009) is an important factor in controlling vegetation type, whereby increased occurrence of periods with deeper water tables may bring about a shift in dominant *Sphagnum spp.* as well as a shift from grasses to sedge cover, and could induce a shift towards vascular communities, as found by Urbanova *et al.* (2012). As WTD was deepest at the bare soil control sites (bare gully and bare flat) it is very difficult for pioneer species, especially with short roots to establish and access the water.

Furthermore, WTD fluctuation influences total biomass production and thus could impact carbon sequestration and hydrological characteristics of bogs (Breeuwer *et al.* 2009). The site with the smallest CV was the bare gully site; the bare soil site with the significantly deepest WTD. The sites with the largest CV are associated with greater annual fluctuation, as found at the geojute soil stabilisation, which was the restoration site with lowest bare soil cover dominance. Bare sites on Bleaklow had deeper water table, no vascular species and less WTD fluctuation than other Bleaklow sites. However, as the WTD at bare sites were significantly deeper than the other sites, the WTD fluctuation would have less interaction or impact on vegetation present at the peat surface.

The study indicated that although treatment of sites raised WTD closer to the surface, as a result of change of vegetation, the hydrological regime was difficult to restore. This is because eroded and intact peats have clearly distinct hydrological regimes (Allott *et al.* 2009), with disturbed and eroded bog having a lower water retention capacity than pristine bogs (Daniels *et al.* 2008, Rothwell *et al.* 2009). Results from the sites used in this study suggest that peat surface treatments (i.e. seeding, liming and fertilisation) either have no or only a small impact on WTDs within a relatively short time scale of years. For a decreased depth in WT at a two year time scale, intervention using gully blocking is recommended (depending on site morphology). Additionally stabilisation techniques (are associated with sedge growth, as sedge dominated plots) had the shallowest WTD after two years and across the five year study. Such techniques would be recommended in future bare peat hydrological and ecological restoration projects.

3.5.4 DOC

3.5.4.1 DOC (2008)

After two years of monitoring, in 2008 there was an indication of site DOC fluctuation (greater CV indicated greater fluctuation). The site with greatest success in reducing bare soil dominance in the first two years of the study, SL.Ge-G, had the largest CV. The lowest CV was at bare control B-G, where no restoration efforts were made and the site remained dominated by bare soil cover. Analysis of variance found that in 2008 monthly variation was more important than site. The addition of mean temperature of the month preceding that of sampling month, was found to be a more important variable in determining DOC concentrations than site. In 2008 peak temperatures were reached in August followed by peak total rainfall volumes in September, and then the peak in soil pore water DOC (191.80 mg C/ L) was in November. The lowest concentrations (between 49.80 and 59.01 mg C/ L) were present during the late winter and early spring (January to April).

Seasonality is an important DOC explanatory variable (Clay *et al.* 2012). Temperature averages over long periods of time (of month instead of days) are useful in models explaining DOC variation. Some of the DOC species can be produced in a relatively short time scale (e.g. route exudate) and some DOC over relatively longer time scales (related to peat break down).
The use of the correct covariates at the correct time scale when developing models for DOC, can allow differences in DOC attributable to site variation to become more apparent. It was found that the temperature of the month preceding sampling month was important in explaining DOC concertation. This lag effect of mean temperature on Bleaklow DOC concentrations can be explained by: a) timing of sampling, as 65.5% of field monitoring and sampling was conducted in the first 2.5 weeks of each month; b) soil pore water DOC data is a measurement of concentration not flux, thus there is a potential lag in DOC production in

relation to weather events (in the days prior sampling). Temperature can influence the microbial activity and DOC production (Mitchell *et al.* 2008, Wallage and Holden 2010), and rainfall events 'flush out' and DOC adsorbed onto the peat soil particles (Clark *et al.* 2007, Worrall *et al.* 2002). Unfortunately the rainfall data is an accumulative value, therefore detail of individual rainfall events are lost, this is possibly a reason why total monthly rainfall was not a significant factor during multivariate analysis.

The importance of the temperature of the months preceding the sampling month indicted a lag between DOC productions and the system flush. This flush mechanism is discussed by Worrall *et al.* (2006) as the autumn flush, in which the labile organic matter produced during the summer is flushed out during autumn months due to increased rainfall. In 2008, the DOC concentrations at the treatment sites were not significantly different to either the vegetated control sites or the bare soil controls. Clay *et al.* (2012) discussed findings indicating the vegetated controls were the only two sites which significantly differed to one another (naturally revegetated soil pore water DOC greater than the vegetated control). The variation in DOC concentrations was greatest during the late summer period in particular at the naturally revegetated site. Although there are observational differences, there was a lack of statistical difference between concentrations of DOC between the bare sites and the treated sites. This is could be due to the treated sites slow establishment of vegetation cover, particularly in the case of the seeded and limed and the heather brash which were dominantly bare. The lack of an active vegetation layer to stimulate a soil microbial community on bare and restoration sites, means there was likely to be little activity driving DOC production (Aguilar and Thibodeaux 2005). Analysis of DOC vegetation in 2008 found that without the use of temperature as a covariate, significant differences between DOC were not apparent. In 2008 the bare control DOC did not differ to the vegetated controls or the sites with treatment. However analysis by vegetation dominance revealed that *Sphagnum spp.* dominated plots were associated with relatively shallow water table depths and higher soil pore water DOC (as found at the naturally revegetating site) than the bare soil control and treatment sites. Grass and sedge species had significantly lower soil pore water DOC (as found the vegetated control). The plots dominated by bare soil cover and shrub plants did not significantly differ. Note the vegetated control sites had the with the shallowest water table were found to significantly differ.

3.5.4.2 DOC (2007 – 2011)

The Bleaklow study sites exhibited a general decreasing trend in DOC concentrations between 2007 and 2010, and in 2011 there was a significant increase in concentrations. The least disturbed vegetated control however had less fluctuation in DOC concentrations over time. It was found that inter-annual seasonality variation was important, as was the interaction of site with year. Furthermore the variation in DOC between site treatment plots was greater than it was between the treatment sites.

Temperature at several time steps was identified as being significantly relating to soil pore water DOC, which is supported the findings of Heathwaite (1993), in which elevated temperatures can elevate decomposition, thus increase in temperatures can result in increased soil pore water DOC (Worrall and Burt 2004). Bonnett *et al.* (2006) stated the seasonal effect of temperature on DOC may be explained by increased plant and microbial activity. Temperature of bimonthly mean was found to best explain the soil pore water variation. However, month nested within year remained the most important factor. Although it explained a very small portion of variation, temperature was found to be more important in influencing DOC concentrations on Bleaklow than inter-site variation was. It was revealed that the bare gully control had significantly higher DOC than the bare flat interfluve. The bare gully control site had relatively high DOC concentrations and deep water table depth. At dominantly bare sites it would be expected that oxidative and microbial biological processes drive the production of the DOC (Mcknight *et al.* 1985). Of the three sites with treatments, two (seeded and limed and the heather brash) did not differ from the vegetated control. They also had lower DOC concentrations than the bare gully, but not bare interfluve site. Seasonal variation in DOC production is associated with microbial communities and root exudation (Mitchell *et al.* 2008, Wallage and Holden 2010). Despite the reduction in bare cover associated with restoration treatments at sites, the concentration of DOC in soil pore water was not significantly different to the controls. Furthermore no significant difference was found between DOC at the different dominate vegetation types.

As previously mentioned, temperature variation was important in explaining DOC concentrations. The lowest soil pore water DOC concentrations for all sites monitored were observe during winter months. As the target of restoration is to encourage sites revegetation and reduction of bare cove sites; it is important to refer to changes over time and how the changes are relative to the controls. It was found that at the end of the study (2011) DOC was significantly greater than in 2007. Bleaklow soil pore water DOC concentrations significantly dropped in 2010 (the year with lowest rainfall and high summer temperatures), followed by a small increase in 2011 (the year with the highest annual rainfall and low temperatures between summer and winter months). The drop in observed DOC in 2010 could be a result of smaller sample size during the summer; however given that the models accounted for monthly temperature variation, the drop in DOC in 2010 is most likely due to differences between years instead of changes between the sites

Despite the changes in vegetation on Bleaklow, restoration sites displayed little evidence of changes in DOC concentrations. Site pH was lower at sites dominated by bare soil cover; however pH was not a significant covariate in explaining DOC concentrations. This could be explained by changes in DOC production but not DOC mobility. Palmer *et al.* (2013) found: a) a negative correlation between pH and DOC concentrations; b) buffering of acid deposition (i.e. sulphates) varied depending on soil base components. Upon addition of sulphate, soils with a greater sulphates buffering capacity result in a decrease in DOC concentrations. Blanket bogs such as Bleaklow are less sensitive to sulphate input variation than shallow confined peat according to Clark *et al.* (2011). This could explain why the use rainfall sulphate concentrations as a covariate within ANCOVA did not improve the model or the ability variate in predict DOC concentrations. Clark *et al.* (2005) found that drought conditions and shallow water table depth were coupled with increased soil sulphate. These conditions were linked to increased pH and increased ionic strength which suppressed the release of DOC. It is a possibility that the low water table conditions simulated draught conditions resulting in reduced DOC solubility therefore suppression of soil pore water DOC concentration. Soil pore water sulphate concentrations were not investigated, and although pH was not a significant covariate in explaining DOC concentrations it was it terms of E_4/E_6. The differences in DOC species were investigated using the UV-Vis absorbance proxy data (E_4/E_6).

3.5.5 E_4/E_6

3.5.5.1 E_4/E_6 (2008)

The ratio of E_4/E_6 was used as a general indicator of organic molecules MW. Specifically high E_4/E_6 is associated with relatively lower MW (fulvic components), and lower E_4/E_6 is associated with relatively higher MW (humic components) (Carlsen *et al.* 2000). In 2008, E_4/E_6 variation was better explained through variation between sites than between months (unlike DOC concentration which were largely determined by seasonality). A correlation between pH and E_4/E_6 indicated that increases in pH would result in a decrease in E_4/E_6 toward the humic range. The findings on Bleaklow in 2008 are supported by that of Scott *et al.* (1998) who found a significant relationship between pH and DOC. The relationship between DOC concentrations and E_4/E_6 was not clear between sites. This result can be explained by Chen *et al.* (1978), who found that E_4/E_6 ratios were not concentration dependent, but in fact the ratio were correlated with acidity and pH (a measure of hydrogen protons which were related to acidity).

Kukkonen *et al.* (1992) determined the ratios of E_4/E_6 humic acids ranged from 5.44 to 5.7 and fulvic acids ranged from a ratio of 8.88 to 9.9. Other values for humic were given as 3.8 to 5.8 and fulvic as 7.6-11.5. Carlsen *et al.* (2000) gave a ratio of >11 for fulvic components. These values are variable according to sample sources. For example, a high amount of lignin from decaying plant material would results in a higher molecular weight and lower E_4/E_6. Based on the research during analysis of E_4/E_6, a 6.5 ratio was used as an indication samples were within a humic range, while anything above a ratio of 7 was considered within the fulvic range.

In 2008, analysis of soil pore water E_4/E_6 found that the least disturbed vegetated control had the lowest ratio followed by the bare (gully) control sites. E_4/E_6 of the heather brash and gully block samples were greater (more fulvic range) than the bare (gully) controls. Analysis by vegetation functional group indicated that sedge dominated plots had the lowest E_4/E_6 ratio. The higher (more fulvic range) E_4/E_6 at sites with treatment indicated that differences in DOC compounds could be attributed produced to the recently established vegetation. However, the lack of vegetation at the bare flat site (also with more fulvic E_4/E_6) did not support this rational. Instead the differences found in previously bare restored sites could potentially be changes in DOC production and mobility. The least disturbed vegetated control exhibited the lowest mean E_4/E_6 (within a humic range), this is possibly related to the litter layer development and peat hummification. Furthermore the relatively low E_4/E_6 found at the bare (flat) control could indicate a the production and mobility of humic compound derived from the decomposition of the deeper layers of the hummified bare eroded peat.

3.5.5.2 E_4/E_6 (2007 – 2011)

Between year 2007 and 2011, all the sites observed a general decrease in E_4/E_6 and shift from a more fulvic to humic range DOC. This finding indicated an effect at locality scale. Analysis of variance found that monthly and inter-annual variation were important factors identified. Site and water table depth were equally important in determining E_4/E_6 variation, followed by soils pore water DOC concentration and pH. Temperature was not related to the speciation of the DOC, although important in its production related to erosion and biological activity (Mcknight *et al.* 1985). The site with the highest E_4/E_6 was the seeded and limed site, also the restoration site with greatest bare soil cover dominance. As indicated in 2008, the fulvic DOC production could be due to the relatively newly established vegetation in addition to microbial activity related to the peat breakdown. The sites with the lowest E_4/E_6 over the five years was the vegetated control (sedge dominated), and the two bare soil controls (bare gully and bare flat). The soil pore water humic range in DOC is likely as a result of litter layer and bare peat breakdown. Furthermore, analysis of E_4/E_6 by most dominated plot cover revealed that vegetation cover impacted the ratio. Plots dominated by sedge had significantly lower E_4/E_6 than both bare and moss dominates sites. Thus across the five years, the geojute site did not significantly differ to the heather brashed site. However, the heather brash site, a vegetation cover most similar to the vegetated control (sedge dominated), showed the greatest rise in water table to the surface. The significant reduction of E_4/E_6 over time, significantly lower periods coincided with the lowest DOC measured in the soil pore water on Bleaklow.

Investigation into the importance of pH on DOC speciation and concertation was conducted through ANOVA. The ANOVA indicated that pH of soil pore water was important in explaining variation of E_4/E_6, therefore pH was related to DOC speciation but not concentration. The importance of acid forming S and N species where excluded from the DOC concentration ANCOVA as they were relatively unimportant compared to WTD, temperature, site and year. The covariates S and N were also unimportant in relation to E_4/E_6.

That variation in the ratio of E_4/E_6 followed a similar annual trend to the variation in pH. Overall Soil pore water pH was lower at bare sites than at the more vegetated site (control and site with soil stabilisation). By the end of the study period in 2011, both the bare gully and bare flat sites had a pH <4; such low pH conditions are sub optimal for soil microbial communities and seed germination (Andrus 1986, Caporn *et al.* 2007, Smith and Read 1997). The least disturbed vegetated control exhibited a slight decrease in pH over time. This indicates an overall effect at locality scale. Furthermore, on Bleaklow the presence of vegetation and low bare peat cover dominance is associated with a higher pH than bare peat dominated soil. Sites with stabilisation techniques increased pH by (increased by: 0.06 at heather brash site, by 0.33 at geojute sites).

As an effect at locality scale was identified, rainfall acid deposition and pH concentrations were investigated. No direct link was established between rainfall acid species and soil pore water DOC concentrations or speciation using ANCOVA. Soil pore water pH was an important covariate in explaining E_4/E_6 variation. At the least disturbed vegetated control and the sits with stabilisation techniques (heather brash and geojute), soil pore water pH was more similar (20% lower than) rainfall pH at that at the bare beat dominated sites, at which soil pore water pH was 30% lower than the rainfall. This indicates the sites less dominated by bare soil cover were more able to resist reduction in pH. The late summer periods are associated lagging behind a peak in a seasonal peak in E_4/E_6. This is likely due to the inputs of root exudates produced in periods of high photosynthesis in addition to break down of peat during drier condition warm conditions. Thus, the release of DOC and amino acids results in the drop in pH during the summer period. The difference between the rainfall pH and the soil pH during the summer period is increased indicting the rainfall is not the primary influence to soil pore water pH and $E_4/E_{6.}$ Furthermore, without intervention all sites could have a drop in pH in the future, ultimately reducing the capacity for vegetation to establish as found at the untreated bare control sites.

The seasonal fluctuation in E_4/E_6 coincide with so called ‘autumn and spring flush’, in which ground thaw during spring and increased rainfall could be responsible for potential increases in DOC and flush out more soluble fulvic range material during that period. The importance of water table in the model could be due to the solubility of fulvic acids being greater than humic acids. The vegetated control had the shallowest water table depth and E_4/E_6 ratio within the humic range. Hence, despite the lower solubility of humic components the high water table allowed for greater humification of the peat. Having considered the complex interactions and variables, supports a rational that disruption (restoration) causes: a) a shift in bio- hydro- chemical- conditions b) increased microbial activity related DOC production. These two mechanisms are potentially the reason why the least disturbed vegetated control has similar E_4/E_6 to the bare soil controls but not the sites where restorative treatments have been implemented.

The use of E_4/E_6 ratio is insufficient to describe changes in specific species; however it does give a good indication of a shift in DOC composition. Based on the evidence on Bleaklow, soil pore water, concentration of DOC production within sites are difficult to manipulate. However it would be expected that sites previously bare then revegetated would have an increased fulvic range of DOC. Once a stable ecosystem is established with a shallower water table (within the range of the vegetated control), the establishment of a healthy depositing litter layer could be expected to shift the bio- hydro- chemical environmental variable and promote a shift in DOC and greater humic range species in the soil pore water. There are benefits for bare peat revegetation, specifically in the use of stabilisation techniques. They are are not to bring about immediate reduction in DOC, but to change the DOC composition. As well as raise the water table depth and increased soil pH to a level in which vegetation can more easily establish and stabilise the peat to potentially reduce loss of carbon through aerobic break down and surface erosion.

This study emphasised the importance of multiannual monitoring. The general trends and statistical analysis of data are important when considering the long term effects of revegetation treatments on: vegetation cover, water table depth and soil pore water quality. The value of long term monitoring was emphasised in the final year of study (2011), as there were significant shift in DOC, E_4/E_6 ratio and WTD which contradicted the general trend observed in the four previous years. It is therefore vital to use covariates to explain the data variation attributed to natural variation (e.g. PAR and air temperature). The use of such covariates enables the explanation data variation attributed the variation between sites with and without restorative treatments

3.6 Conclusion

The study was conducted to assess the success of bare peat restoration methods used on Bleaklow; on soil pore water quality related predictors of DOC and absorbance. This study put the small spatial site variations (at plot scale) in context of the larger temporal changes.

Analysis showed that:

- Sites with restorative treatments (previously bare) had reduced dominance of bare soil cover relative to the bare soil controls (in 2008, within three years of treatment). The sites treated with soil stabilisation techniques had lower bare cover dominance than the site in which only seeding and liming was conducted (In 2011, the end of five years study; eight years after treatment).
- The water table fluctuated seasonally and annually. Site morphology and treatment had an important role in influencing WTD.
- The hydrological regime was difficult to restore within three years without targeted intervention (gully blocking).
- In the long term (5 years), the least disturbed vegetated control had the shallowest WTD. The bare control sites had a WT x10 deeper than the control, while the site with restoration treatment were between x8-9 times deeper than the least disturbed vegetated control. The site treated with heather brash had the best improved WTD.
- Concentrations of soil pore water DOC are influenced more greatly by temperature as a function of seasonality than by site restoration.
- Disruption through restoration created a shift in DOC composition. The promotion of a stable ecosystem (like the least disturbed vegetated control site) is associated with less soluble DOC within the humic range. The relatively more mobile fulvic range is associated with newly established vegetation and DOC production in summer months. A shift in DOC composition can be achieved through raising water table depth and pH, and promoting sedge vegetation functional group establishment.

- Sites treated through seeding and liming (previously bare) alone could not maintain a pH >4, required for a seed germination. Sites treated with seeding and liming in addition to soil stabilisation techniques (heather brash or geojute mesh) were better protected from over acidification.

The study findings emphasise the importance of long term monitoring. It is proposed that changes in DOC were attributed to, restoration and consequent to changes in site chemo-hydrology which are not fully restored by revegetation methods alone. To increase the rate of revegetation and reduce water table depth; peat surface stabilisation techniques such as geojute and heather brash should be coupled with water table restoration methods such as gully blocking (depending on site morphology).

4 *Bare peat restoration effects on CO_2 fluxes*

4.1 Rationale

Many nations now conduct restoration to improve degraded peatlands (Verhoeven 2014), as restoration is largely beneficial to many ecosystem services (Parry *et al.* 2014). The benefits and reduction in carbon losses, justify the costs of restoration (Moxey and Moran 2014). To combat the impacts of blanket peat degradation on carbon fluxes, hydrology and ecology (Holden *et al.* 2011, Holden *et al.* 2007, Lindsay 2010, Parry *et al.* 2014), work in the Peak District and South Pennines has been conducted to revegetated bare and degraded bog.

The research presented in this chapter formed part of two published papers[1,2]. Existing research has so far demonstrated the largest fluxes of carbon from a degraded upland peat system is in the form of POC (Evans *et al.* 2006, Pawson *et al.* 2008). These losses of POC in upland catchments have resulted in the significant reductions in reservoir water storage capacity (Labadz *et al.* 1991, Yeloff *et al.* 2006). Research on revegetation and bare peat restoration has focused on erosion (Evans and Lindsay 2010), and water table depth links to carbon fluxes (Allott *et al.* 2009, Daniels *et al.* 2008). Clay *et al.* (2012) found that soil pore water at gully sites had higher DOC concentrations than at interfluves. Deeper water tables are associated with erosion and the drainage of peatlands which in turn are thought to increase the exports of DOC from peatlands to surface water (Clay *et al.* 2009, Strack *et al.* 2011).

[1] Dixon S. D., Qassim S. M., Rowson J. G., Worrall F., Evans M. G., Allott T. E. H., Boothroyd I. M. 2014. The impact of peatland restoration on CO_2 fluxes and water table depths from a climatically marginal upland blanket bog. Biogeochemistry 118 159-176.

[2] Qassim S. M., Dixon S. D., Rowson J. G., Worrall F., Evans M., Bonn A. 2014. A 5-year study of the impact of peatland revegetation upon DOC concentrations. Journal of Hydrology 519: 3578-3590.

Previous studies on CO_2 fluxes have found that solar radiation (Kim and Henry 2013), soil moisture (Gomez-Casanovas *et al.* 2012), water table depth (von Arnold *et al.* 2005), air temperature (Lloyd and Taylor 1994), soil temperature (Danevčič *et al.* 2010), and vegetation cover are important drivers for CO_2 exchange (Glatzel *et al.* 2006). Caporn *et al.* (2007) found revegetation (using lime and fertiliser) of bare peat, with interventions, led to increased rates of CO_2 emissions compared with untreated bare peat sites. Biasi *et al.* (2008) found, on Finnish cultivated peat, that in the first 2–4 months after the application of lime, a maximum of 12% of monthly CO_2 emissions were as a result of lime derived CO_2. A study by Waddington *et al.* (2003), on bare extracted lowland bogs in Eastern Quebec, found that the use of mulch to restore vegetation resulted in an initial carbon loss from the system as a result of mulch decomposition. Thus intervention can result in the increase of CO_2 in short term after restoration.

The evidence in the previous chapter (Chapter 3) on bare peat restoration and DOC, found that temperature and annual variation had influenced DOC concentration more than revegetation treatments did at a relatively short time period of five years The finding discussed in Chapter 3 of this thesis explained that vegetation does not easily re-establish on bare sites without intervention. Bare soil sites incur higher losses of carbon in the form of DOC. At bare gully unrestored sites DOC losses were 2 times greater than least disturbed vegetated sites. Out of the three sites with restoration, those with stabilisation techniques had greater reduction in bare surface area. However, DOC concentration did not significantly differ to the bare controls and only the site with geojute mesh installed had higher DOC losses than the least disturbed site. Treatment through revegetation had an important impact on DOC composition, with a higher ratio of fulvic components relative to both bare and least disturbed vegetated controls. The interventions and restoration techniques used to restore the degraded peatlands are discussed in detail in Chapter 3. The specific aims (and methods) of restoration share a common aim: to modify the peatland so that it starts to approach some conception of 'pristine' condition. Importantly, a functioning pristine peatland is one in which accumulation rates exceed the loss of organic material.

Chapter 3 considered site losses through the fluvial pathways in the form of DOC. This chapter focuses upon CO_2 fluxes. As almost all carbon enters peatlands through photosynthesis, peatlands rely on the rate of photosynthesis being greater than all outward carbon fluxes to accumulate carbon and grow. It is therefore important to monitor CO_2 fluxes on restored peatlands to assess the impact of restoration on this vital carbon pathway. The results are then used to assess which restoration method or scenario would provide the least carbon losses, and to determine whether areas of peat subject to restoration are in a more favourable condition, from the perspective of CO_2 flux and water table depth, than unrestored areas of degraded peat. Moreover, this research compares different restoration techniques with 'least disturbed' areas.

4.2 Aim

The aims of this study were to investigate the effects of re-vegetation upon CO_2 fluxes from the peat, and assess which bare soil site treatment technique would give rise to reduction of carbon losses and improve potential for a site to become a net sink. The null is that there is no significant difference in CO_2 flux between revegetation treatment sites relative to the bare control sites. This hypothesis will be tested through analysis of a multiannual data set of CO_2 flux, R_{eco}, NEE and P_g, and WTD; all relative to a bare and vegetated controls. Analysis of seasonal and annual data trend will be conducted using graph, comparative analysis of the data, ANCOVA GLM models and post hoc tests to find where differences lie.

4.3 Methodology

4.3.1 Study Area

The sites used in this study are all located on the Bleaklow Plateau (clustered around 53° 46’ N 1° 84’ W) within the Peak District National Park in northern England. Chapter 2 gives detail on the site experimental design and research methodology used in all research chapters.

4.3.2 Field monitoring

The field monitoring follows the detailed methodology section in Chapter 2. Data gathering for this study began in January 2007 and finished in December 2011.The data were gathered once each calendar month from each of the monitoring site (8 sites during 2008 and 6 sites between 2007 and 2011). Additional weather data from Manchester University and Defra (detailed in chapter 3) were used. Recession data were gathered using an infield pin grid method (obtained from Manchester University), available for all but two of the sites (SL.B-G and NRv-G). The erosion pins were used to measure recession rates over time which are indicative of peat soil erosion and deposition.

This study employed a portable infra-red gas analyser (EGM-4, PP-Systems, Hitchin, UK) with a clear 20 cm tall, 15 cm diameter acrylic closed chamber (CPY-2, PP-Systems, Hitchin, UK) to measure fluxes of CO_2 from the permanently installed gas collars. The protocols employed in the measurement of CO_2 fluxes were in line with previous research in the area (Clay *et al.* 2012, Rowson *et al.* 2010, Worrall *et al.* 2011) This method was selected as it enabled the measurement of above and below ground productivity simultaneously (Streever *et al.* 1998).

Ecosystem respiration (R_{eco}) fluxes were measured in the absence of light by covering the CPY-2 chamber with a tightly fitting u-PVC sleeve that blocked all photosynthetically active radiation (PAR) from entering the chamber. Net ecosystem exchange (NEE) fluxes were measured using the CPY-2 chamber without the u-PVC sleeve, i.e. with full sunlight entering the chamber. While CO_2 fluxes were being measured PAR (µmol m^{-2} s^{-1}) and air temperature (K) probes were in operation within the chamber. The chamber was sealed to the gas collar by a tapering metal skirt so as to prevent interaction of chamber and ambient air during measurement. The air within the chamber was circulated by a 12 V fan to keep it well mixed and prevent its stratification. Fluxes of CO_2 (R_{eco} and NEE) were measured over two minute intervals in which CO_2 concentration data were recorded at four second sampling intervals. The fluxes of CO_2 were calculated by using the gradient of the linear regression of chamber CO_2 concentration with time, and are reported in units of CO_2 m^{-2} h^{-1}. Direct measurement of gross photosynthesis (P_g) was not possible; it was derived by subtraction between the R_{eco} and NEE fluxes. When referring to absolute data, the sign convention used for reporting CO_2 fluxes throughout this chapter is that all fluxes are considered relative to the atmospheric pool (i.e. R_{eco} fluxes are positive and P_g fluxes are negative). After measuring CO_2 flux water table depth (WTD) measurements were taken on each plot using a conductivity probe (details of how WTD is measured is in Chapter 2; absolute WTD data are negative, more negative values indicate greater depth from the surface).

4.3.3 Statistical methods

This study had several limitations. Firstly, direct consideration of restoration treatment is a factor, as it was not possible to repeat treatment between sites. There was no combination of site and treatment available on the Bleaklow Plateau. Secondly, no measurements were collected prior to the wildfire; the differences identified between sites could therefore be ascribed to pre-existing differences between sites rather than to restoration. To assess if a significant shift in CO_2 fluxes occurred, the data were analysed relative to the controls.

As per the thesis methodology section, all data were quality checked for normal distribution and outliers. Data outside of 4 standard deviations of the mean were removed. Analysis was conducted in terms of 'absolute' (units of measure CO_2 m^{-2} h^{-1} for fluxes and mm for WTD) and 'relative data' (no units). The untreated sites as 'controls' are comparators to treated sites. There were four controls sites: bare flat interfluve (B-F), bare gully (B-G), least disturbed vegetated (LD-F), naturally revegetated (NRv-G), and four sites with treatment: seeded and limed (SL-F), seeded and limed (SL.HB-G), seeded, limed and geojute (SL.Ge-G) and seeded limed and gully blocked (SL.B-G). Sites NRv-G and SL.B-G had a shorter data record; therefore analysis on all eight sites was only conducted for 2008. Analysis was also carried out on a multiannual five year data set, conducted using data for six sites.

The same methods for statistical analysis (ANOVA/ ANCOVA) were used for both absolute and the relative data. A relative data set was calculated using monthly mean data. Site monthly mean values (for CO_2 fluxes and WTD) were calculated; the restoration treatment site data were then expressed relatively to the control sites (Bare sites: B-G and B-F) monthly mean values. Successful restoration would mean: i) that restored sites become increasingly less like the bare soil, unrestored controls, ii) the site is losing less carbon than it is taking in. Success could also be sites having no significant difference to the vegetative control (LD-F).

The least disturbed vegetated control was considered the ideal WTD scenario for Bleaklow. It was found that WTD was significantly shallower at the vegetated control sites than the bare control sites (Chapter 3). Therefore analysis of WTD was conducted using relative data (relative to the vegetated control). Analysis was conducted relative to the bare control as

WTD. However as observations were frequently too deep to detect at the bare sites there were data gaps or a possible underestimation of mean WTD during drier months. This could have resulted in a slight misrepresentation of values relative to the bare control. Therefore focus of the analysis was on the relative to vegetated controls data and graphs of mean monthly WTD relative to LD-F were used to illustrate the variation in WTD over the study period (5 years).

Flux data relative to the bare soil controls (B-F and B-G) were analysed using ANOVA. Each of these ANOVA was considered with and without nested factors and covariates. The inclusion of the interaction between the site and project year factors within the models was important due to the absence of a pre-fire and pre-restoration control. Thus restoration on a site would be considered a success if there were significantly greater influx of CO_2 than the control sites. A good result would be a negative flux which would indicate a CO_2 influx from the atmosphere. The interaction was assessed as follows: If a site had a CO_2 flux equal to the control (bare flat and bare gully mean flux), the relative value would equal to one; If the a relative value was greater than one, the site flux was greater than the bare control; if the value was smaller than zero the site flux is smaller than the bare control. Site WTD relative to the control was also analysed in this way. However the relative values were compared to negative one (instead of one) as WTD is a negative value. The output and post hoc analysis results for the sites relative to bare control were included in a summary table and within a graph (along with relative flux post hoc results) for reference during discussion of CO_2 fluxes at sites relative to the bare controls.

Absolute data were first analysed, then compared to data relative to bare control. Both the absolute and relative data were primarily investigated using General Linear Model Analysis of Covariance (GLM-ANCOVA) (detailed in Chapter 2). Predictors (i.e. absolute and relative WTD, R_{eco}, P_g and NEE) were investigated using GLM: factors (e.g. site, month, year), nested factors (i.e. month within year and plot within site), factor interactions (e.g. treatment; gully or flat, and plot cover dominance with month and year) and covariates (e.g. WTD; air temperature and PAR from inside chamber during flux measurement; daily and monthly mean temperature and rainfall from weather monitoring station; and percentage vegetation cover). Models were required to meet a minimum benchmark for statistical significance (i.e. $p = 0.05$). A best fit model was determined by finding the combination of significant predictors that yields the greatest coefficient of determination (R^2). The adjusted R^2 ($adjR^2$) was used in place of R^2 as it is adjusted for the number of explanatory terms in a model. Model residuals were tested for a normal distribution and the residuals were plotted against the fitted values to show that no correlations were still present. If these tests failed then the data were log-transformed and the re-tested for normality and re-analysed. To understand the importance of each significant factor, interaction and covariates effects on the predictor were calculated using generalised ω^2 (Olejnik and Algina 2003). Post-hoc testing of the results between factor levels, using Tukey's pairwise comparisons, was conducted to assess where significant differences lay between factor levels. The least mean squares outputs of the post hoc tests were also reported. Where a dataset was logged or the square root of the data were analysed, the least mean squares were converted back to a non-logged values using exponential function and squared (respectively). This allowed a quantitative comparison of the sites relative to bare values.

The NEE absolute data were very positively skewed. The data were squared to produce a positive flux, and then log transformed to normalise the data. Analysis of the data was conducted after which the least mean squares and their standard deviation where converted back into untransformed relative data and reported. The NEE dataset taken relative to the bare soil, contained many negative values which would have been removed during logging. To log the data, the data was shifted towards positive values while also retaining the relative differences within the data. This was achieved by adding the most negative value within the data set to each of the data points. The natural log of the data were then analysed using ANOVA. In the case of relative to bare soil control NEE, it was not possible to normalise the data and a non-parametric alternative was used. The Kruskal Wallis test was used to test the relationship of one factor (site, PAR, air temperature, WTD and rainfall) at a time against relative NEE. Peat surface recession data (obtained from Manchester University erosion pins study) was not included in the models for the following reason. The import explanatory factor 'plot nested within site' could not be included in the same model as the erosion rate, as the erosion rate was measured for site (not plot). The erosion data were referred to qualitatively in relation to the findings on WTD and DOC. Significant covariates relationship to WTD and the gas fluxes were investigated using the Pearson correlation. The correlation gave negative or positive correlation which indicated in which direction (increase or decrease) the covariate would influence the WTD and the gas fluxes. Where no significant relationship with WTD was observed this term was removed and the model refitted. Results output summary is given in tables at the end of the results section.

4.4 Results

4.4.1 Water table

4.4.1.1 WTD relative to LD-F (2008)

In 2008 relative to WTD there are 419 data points (Table 4.1). Analysis of 2008 WTD relative to the vegetated control (LD-F) was able to explain R^2 86.94% of the WTD variation. This fit was 30.92% greater than that possible using the raw data analysis R^2 56.02% (Chapter 3). It was found that unlike monthly raw data analysis variation, the most important factor in explaining variation in WTD variation relative to the vegetated control was monthly variation ($p < 0.0001$; $\omega^2 = 52.39\%$), which was then followed by site variation ($p < 0.000$; $\omega^2 = 18.79\%$) and then plot nested within site ($p < 0.0001$; $\omega^2 = 15.74\%$). Residuals were normal. Post hoc analysis revealed that naturally revegetated (-0.86 ± 0.06) had significantly shallower WTD relative to vegetated control, followed by the gully blocked site (-1.26 ± 0.06). The bare flat (-2.43 ± 0.06) and bare gully (-2.23 ± 0.06) sites had significantly higher relative WTD than the naturally revegetated and gully blocked sites; as did the treatment sites seeded and limed (-2.31 ± 0.06) and geojute (-2.29 ± 0.07) sites, which did not significantly differ to the bare control sites.

Measure	Absolute data			Relative data	
	N.	% removed	No. removed relative	% removed	No. removed relative
R_{eco}	1630	1.17	19	0.37	6
NEE	1843	0.38	7	1.03	19
P_g	1336	0.90	12	0.82	11
WTD	1890	0.00	0	0.00	0

Table 4.1: Absolute and relative data quality control. Includes detail on number data point (N), percentage of data points removed and number.

The heather brash site (-2.07 ± 0.07) did not significantly differ to the two other treatment sites seeded and limed nor geojute, however it was found to have relatively shallower WTD than bare flat control site, but deeper relative to the vegetated control WTD depth. The addition of covariates PAR was significant, but did not improve the overall R^2. Thus the model for the 2008 relative to LD-F WTD ANOVA was accepted (Figure 4.1).

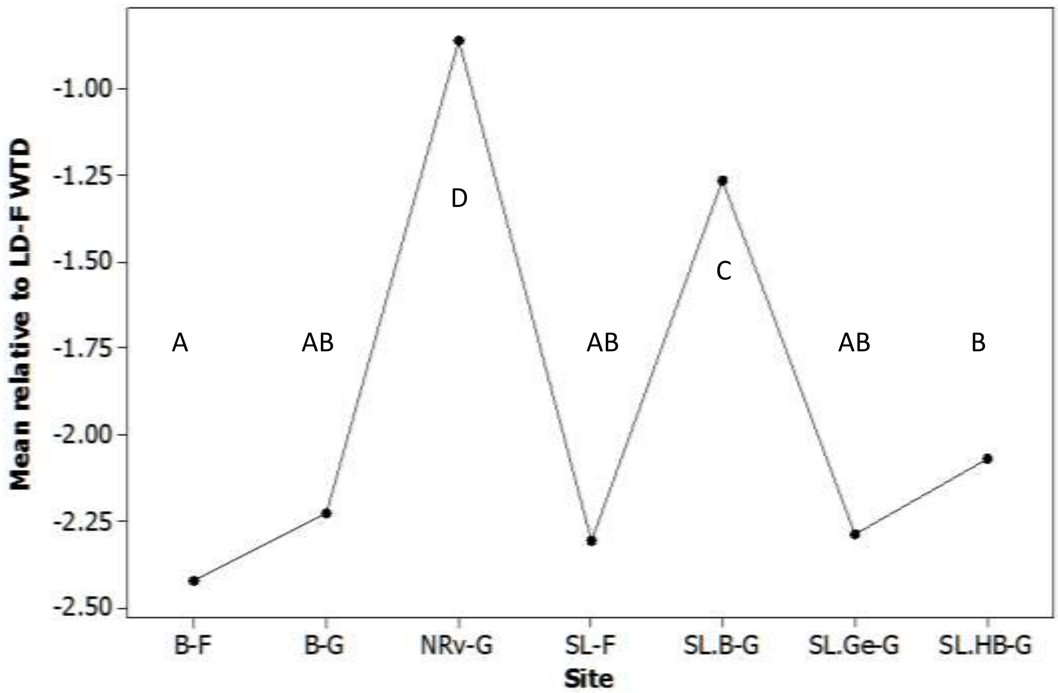


Figure 4.1: Relative to LD-F (n 314) water table depth ANOVA main effects plot by sites (2008). The letters on the plots represent the post hoc test results.

4.4.1.2 WTD relative to LD-F (2007-2011)

Over the five year study 1232 data points were used to consider the hydrological restoration success between sites, through comparison of sites WTD shift toward becoming more or less similar to the vegetated control over the study period (Figure 4.2). More positive values indicate vegetated control had a relatively shallower WTD than treated site. More negative values indicate vegetated control had a deeper WTD than the treated site. It is clear from the plots that both the gully blocked and naturally revegetated sites had the best hydrological outcomes of the eight sites monitored, as they had the lowest WTD relative to the vegetated control. A clear observation is not easily made due to monthly fluctuation. Thus ANOVA was required to find significant differences. The bare sites and the seeded and limed site had the highest WTD relative to the vegetated control in comparison to the other sites. In 2009 a peak in the relative WTD was observed in the bare and restored sites.

Analysis of the five year WTD relative to the vegetated control was conducted using ANOVA R^2 82.18%. The most important explanatory variable was month within year ($p < 0.0001$; $\omega^2 = 70.39\%$), annual variation is the second most important factor ($p < 0.0001$, $\omega^2 = 7.39\%$). Intra-site variation, plot within site, was more important ($p < 0.0001$; $\omega^2 = 2.96\%$) than between sites variation ($p < 0.0001$; $\omega^2 = 1.04\%$) There is a significant but small interaction between year and treatment ($p < 0.0001$; $\omega^2 = 0.46\%$). Residuals were normal. Post hoc analysis revealed that the bare sites (B-F and B-G) had relatively highest WTD and it was significantly higher than the heather brash site. The seed and limed and the geojute sites were not significantly different to either the bare site controls or the heather brash site.

The addition of covariates in an ANCOVA increased the R^2 by 2.57% to 84.75%. The addition found the importance of month within year was very slightly reduced ($p < 0.0001$; $\omega^2 = 69.64\%$). Annual variation remained the second most important factor ($p < 0.0001$, $\omega^2 = 7.34\%$). Intra-site variation, plot within site, became relatively more important ($p < 0.0001$; $\omega^2 = 3.41\%$), remaining more important than between sites ($p < 0.0001$; $\omega^2 = 0.54\%$). The significance for interaction between year and treatment remained small ($p < 0.0001$; $\omega^2 = 0.60\%$). Air temperature variation was more important than site ($p < 0.0001$; $\omega^2 = 2.00\%$) and positively correlated to relative WTD (Pearson correlation coefficient 0.113; $P < 0.0001$), as was PAR ($p < 0.0001$; $\omega^2 = 1.23\%$) (Pearson correlation coefficient; 0.161; $P < 0.0001$). Residuals were normal.

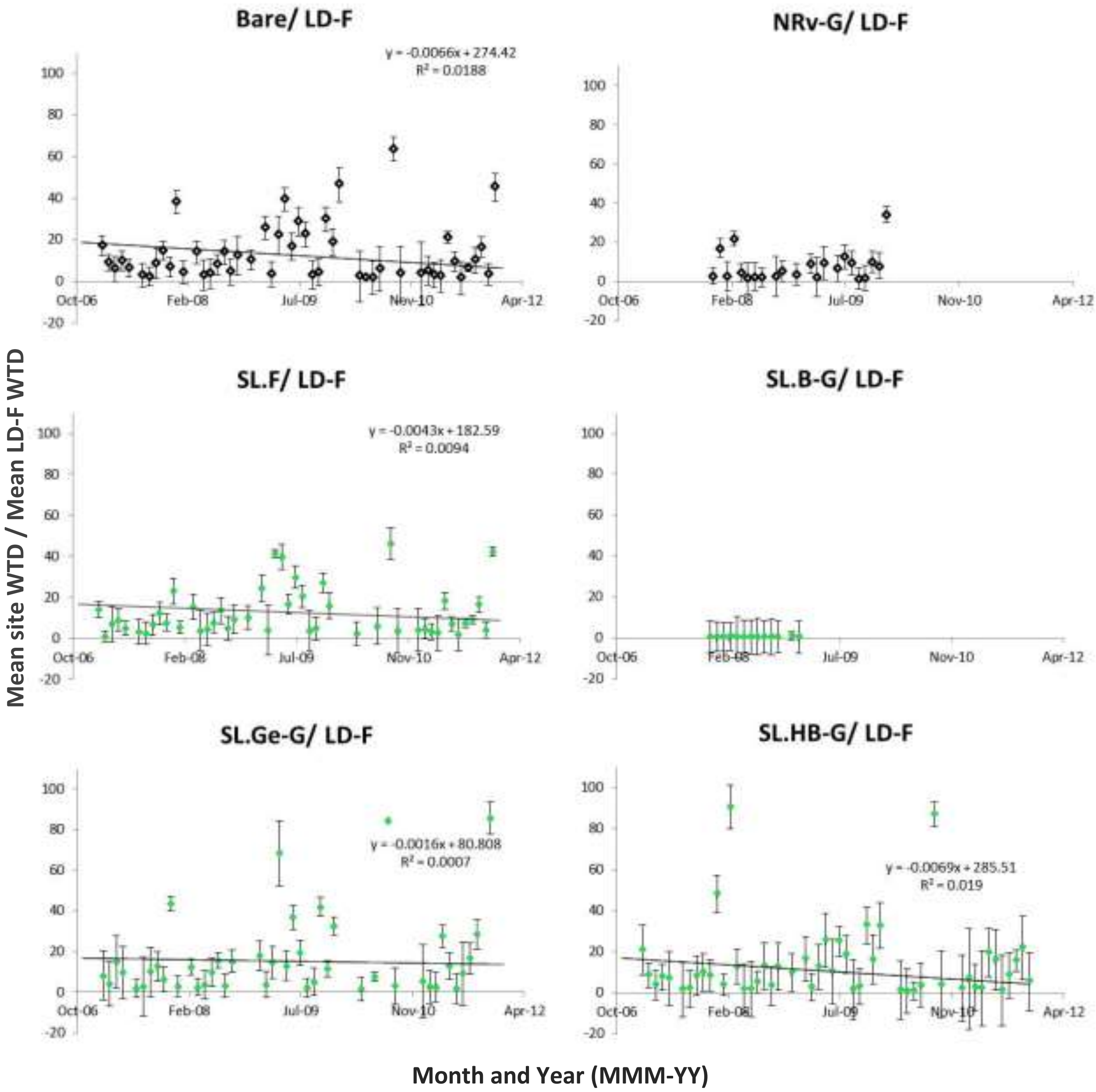


Figure 4.2: Six plots of site water table depth relative to the vegetated control (2006–2011). Bare signifies the average of B-F and B-G. The black-hollow diamonds represent untreated sites, and green diamonds represent treated sites. Values greater than one indicate site flux is greater than LD-F, values lower than one are smaller than LD-F. The error bars are the SE mean. A linear trend demonstrates the similarity between site and LD-F over time.

Post hoc analysis revealed (Figure 4.3) that the WTD was deepest at the bare gully (-2.13 ± 0.04) and bare flat (-2.12 ± 0.03). Both bare soil controls did not significantly differ to geojute (-2.05 ± 0.06). The seeded and limed (-1.96 ± 0.04), heather brash (-1.88 ± 0.04) and geojute had shallower WTD relative to the bare soil controls and although they had varying degrees of vegetated success, they did not significantly differ to each other.

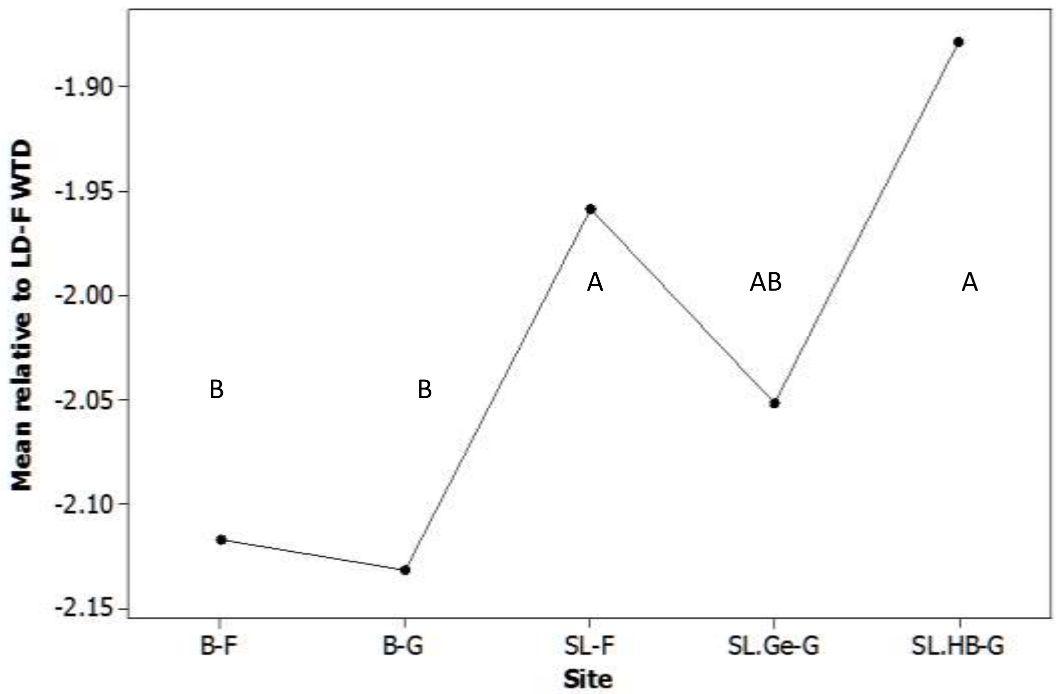


Figure 4.3: Main effects plot by sites (2007-2011) for mean relative to the vegetated control water table depth ANCOVA. The letters on the plots represent the post hoc test results.

Analysis of sites WTD relative to the bare control were calculated and analysed using ANOVA/ ANCOVA, the ANCOVA output are included at the end of the results section (Table 4.4). Post hoc results are presented in Table 4.2 and are included in Figure 4.8.

2008						5 year						
Site	Mean	SE Mean	Post hoc results			Site	Mean	SE Mean	Post hoc results		
LD-F	-0.15	0.03	A			LD-F	-0.28	0.03	A		
NRv-G	-0.41	0.03		B				N/A			
SL.B-G	-0.50	0.03		B							
SL.Ge-G	-0.92	0.03			C	SL.Ge-G	-1.11	0.04			B
SL.HB-G	-0.94	0.04			C	SL.HB-G	-1.07	0.02			B
SL-F	-1.01	0.03			C	SL-F	-1.04	0.03			B

Table 4.2: Site WTD relative to the bare control (mean B-F and B-G) in 2008 and over the 5 year study.

4.4.2 Carbon dioxide fluxes

4.4.2.1.1 *Ecosystem respiration (2008)*

Outputs from an ANOVA of natural log of absolute R_{eco}, of the 2008 dataset demonstrated that there were significant differences in R_{eco} fluxes between sites (p = 0.001, ω^2 = 17.82%) and months (p = 0.001, ω^2 = 20.60%). *Post-hoc* testing (Figure 4.5.a) demonstrated that the sites were distributed into two distinctly different groups, the first with the bare controls (B-F and B-G) and the seeded and limed only site (SL-F) having significantly lower R_{eco} fluxes than the other sites. The site B-G (0.0275 ± 0.0050 g CO_2 m^{-2} h^{-1}) had the lowest mean R_{eco} fluxes in 2008; its fluxes were 13.58% of the magnitude of the site SL.Ge-G, with the highest R_{eco} fluxes (0.2021 ± 0.0235 g CO_2 m^{-2} h^{-1}).

In an ANCOVA, R^2 increased from 40.92 to 41.46%. The importance of both site (p < 0.0001, ω^2 = 14.22%) and month were reduced (p < 0.00, ω^2 = 10.22%). The covariate air temperature (p < 0.038, ω^2 = 14.38%) was of equal importance to sites. The differences revealed in the ANOVA remained the same with the addition of the air temperature covariates. The dominantly bare soil sites (B-F, B-G and SL-F) had the significantly lowest R_{eco} of all the sites in 2008 (Figure 4.5).

4.4.2.1.2 *Ecosystem respiration (2007-2011)*

The ANOVA, of natural log of absolute R_{eco} output, of the full (six site) five year dataset shows that monthly (p =0.001, ω^2 = 14.92%) and inter-site (p =0.001, ω^2 = 16.99%) variation was more important than inter-annual variation (p =0.001, ω^2 = 4.01%). There were also significant interactions between site and year (p =0.001, 2.73%) and site and month (p =0.006, F = 1.56, ω^2 = 1.55%). This represents the relatively flat temporal trend of the bare/poorly re-vegetated sites (e.g. B-F, B-G, SL-F) relative to the more pronounced seasonal trend of the well vegetated sites (e.g. SL.Ge – G, LD – F) (Figure 4.4). The five year ANOVA output, unlike the 2008 output, indicated that the vegetated site groupings have started to fragment in terms of mean R_{eco} (Figure 4.5), SL.Ge-G had the greatest R_{eco} fluxes across the full five year dataset (0.2114 ± 0.0118 g CO_2 m^{-2} h^{-1}) with B-G having the lowest R_{eco} fluxes (0.0519 g CO_2 m^{-2} h^{-1}) at 24.54% of SL.Ge-G.

In both the full five year ANCOVA models R^2 increased from 44.69 to 47.08%. The models output had decreased importance of site ($p < 0.0001$, ω^2 15.96%) and month ($p < 0.0001$, ω^2 4.30%) in the ANCOVA model. The importance of year and the interaction between site with month and site with year remained of similar importance in the ANCOVA as they were in the ANOVA. The covariate air temperature was the second most important explanatory variable after site ($p < 0.0001$, ω^2 12.99%). Over the five years (Figure 4.5.b), the sites which were found (in Chapter 3) to have bare soil as the most dominant cover type (bare flat, bare gully and seeded and limed) had the lowest R_{eco}. The vegetated control had higher R_{eco}, not significantly different to heather brash. The two sites with stabilisation heather brash and geojute had the equally greatest R_{eco}.

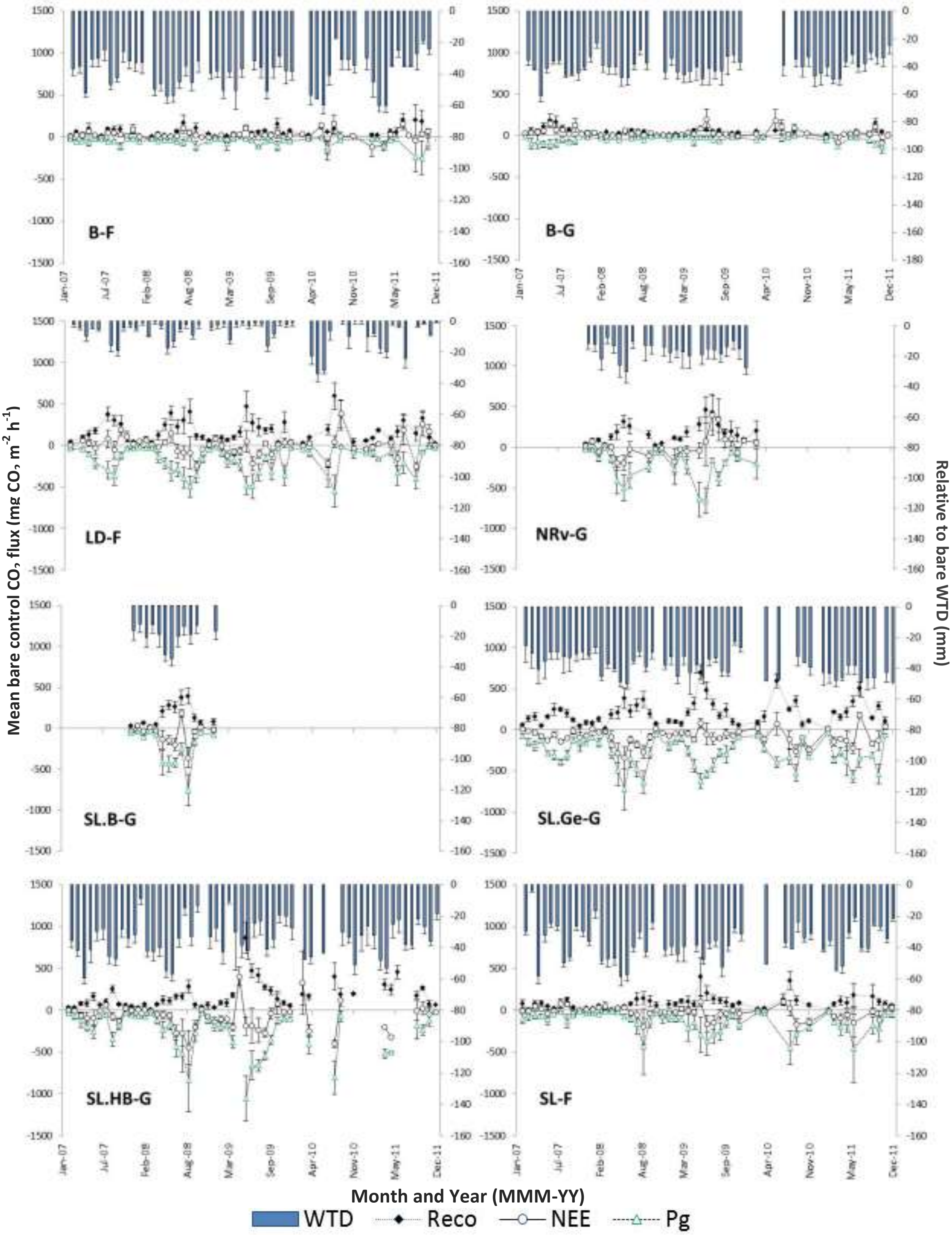


Figure 4.4: Bleaklow absolute data (Jan 2007 – Dec 2011): mean water table depth (WTD – right y-axes) and CO_2 fluxes (Reco, NEE, Pg – left y-axes) by month and site. All data-points represent the monthly site mean, with error bars defining SE mean. Blue-drop-boxes denote WTD, black-diamonds with a thin dashed connect line denote R_{eco}, white-circles with a solid black connect line denote NEE, and green-triangles with a thick dashed connect line denote P_g. (Figure adapted from Dixon et al. (2014)).

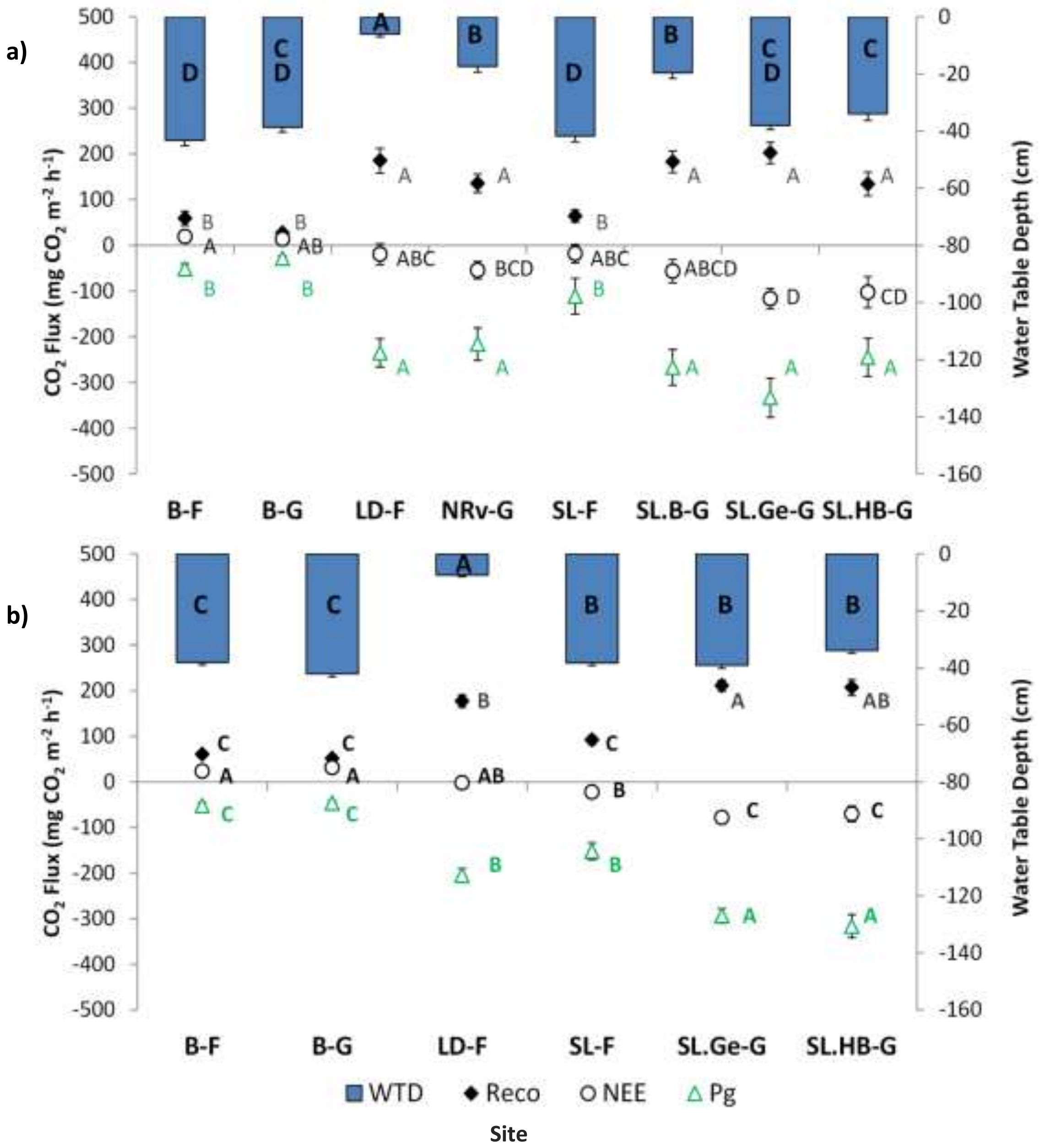


Figure 4.5: Mean water table depth (WTD – right y-axis) and CO_2 fluxes (R_{eco}, NEE, P_g – left y-axis) a) by site for the one year dataset (2008); b) for the full five year dataset (2007 - 2011). Error bars denote SE mean, in some cases the errors are smaller than the data-points themselves. The letters within/next to each bar/data-point represent the post-hoc tests results. Sites with different letters for a given dataset (e.g. R_{eco}) are significantly different in terms of the magnitude of that flux. Grey-drop-boxes denote water table depth, black-diamonds denote R_{eco}, white-circles denote NEE, and white-triangles denote $P_{g.}$ (Figure adapted from Dixon et al. (2014)).

4.4.2.1.3 *Ecosystem respiration (Relative to bare – 2008)*

Analysis of absolute R_{eco} in 2008 found that the bare control sites had the significantly lowest flux of the eight sites monitored (Figure 4.5). Analysis of the R_{eco} data relative to bare soil sites (mean of B-G and B-F) was conducted. The mean R_{eco} for the bare soil control sites was calculated for each month (Figure 4.6). The relative data to the bare site mean was calculated for site data (Figure 4.9) Sites with relatively larger R_{eco} values than the bare soil sites will be > 1, and those relatively smaller than bare soil site will be <1.

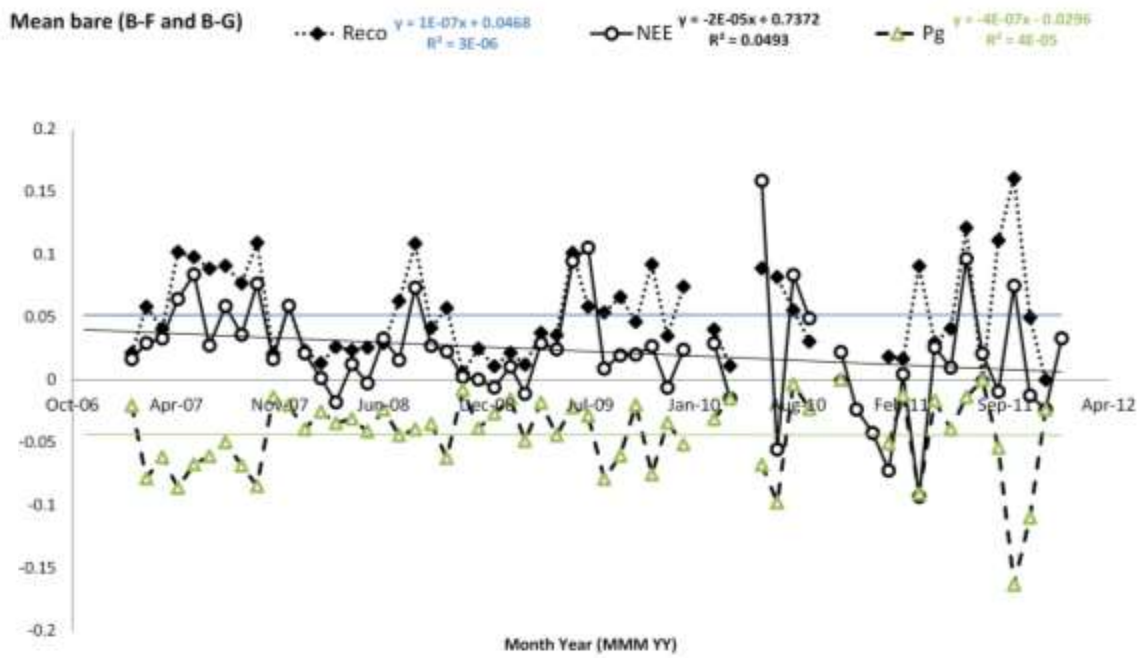


Figure 4.6: Multi annual (2006-2012) bare site mean DOC and NEE. The hollow black circles represent NEE; the hollow diamonds represent soil pore water DOC. The blue diamonds represent R_{eco}, the black circles represent NEE, and the green triangles represent P_g.

An ANOVA found that, for natural log of relative R_{eco}, the most important factor was month ($p < 0.0001$, $\omega^2 = 18.73\%$) followed by collar nested factor within the site ($p < 0.0001$, $\omega^2 = 16.76\%$). The differences between collars within a site were more important than between sites ($p < 0.0001$, $\omega^2 = 13.35\%$) in explaining relative R_{eco} in 2008. Residuals were normal. The site with the lowest R_{eco} relative to the bare control was SL-F (0.85 ± 1.16 g CO_2 m-^{2} h^{-1}), however SL-F R_{eco} was most similar to the bare controls. Three sites: naturally revegetated (1.98± 1.13 g CO_2 m-^{2} h^{-1}), the vegetated control (3.04 ± 1.13 g CO_2 m-^{2} h^{-1}) and gully blocked (2.96 ± 1.14 g CO_2 m-^{2} h^{-1}) had significantly higher relative R_{eco} than the seeded and limed, and did not significantly differ to each other. The vegetated control and gully blocked sites were significantly different, with respect to relative R_{eco}, to the two sites with the highest relative to bare R_{eco} which were the geojute (5.19 ± 0.01 g CO_2 m-^{2} h^{-1}) and heather brash (5.93 ± 0.03 g CO_2 m-^{2} h^{-1}) sites.

The inclusion of covariates ANCOVA increased the models R^2 from 48.92% up to 51.31%. When accounting for covariates the most important factor was differences within sites. Collar nested within sites ($p < 0.0001$, ω^2 = 17.91%) became more important than monthly variation importance which was decreased ($p < 0.0001$, ω^2 = 10.61%), however importance of variation between sites ($p < 0.0001$, ω^2 = 13.38%) become more important than month. Air temperature explained the smallest portion of variation ($p < 0.0001$, ω^2 = 8.98%). Residuals were normally distributed. There was a positive correlation between R_{eco} and air temperature (Pearson correlation coefficient 0.302, $P < 0.0001$). Post hoc analysis revealed (Figure 4.8.a) that with the addition of covariates, the site with lowest R_{eco} relative to the bare control remained the seeded and limed (0.95 ± 1.16 g CO_2 m-^{2} h^{-1}). The naturally revegetated (2.07 ± 1.13 g CO_2 m-^{2} h^{-1}), vegetated control (2.89 ± 1.13 g CO_2 m-^{2} h^{-1}) and gully blocked (3.22 ± 1.14 g CO_2 m-^{2} h^{-1}) sites had significantly higher relative R_{eco} than the seeded and limed site, and they did not significantly differ from each other. The highest R_{eco} relative to the bare soil controls were the geojute (4.17 ± 1.13 g CO_2 m-^{2} h^{-1}) and heather brash (5.23 ± 1.28 g CO_2 m-^{2} h^{-1}) sites.

4.4.2.1.4 *Ecosystem respiration (Relative to bare – 2007-2011)*

Figure 4.10 displays the sites CO_2 fluxes relative to the bare control (mean B-G and B-F) for the study monitoring period. Site relative to bare flux fluxes closest to 1, had fluxes most similar to the bare controls. The site SL-F had the lowest relative to bare R_{eco} over the five years with a very slight gradually increasing trend. Both sites SL.Ge-G and LD-F had consistently greater R_{eco}, also with a slightly increasing trend. The site SL.HB-G had the greatest starting point and was the only site to display a slightly decreasing trend in which the R_{eco} was being reduced over time and the difference between SL.HB-G and the bare control was becoming smaller.

Analysis of relative R_{eco} over the five year study between 2007 and 2011, found that like the 2008 output, the most important factor was related seasonality, month within year ($p < 0.0001$, $\omega^2 = 18.48\%$). Variation between site ($p < 0.0001$, $\omega^2 = 9.15\%$) was less important than between collar nested within site ($p < 0.0001$, $\omega^2 = 11.21\%$). Annual variation was a significant factor ($p < 0.0001$, $\omega^2 = 4.59\%$), but only a small interaction existed between year and site ($p < 0.0001$, $\omega^2 = 2.09\%$). The residuals were normal. Post hoc analysis revealed the restoration site SL-F (1.18 ± 1.08 g CO_2 m-^{2} h^{-1}) had the lowest R_{eco}, relative to bare as it was also in 2008. The vegetated control LD-F (2.32 ± 1.07 g CO_2 m-^{2} h^{-1}) had a higher relative to bare R_{eco} than SL-F and did not significantly differ from SL.HB-G (2.71 ± 1.07 g CO_2 m-^{2} h^{-1}). Site SL.Ge-G (3.46 ± 1.07 g CO_2 m-^{2} h^{-1}) had the highest relative R_{eco}, and did not significantly differ to SL.HB-G. Post hoc testing found that on Bleaklow there was an increase in relative to bare R_{eco} from 2007 (1.42 ± 1.07 g CO_2 m-^{2} h^{-1}) to 2008 for the three following years 2008–2010 (2.13 ± 1.09; 2.71± 1.06 and 2.71 ± 1.07 g CO_2 m-^{2} h^{-1}).

The addition of a covariate increased the R^2 from 46.57% to 53.13%. Analysis of the relative to bare soil site R_{eco} data, between 2007 and 2011 using ANCOVA found that, like the 2008 ANCOVA output, the most important factor was collar nested within site ($p < 0.0001$, ω^2 =

29.30%) followed closely by the factor month nested within year ($p < 0.0001$, $\omega^2 = 26.87\%$). Variation between site ($p < 0.0001$, $\omega^2 = 4.58\%$) was less important than that between collars nested within site. Figure 4.7 demonstrates the similarity of collars in each site to the bare controls. There was an observed difference in R_{eco} relative to the bare controls particularly at each of sites (Observed at: LD-F 6, SL-F 6, SL.Ge-G 6, SL.HB-G 2). Annual variation was a significant factor ($p < 0.0001$, $\omega^2 = 14.26\%$), but only a small interaction existed between year and site ($p < 0.0001$, $\omega^2 = 0.000\%$). The residuals were normal. Post hoc test found that (Figure 4.8.b) the addition of the covariate removed the overlap in significance but did not change the levels of significances between sites. The seeded and limed restoration site (1.27 ± 0.004 g CO_2 m^{-2} h^{-1}) had the lowest R_{eco} relative to the bare soil controls of sites monitored in both 2008 and across the five years. Therefore seeding and liming of a site gives the least change over a five year period. The vegetated control (3.15 ± 0.004 g CO_2 m^{-2} h^{-1}) and the heather brash (2.26 ± 1.08 g CO_2 m^{-2} h^{-1}) sites did not significantly differ to each other, and both had higher relative R_{eco} than the seeded and limed site did. They also had lower relative R_{eco} than the geojute (2.45 ± 1.07 g CO_2 m^{-2} h^{-1}) which was the site with highest R_{eco} relative to bare. Thus geojute site R_{eco} flux was most different to the bare controls over the five year study, while the seeded and limed site was the most similar. Additionally post hoc analysis found that on Bleaklow sites there was an increase in relative to bare R_{eco} from year 2007 (1.41 ± 1.07 g CO_2 m^{-2} h^{-1}) to 2008 for the duration of three years between 2008 and 2010 (Between 2.15 ± 1.09 and 2.79 ± 1.07 g CO_2 m^{-2} h^{-1}).

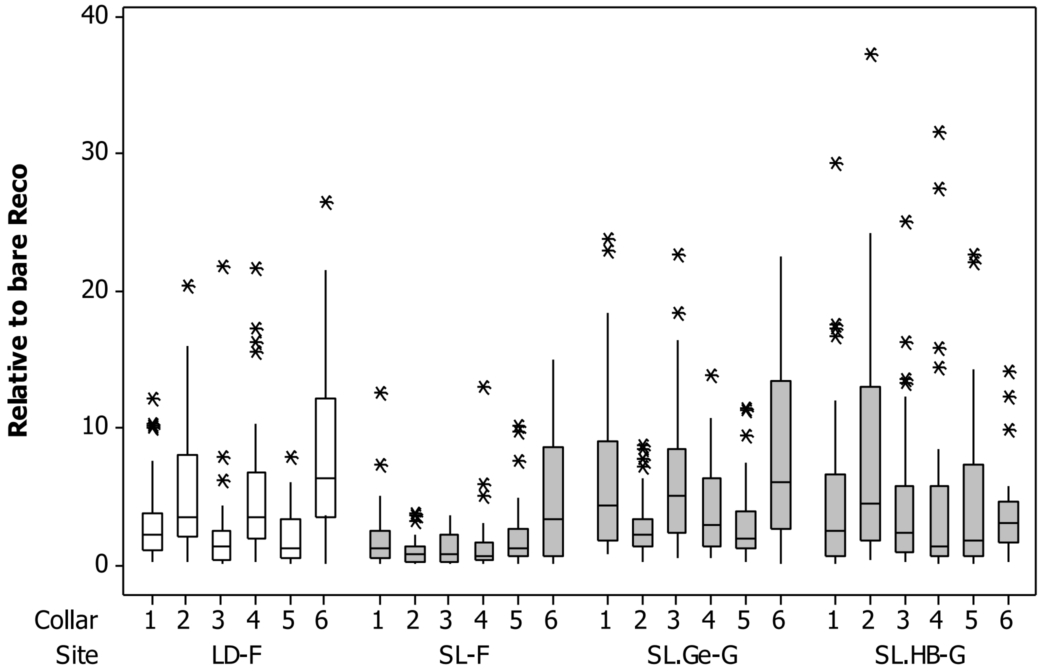


Figure 4.7: Box plot of relative to bare ecosystem respiration (g CO_2 m^{-2} h^{-1}) (2007–2011), by site and collar.

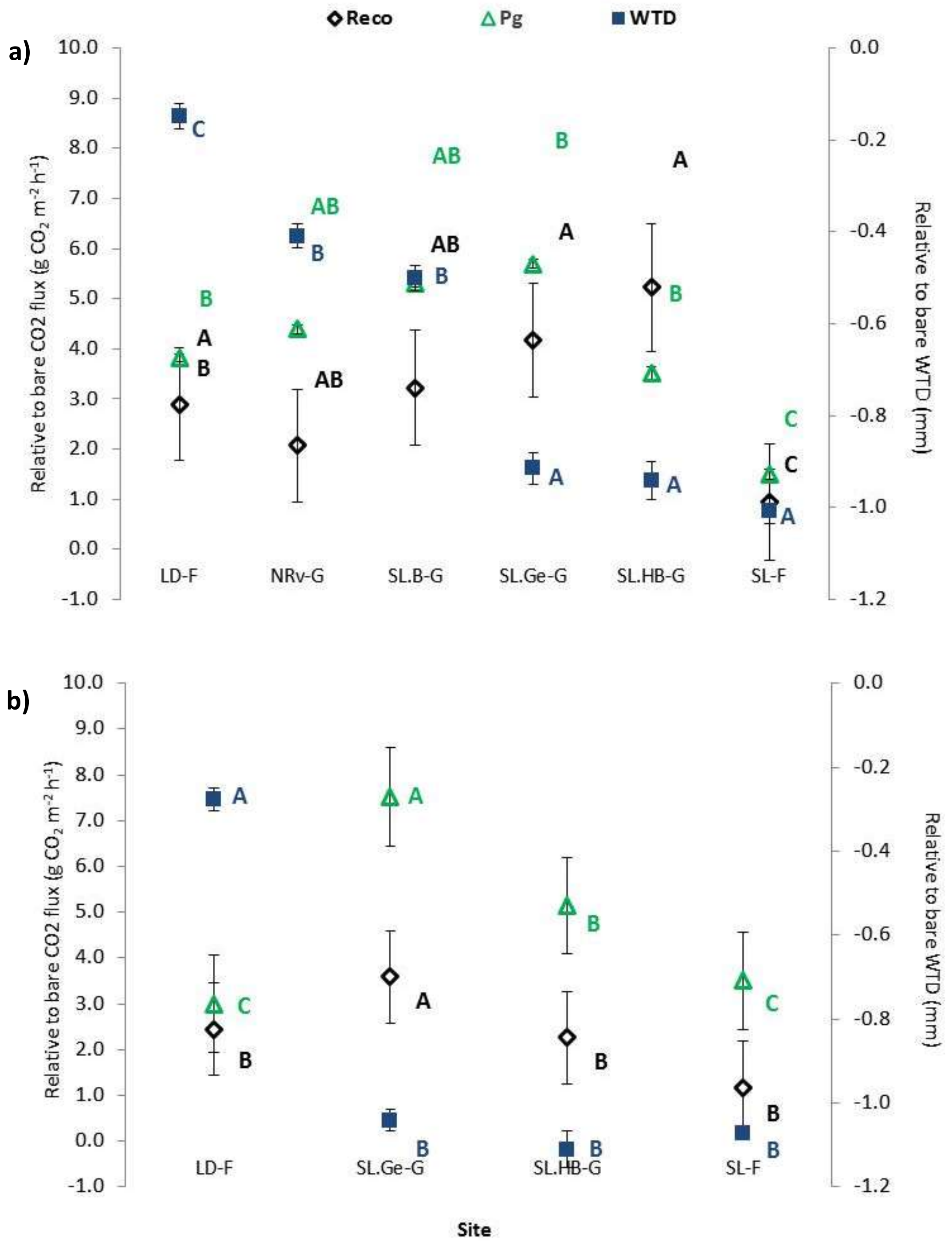


Figure 4.8: Main effects plots of relative to bare control (mean bare flat and bare gully): CO_2 fluxes (Left axis) - R_{eco} (black diamonds and letters) and P_g (green triangles and letters), and water table depth (blue boxes and letters- right axis) by site. a) At one year duration (2008), b) full five year study (2007-2011). Relative flux values >1 indicate site values were greater than bare control; <1 indicate site values were smaller than the bare control. WTD is a negative value, therefore relative values closer to >-1 are WTs nearer the surface and most different to the bare control Error bars denote SE error mean. The capital letters represent the significant differences between factor levels determined by post-hoc tests. The post hoc results are specific to each data set (e.g. R_{eco}).

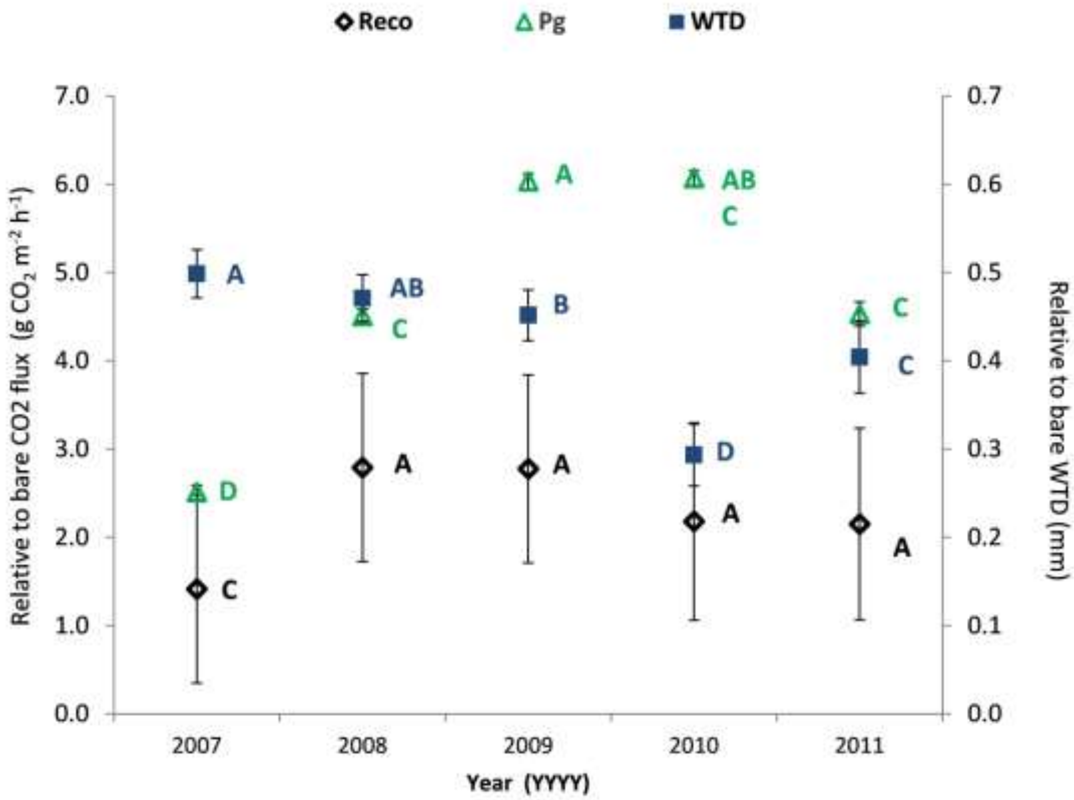


Figure 4.9: Main effect plot of pooled data (treatment sites and LD-F) relative to the bare controls: water table depth (Blue squares, Right hand axis) and relative to bare CO_2 R_{eco} (black-diamonds) and P_g (green triangles) fluxes (Left hand axis) by year (2007-2011) across the Bleaklow sites. Values closer to zero indicate the measure (relative to bare flux or WTD) were more similar to the bare site. Large relative values indicate the measure at a site was more different to the bare site control. Standard errors are denoted by error bars. The capital letters next are post-hoc tests results; the different letters represent significant differences between years (5 factor levels).

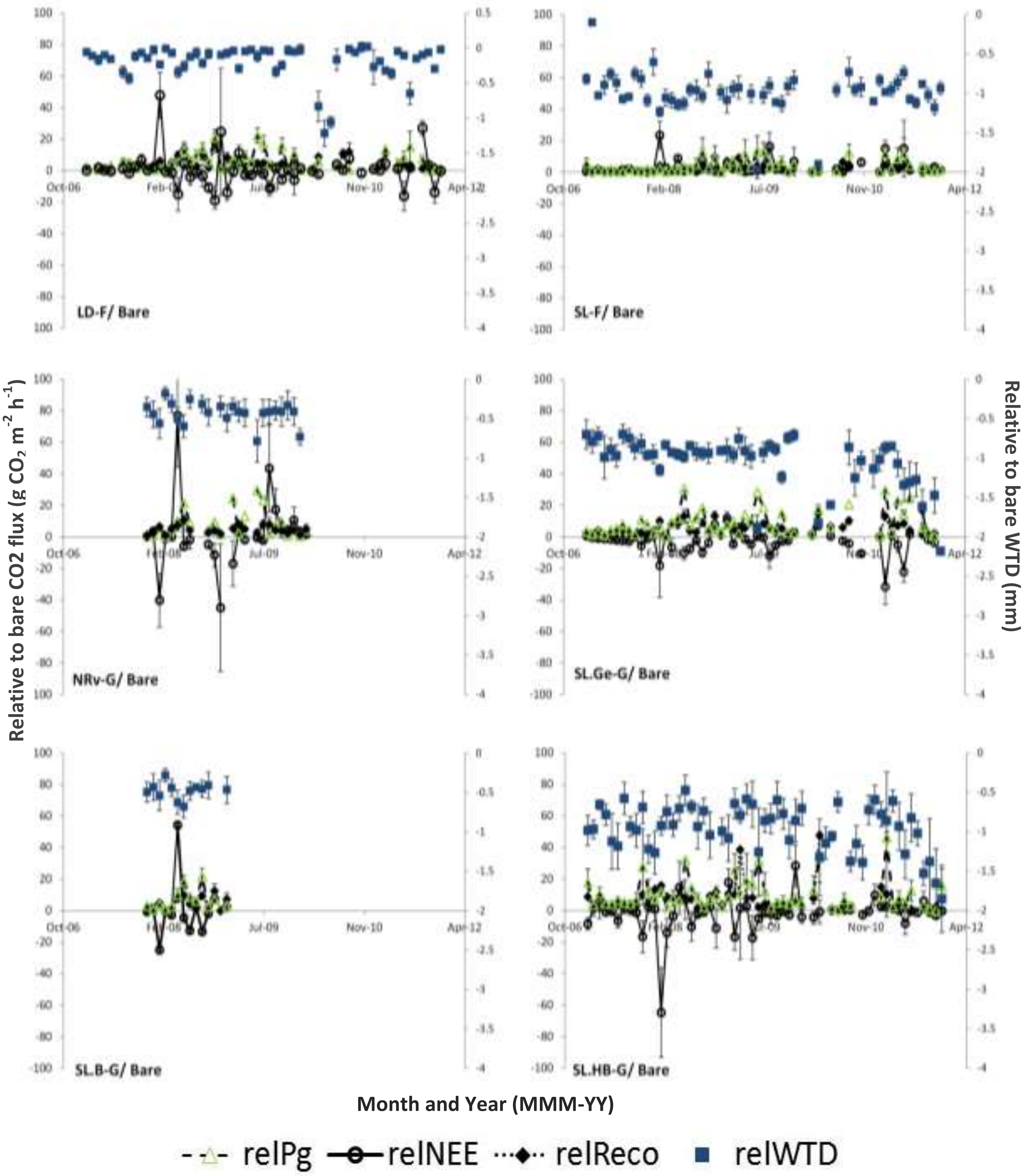


Figure 4.10: Mean site data relative to bare control (B-G, n 364; B-F n, 363) (2007-2012)-: mean water table depth (WTD blue squares – right hand axes,) and CO2 fluxes (Reco black-circles; NEE, black-diamonds; Pg, green-triangles – left hand axes) by month and site. All data-points represent the mean of the plot replicates on each site with error bars defining the standard error of the mean.

4.4.2.2 Gross photosynthesis

4.4.2.2.1 *Gross photosynthesis (2008)*

Analysis of absolute R_{eco} 2008 data using ANOVA demonstrated significant differences between sites ($p = 0.001$, $F = 26.83$, $\omega^2 = 19.83\%$) and months ($p = 0.001$, $F = 21.97$, $\omega^2 = 28.42\%$). Post hoc results (Figure 4.5) revealed that the bare/poorly vegetated (e.g. B-F, B-G and SL-F) sites' P_g fluxes are significantly lower than rest of the sites with lower bare cover. The bare gully control site had the lowest overall rates of P_g (-0.0285 ± 0.0051 g CO_2 m^{-2} h^{-1}), while the geojute site (-0.3332 ± 0.0431 g CO_2 m^{-2} h^{-1}) had the highest rates in 2008.

The addition of covariates in the 2008 dataset found WTD ($p = 0.003$, $\omega^2 = 3.46\%$) and PAR ($p = 0.039$, $\omega^2 = 5.26\%$) to be significant. However, the addition for the two covariates did not appear to explain any additional variation (i.e. R^2 decreases slightly from 50.53% in ANOVA to 49.88% in ANCOVA). Instead they explain some of the magnitude of the effect of site (i.e. ω^2 decreases from 19.83 in ANOVA to 13.38% in ANCOVA) and month (i.e. ω^2 decreases from 28.42 in ANOVA to 24.71% in ANCOVA). Post hoc testing revealed (Figure 4.5.a) that the smallest absolute P_g flux was observed at the bare control sites, and did not significantly differ to the seeded and limed site. The seeded and limed site also did not significantly differ to the vegetated control and naturally revegetation sites, both which had a greater P_g flux than the bare control sites. The sites with the significantly highest P_g were the gully blocked and the two sites where stabilisation techniques were used (geojute and heather brash).

4.4.2.2.2 *Gross photosynthesis (2007-2011)*

In the full five year study, ANOVA inter-annual variation was significant ($p = 0.001$, $F = 10.06$, $\omega^2 = 2.30\%$) but is less important than inter-site ($p =0.001$, $F = 50.64$, $\omega^2 = 23.09\%$) and monthly ($p =0.001$, $F = 12.76$, $\omega^2 = 9.83\%$) variation. Significant interactions between site with month (p

= 0.001, F = 1.71, ω^2 = 2.19%) and site with year (p = 0.037, F = 1.65, ω^2 = 0.72%) were identified. The site B-G had the lowest mean rate of P_g (-0.0463 ± 0.0052 g CO2 m^{-2} h^{-1}) over the full five year monitoring period which was 14.64% of SL.Ge-G (-0.3164 ± 0.0255), the site with the highest mean rate of P_g.

In an ANCOVA, the addition of covariates did not improve the R^2 but decreased it slightly from 43.48% to 42.80%. It was found that over the five year monitoring site and month being less important (ω^2 decreases from 23.09 to 19.62% and 9.83 to 8.12% for site and month respectively). WTD (p = 0.001, ω^2 = 2.24%) and PAR (p = 0.001, ω^2 = 2.58%) remained important factors of similar importance to each other. Significant positive correlations were identified with WTD and negative correlations with PAR.

4.4.2.2.3 *Gross photosynthesis (Relative to bare - 2008)*

Analysis of the data relative to the bare controls (mean of the bare flat and gully sites) P_g was conducted using ANOVA. The most significant differences were between month (p < 0.0001, ω^2 = 28.75%), followed by difference between collars nested within site (p < 0.0001, ω^2 = 19.39%) which was a more important factor than differences between sites (p < 0.0001, ω^2 = 10.22%) in explaining the variation of relative to bare soil P_g flux. Residuals were normal. Post hoc tests found five sites (Vegetated site, naturally revegetated, Gully blocked, geojute and heather brash) had greater P_g, relative to the bare soil controls (Between 5.01 ± 1.08 and 6.66 ± 1.08) than the seeded and limed (2.20 ± 1.10). The restoration seeded and limed site, with the highest bare cover, had the most similar P_g flux to the bare controls (Bare gully and flat).

The addition of covariates in an ANCOVA increased the R^2 from 58.45 to 62.71%, increasing the relative importance of collars nested within site (p < 0.0001, ω^2 = 25.98%). The importance of monthly variation (p < 0.0001, ω^2 < 19.80%) remained higher than difference between sites ($p < 0.0001$, $\omega^2 = 8.35\%$). Air temperature inside the gas collar (Positively correlated to P_g; $P < 0.0001$, 0.257) was found to be the most important covariate ($p = 0.001$, $\omega^2 = 5.20\%$). The covariate PAR (Positively correlated to P_g; $P < 0.0001$, 0.271) was also found to have a small significant impact on relative to bare P_g ($p = 0.001$, $\omega^2 = 3.30\%$). Residuals of the nested ANCOVA plotted against fitted values were random and normally distributed. Post hoc analysis again determined that seeded and limed site had the lowest P_g relative to the bare control (1.51 ± 0.10 g CO_2 m-^{2} h^{-1}). Site geojute (5.69 ± 0.08 g CO_2 m-^{2} h^{-1}) had significantly higher relative flux than the vegetated control (3.82 ± 0.08 g CO_2 m-^{2} h^{-1}). Neither differed to the remaining three sites: gully blocked, naturally revegetated and heather brashed (Between 5.31 ± 0.09 and 3.82 ± 0.08 g CO_2 m-^{2} h^{-1}). Therefore the sites with the P_g flux most different to the bare soil control in 2008 were the geojute and seeded limed.

4.4.2.2.4 *Gross photosynthesis (Relative to bare 2007-2011)*

Analysis of the P_g relative to bare soil sites across the five year study found that month within year variation was the most important factor ($p < 0.0001$, $\omega^2 = 24.89\%$). Variation within sites as collars nested within sites ($p < 0.0001$, $\omega^2 = 14.40\%$) was more important than variation between sites ($p < 0.0001$, $\omega^2 = 5.84\%$). Annual variation was also important ($p < 0.0001$, $\omega^2 = 14.40\%$) with a small significant interaction between site and year ($p < 0.0001$, $\omega^2 = 2.64\%$). Residuals were normal. Post hoc analysis revealed that SL-F (-1.55 ± 1.11 g CO_2 m-^{2} h^{-1}) had significantly lowest relative to the bare controls P_g flux over the five years of monitoring. The vegetated control (-0.66 ± 1.09 g CO_2 m-^{2} h^{-1}) had a higher relative P_g flux than the seeded and limed, and lower than the geojute (2.47 ± 1.10 g CO_2 m-^{2} h^{-1}) and heather brash (1.32 ± 1.13 g CO_2 m-^{2} h^{-1}) sites, which both had the greatest P_g flux relative to the bare controls.

The addition of covariates in the ANCOVA increased the R^2 from 52.39% to 56.46%. Month within year variation was most important factor ($p < 0.0001$, $\omega^2 = 20.37\%$). Variation between collars nested within sites remained the second most important factor ($p < 0.0001$, $\omega^2 = 13.95\%$). Figure 4.1 demonstrates the variation between the collars in each site, evident at the least or most well vegetated collars (observed at: LD-F 3,5; SL-F 5, 6; SL.Ge-G 6; SL.HB-G 6). Annual variation was more important ($p < 0.0001$, $\omega^2 = 6.27\%$) than variation between sites ($p < 0.0001$, $\omega^2 = 5.60\%$). An interaction between site and year also existed ($p < 0.0001$, $\omega^2 = 3.86\%$). The most important covariate was air temperature within the chamber at the time of flux measurement ($p = 0.003$, $\omega^2 = 5.00\%$). It was more important than PAR ($p = 0.016$, $\omega^2 = 0.83\%$) and water tale depth ($p = 0.006$, $\omega^2 = 0.53\%$). Residuals were normal. Post hoc testing revealed that, over the five years of monitoring, site P_g flux relative to the bare soil control did not differ between the seeded and limed (3.50 ± 1.05 g CO_2 m-^{2} h^{-1}) and vegetated control (3.00. ± 1.06 g CO_2 m-^{2} h^{-1}) and both had relatively higher than P_g flux the bar The heather brash site (5.16 ± 1.05 g CO_2 m-^{2} h^{-1}) had the highest relative P_g, however the addition of the covariates revealed that in fact the geojute site (7.53 ± 1.08 g CO_2 m-^{2} h^{-1}) had the greatest P_g flux relative to the bare controls.

The sites with significantly greater P_g flux in 2008 and over the five year monitoring are those treated with stabilisation techniques and significantly greater P_g flux (Table 4.3). Two sites, geojute and heather brash, are the only to have successfully deposited peat over the five years of monitoring. The remaining four sites have incurred losses of peat observed as surface recession. The greatest rate in peat recession was observed at the bare control sites specifically at the bare flat site (although the bare gully site had greater variability), followed by the seeded and limed site which received treatment without stabilisation techniques. The least disturbed vegetated control also had some surface recession. The site with the greatest P_g relatively to that of the bare soil controls was the geojute site which notably had greater surface deposition than the heather brash site.

Using the Pearson correlation test significant correlations were identified between relative to bare Pg and: PAR (Positive; $p < 0.0001$, 0.155), air temperature in chamber (Positive; $p < 0.0001$, 0.201) and WTD (Positive; $p < 0.0001$, 0.014). Post hoc also found that there was an increase of relative P_g from year 2007 (2.15 ± 1.04 g CO_2 m-^{2} h^{-1}) to its peak in years 2009 and 2010 (6.03 ± 1.04 and 6.07 ± 1.11 g CO_2 m-^{2} h^{-1}) then dropped in 2011 (4.53 ± 1.05 g CO_2 m-^{2} h^{-1}) to an equal level no different than of 2008, but higher than in 2007.

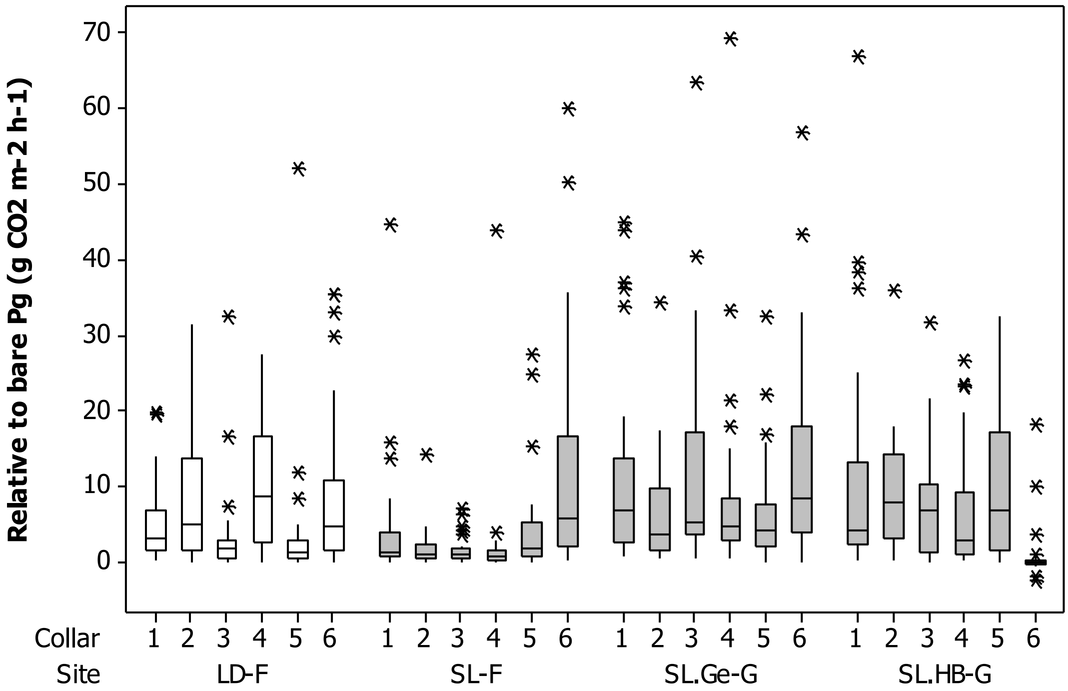


Figure 4.11: Box plot of relative to bare gross photosynthesis (2007–2011) by site and collar. The grey box plots represent collar within with treated sites, white box plots are collar within the untreated least disturbed vegetated control.

Site	Absolute WTD (cm)			Absolute R_{eco} ($g\ CO_2\ m^{-2}\ h^{-1}$)			Absolute NEE ($g\ CO_2\ m^{-2}\ h^{-1}$)			Absolute P_g ($g\ CO_2\ m^{-2}\ h^{-1}$)			Surface recession ($cm\ y^{-1}$)
	N	Mean	S.E.	N	Mean	S.E.	N	Mean	S.E.	N	Mean	S.E.	Mean (St.Dv.)
B-F	264	-38.07	0.91	220	0.0646	0.0075	277	0.0239	0.0049	160	-0.0522	0.0066	2.53 (0.32)
B-G	280	-42.02	0.99	201	0.0525	0.0052	264	0.0315	0.0053	126	-0.0469	0.0053	1.57 (1.09)
LD-F	315	-7.47	0.6	259	0.1792	0.0134	264	-0.0018	0.0114	225	-0.2059	0.0155	0.18 (0.30)
NRv-G	175	-17.22	1.36	160	0.1819	0.0194	195	-0.0043	0.0233	133	-0.2524	0.0273	-
SL-F	300	-38.17	0.94	229	0.0927	0.0097	259	-0.0222	0.0087	187	-0.152	0.0183	1.51 (0.46)
SL.B-G	72	-19.13	1.78	69	0.1641	0.0213	79	-0.0513	0.0214	64	-0.2441	0.0357	-
SL.Ge-G	197	-39.09	1.07	254	0.2108	0.0118	260	-0.0779	0.0095	242	-0.2938	0.016	-0.31 (0.34)
SL.HB-G	287	-33.86	0.95	219	0.2112	0.0177	238	-0.0555	0.0198	187	-0.3123	0.0254	-0.14 (0.64)

Table 4.3: Absolute data, number of data points used, mean, standard error mean (S.E.) for water table depth and gas fluxes (R_{eco}, NEE and P_g). Surface recession for the sites monitored for the full five years (2007–2011) (adapted from Dixon et al. (2014)).

4.4.2.3 Net Ecosystem Exchange (2008)

Outputs of the 2008 ANOVA demonstrated that there are significant differences between sites ($p = 0.001$, $F = 6.10$, $\omega^2 = 5.20\%$) and months ($p = 0.001$, $F = 5.04$, $\omega^2 = 6.83$), however, these factors were not as important as in the equivalent R_{eco}/P_g models. Post hoc testing suggested that many sites overlap in terms of their NEE magnitude (Figure 4.5.a) however significant differences are evident between the bare sites (mean of the bare flat and gully sites) which are mean net sources, unlike the revegetated geojute and heather brash sites which are mean net sinks. The site bare flat was the largest net source of CO_2 (0.0195 ± 0.0081 g CO_2 m^{-2} h^{-1}), whereas the geojute site hade largest mean net sink of CO_2 (-0.1167 ± 0.0222 g CO_2 m^{-2} h^{-1}) in 2008. The introduction of covariates slightly improved model fits (R^2 increased from 14.82 to 16.95%). In 2008 a significant, but weak, positive correlation with WTD ($p = 0.002$, $\omega^2 = 0.91\%$) was observed, and negative correlation with PAR ($p = 0.001$, $\omega^2 = 1.44$).

4.4.2.4 Net Ecosystem Exchange (2007 -2011)

Outputs of the full five year ANOVA showed that site was the most important factor predicting NEE magnitude ($p = 0.001$, $\omega^2 = 6.16\%$) with inter-annual ($p = 0.001$, $\omega^2 = 0.84\%$) and inter-monthly ($p = 0.016$, $\omega^2 = 0.38\%$) variation being significant but relatively unimportant. A significant interaction between site and month ($p = 0.001$, $\omega^2 = 3.22\%$) was identified, reflecting the relatively flat trend of the bare control sites and dominantly bare seeded and limed site when compared to other sites with greater vegetative cover across the year. Post hoc testing (figure 4.5) revealed that the restored geojute (-0.0781 ± 0.0095 g CO_2 m^{-2} h^{-1}) and heather brash (-0.07014 ± 0.0172 g CO_2 m^{-2} h^{-1}) sites were the largest mean net sinks of CO_2, while the bare gully (0.0315 ± 0.0053 g CO_2 m^{-2} h^{-1}) site was the largest mean net source. In the full five year monitoring (Figure 4.5.b), the addition of covariates increased the R^2 increased from 14.82 to 16.95%. Significant correlation was again found as in the 2008 data set. A positive correlations with WTD ($p = 0.001$, $\omega^2 = 0.70\%$) and negative correlations with PAR ($p = 0.001$, $\omega^2 = 0.62\%$)

4.4.2.4.1 *Net ecosystem exchange (Relative to bare - 2008)*

Normalisation of the relative to bare NEE data was not possible. The data had a higher Anderson darling value (42.77) and although the data had a bell shaped curve histogram, there were many legitimate negative values, which resulted in a variance (1308.3) 40 times larger than relative R_{eco}, and 13 times more than relative P_g. As normal data distribution is required for ANOVA, analysis could only explain a very small portion of the data. Using ANOVA demonstrated that the same variables were significant however the importance of month ($p < 0.0001$, $\omega^2 = 6.07\%$) and collar nested within site ($p < 0.0001$, $\omega^2 = 3.11\%$). Residuals of the nested ANOVA plotted against fitted values were random and normally distributed. Post hoc analysis revealed no significant differences which contradicted the findings of the absolute data analysis. Addition of covariates in an ANCOVA increased the R_2 from 8.94% to 10.53%. Variation between months remained the most important factor ($p < 0.0001$, $\omega^2 = 6.07\%$), followed by collar nested within site ($p < 0.0001$, $\omega^2 = 3.78\%$). However the site itself was not significant. The covariate PAR was found significant although of low importance ($p < 0.0001$, $\omega^2 = 0.36\%$). Using the Kruskal Wallis test found month was significantly related to relative NEE ($P = 0.008$). Other factors (Site) and covariates, PAR, air temperature, WTD and rainfall were not significant.

4.4.2.4.2 *Net ecosystem exchange (Relative to bare – 2007-2011)*

Analysis of the NEE relative to the bare soil control using ANOVA was not possible due to abnormal distribution of data. The Kruskal-Wallis test found that relative NEE was significantly related to month ($p = 0.001$) and natural log of air temperature ($p = 0.047$).

	Responsible Variable	Test	Factor (F)/ covariate (C)/ interaction (I)/ nested factor (N)	Variable	P	ω^2 (%)	$_{adj}R^2$ (%)
2008	Relative to LD-F WTD	ANOVA	F	Site	< 0.0001	18.79	86.94
			F	Month	< 0.0001	52.39	
			N	Collar (Site)	< 0.0001	15.74	
		ANCOVA			N/A		
	Relative to Bare WTD	ANOVA	F	Site	< 0.0001	47.74	76.91
			F	Month	< 0.0001	1.84	
			N	Collar (Site)	< 0.0001	27.29	
		ANCOVA	C	lnPAR	< 0.0001	2.10	78.65
			F	Site	< 0.0001	47.46	
			F	Month	0.001	1.51	
			N	Collar (Site)	< 0.0001	27.53	
2007 - 2011	Relative to LD-F WTD	ANOVA	F	Year	< 0.0001	7.39	82.18
			F	Treatment	< 0.0001	1.04	
			N	Month (Year)	< 0.0001	70.32	
			N	Collar (Site)	< 0.0001	2.96	
			I	Year*Site	< 0.0001	0.46	
		ANCOVA	C	PAR	< 0.0001	1.23	84.75
			C	Air temperature	< 0.0001	2.00	
			F	Year	< 0.0001	7.34	
			F	Site	< 0.0001	0.54	
			N	Collar (Site)	< 0.0001	3.41	
			N	Month (Year)	< 0.0001	69.64	
			I	Site*Year	< 0.0001	0.60	
	Relative to Bare WTD	ANOVA	F	Site	< 0.0001	35.63	70.92
			F	Year	< 0.0001	3.81	
			N	Month (Year)	< 0.0001	28.11	
			N	Collar (Site)	< 0.0001	2.53	
			I	Site*Year	< 0.0001	0.82	
		ANCOVA	C	lnPAR	0.002	0.45	75.06
			C	Air temperature	< 0.0001	2.56	
			F	Site	< 0.0001	35.66	
			F	Year	< 0.0001	3.72	
			N	Month (Year)	< 0.0001	29.53	
			N	Collar (Site)	< 0.0001	2.20	
			I	Site*Year	< 0.0001	0.92	

Table 4.4: Output for WTD ANOVA and ANCOVA (2008 and 2007-2011). Analysis of site WTD relative to the least disturbed vegetated control.

Responsible Variable	Test	Factor (F)/ covariate (C)/ interaction (I)/ nested factor (N)	Variable	P	ω^2 (%)	$_{adj}R^2$ (%)
2008						
Relative to Bare lnR_{eco}	ANOVA	F	Site	< 0.0001	13.35	48.92
		F	Month	< 0.0001	18.73	
		N	Collar (Site)	< 0.0001	16.76	
	ANCOVA	C	AT	0.001	8.98	51.31
		F	Treatment	< 0.0001	13.74	
		F	Month	< 0.0001	10.61	
		N	Collar (Site)	< 0.0001	17.91	
Relative to Bare lnP_g	ANOVA	F	Site	< 0.0001	10.22	58.45
		F	Month	< 0.0001	28.75	
		N	Collar (Site)	< 0.0001	19.39	
	ANCOVA	C	Air temperature	0.001	5.20	62.71
		C	lnPAR	0.015	3.30	
		F	Treatment	< 0.0001	8.35	
		N	Collar (Site)	< 0.0001	25.98	
		F	Month	< 0.0001	19.80	
Relative to Bare NEE	ANOVA	F	Site	0.668	-0.26	8.94
		F	Month	< 0.0001	6.07	
		N	Collar (Site)	0.070	3.11	
	ANCOVA	C	PAR	0.046	0.36	10.53
		F	Site	0.456	-0.37	
		F	Month	< 0.0001	6.74	
		N	Collar (Site)	0.048	3.78	

Table 4.5 : Output for CO_2 gas fluxes (R_{eco},P_g and NEE) ANOVA and ANCOVA (2008). The data were analysed as values relative to the bare soil control sites (mean of B-F and B-G).

2007 - 2011						
Responsible Variable	Test	Factor (F)/ covariate (C)/ interaction (I)/ nested factor (N)	Variable	P	ω^2 (%)	$_{adj}R^2$ (%)
Relative to Bare lnR_{eco}	ANOVA	F	Site	< 0.0001	9.15	46.57
		F	Year	< 0.0001	4.59	
		N	Month (Year)	< 0.0001	19.51	
		N	Collar (Site)	< 0.0001	11.21	
		I	Site*Year	< 0.0001	2.09	
	ANCOVA	C	Air temperature	< 0.0001	5.98	46.57
		F	Site	< 0.0001	8.95	
		F	Year	< 0.0001	5.15	
		N	Month (Year)	< 0.0001	17.39	
		N	Collar (Site)	< 0.0001	12.38	
		I	Site*Year	< 0.0001	2.84	
Relative to Bare lnP_g	ANOVA	F	Site	< 0.0001	5.84	52.39
		N	Collar (Site)	< 0.0001	14.40	
		F	Year	< 0.0001	4.59	
		N	Month (Year)	< 0.0001	24.89	
		I	Site*Year	< 0.0001	2.64	
	ANCOVA	C	ln air temperature	0.003	5.00	56.46
		C	WTD	0.006	0.53	
		C	PAR	0.016	0.83	
		F	Site	< 0.0001	5.60	
		N	Collar (Site)	< 0.0001	13.95	
		F	Year	< 0.0001	6.27	
		N	Month (Year)	< 0.0001	20.37	
		I	Site*Year	< 0.0001	3.86	
Relative to Bare NEE	ANOVA	N/A				
	ANCOVA					

Table 4.6: Output for relative to control ANOVA and ANCOVA (2008)

2007 - 2011						
Responsible Variable	Test	Factor (F)/ covariate (C)/ interaction (I)/ nested factor (N)	Variable	P	ω^2 (%)	$_{adj}R^2$ (%)
Relative to Bare lnR $_{eco}$	ANOVA	F	Site	< 0.0001	9.15	46.57
		F	Year	< 0.0001	4.59	
		N	Month (Year)	< 0.0001	19.51	
		N	Collar (Site)	< 0.0001	11.21	
		I	Site*Year	< 0.0001	2.09	
	ANCOVA	C	AT	< 0.0001	5.98	46.57
		F	Site	< 0.0001	8.95	
		F	Year	< 0.0001	5.15	
		N	Month (Year)	< 0.0001	17.39	
		N	Collar (Site)	< 0.0001	12.38	
		I	Site*Year	< 0.0001	2.84	
Relative to Bare lnP $_g$	ANOVA	F	Site	< 0.0001	5.84	52.39
		N	Collar (Site)	< 0.0001	14.40	
		F	Year	< 0.0001	4.59	
		N	Month (Year)	< 0.0001	24.89	
		I	Site*Year	< 0.0001	2.64	
	ANCOVA	C	lnAT	0.003	5.00	56.46
		C	WTD	0.006	0.53	
		C	PAR	0.016	0.83	
		F	Site	< 0.0001	5.60	
		N	Collar (Site)	< 0.0001	13.95	
		F	Year	< 0.0001	6.27	
		N	Month (Year)	< 0.0001	20.37	
		I	Site*Year	< 0.0001	3.86	
Relative to Bare NEE	ANOVA	N/A				
	ANCOVA					

Table 4.7: Output for relative to control ANOVA and ANCOVA (2007-2011)

4.5 Discussion

4.5.1 Water table depth

Analysis of the Bleaklow WTD (absolute data) (Chapter 3) revealed that WT was shallowest at the least disturbed vegetated control site, both in 2008 (earlier on during the study) and across the five years of monitoring. As the vegetated control was considered the ideal WTD on Bleaklow, analysis of the WTD was conducted relative to the vegetated control. This comparison was used to: put the changes at sites in context with the CO_2 fluxes; determine if the sites with treatment were becoming more similar to the controls over time. The bare site controls had a smaller n number than the least disturbed vegetated control (16% fewer data points) this is due to occasions in which the WT at the bare control was deeper than the detection limit (>80 cm). This difference in n number could have resulted in the underestimation of WTD at the bare controls, in addition to a greater similarity of bare controls to the sites with treatment. Therefore analysis of WTD was focused more on the use of the vegetated control as a comparator when using relative values in place of absolute.

Water table depth was significantly influenced by site variation in both 2008 and the five year analysis. Site water table deviation from the vegetated control mean varied both seasonally and over time regardless of which site. Across the five years of monitoring the variation between plots within a site as more important than variation between treatment sites. Site morphology had an important part to play in the variation. This is evident at the naturally revegetated and gully blocking sites. These sites had the most similar water table to each other and the vegetated control. There was a lack of WTD recovery through the three treatments at the seeded and limed, heather brash and geojute sites, which have significantly deeper WTD than the least disturbed vegetated control. The three treatment sites were also found to have the most similar WTD to that of the bare soil site controls, both in 2008 and across the five years. Thus after two years of monitoring in 2008, without hydrogeological targeted restoration, the recovery of the WTD to a shallower depth to that of a 'pristine' site was unlikely without gully blocking.

The importance of site variation on WTD was reduced over the five years of monitoring. The relationship between the depth to water table at sites relative to the vegetated control and relative to the bare soil control sites did not change with time in the project. Thus this indicated the differences between the sites were consistent over time. The very small interaction between site and project years indicated that there was little improvement in the water table of the restored sites over the five years, and the sites remained different to the vegetated control. Given that the monitoring began after the restoration treatments were applied, it is possible that the impact of restoration occurred before the monitoring started. However, the changes in WTD were not sufficient to show significant continuing improvement over the five year period.

In 2008, temperature was not significantly related to site WTD, however during the five years of monitoring (2007–2011) there were small positive correlations of both PAR and air temperature to deeper site WTD. Kettridge *et al.* (2012) wild fires alter near surface temperatures and soil water evaporation rates, therefore influencing peat thermal hydrological conditions and post fire peatland recovery. The damage to peat properties post a wildfire can contribute to explaining why without intervention the bare sites remain bare and their hydrology is difficult to restore. Furthermore areas of peat with low vegetation cover are less protected against desiccation, such as the seeded and limed site with a greater similarity in WTD to the bare control (bare gully and flat). In the case of sites with a shallower WTD, closer to the surface peat, the water table is more greatly influenced PAR and temperature variation. The vegetated sites would expect greater loss of water via transpiration related to the process of photosynthesis which would require PAR. On Bleaklow over the five year period annual variation in weather, specifically why PAR was significantly related to increase in WTD The importance of peat soil properties and vegetation cover to WTD and their link to potential transpiration; is supported by the evidence. Between 2006 and 2011, year 2009 had the second lowest annual rainfall (other than 2010) and highest recorded temperature range (as explained in Chapter 3). In 2008, the shallowest water table depths were found at the vegetated control naturally revegetated and gully blocked sites. These sites were also more resistant to seasonal or annual changes as they had little WTD fluctuation. On the other hand, the heather brash site had the greatest fluctuation of WTD and over time and was becoming less similar to the bare soil control. However, over the five years the three sites (heather brashed, seeded and limed and geojute site) all had WTD more similar to the bare soil control sites than the least disturbed vegetated control.

The effect of PAR on variation of WTD was important, but was less relevant in variation of site water table relative to the vegetated control. This means that, relative to the pristine site (the site with the shallowest WTD), the WTD was influenced less by PAR. The importance of month within year indicates that seasonality was more important than the differences between sites in influencing changes in WTD relative to the changes occurring at the vegetated control. However the differences in WTD relative to the bare control are attributable to site variation, thereby supporting the need for restoration.

4.5.2 Carbon outflux (R_{eco})

Significant differences in R_{eco} were found between sites earlier on in the study in 2008 and over the five years. The sites with the highest bare surface area specifically the bare soil control and seeded and limed site had the lowest R_{eco} in 2008. The seeded and limed site had the most similar R_{eco} flux to the bare sites in 2008 and differed to the more vegetated sites (least disturbed control, heather brash and geojute). However, the R_{eco} flux over the five years of monitoring, at three sites (the seeded and limed, heather brash and the vegetated control) were all higher than the bare soil control. In 2008 the flux was highest at the well vegetated sites regardless of WTD. The relatively lower R_{eco} at Bleaklow's dominantly bare sites than at vegetated site (both control and restoration sites), can be explained by the bare sites absence of vegetation, which reduces the potential for above ground autotrophic plant respiration. Above ground respiration has been estimated at between 35 to 50% of R_{eco} (Crow and Wieder 2005, Moore *et al.* 2002).

Restoration techniques that successfully reduced bare soil cover did not have reduced R_{eco}. Furthermore site with the soil stabilisation methods (geojute) site had the highest R_{eco} relative to the bare soil control, earlier on in the in 2008 and throughout the 5 year study. Quin *et al.* (2014) also found that upland heath restoration did not lower the rate of soil respiration below that of degraded areas. The findings for the Bleaklow sites support a relationship between R_{eco} and site vegetation. In 2008, the heather brash site R_{eco} did not significantly differ from the geojute. Over the five years, the geojute R_{eco} flux increased and became progressively different to the bare soil controls. Geojute R_{eco} over took the heather brash site flux as being the site flux most different to the bare soil. Over the 5 year study, the heather brash R_{eco} flux became increased similarity to the bare site R_{eco}. This is possibly due to the decomposition of heather brash earlier on in the study which would have had an initial increase in R_{eco}, as found for mulches on restored sites in Canada by Waddington *et al.* (2003). There is therefore evidence that the use of heather brash can initially increase the CO_2 outflux from a site due to an increased microbial activity. The changes in vegetative cover at the restoration sites also have induced higher R_{eco} flux, as a result of a change in productivity within the upper peat (root zone)(Rowson *et al.* 2013). The presence of vegetation encourages the production of CO_2 from the rhizosphere of peatlands through root and microbial respiration of root products (exudates, mucilage, or dead tissue) in addition to decomposition of peat (Cheng *et al.* 1993, Cheng *et al.* 1996, Kuzyakov 2002, Rowson *et al.* 2013).

When compared to the bare control, the differences between each collar within each site were important in explaining the changes in site R_{eco}. This finding on the Bleaklow sites is supported by Soini *et al.* (2010). In Canada Soini *et al.* (2010) found that R_{eco} flux, post restoration of cut over peatlands, was greater at the restored site due to the heterogeneity of the vegetation. On Bleaklow, air temperature was an important covariate both early on in the study in 2008 and throughout the five years of monitoring, with greater R_{eco} fluxes during the summer periods, indicating a sensitivity of R_{eco} to near-surface temperatures as found by Samaritani *et al.* (2011) and Lafleur *et al.* (2005). There was a greater variation relative to the bare soil site which would suggest the change in vegetation and microbial community had an important role. The importance of difference in vegetation between the sites is evidence as although the importance of air temperature was reduced over time, the importance of monthly variation and the differences between site collars became more important in influencing the changes in R_{eco} which occurred relative to the bare controls.

Ecosystem respiration was measured in the dark, hence it follows PAR was not a significant covariate. In 2008, near soil air temperature measured inside the chamber, was the biggest explanatory variable of R_{eco} flux although air temperature and site had similar importance. Quin *et al.* (2014) also found R_{eco} was significantly related to air temperature. The flux magnitudes were lowest at sites dominated by bare soil cover, specifically the two bare soil controls (bare flat and bare gully) and the seeded and limed sites. This finding is not surprising and has been reported from cutover peatlands in Canada (Waddington *et al.* 2003). Relative to bare peat, the presence of vegetation on the soil surface stimulate both the autotrophic and heterotrophic components of ecosystem respiration (Knorr *et al.* 2008). The presence of vegetation increases the microbial biomass within the soil, and therefore alters the community structure of heterotrophs through provision of root exudates (Crow and Wieder 2005). Additional substrates for heterotrophs will be provided via litter accumulation, the effect of which will differ with the species present (Bragazza *et al.* 2007, Zhang *et al.* 2001). The apparent effect of the introduction of these new sources of labile substrate onto the bare peat was to increase the temperature sensitivity of peat surface R_{eco}. The most effectively revegetated sites were the one where gully blocking was employed and the two in which stabilisation techniques were used. Waddington *et al.* (2001) demonstrated that Q_{10} factors at bare cut-over peat soil substrates were greater than the comparable intact soil substrates. Therefore, the site modification at the restored sites was due to the presence of surface vegetation and the substrates it produces. The difference between the bare soil controls to the vegetated control and treated site R_{eco} increased from year 2007 to 2008 to 2011 which supports the finding that R_{eco} was sensitive to change in air temperature.

Many studies indicate there is an important relationship between WTD and R_{eco}, for example: Dimitrov *et al.* (2010) found that R_{eco} was significantly influenced during periods of drought. Knorr *et al.* (2008) found that after periods of draught respiration was impacted at lower depth of peat even after the rewetted of a site. On Bleaklow it was found that (of both treatment sites and vegetated control) R_{eco} variation relative to the bare soil controls, was not significantly related to WTD. This is evident despite the raised WTD at the gully blocked site, the only site with hydrological manipulation. The gully blocking R_{eco} was no different to the remaining restoration sites or vegetated controls. Thus variation in R_{eco} relative to the bare soil control was not significantly explained by WTD. Furthermore, the three sites with the shallowest WTD in 2008 (vegetated control, revegetated and gully blocked sites) had greater R_{eco} than the bare soil control, and over the five years of monitoring, despite the least disturbed vegetated control continuing to have the shallowest WTD, most different to that of the bare soil control, the vegetated control R_{eco} did not significantly differ to the seeded and limed site (dominantly bare). The vegetated control R_{eco} flux did not differ to the bare soil control; the heather brash site had higher R_{eco} flux than both controls. These findings on Bleaklow are supported by Lafleur *et al.* (2005) in which R_{eco} was not significantly influenced by WTD variation in dry bogs, with WTD varying between -30 and -75 cm below the surface. The specified WTD is similar to that found at all but three Bleaklow sites (vegetated control, naturally revegetated and gully blocked). At these depth, specifically between -40 and -50 cm below the surface, the peat is highly recalcitrant and as such there is little available substrate to decompose (Frolking *et al.* 2001). This point may also explain why some findings such as that by Crow and Wieder (2005) which indicate there is greater peat and roots CO_2 production under dry conditions than wet conditions. The high level of erosion on Bleaklow, especially at the bare soil sites, implies it has lost in the region of 13 cm of peat in five years 2007 -2011.

The fire occurred in 2003. Therefore these sites would have been bare at that point and could have lost up to 26 cm between 2003 -2011. Given the sites had already been subject to degradation, it is likely the peat at surface was in fact lower in the peat profile, and therefore had high recalcitrant material. Hence, that the bare soil control would therefore have low R_{eco} as the recalcitrant material is less labial or able to decompose, therefore resulting in less atmospheric carbon loss as R_{eco}.

The true night time R_{eco} where measured by Dixon (2011) at the least disturbed vegetated site. Dixon (2011) found that daytime variability in R_{eco} was not great discernibly greater than night time fluxes at the least disturbed control and daytime R_{eco} measurements could be used to adequately estimate daily (day and night) R_{eco}. However the presence of vegetation at the peat surface can reduce the surface albedo relative to bare peat which has a lower WTD (and surface water content) associated with higher albedo (Idso *et al.* 1975). So the differences in soil temperature fluctuation from day to night at dominantly bare site and vegetated sites may differ. Therefore in addition to differences in daytime and nigh time temperature may vary between sites and be reflected in difference between site R_{eco}

4.5.3 Carbon influx (P_g)

Gross photosynthesis was derived from NEE measured during light and R_{eco} was measured during simulated night. There is significant difference in terms of gross (and net) productivity both earlier on in the study and over the five years of monitoring. Previously bare sites revegetated through treatment, or as a result of morphology as on the naturally revegetated site, had significantly greater rates of gross photosynthesis than untreated bare soil sites. In 2008, monthly variation was the most important explanatory variable. This implies that slope stabilisation (mulch/geojute) in addition to seeding, liming and fertilisation is advisable to maximise restoration benefits on CO_2 fluxes which supports other findings including Price *et al.* (1998), Sliva and Pfadenhauer (1999), Waddington *et al.* (2003). The use of mulch reduces the surface runoff and erosion (Meyer *et al.* 1970). This was evident at the heather brash site, which deposited at an average of 0.14 cm of peat per year.

According to Sliva and Pfadenhauer (1999) the use of mulch as a protective layer provides seed-stock, assisted seed germination, and maintenance/restoration of target species. Left to decay onsite, the mulch contributed to higher vegetation establishment and results in an initial increase in R_{eco}. Sliva and Pfadenhauer (1999) also found mulch was more successful at revegetating peat than geojute was. On Bleaklow the naturally revegetating site was the only site where *Sphagnum spp.* was observed. This formed behind peat bars in the gully floor, and was stabilised by vegetation from the sedge functional group (such as *Eriophorum angustifolium*). This revegetation pattern matched the findings of Crow and Wieder (2005). Given sufficient time, it is possible that *Sphagnum spp.* could re-establish at the heather brash site, providing that the environment could progress to that of the revegetating gully in which the sedge (e.g. *Eriophorum spp*.) continues to regenerate and the water table continues to rise. Additionally, the use of seeding and liming is preferred over leaving the bare sites untreated. However, it is evident that mulch increased the likelihood of seed germination as found at restoration of cut-over bogs in Southern Germany (Sliva and Pfadenhauer 1999).

All the treated sites and both the vegetated control and naturally revegetated site had higher P_g in comparison to the bare soil controls. At the geojute site P_g was greatest relative to the bare soil controls, which was arguably the most successfully at moving away from the low magnitude of the P_g on the bare soil flux. Although it was insignificantly different to the heather brash, over the five year times scale it remained the most different to the bare site. The seeded and limed site had a greater magnitude of flux than the bare site and failed to revegetate at the same rate as the other treated sites. The seeded and limed P_g flux was most similar to the bare controls in 2008. Interestingly, despite the difference in bare soil dominance between the vegetated control and the seeded and limed site over the five year period both had a similar P_g to the bare sites. This is likely due to the low vegetation cover on the seeded and lime and the lack of new vegetative go at the LD-F.

The significance of the site factor can be explained through the dependency of P_g flux magnitude on the vegetation cover as found by Burrows *et al.* (2005) and vegetation composition by Lafleur *et al.* (2003), which found that vegetation from the shrub functional groups for example had the largest portion of photosynthesis when compared to sedges and grasses. The effect of vegetation change is further evident when comparing the P_g flux of sites to that of the bare soil controls. It was found that the differences between sites were less important than differences within sites at explaining the relative fluxes. This difference is likely the influence of increased variation within restored sites than at unrestored sites, as found by

Soini *et al.* (2010). The newly vegetated sites with greater vegetation cover have both the highest P_g and R_{eco}. The collars dominated by bare soil in the seeded and limed (collars 1-4), grass heather brashed (collar 6) and vegetated control (collars 3 and 5) have a similar R_{eco} and P_g to that of the bare soil controls (refer to site photos: appendix B). The variation between the collars was greatest between seeded and lime and geojute site. Interestingly a collar dominated by bare soil at SL.HB-G no.6 was the collar with the lowest relative P_g. However, it had a $R_{eco,}$ similar to the bare soil controls than the SL-F restoration site with least successful restoration. This indicated that there was a difference in the microbial community at the heather brash site as there is a lack of vegetation to conduct photosynthesis. There was a change which caused the R_{eco} flux to become less similar to the bare controls. Additionally, the presence of vascular plant roots (i.e. *Calluna*) may be responsible in causing an increase in the microbial activity of the rhizosphere, hence resulting in an increase in CO_2 emission due to microbial respiration as found by Crow and Wieder (2005).

In the full five year analysis of the revegetated sites, where slope stabilisation techniques were used (at the geojute and heather brash site), had greater productivity than the least disturbed vegetated control site. Two years into the monitoring in 2008, the two stabilised sites had the greatest P_g flux, higher than that of the gully blocked site (despite the hydrological intervention). According to Dimitrov *et al.* (2010), P_g is sensitive to variation in WTD which supports the correlation between WTD and P_g evident at the gully blocked site. The gully blocking allowed WTD to rise, thereby decreasing plant stress and encouraging photosynthesis and suppressing R_{eco} (Mezbahuddin *et al.* 2014).

Other than the variation in vegetation or water table over time, the other important variables to consider were related to climate. The air temperature was not found to be significantly related to P_g flux in 2008, however PAR and WTD were. Over the five years, the

WTD and PAR became more important. When comparing the changes occurring in the vegetated control, naturally revegetating and four treatments sites to that of the bare soil controls, it is clear that the importance of WTD and PAR and decreased in 2008. However, over the full period of the study, air temperature, WTD, and PAR all have some importance. This suggests that the vegetative and microbial communities at those sites are more influenced by environmental changes than the bare soil site, and changes in those environmental variables influence their capability of taking in carbon. In 2009, the annual temperature ranges with the highest and lowest daily mean temperature was measured. It was also during the summer period of 2009 that sites with vegetation had greater photosynthesis. Findings by Bubier *et al.* (2002) support the importance of temperature on P_g, as they found it occurred only when ground temperatures were above 0°C. The annual and seasonal changes are therefore important for the future of carbon storage in peatland.

4.5.4 Net ecosystem exchange (NEE)

Net ecosystem exchange is measured in during daylight hours. It is a function of CO_2 influx (gross photosynthetic activity) and outflux (ecosystem respiration). Negative NEE fluxes indicated greater influx than CO_2 outflux and would indicate a peatland is a sink. However, where R_{eco} exceeds the rate of P_g, this results in a positive NEE such as that found at the bare soil controls. As discussed earlier the R_{eco} is best explained as CO_2 productivity at two zones: the upper peat (root zone), in which CO_2 export is related to microbial production and labile carbon from plant root exudates and root respiration, and the lower peat (below root zone), in which CO_2 production is lower with less labile carbon (Rowson 2007).

Surface recession and deposition data (Table 4.3) is also indicative of a sites sink or source behaviour. Sites with slope stabilised had accumulated material over time, whereas all other sites (total of 6 monitored for the five year time series) had lost material. These results appear to demonstrate the greater effectiveness of the slope stabilisation measures undertaken on the Bleaklow Plateau when compared to revegetation measures alone.

In 2008 the sites with the shallowest WTD (vegetated control, naturally revegetated and the gully blocked sits) were small sinks. On the other hand, sites in which stabilisation techniques were employed (geojute and heather brash) were the greatest net sinks and continued to be net sink over the five year duration study. These sites had the greatest outflux through R_{eco}. However, the loss was offset through their productivity, which was evident in their successful peat deposition over the study period. The NEE flux of the seeded and limed site was not significantly different to the bare soil site earlier on in the study, as it had a small R_{eco} outflux. The influx through P_g was also too small for the site to be a sink in 2008. The site did however become a net sink over the five year duration. Thus, although seeding and liming was less successful than other sites with stabilisation techniques, it would be preferred to no treatment of the bare soil. Interestingly the vegetated control did not significantly differ to the bare soil controls, and despite it being a small sink in 2008, it was neither a sink nor source over the five years and incurred some peat losses.

Many interacting variables contribute to NEE including PAR, temperature, water table, plant biomass, and species composition (Burrows *et al.* 2005). The notion of a link between water table depth or surface moisture and net exchange of CO_2 is reported elsewhere (Petrone *et al.* 2004, Waddington and Price 2000, Waddington *et al.* 2001). Dimitrov *et al.* (2010) also found NEE is sensitive to WTD, as did Soini *et al.* (2010) who found that once the peatland was in a good hydrological condition, restored peatlands were able to become a large CO_2 sink during earlier stages of revegetation, which to a small extent support the finding at SL.B-G, although the sites with stabilisation techniques had greater success as net sinks. Gully blocking was found to significantly raise the WTD, thus it could be used to allow increased chances of creating net CO_2 sinks in these areas of degraded peatland hydrological function (Allott *et al.* 2009). It is possible that given time the WTD of the heather brash site (the site with the most similar vegetation dominance to the vegetated control) will continue to rise which should aid the reduction of respiration. Considering the importance of vegetation on a sites carbon fluxes, there is uncertainty in relation to the future of the sites vegetation succession towards a greater presence of S*phagnum spp.* and 'good' peatland ecosystem. To address this issue, the restoration and control sites may require future monitoring in order to assess long term effects of the restoration (+10 years) on the ecosystem community dynamics alongside CO_2 flux and water table depth variation.

4.6 Conclusion

This chapter had the research aim of investigating the effects of bare peat re-vegetation upon CO_2 fluxes from the peat; in order to assess which of the treatment techniques would provide the best results in terms of decreasing land to atmosphere carbon losses. Analysis of the data found that:

- Sites with treatment had significantly differing vegetation cover and CO_2 fluxes to the locality controls (bare and vegetated).
- Sites treated through seeding and liming and through the use of stabilisation techniques, had both increased R_{eco} and P_g to a magnitude greater than the bare soil controls and the least disturbed vegetated control.
- Variation in site CO_2 fluxes was due to multiple factors including: site morphology, hydrology, vegetation cover (reduced bare soil cover dominance), and seasonal variation (near surface air temperatures and PAR).
- Variation in WTD was significantly linked to the potential of sites to become C sinks. The sink potential can be more greatly improved through new vegetation establishment and development of a litter layer that allowed for peat deposition, as found at sites in which soil stabilisation techniques were used.

In order to gain the maximum benefits of reduced atmospheric carbon losses, vegetative restoration of bare peat should be conducted using a combination of seeding, liming and stabilisation techniques.

5 The effects of heather management, through cutting and burning, on DOC and water quality

5.1 Rationale

Anthropogenic interventions in the uplands have been widespread since Mesolithic times (Reed *et al.* 2009). The UK peatlands are intensely managed; these peatlands have also been impacted by a legacy of atmospheric pollution, tourism, overgrazing and wildfire (Holden *et al.* 2007). However anthropogenic impacts (through mismanagement) have been identified as the biggest causes of peatland degradation according to a report by Natural-England (2010). Yallop *et al.* (2006) found that approximately 40% of the uplands surveyed displayed evidence of managed burning between 1996 and 2000. Current methods employed to manage upland peatland ecosystems include: drain blocking, cattle and sheep grazing, managed heathland burning and vegetation cutting (Mitchell *et al.* 2008).

Prescribed burning is the main management practice used in the UK in order to rejuvenate heather (*C. vulgaris*) (Ross *et al.* 2003), increase productivity for sheep and game (i.e. grouse), and achieve a mosaic of multi aged heather stands favourable for grouse (Tharme *et al.* 2001). A varied heather habitat is required by the grouse, specifically Juvenile heather as a grazing habitat whilst mature heather is required for nesting.

Studies by Clark *et al.* (2005; 2008), documented the link between peat soil pore water DOC concentrations and catchment stream-water. Worrall *et al.* (2003b) found a widespread increase in surface and soil pore-water, colour and DOC concentrations, within blanket peat dominated catchments. These increases have been attributed to various drivers including increased air temperature, decreasing soil pH, increased variability in hydrological inputs and rising atmospheric CO_2 concentrations (Freeman *et al.* 2004, Worrall *et al.* 2003a, Worrall *et al.* 2003b). Furthermore, all of these factors interact with land management (Mitchell *et al.* 2008, Wallage and Holden 2010). Given that many of these peat covered catchments also provide drinking water to a large percentage of the UK population (Bonn *et al.* 2009), concern has been raised about the release of carbon into headwaters in the form of dissolved organic carbon (DOC) and links to burning as a management practice (e.g. Yallop and Clutterbuck 2009).

5.1.1 Vegetation management effects on fluvial C in peatlands

Both Neff and Hooper (2002) and Vestgarden *et al.* (2010) found evidence that vegetation management may provide a means of reducing the increasing trend in DOC. These findings were supported by Armstrong *et al.* (2012) who found that higher soil pore water DOC concentrations were more associated with heather than with sedges and *Sphagnum spp*. vegetation. Furthermore, increased vascular plant dominance (shrubs such as heather and bilberry) is associated with an increase in net carbon fluvial output in the form of DOC (Fenner *et al.* 2009). In Chapter 3, it was found that interventionist vegetation management through bare peat restoration significantly influenced soil pore water pH and DOC speciation in addition to WTD. The research on Bleaklow provided supporting evidence that vegetation management can be used to influence water quality. There is little research into the effects of heather management through cutting on DOC. On the other hand there has been much debate as to the impact of burning on DOC in soil pore and surface water with the exception of (Worrall *et al.* 2012) who found that in a short term study, both cutting and burning lead to raised water tables and a decline in soil water DOC concentration (Worrall *et al.* 2012). These differences in DOC were linked to water table position and soil water conductivity. It was also found that burning of *heather* lead to an increase in both surface runoff frequency and E_4/E_6 ratio relative to site in which heather was cut.

The research presented in this chapter uses some data from Dixon (2011) and is in part a continuation of the study by Worrall *et al.* (2012). The research in this chapter is focused on the impacted of: vegetation management using fire for the purpose of maintenance on water quality and DOC, unlike the Bleaklow study which focused on the effects of restorative management methods used to counter the negative impact associated with of wildfire. It is important to conduct this research as there is a literature gap with respect to the comparison between the long term effects of cutting and burning in the UK, particularly on the effects relating to water quality and DOC.

5.2 Aim

The aim of this study was therefore to investigate:

- heather burning *vs.* cutting on dry peat

The effect of these management practices on soil pore water DOC concentration and composition will be explored. Burning and cutting were compared between locations with contrasting water tables. The null hypotheses of this study area is that there is no significant difference in soil water or runoff water DOC concentrations between untreated heather control sites and sites in which the heather was burnt or cut, at wet or dry peat conditions. The hypothesis will be tested through analysis of a multi annual data set of soil water table, soil pore water sample DOC concentration, UV-Vis absorbance, pH, and conductivity. Analysis of seasonal and annual data trend will be conducted using graphs, comparative analysis of the data, ANCOVA GLM models and post hoc tests to find where differences lie.

5.3 Methodology

5.3.1 Study sites

The study sites were installed in the Goyt valley 4 km west of Buxton, in the South West Peak across two localities named: Big Moss (a dry locality) and Ravenslow (a wet locality) (Chapter 2, section 2.2). Monitoring in the Goyt Valley was conducted was conducted between May 2008 – May 2013 The study sites used in this thesis research are the same as those used by Dixon (2011), as such Dixon (2011) is the source of data (analysed in this thesis) collected until July 2011. Hence the study is in part a continuation of his work with a change in focus and temporal scale of the study.

5.3.2 Experimental design

The experiment was designed to compare cutting as an alternative to burning of heather. Refer to the experimental site details in Chapter 2.2. When heather is cut, short stick ~100mm is left behind compared to ~126 mm when it is burnt (as measured across five freshly cut sites and one burn site in the Goyt Valley, March 2012). The cutting of heather was performed in two ways: a) cut and lift, b) cut and leave. In both cases the vegetation was flailed to the ground level but in the former case (a) the cuttings were removed from the site, while in the latter (b) the cuttings were left where they fell.

The treatments available to the study were: new cut and leave (cut at the outset of the experiment), old cut and lift (cut 1 year before the start of the experiment), new managed burn (burnt at the outset of the experiment), old managed burn (burnt 1 year before the start of the experiment), and a heather untreated control (not cut or burnt in over 15 years). The control site vegetation cover was dominated by mature to degenerate phase heather. The canopy structure was open, allowing mosses and lichens to develop, typical of a site where there has been no burn management for more than 15 years.

The sites were chosen for their particular management; each treatment site had six plots, divided into duplicate sets of three nested plots. Within in each plot there was equipment to sample soil pore water and surface runoff water. Soil water from below the water table was accessed via a series of dipwells from the surface. In each plot three dipwells were placed to at least 90 cm depth with openings along their entire length. Water table depth was measured and soil pore water samples were collected from site dipwell, as per the methodology detailed in Chapter 4. The sample from the dipwell was indicative of soil pore water quality along across a ~1m soil depth.

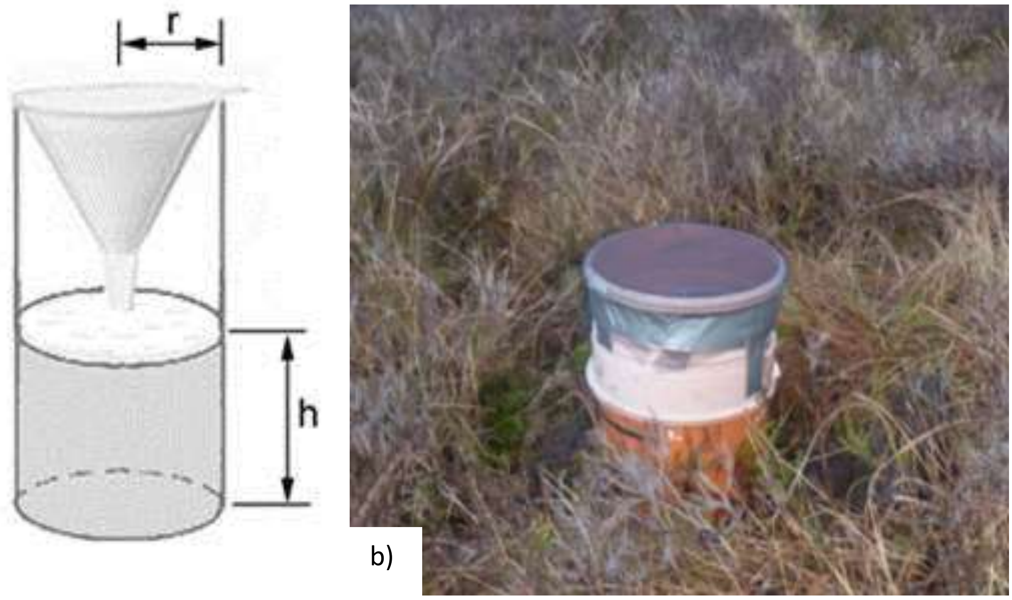
In addition to the dipwell (detailed chapter 2, section 2.3.1.1), crest-fall runoff traps were used to collect monthly, accumulative, intercepted surface runoff water. The trap was made of ~30 cm UPVC long tubing (The same tubing used for the dipwell), with 4 symmetrical holes drilled at 10 cm down the tube, on opposite sides. This allowed water to enter from different directions. A bung was inserted into the bottom and at the top to exclude the entrance of ground or rain water, environmental degradation (photo degradation), and airborne contamination of the sample. The trap was inserted into the ground with the holes just above the ground surface so that holes were aligned with, and perpendicular to the local slope, allowing only over ground surface flow to enter the trap. Sample presence was noted on monthly site visits, after which the runoff trap samples were collected. This was done by the removal of the trap from the ground (with minimum disturbance). The water sample was then poured into a sample bottle and the trap placed back into the ground, while ensuring the inlet holes were flush with the ground. To reduce cross contamination of samples a pump was not used.

In total the Goyt valley study consisted of 48 dipwells and 48 runoff traps. Over the study time period it was a possible to analyse DOC concentration variation in both soil and runoff water. However, due to low water tables or absence of water runoff it was not always possible to sample soil or runoff water, particularly in the drier summer period. For example between April 2011 and May 2013, it was not possible to collect runoff water 57% of the time. Water sample absorbance was measured at 400 nm for a basic colour reading (Thurman 1985). The DOC concentrations were measured colorimetrically using the method by Bartlett and Ross (1988). By measuring both absorbance at 400 nm and DOC, specific absorbance can be evaluated and thus the nature of the DOC could be tested. Furthermore, the E_4/E_6 ratio (the ratio of absorbance at 465 nm to absorbance at 665 nm) was also measured as an additional assessment of DOC composition. Chen *et al.* (1977) show that the E_4/E_6 ratio is an indicator of humic acid and fulvic acids. The pH and conductivity were measured by electrode methods.

5.3.3 Site characterisation

A Volume measure method was use to collect rainfall. In February 2012, four rain gauges were installed, two at each locality: At Ravenslow (‘Kra’ - heather control and ‘Nep’ - the old burn) and Big Moss (‘OB’ - new burn and ‘Ben’ - new cut and leave. Monthly mean rainfall data (chemistry and volume) was also obtained from the Sustainable Catchment Monitoring Program (SCaMP) (2008-2012).

The rain gauges were installed adjacent to sites, away from high vegetation or fencing. The rain gauge was composed of a funnel placed over a vessel (with narrow inlet). The funnel and vessel were placed inside a cylinder unit (Figure 5.1.a) and buried 20cm deep within the peat (Figure 5.1.b) to protect the sample from being blown over and to reduce temperature fluctuations and evaporation. The top of the unit was then covered with mesh netting to prevent debris or leaf litter being collected and contaminating the sample. On a monthly basis, the volume of the rain water calculated by measuring the height of the water within the vessel. The rain water was then sampled and the remaining water vigorously emptied to clean out any potential algae build up. The vessel height (h) and the cylinder radius (r) were used to calculate the volume and monthly total rainfall in (mm/month).



a) b)

Figure 5.1. Rain gauge, a) cross section b) example of gauge in field (Ravenslow, Nep)

Vegetation was surveyed at each plot and its collar as conducted on the Bleaklow Plateau (Chapter 3). The classifications used in the survey were more specific than the functional groups use previously in Chapter 3. The cover group types used in the vegetation survey in the Goyt were: bilberry (i.e. *Vaccinium*), heather (*C. vulgaris*); cotton grass (i.e. *Eriophorum spp.* - sedge), molinia (i.e. *Molinia* - grass), lawn grass (i.e. other), non-*sphagnum spp.* moss, S*phagnum spp.,* stick (dead shrub) and bare peat. As shrub species are the target cutting and burning management, the heights of both heather and bilberry were also measured using a tape measure. A replicate of up to three height measurement were attempted per plot to be used in calculating a mean plot heather or bilberry height. Depending on the extent of the vegetation cover the number of measurements varies. For example in cases where heather cover of the plot was zero the heather height would also be zero.

To characterise the differences between sites, peat soil bulk density and peat depth were also measured. In August 2011, peat core samples were collected (n= 32), four from each site in the Goyt Valley. Using an auger, 10 cm by 2.5 cm diameter depth core samples were collected. To calculate bulk density (Equation 5.1), the samples were dried in an oven at 105 °C overnight, so that the water would evaporate and the dry weights could be measured and recorded. Due to the physiochemical properties of peat (e.g. low conductivity), water occupies a high volume in peat. Peat sample bulk density (Equation 5.1) would typically be below 0.5 g/ cm^3 and is a measure of sample mass (sample dry weight) divided by sample volume (Equation 5.2). Bulk density can be used as a measure of decomposition (Boelter 1986).

$$Bulk\ density\ = \frac{dry\ sample\ mass\ (g)}{Sample\ volume\ (cm^3)}$$

Equation 5.1

$$Sample\ volume\ =\ sample\ height\ x\ \pi r^2$$

Equation 5.2

There is a significant relationship between peat depth and slope (Holden and Connolly 2011). This is due to the importance of slope as a factor explaining water drainage and erosion (Holden *et al.* 2007). A Digital Elevation Model (DEM) map with slope and elevation information can be used to predict peat depth. Such predictions are more accurate in deeper blanket peats as found on blanket bogs (Parry *et al.* 2012). The peat in the Goyt valley is relatively shallow compared to other upland bogs, therefore infield measurements were required to ascertain better quality peat depth measures using manual probing. Parry *et al.* (2012) recommended the use of one central measurement and four at right angles 4m from the centre to allow for localised variability, however this method was not possible due to time constraints. Parry *et al.* (2014) recommended that probes be conducted at 2–3 m apart to avoid large artefacts or clusters of woods, this was possible in this study as the sites are composed of replicate plots, and therefore the peat depth measure per management type is replicated within plot. Six peat depth measurements were taken per site. The peat depth measurements were used to calculate water table height (WTH) up from the mineral soil (Equation 5.3). This was used instead of WTD down from the surface to consider the importance of peat depth and the water table on DOC concentrations.

$$Water\ table\ hight\ = Water\ table\ depth - peat\ depth$$

Equation 5.3

5.4 Statistical methodology

Statistical analytical methods are presented in the following order: Site characterisation, in terms of weather variables (surface ground water temperature and rainfall (mm)); peat depth and bulk density, WTD, plot heather (height (mm) and cover (%)); followed by water samples analysis for DOC and E_4/E_6 ratio. The analysis was conducted in this sequence in order to characterise the study sites and better understand the variables influencing the water sample composition.

5.4.1 Site characteristics

Descriptive statistics and ANOVA were used to analyse the peat soil properties across the Goyt monitoring sites, which were analysed by site to account for site specific variation between plots. Factors included were localities (wet or dry), sites (by treatment type), and water sample type (surface runoff or soil pore water). Locality and site specific variables investigated were environmental (temperature and rainfall), peat (depth, bulk density and wet weight), and plot vegetation (cover and canopy height). Vegetation cover type was analysed using the factors sites by treatment type, locality and vegetation class (functional groups). The functional groups used were: bare, bilberry, heather, cotton grass, lawn grass, non-*sphagnum spp.* moss, *sphagnum spp.,* stick and bare soil. The dominant vegetation type was further investigated between wet and dry locality using ANOVA. Mean height plot values were used in the ANOVA, because depending on the extent of the heather plot cover a different number of heather height measurements were collected (between 0 – 3). After which the target vegetation (*C. vulgaris*) height and cover was investigated. Site peat (peat depth and surface bulk density), WTD, soil temperature and rainfall are also analysed. The difference between the sample wet weight and dry weight was calculated for the sample used to measure bulk density and analysed using ANOVA. The importance of site was further investigated using Pearson correlation and regression between WTD and peat depth and bulk density.

5.4.2 Surface and soil pore water DOC

The study was designed to include a number of factors. Each site (and its plots) had a unique treatment (e.g. cut) and no additional treatments were applied to any individual plot within the study period and so a repeated measures design was not required. Analysis of variance was conducted to assess the differences between site water sample (soil pore and runoff) DOC concentrations. The data by a) wet burn verses dry burn sites (verses controls), and b) on dry locality cuts sites verses burns (verses control). Factors included in the ANOVA were: Site location (two levels), which represented the difference between the wet (Ravenslow) and dry heath sites (Big Moss). Site treatment included the following eight levels: new managed burn, old managed burn, fresh cut and leave, old cut and leave, old cut and lift, and a heather control. Project year (five levels 1-5; each project years runs May - April), did not represent calendar years as the experiment started in June. Month (12 levels, January till December), which was indicative of seasonality. Finally plot was a factor nested within site.

Analysis of water table depth, runoff and soil pore water DOC concentrations, and E_4/E_6 ratio were conducted in a number of ways. Firstly, DOC concentrations absolute data were analysed across sample type (soil pore water and runoff water samples), and were tested for differences across all sites (in wet and dry sites), treatments (all eight levels) and years (five project years). Secondly, the data was analysed relative to the heather control. This was conducted separately for the soil pore water and then by surface runoff by both wet and then dry site.

Analysis of variance was conducted similar to that included in Chapter 3 and 4. The quality checked untransformed data were analysed first, after which logged and square root values of the data were investigated. The Anderson-Darling test was used to select which transformation (if any) was most appropriate. In addition to using the factors mentioned, covariates were also used to produce best fit data models. Covariates used in the ANCOVA included were: water pH and conductivity, WTD, WTH, surface soil temperature, rainfall, heather height, heather plot cover, peat depth, bulk density, DOC and E_4/E_6. Pearson's correlation coefficient was used to find the direct relationship between the predictor and significant covariates; some were investigated further using regression plots. As in the previous chapters' statistical analysis methods (Chapter 3 and 4), models in which all inputs were significant (all significant differences are assessed at the 95% probability of not being zero) were accepted on the condition the residuals were 'normal'. Normal residuals were also prioritised over having the highest adjusted R^2 (referred to as R^2). The magnitude of the differences between the factors and covariates used in the ANOVA were calculated using ω^2. The differences between the levels, of factors found to be significant, were compared using the post hoc Tukey test. Results were expressed as least square means (mean standard error is also included), as these provide estimates of the mean for factor levels, having taken account of the other factors (interactions and covariates that were included in the analysis). Tables of results output summary are located at the end of the results section.

5.5 Results

5.5.1 Weather variables

During the study period, Goyt Valley surface temperature and rainfall volumes (Figure 5.2) followed a similar pattern as that observed on the Bleaklow Plateau (Chapter 3). Surface ground temperature fluctuated less than air temperature. In the Goyt, the largest differences, between peak summer and lowest winter temperatures, was observed in 2010 with a range of 5.5 °C. In comparison the largest range in air temperature on Bleaklow was 12.39 °C in 2009. Mean monthly ground temperatures did not fall below 0 °C. An ANOVA found a significant difference between annual ground temperatures ($p < 0.0001$; $\omega^2 = 2.97\%$). Post hoc analysis revealed that the highest annual mean temperature was in year 2008 (7.77 ± 0.02 °C), and the lowest was in 2010 (7.13 ± 0.02°C).

Analysis of rainfall volumes data (Figure 5.3) collected at the four Goyt rain gauges in the fifth project year, using an ANOVA (R^2 82.71%), found that month was an important factor ($p < 0.0001$; $\omega^2 = 77.92$ %) as was site locality ($p = 0.057$; $\omega^2 = 4.47$ %). Post hoc analysis did not reveal differences in monthly rainfall volumes between the dry and wet site locality. Analysis of the SCaMP mean monthly rainfall (Figure 5.2,Table 5.1) significantly varied between months ($p < 0.0001$; $\omega^2 = 25.86\%$). Year was a significant factor of less importance ($p < 0.0001$; $\omega^2 = 7.70\%$). Post hoc analysis revealed months July and October to November had highest rainfall (Between 122.77 ± 3.39 and 118.87 ± 3.39 mm). Month February and March had the lowest rainfall (34.18 ± 3.39 and 30.17 ± 4.80 mm).

Year	Rainfall Sum (mm)	Temperature mean (C°)
2008	1150.1	7.78
2009	1219.42	7.53
2010	988.756	7.14
2011	726.9	7.35
2012	1047.7	7.31

Table 5.1: Total annual rainfall (mm) and mean surface ground water temperature (°C) data obtained by SCaMP in the Goyt Valley (2008-2012).

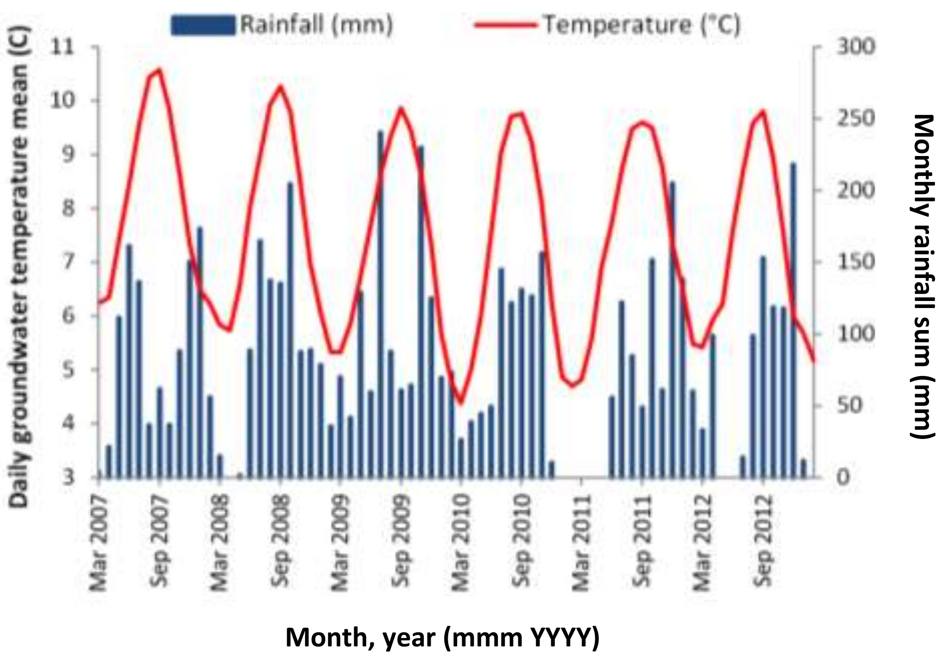


Figure 5.2: Monthly total rainfall (mm) and mean ground water temperature (°C) (March 2007- Jan 2013). The data is a compilation of Goyt SCaMP data.

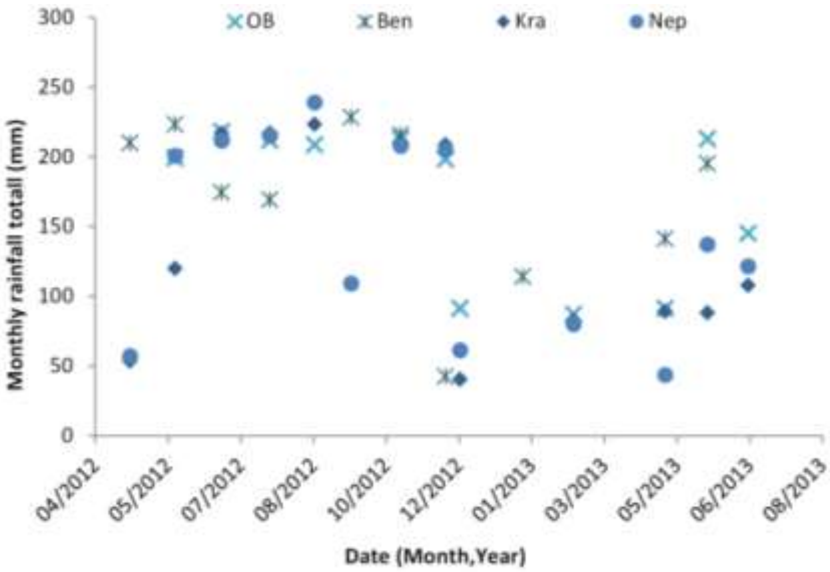


Figure 5.3: Goyt study site monthly rainfall sum (mm) (April 2012 - June 2013).

5.5.2 Site characterisation

5.5.2.1 Peat depth

Peat depth measurements varied between 38.8 and 382.1 cm. Analysis of peat depth across the Goyt sites using an ANOVA (R^2 86.30%) found that variation between sample localities was more important ($p < 0.0001$; ω^2 = 46.86 %) than site nested within locality ($p < 0.0001$; ω^2 = 39.19 %). Post hoc analysis revealed the wet locality (233.13 ± 8.83 cm) had significantly higher peat depth than the dry locality (101.4 ± 7.25 cm). Post hoc analysis revealed (Figure 5.4) that the controls at both the wet and dry locality were the sites with the lowest peat depth. The site with the highest peat depth was the wet old burn (Nep; 329.0 ± 15.30 cm); which did not differ to the dry site old burn (BN; 24.30 ± 21.64 cm). Site Nep had significantly higher peat depth than the wet new burn site (Pos; 234.07 ± 15.30 mm).

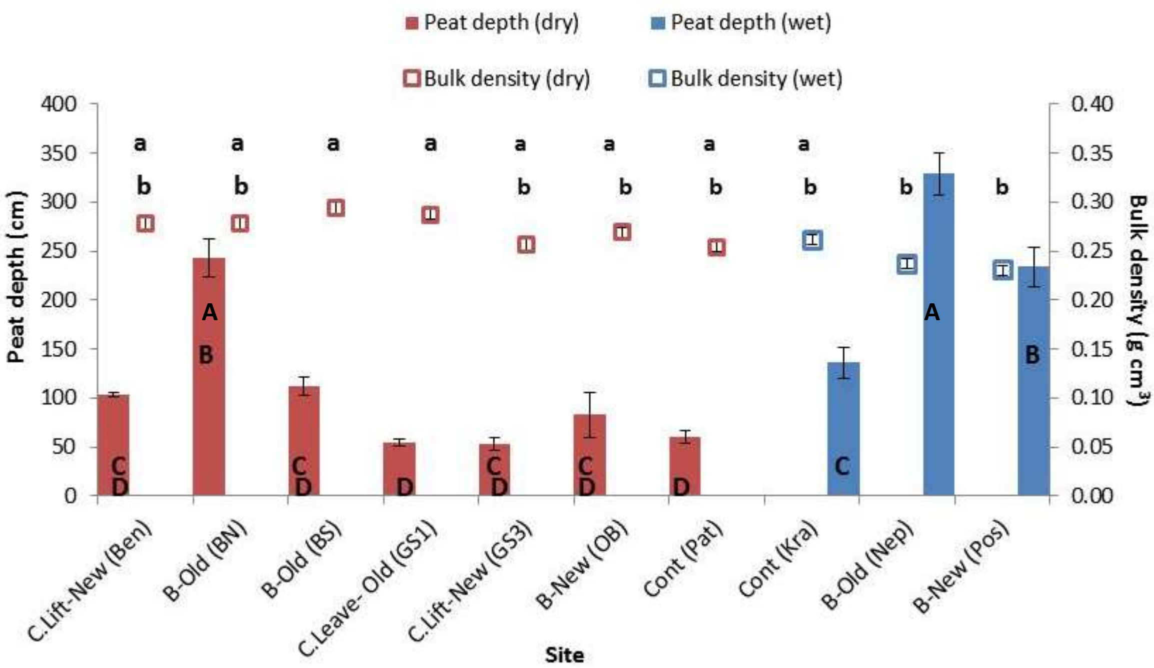


Figure 5.4: Goyt sites peat depth (bars: red, dry locality; blue wet locality) and bulk density (hollow blocks) raw data. The lettering represents post hoc analysis results. Lower case letter are the ANOVA post hoc results for bulk density (n= 32). The capital letters belong to the peat depth ANOVA post hoc test (n= 48). Site factor levels with shared letters indicate no significant difference; the opposite is true for different letters.

The wet heather control site (Kra; 136.37 ± 15.30 cm) had significantly shallower peat depth than the wet burn sites (Nep and Pos. The wet heather control also had greater peat depth than the dry locality heather control (Pat; 60.48 ± 15.30 cm). The sites with the significantly shallowest peat depth were the dry control and the old cut and leave (GS1; 54.65 ± 15.3 cm).

5.5.2.2 Slope

Analysis of site slope position using an ANOVA found that locality ($p < 0.0001$; R^2 75.98%) was a significant factor. Post hoc analysis revealed (Figure 5.5) that slope at the dry locality was significantly steeper (6.37 ± 0.594) than the wet locality (1.38 ± 0.55).

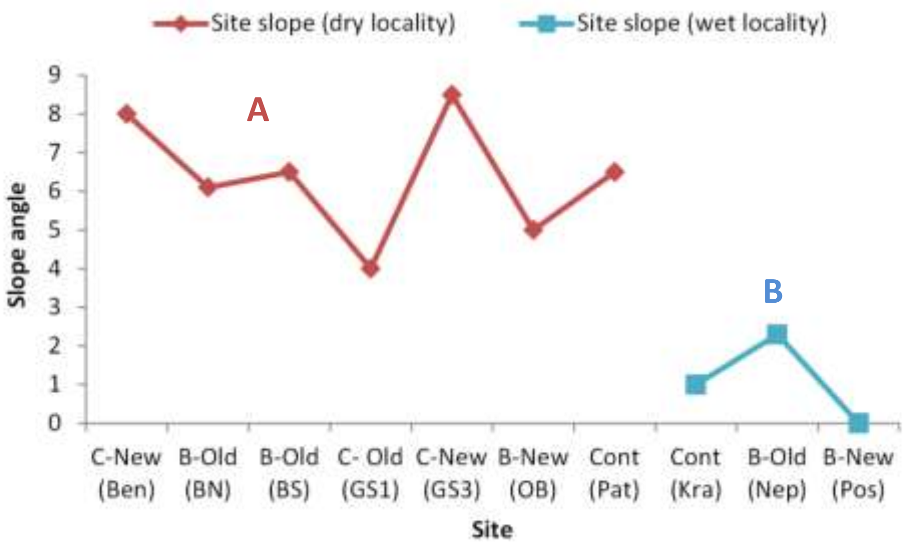


Figure 5.5: Goyt sites slope angle (red, dry locality, blue, wet locality).There are no error bars as no replicate measures were taken. The letters represent the results of an ANOVA tukey post hoc test; they indicate the significant difference between the locality levels (wet and dry).

5.5.2.3 Surface soil bulk density

Bulk density measurements ranged between 0.200 and 0.313 (g/cm^2). The wet sites had higher bulk density than the dry sites (Figure 5.6). An ANOVA (R^2 48.24%), found locality was a slightly more important factor ($p < 0.0001$; $\omega^2 = 24.38$ %) followed by site nesting within locality ($p < 0.0001$; $\omega^2 = 23.93$%). Post hoc analysis revealed (Figure 5.7) the wet locality (0.244 ± 0.01 mm) had significantly lower bulk density than the dry locality (0.270 ± 0.00 g/cm^2). The wet old burn (Nep: 0.237 ± 0.01 g/cm^2) and new burn had lower bulk density (Pos: 0.230 ± 0.01 g/cm^2) did not significantly differ to the wet locality controls site (Kra: 0.264 ± 0.01 g/cm^2). The burn treatment sites however did have lower bulk density than the dry old burn (BS: 0.294 ± 0.01 g/cm^2) and the old cut site (GS1: 0.288 ± 0.01 g/cm^2) in the dry locality (Figure 5.4)

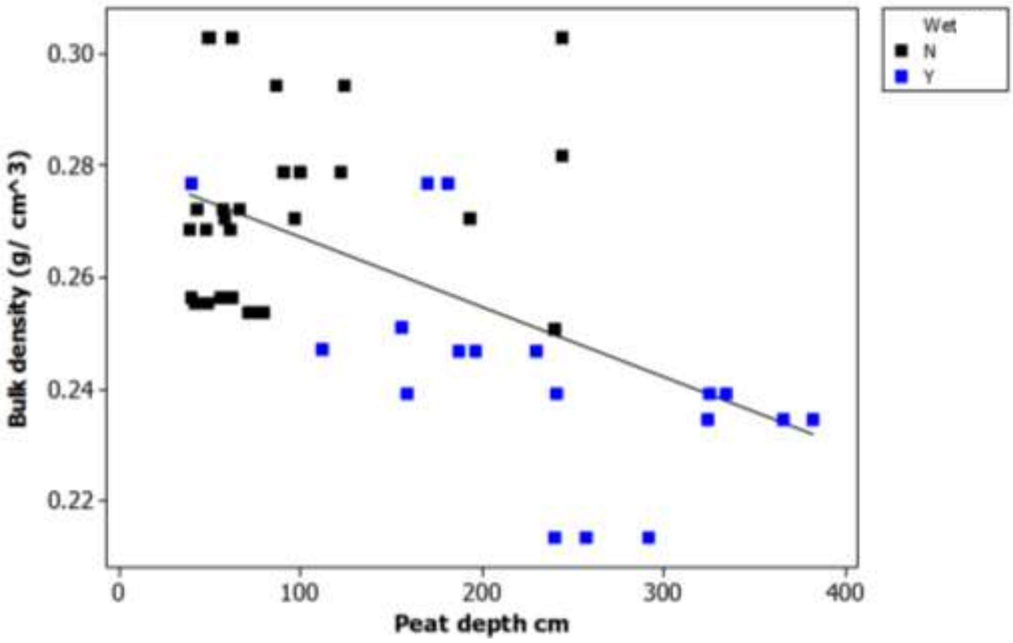


Figure 5.6: Regression plot of peat depth against bulk density, by site locality (wet vs. dry).

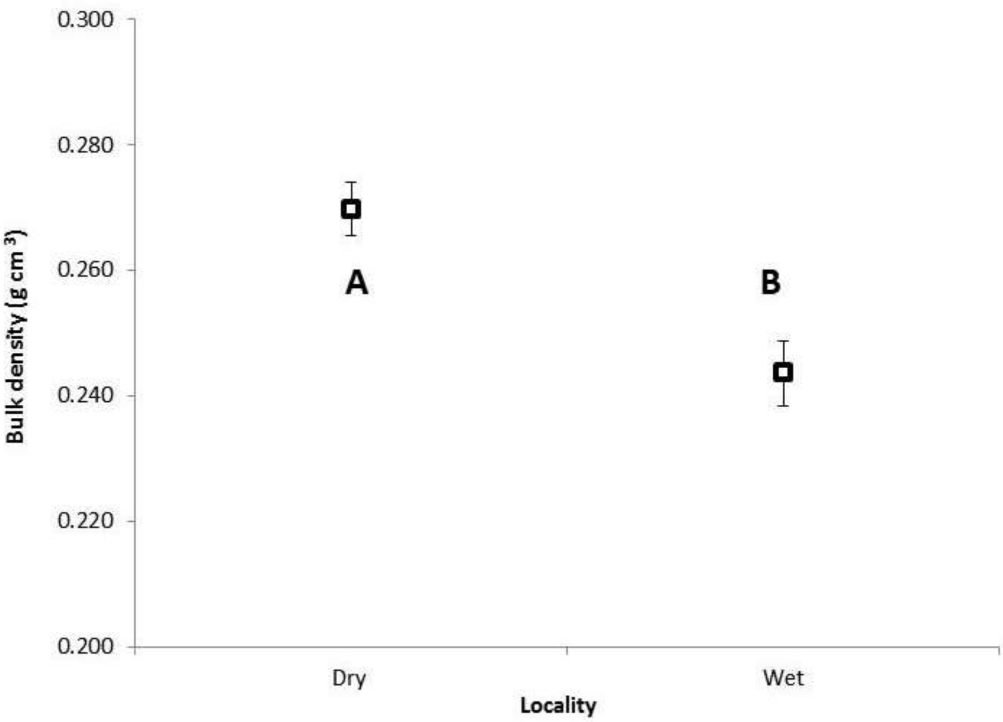


Figure 5.7: Main effects plot for bulk density (black hollow squares) by locality. Post hoc analysis results are displayed as letters. Different letters denote significant differences between localities.

5.5.2.4 Vegetation cover

Analysis of plot vegetation percentage cover using ANOVA (R^2 40.51%) found that vegetation functional cover type ($p < 0.0001$; $\omega^2 = 28.13\%$) was a significant factor explaining variation in plot cover. There was also a significant interaction between cover function groups with locality. ($p < 0.0001$; $\omega^2 = 7.48\%$) and between interaction between cover function groups and treatment ($p < 0.0001$; $\omega^2 = 5.27\%$). It was revealed that the functional group accounting for the greatest plot cover across the Goyt sites were (Figure 5.8) sedge-cotton grass (BS: 94.8 ± 0.73%) and shrub-bilberry (7.13 ± 0.73%).

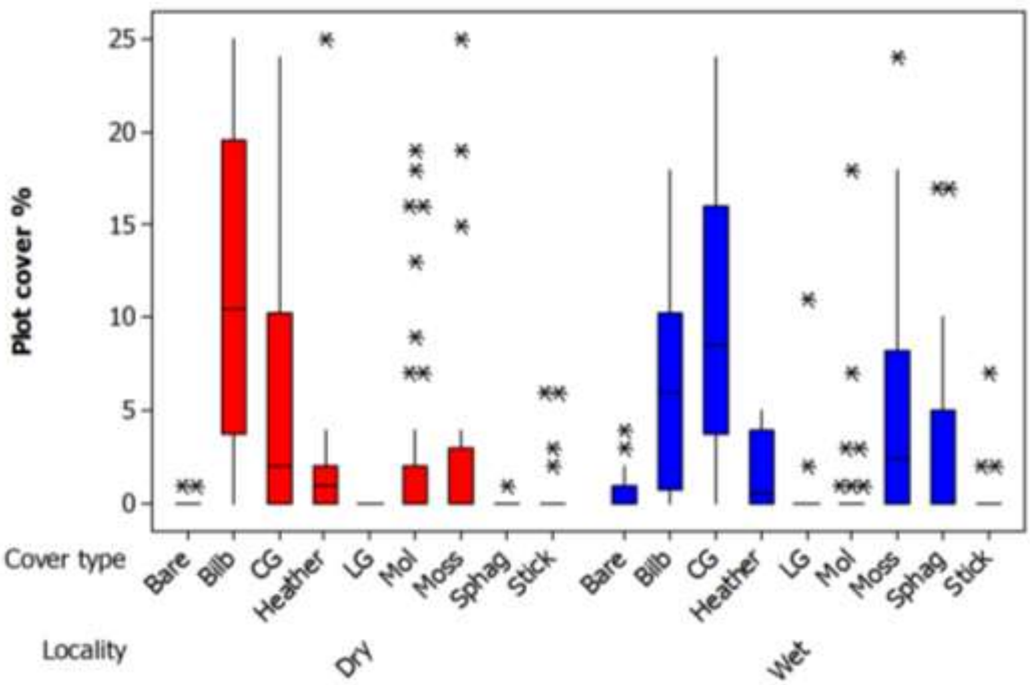


Figure 5.8: Box plot of Goyt valley vegetation plot cover (%) by location. The plot includes nine abbreviated vegetation functional groups (bare - bare soil; Bilb - bilberry, CG - cotton grass, Heather, LG - law grass and other, Mol – Molinia, Moss – non-Sphagnum spp. moss , Sphag - Sphagnum spp., and Stick - dead woody littler). Red boxes represent dry locality, and blue boxes represent wet locality.

The dominance of sedge-cotton grass was investigated further as a response factor using ANOVA. It was found that variation was more important between treatment nested within locality ($p < 0.0001$; $\omega^2 = 31.33\%$) than between localities ($p = 0.042$; $\omega^2 = 5.14\%$). It was revealed that the new treatment sites had higher sedge-cotton grass cover, specifically the wet new burn and the dry new cut. The addition of covariates revealed that treatment was the most important factor ($p < 0.0001$; $\omega^2 = 5.27\%$) and the differences between locality were insignificant. However, variation in plot peat depth was a significant covariate as was water table depth. Bulk density was significant although of relatively lower importance ($p < 0.0001$; $\omega^2 = 5.27\%$). Post hoc analysis revealed (Figure 5.9) that the wet control (26.26 ± 1.41%) had higher sedge – *Eriophorum spp.* cover than both the dry control (10.86 ± 2.22%) and dry new burn. (6.13 ± 2.27%). The treatment sites with highest sedge – *Eriophorum spp.* cover were new cut (63.11 ± 1.88%) followed by the old cut (39.43 ± 1.65%) and the wet new burn (45.85 ± 1.88%). The dry old burn (35.11 ± 1.71%) had higher sedge – *Eriophorum spp.* cover than the dry new burn and both the wet and dry control sites.

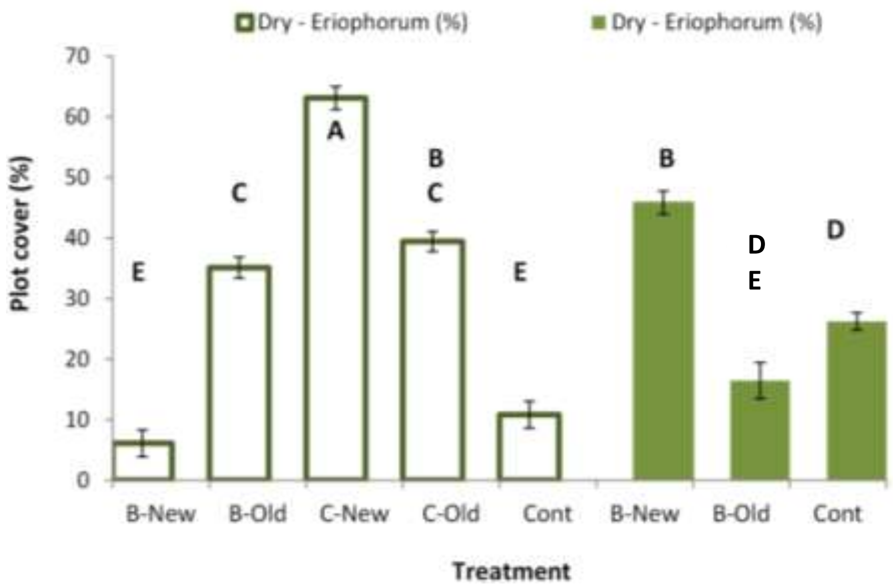


Figure 5.9: Main effect plot. Percentage plot cover cotton grass sedge – Eriophorum spp. analysis by treatment in wet (green bars) vs. dry (hollow bars) locality. Post hoc analysis results are represented by letters. Differing letters denote significant differences between treatment factors levels.

5.5.2.4.1 *Heather plot height and cover*

Heather plot height was analysed using an ANOVA. It was found that site nested within local was a significant factor ($p < 0.0001$; $\omega^2 = 60.81\%$), however locality was not significant ($p = 0.214$; $\omega^2 = 0.11\%$). Post hoc analysis revealed (Figure 5.10) that heather height at the wet (Kra: 279.25 ± 34.46 mm) and dry heather controls (Pat: 299.75 ± 6.81 mm) did not differ. The wet new burn (Pos: 67.75 ± 52.66 mm) had the lowest heather height of the sites in the wet locality. While on the dry locality in addition to the new burn (OB: 133.03 ± 63.99 mm) the old cut (GS1: 155.54 ± 16.75 mm) also had lower heather height than the heather controls.

Analysis of heather plot cover using ANOVA, found that variation of heather cover between sites nested within locality ($p < 0.0001$; $\omega^2 = 36.98\%$) was more important than between locality ($p = 0.003$; $\omega^2 = 10.29\%$). Post hoc analysis revealed (Figure 5.10) the heather cover at the wet new burn (Pos: 6.67 ± 9.02%) was significantly lower than the dry old burn (BS: 61.33 ± 12.75%) and the dry new burn (OB: 83.33 ± 9.02%). The remaining site did not significantly differ to each other. Both the old cut (GS1: 12.67 ± 9.02%) and the new cut (GS3: 26.667 ± 12.75%) had lower heather cover than the dry new burn.

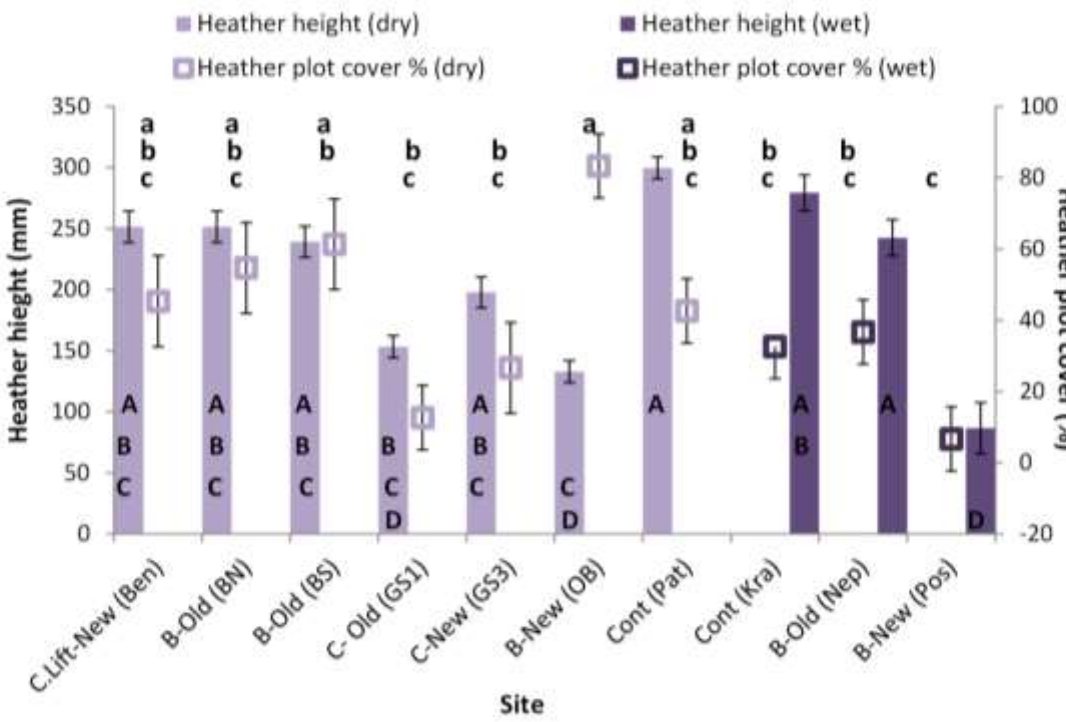


Figure 5.10: Goyt sites heather ANOVA main effects plot of: heather percentage plot cover (n= 48) (bars) and height (n= 48) (hollow blocks) by site and locality (light purple, dry locality; dark purple, wet locality). The lettering represents post hoc analysis results. The letters signify post hoc test results (capital letters, percentage plot cover; lower case letter, heather height). shared letters indicate no significant difference between factor levels.

5.5.3 Water table depth

5.5.3.1 Water table depth – managed cut vs. burn (dry locality)

The dry locality cut vs. burn, WTD (Table 5.2) (square root transformed) data (Figure 5.11) were analysed using ANOVA. Variation between the factor site was the most important ($p < 0.0001$; $\omega^2 = 38.16\%$), variation between plots nested within sites was also important ($p < 0.0001$; $\omega^2 = 25.61\%$). Month nested within project year ($p < 0.0001$; $\omega^2 = 4.69\%$) was more important than project years ($p < 0.0001$; $\omega^2 = 0.63\%$). An interaction between factors year and site was also significant but relatively unimportant ($p < 0.0001$; $\omega^2 = 0.30\%$). Residuals were normal. The addition of covariates did not improve the model R^2, hence the ANOVA (R^2 66.03%) was accepted. Water table depth was negatively correlated with peat depth (Pearson's correlation = - 0.084; P = 0.001). A deeper WTD is observed in first year of study at the dry locality new and old burn sites (Figure 5.11). Post hoc analysis revealed (Figure 5.12.a) the heather control site had the greatest WTD (-53.2 ± 0.06 mm). The old burn (-36.4 ± 0.06 mm), new burn (-37.4 ± 0.06 mm), and old cut (-38.6 ± 0.06 mm) did not significantly differ from each other and had shallower WTD than the heather control. The new cut site had the shallowest WTD (-18.7 ± 0.06 mm) visible throughout the study monitoring period (Figure 5.11). The first project year (Figure 5.12.b) (-37.54 ± 0.06 mm) had significantly deeper WTD than project years two (-33.84 ± 0.06 mm) and five (-34.14 ± 0.05 mm).

Variable	Water sample	n	No. removed	Mean	SE Mean
pH	Soil pore	1774	2	4.7	0
	Runoff	884	1	5.8	0
Conductivity	Soil pore	1748	1	56.7	0.9
	Runoff	853	0	105.4	4.5
abs_{400}	Soil pore	1706	0	0.3	0
	Runoff	856	0	0.1	0
E_4/E_6	Soil pore	1691	1	16.3	0.2
	Runoff	819	0	6.2	0.2
DOC	Soil pore	1473	0	115.3	2.1
	Runoff	809	0	106.3	3.3
WTD	-	2312	0	23.1	0.4

Table 5.2: Chapter dataset details (5 project years, between May 2008 and Jun 2013).

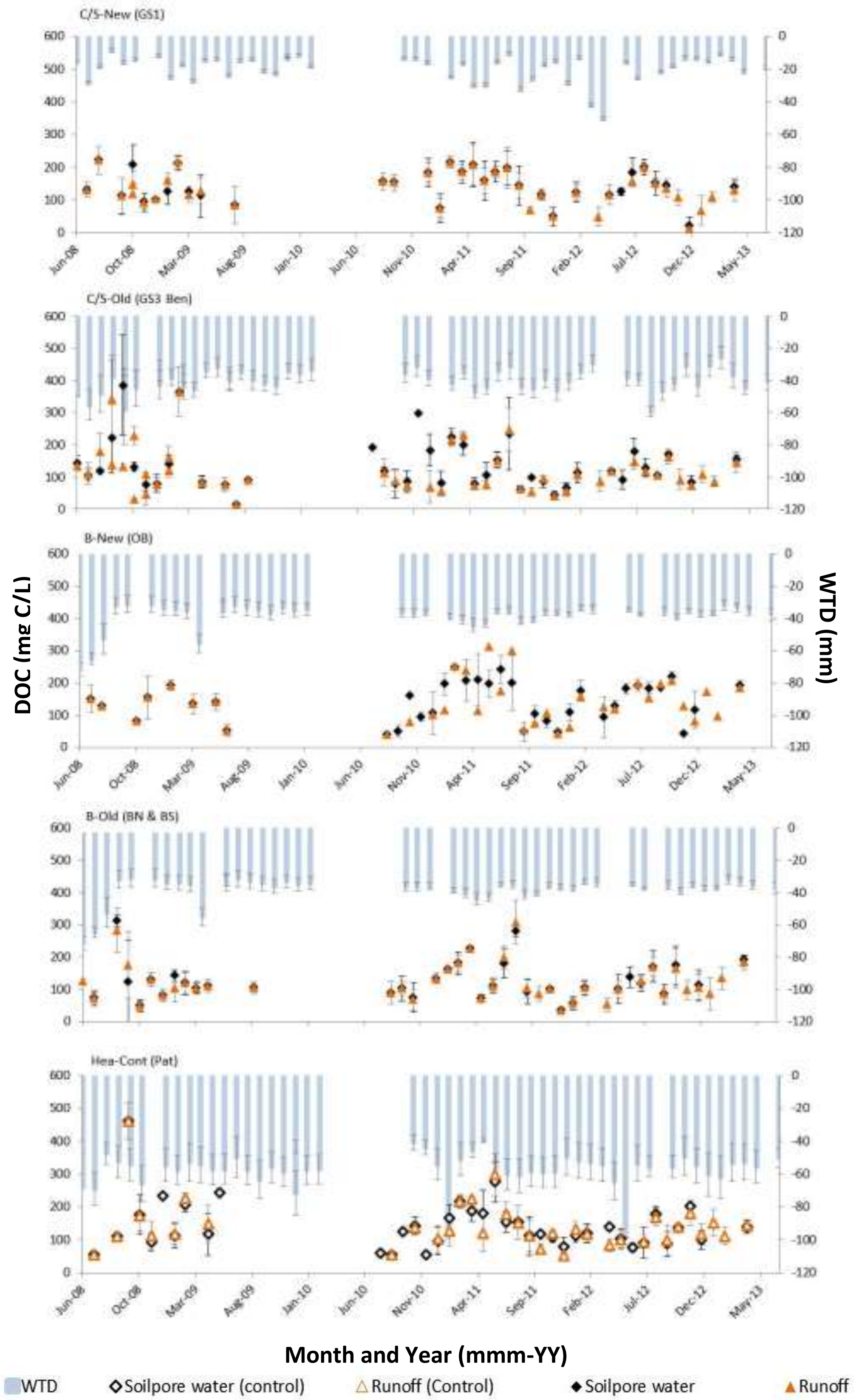
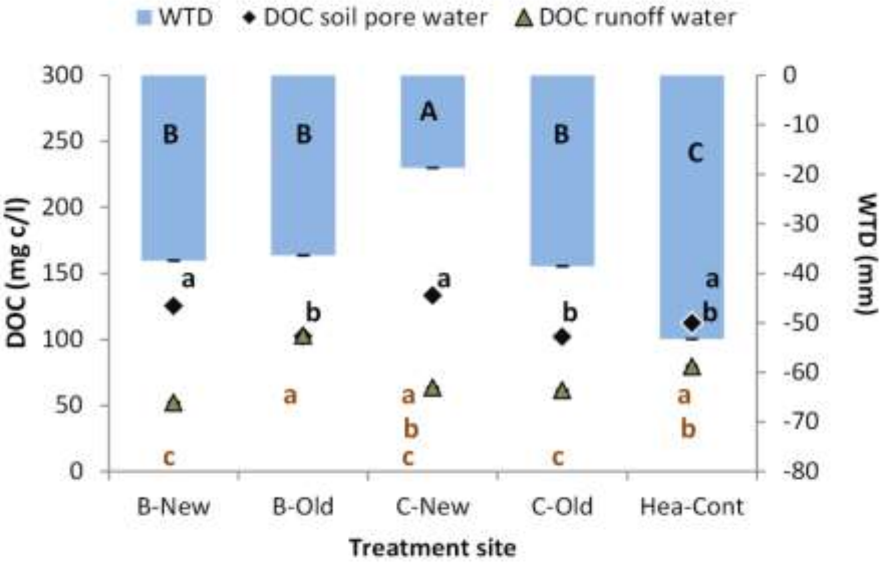


Figure 5.11: Goyt valley cut verses burn study (dry locality Big Moss) absolute data (Jun 2008 – May 2013): mean WTD (right y-axes, blue-drop-boxes) and DOC concentrations (left y-axes) in surface runoff (yellow triangles) and soil pore water (black diamond), by site, month and year. All data-points represent the monthly site mean with error bars defining the SE mean. Refer to methodology (section 2.2) for site details.

a)



b)

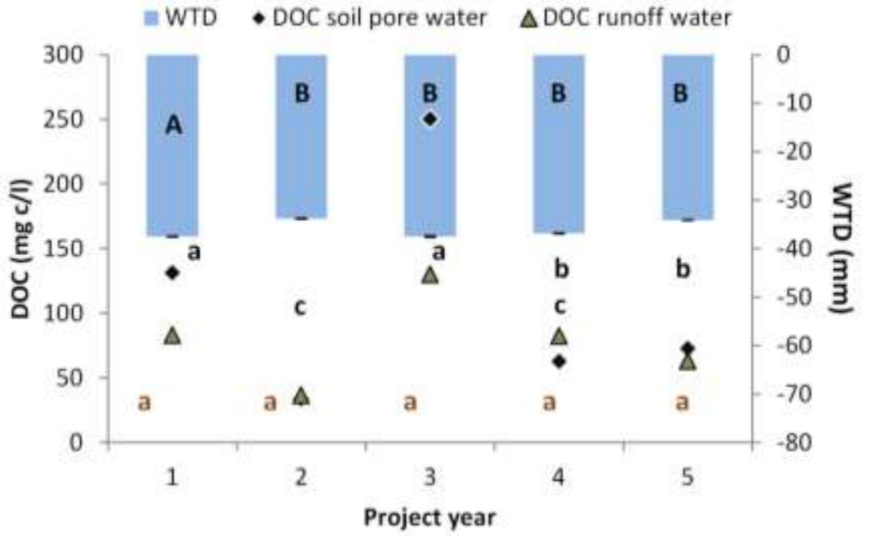


Figure 5.12: Managed heather *cut verses burn study, ANOVA main effect plots for WTD (right y-axis) (blue bars - capital letters) and DOC concentrations WTD (left y-axis) (soil pore water - black diamonds, black small letters; runoff water - yellow triangles, brown letters): a) by treatment type; b) by project year. The letters represent tukey post hoc results; shared letters indicate no significant differences between factor levels.*

5.5.3.2 Water table depth - managed burn vs. dry burn (wet locality)

Analysis of the wet burn vs. dry burn WTD data (Figure 5.11 and Figure 5.13) (square root) using an ANOVA found that locality was the most important factor ($p < 0.0001$; $\omega^2 = 35.77\%$), followed by treatment site within locality ($p < 0.0001$; $\omega^2 = 15.52\%$). Variation between month nested within project year ($p < 0.0001$; $\omega^2 = 4.45\%$) was more important than between project years, which was not a significant factor ($p = 0.666$; $\omega^2 < 0.0001\%$). There was a small interaction between locality and project year ($p = 0.018$; $\omega^2 = 0.20\%$). Residuals were normal. Post hoc analysis revealed that the wet locality had shallower WTD than the dry locality. Furthermore, the dry site heather control had significantly deeper WTD (-53.88 ± 0.07 mm), followed by the dry site new burn (-37.27 ± 0.07 mm) and old burn (-36.39 ± 0.07 mm) which did not significantly differ to each other. The wet site heather control (-29.67 ± 0.07 mm) had shallower WTD than the dry control, which had the deepest WTD at the wet locality. The wet site new burn (-15.05 ± 0.07 mm) and old burn (-14.95 ± 0.07 mm) had the shallowest WTD of all treatment sites. No difference in WTD was found between project years.

The addition of covariates improved the model R^2 from 55.89% to 60.56% (Figure 5.14.a). The covariates had greater importance in the ANCOVA than the factors originally included in the ANOVA. The importance of locality as a factor was reduced ($p < 0.0001$; $\omega^2 = 3.26\%$) and exceeded by variation between treatment site nested within locality ($p < 0.0001$; $\omega^2 = 4.86\%$). Variation between month nested within project year ($p < 0.0001$; $\omega^2 = 3.82\%$) was more important than between project years, which was not a significant factor ($p = 0.217$; $\omega^2 = 0.030\%$). There was a small interaction between locality and project year ($p = 0.045$; $\omega^2 = 0.13\%$). The ratio of peat depth to surface peat soil bulk density was the most important covariate and explanatory variable ($p < 0.0001$; $\omega^2 = 25.21\%$), followed by variation between altitude (ASL m) ($p < 0.0001$; $\omega^2 = 19.07\%$). Plot heather height was of similar importance to locality ($p < 0.0001$; $\omega^2 = 19.07\%$), and daily rainfall was significant although of low importance

(p = 0.217; ω^2 = 0.94%). A positive correlation was found between WTD (square root transformed) and the ratio of peat depth to bulk density (Pearson's correlation 0.988; $p <$ 0.0001). Deeper water tables were also positively correlated to increase in slope (Pearson's correlation 0.578; $p < 0.0001$). A regression (Figure 5.15) found a monthly WTD was correlated to both peat depth ($p < 0.0001$) and top 10 cm of peat bulk density ($p < 0.0001$) (Figure 5.15), together they were gave an R^2 22.60%. The wet locality heather control had a distinct hydrology to that of the wet locality burn sites (Figure 5.13)Post hoc analysis of the ANCOVA revealed the wet locality had significantly shallower water table (-35.69 ± 0.07 mm) than the dry locality (-24.07 ± 0.06 mm). Furthermore, the dry site new burn had the deepest WTD (-42.93 ± 0.11 mm) (Figure 5.14.a) within the wet locality, the heather control had deeper WTD than the new burn and old burn. The wet heather control (-35.58 ± 0.06 mm) was also insignificantly different to the dry heather control (-31.83 ± 0.06 mm) and dry old burn (-32.80 ± 0.06 mm). The addition of covariates did not reveal a significant difference between project years WTD.

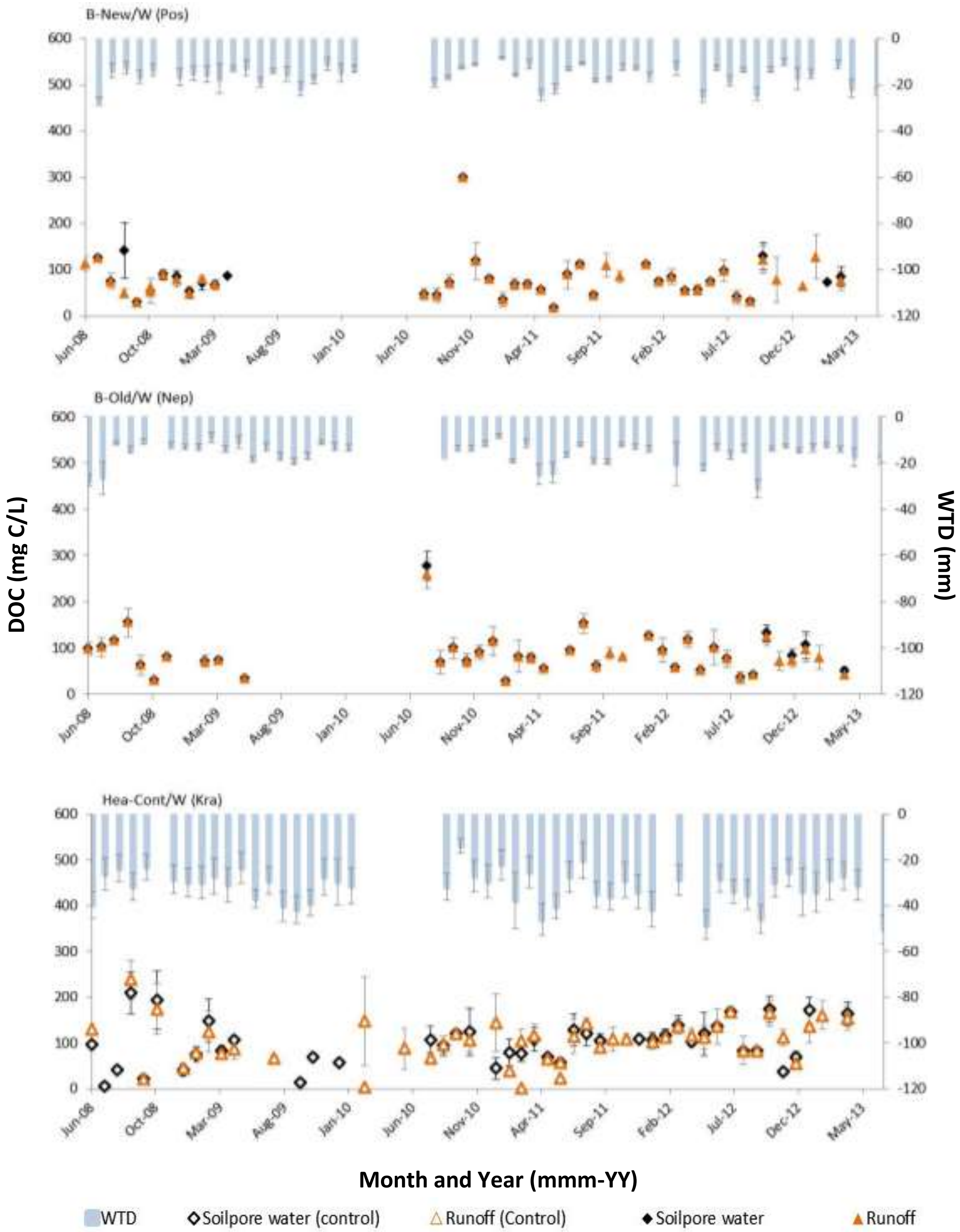
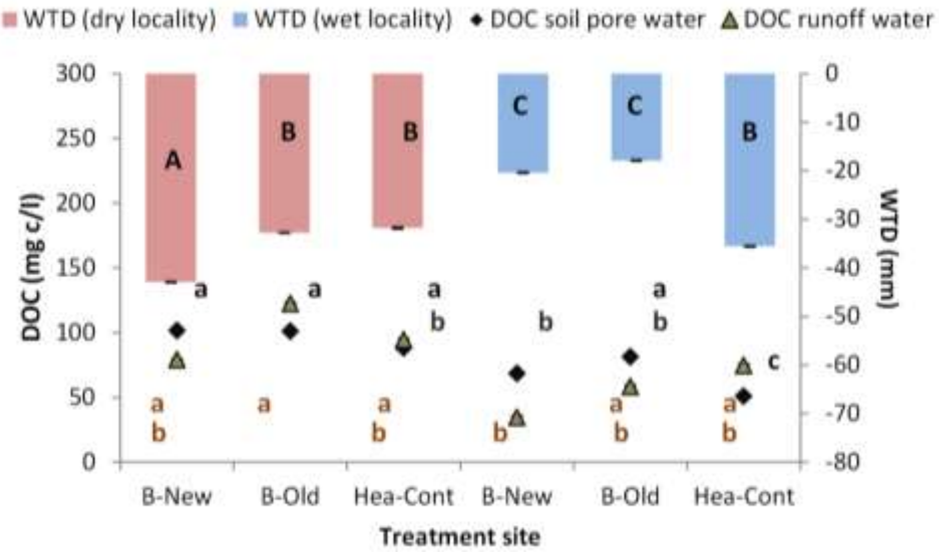


Figure 5.13: Goyt valley burn study in wet locality (Ravenslow) absolute data (Jun 2008 – May 2013): mean WTD (right y-axes) and DOC (left y-axes) by site, month and year. All data-points represent the monthly site mean with error bars defining the SE mean. Blue-drop-boxes denote WTD. Refer to methodology (section 2.2) for site details.

a)



b)

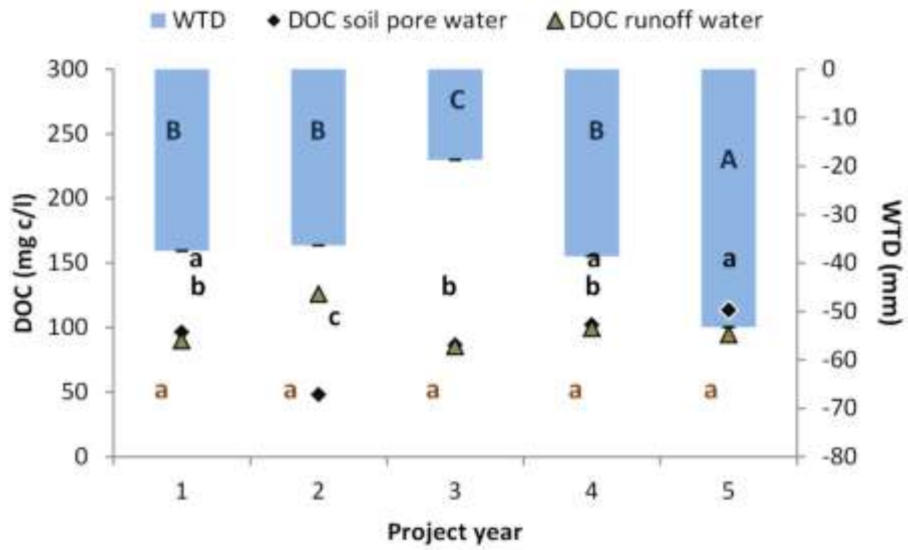


Figure 5.14: Managed Heather burns in wet verses dry study ANCOVA main effect plots for water table (blue bars – black capital letters) and DOC (in soil pore water - black diamonds – black lower case letters; surface water runoff - yellow triangles, brown lower case letters): a) by treatment site; b) by project year. The lettering represents post hoc analysis results and differences between site factor levels; shared letters indicate no significant difference.

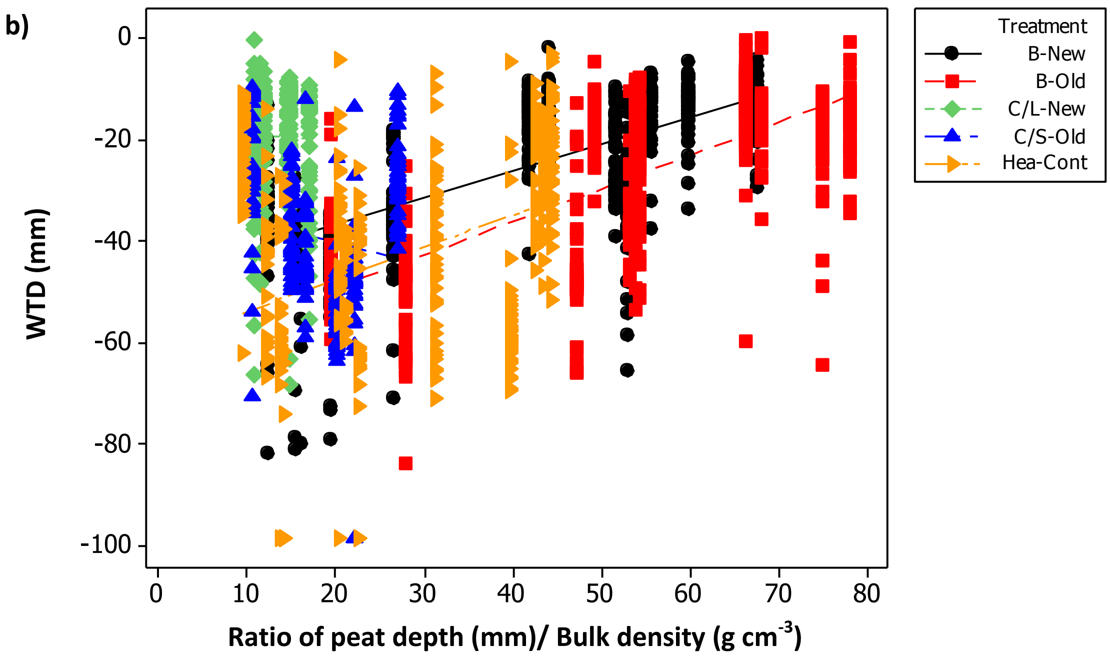


Figure 5.15: Goyt Valley (wet and dry locality) regression plots describing relationship between WTD (pooled data) and peat soil depth and bulk density (top 10 cm²) ratio, to WTD (data over 5 project years). The sites are displayed by treatment type: new burn sites (black circle), old burn (red square), new cut and leave (green diamond), old cut the scatter (blue triangle) and Heather control (yellow triangle).

5.5.4 Dissolved organic carbon

Analysis of pooled Goyt Valley DOC concentrations data revealed that runoff water DOC had a lower median value than soil pore water. Additionally runoff water DOC concentrations at the wet locality (75.44 mg C/L) were lower than the dry locality (82.98 mg C/L). Soil pore water median DOC concentrations were higher at the dry locality (113.29 mg C/L) than the wet locality (80.39 mg C/L).

Analysis of DOC concentrations (normalised square root) was conducted for all Goyt valley samples (both runoff and soil pore water) across the five project years between June 2008 and June 2013, using an ANOVA. It was found that monthly variation nested within project year was the most important explanatory factor ($p < 0.0001$, $\omega^2 = 11.05\%$), however variation between project years was relatively unimportant ($p = 0.017$, $\omega^2 = 0.24\%$). Variation between locality (wet or dry) was of similar importance ($p < 0.0001$, $\omega^2 = 3.00\%$) to variation between sites within locality ($p < 0.0001$, $\omega^2 = 3.25\%$). Sample type (soil pore water or runoff) was a significant factor ($p < 0.0001$, $\omega^2 = 0.54\%$) and there was a significant (albeit small) interaction between sample type and locality ($p < 0.0001$, $\omega^2 = 0.38\%$). There was also a small interaction between locality and project year ($p = 0.021$, $\omega^2 = 0.27\%$) and an interaction between the water sample type and the locality ($p = 0.001$, $\omega^2 = 0.45\%$). Post hoc analysis revealed DOC concentrations were higher in soil pore water sample (96.29 ± 0.03 mg C/L) than in runoff water (86.58 ± 0.03 mg C/L) (Figure 5.16). The sites with significantly highest DOC concentrations were the old burn (BN: 134.79 ± 0.11 mg C/L), dry locality control (Pat: 121.24 ± 0.08 mg C/L), new cut (GS3: 119.09 ± 0.12 mg C/L) and the old cut (GS1: 111.30 ± 0.08 mg C/L). The sites with the lowest DOC concentrations were the new cut (Ben: 82.03 ± 0.10 mg C/L) and the burn sites within the wet locality, the wet old burn (Nep: 76.14 ± 0.08 mg C/L) and dry new burn (Pos: 61.06 ± 0.08 mg C/L).

The addition of covariates improved the R^2 from 18.74% up to 21.21%. The ANCOVA found that variation between month nested within year remained the most important factor ($p < 0.0001$, $\omega^2 = 11.35\%$). Variation between sites within locality was important ($p < 0.0001$, $\omega^2 = 5.46\%$), however locality was not ($p = 0.726$, $\omega^2 = 0.00$ %). Sample type remained a significant factor ($p < 0.0001$, $\omega^2 = 0.90\%$) and there was a small interaction between sample type and locality ($p < 0.0001$, $\omega^2 = 0.69\%$) in addition to an interaction between locality and project year ($p < 0.0001$, $\omega^2 = 0.243\%$). The covariate peat top 10 cm^2 bulk density ($p = 0.020$, $\omega^2 = 2.19\%$) was more important than peat depth ($p < 0.0001$, $\omega^2 = 0.68\%$). Daily rainfall sum was a significant covariate, although of low importance in the ANCOVA ($p < 0.0001$, $\omega^2 < 0.000\%$). Site slope was also a significant covariate. However, its importance was dropped out of the model when the other significant covariates were added. Residuals were normal.

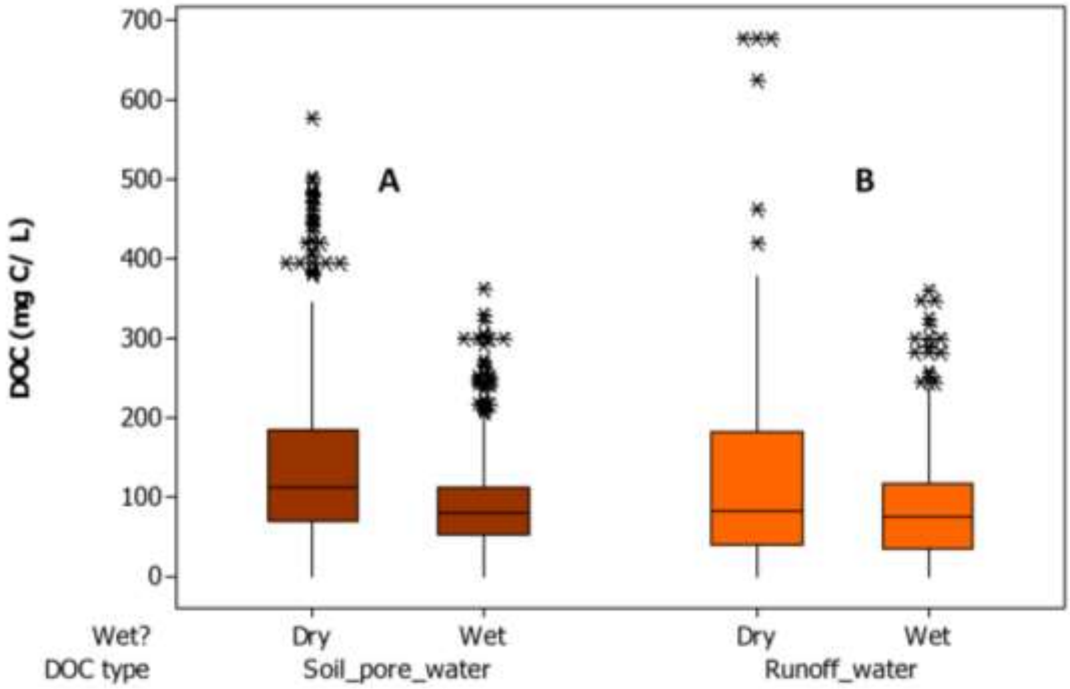


Figure 5.16: Box plot soil pore water (brown) and surface water runoff (orange) absolute data. The lettering represents post hoc analysis results, and the differences between the water samples (i.e. runoff and soil pore water) DOC concentrations.

A positive correlation was found between sample DOC concentration and bulk density (Pearson's correlation 0.153; $p < 0.0001$), and negative correlation between DOC and peat depth (Pearson's correlation -0.164; $p < 0.0001$) (Figure 5.17). Site slope also positively correlated to DOC (Pearson's correlation -0.158; $p < 0.0001$). Post hoc analysis revealed that runoff samples (87.85 ± 0.17 mg C/L) had significantly lower DOC concentrations than soil pore water samples (100.54 ± 0.19 mg C/L. No significant difference in DOC concentrations were identified between the wet and dry locality. The site with the highest mean (runoff and soil pore water) DOC concentration (Figure 5.18.a) was the dry old burn (BN: 159.79 ± 0.41 mg C/L), followed by the dry new cut (GS3: 103.35 ± 0.40 mg C/L), new burn (OB: 85.43 ± 0.30 mg C/L), dry locality control (106.17 ± 0.35), wet control (Kra: 89.91 ± 0.29mg C/L), and wet old burn (Nep: 112.40 ± 0.44 mg C/L). The sites with the lowest DOC concentrations were the new cut (Ben: 64.21 ± 0.34 mg C/L) and wet new burn (Pos: 85.27 ± 0.37 mg C/L). Post hoc analysis also revealed that project year three (67.55 ± 0.39 mg C/L) had significantly lower DOC concentrations than project year two (112.66 ± 0.63 mg C/L). The other years did not significantly differ (Figure 5.18.b).

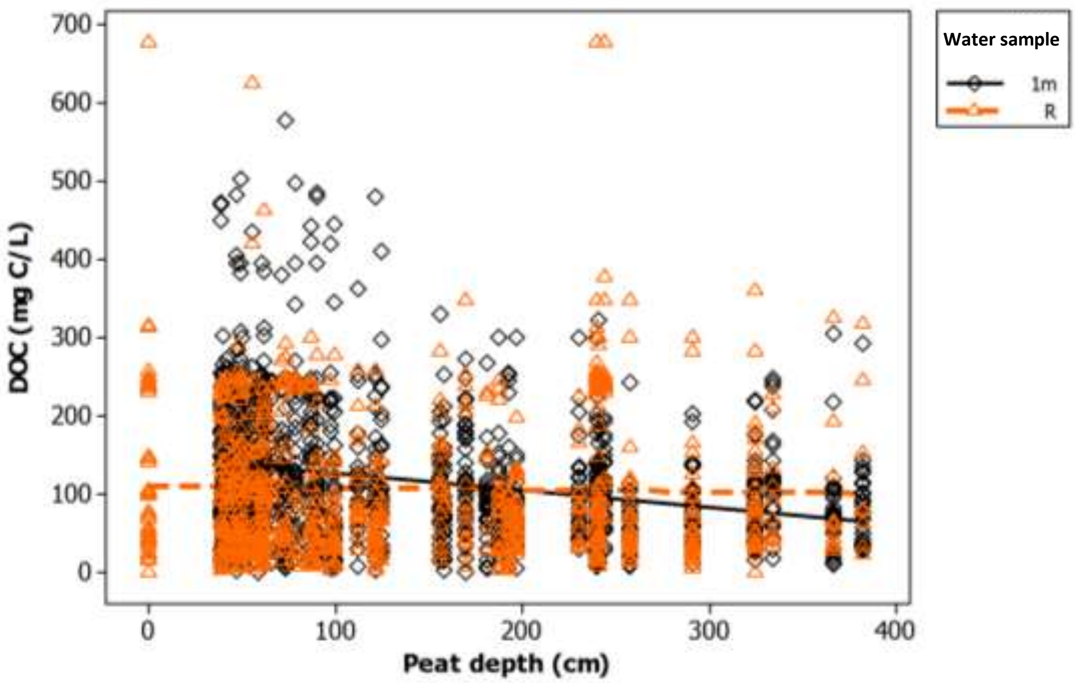


Figure 5.17: Goyt Valley runoff (R, brown triangles) and soil pore water (1m, black diamonds) sample DOC concentration plotted against site peat depth (May 2008 – April 2013).

a)

b)

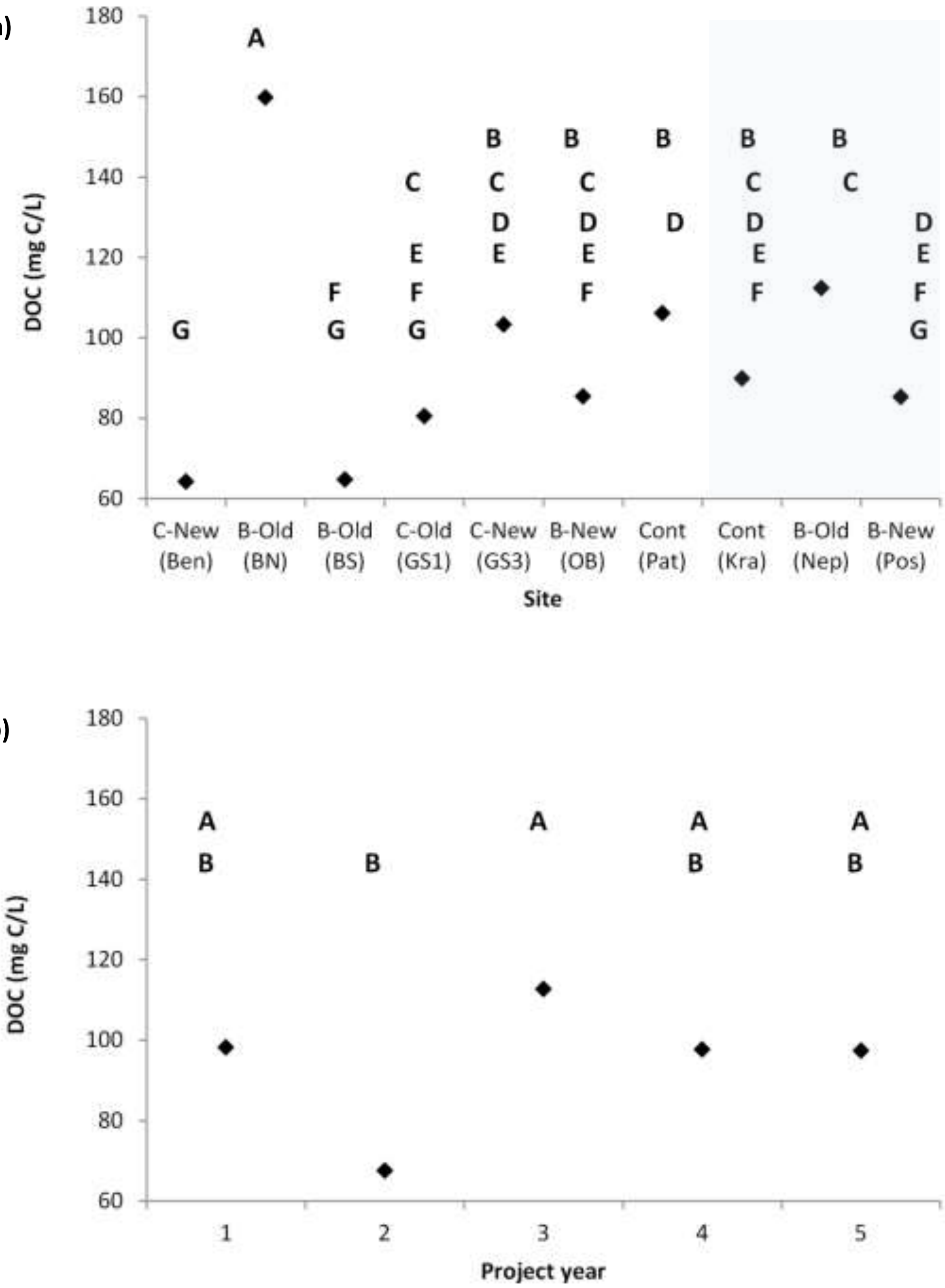


Figure 5.18: Main effects plot for water DOC concentration (pooled absolute runoff and soil pore) ANOVA: a) analysis by site, b) analysis by year. The lettering represents results from post hoc analysis. Shared letters indicate no significant difference between factor levels.

5.5.4.1 Runoff water DOC – managed burn in wet vs. dry locality

Analysis of square root transformed wet vs. dry burn treatment sites surface water runoff DOC concentrations was conducted using ANOVA. It was found that DOC variation between the factor plots nested within treatment site was more important ($p < 0.0001$; $\omega^2 = 13.82\%$) than treatments nested within locality ($p < 0.0001$; $\omega^2 = 8.15\%$). Locality was significant, but of low importance in the model ($p < 0.0001$; $\omega^2 = 1.37\%$). Variation between months nested within project year ($p < 0.0001$; $\omega^2 = 8.86\%$) was of similar importance to treatment within locality, however project year was not a significant factor ($p = 0.201$; $\omega^2 < 0.000$ %). Post hoc analysis revealed runoff DOC concentrations were higher at the dry localities (104.39 ± 0.07 mg C/L) than wet localities (83.56 ± 0.10 mg C/L). Furthermore, runoff water DOC concentrations at the heather control at the wet locality (109.06 ± 0.59 mg C/L) did not significantly differ to the dry site control (123.19 ± 0.59 mg C/L). The dry control site runoff DOC did not differ to the dry old burn (149.57 ± 0.20 mg C/L) or wet old burn (87.24 ± 0.25 mg C/L); both the dry new burn (79.35 ± 0.30 mg C/L) and wet new burn (66.21 ± 0.21 mg C/L) had lower runoff DOC concentrations than the their respective control sites. There was no difference in runoff DOC concentration between project years.

The addition of covariates improved the R^2 from 32.18 to 38.09%. The variation explained by month nested within project year remained important ($p < 0.0001$; $\omega^2 = 13.45\%$), however project year as a factor was insignificant ($p = 0.630$; $\omega^2 = 0.34\%$). Variation between plots nested within sites ($p < 0.0001$; $\omega^2 = 14.01\%$) remained the most important factor. Variation between sites within locality was also significant ($p < 0.0001$; $\omega^2 = 6.58\%$). Locality as a factor was insignificant ($p = 0.172$; $\omega^2 = 0.03\%$). Runoff sample pH was a significant covariate ($p = 0.008$; $\omega^2 = 3.60$ %). Residuals for the runoff DOC concentration ANCOVA were normal. Post hoc analysis revealed that when covariates were accounted for in the model, differences in surface runoff water DOC concentrations by site were reduced. However, the dry site old burn (136.28 ± 0.509 mg C/L) had higher concentrations than the dry new burn (72.52 ± 0.573 mg C/L) and the wet old burn (93.10 ± 0.671 mg C/L) (Figure 5.14.b). The remaining sites did not significantly differ. No significant difference was found between project years.

Runoff DOC concentrations correlated to pH (Pearson's correlation 0.117; $p < 0.0001$). A regression plot of DOC and pH did not reveal any useful information on the variation of pH in relation to treatment. However, a simple one way ANOVA found there was a significant difference between treatment types pH ($p < 0.0001$; $R^2 = 5.10\%$). Post hoc analysis identified (Figure 5.19) that the control sites had the highest pH (6.13 ± 0.05), followed by the old burn (5.80 ± 0.05) and the new burn with the lowest pH (6.13 ± 0.05).

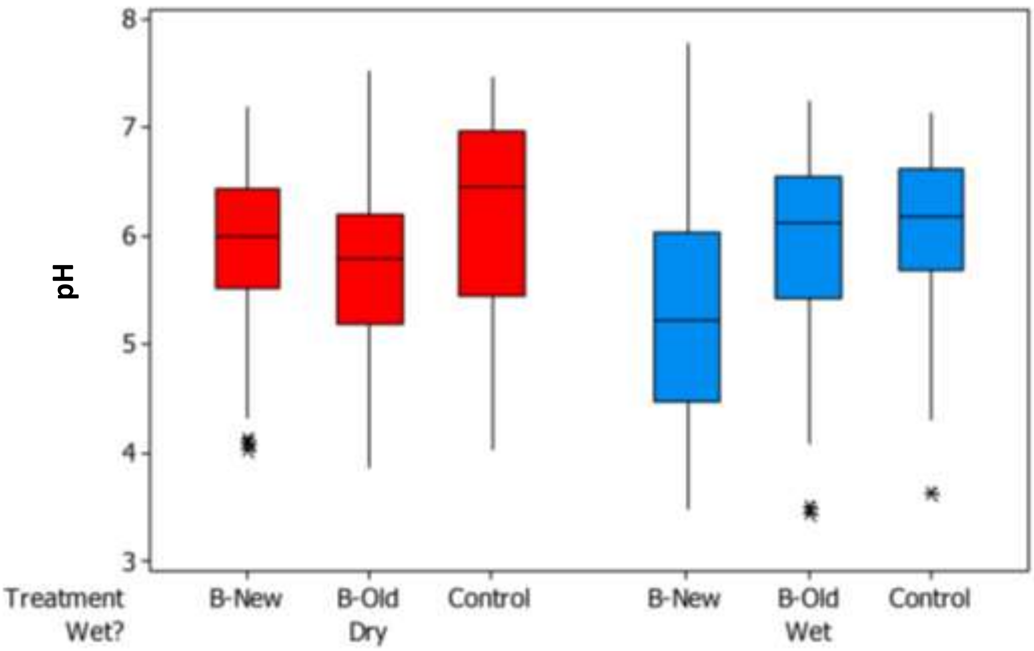


Figure 5.19: Box plot of pH by locality (red, dry locality; blue, wet locality).

5.5.4.2 Soil pore water DOC – managed burn in wet vs. dry locality

Analysis of the burn wet vs. dry soil pore water DOC (square root normalised) data was conducted using an ANOVA. The variation between month nested with project year was the most important factor ($p < 0.0001$; $\omega^2 = 11.59\%$), followed by variation between wet and dry locality ($p < 0.0001$; $\omega^2 = 7.09\%$). Variation between plots nested within site ($p < 0.0001$; $\omega^2 = 4.29\%$) was more important than variation between treatment sites nested within locality ($p < 0.0001$; $\omega^2 = 1.69\%$). An interaction between year and site was also significant but relatively unimportant ($p < 0.0001$; $\omega^2 = 0.30\%$). However, there was a small interaction between locality and project year ($p = 0.018$; $\omega^2 = 0.20\%$). Residuals were normal. Post hoc analysis revealed least squares mean soil pore water DOC concentrations were higher at the dry locality (115.58 ± 0.193 mg C/L) than the wet (72.79 ± 0.18 mg C/L). Furthermore, the soil pore water DOC concentrations were highest at the dry site new burn (124.81 ± 0.29 mg C/L) and did not differ to that of the dry site heather control (121.6 ± 0.28 mg C/L). The dry old burn (101.1 ± 0.27 mg C/L) had significantly lower concentrations than the dry new burn. The dry old burn did not significantly differ to the site at the wet locality with the highest DOC concentrations; the wet heather control (87.44 ± 0.25 mg C/L). The wet new burn (71.06 ± 0.27 mg C/L) had significantly lowest soil pore water DOC concentrations. Concentration were significantly highest in project year five (103.98 ± 0.20 mg C/L) and lowest in the second project year (68.49 ± 0.62 mg C/L).

The addition of covariates improved the R^2 from 24.95% to 29.37%. The ANCOVA found the addition of covariates increased the importance variation between month nested within project year increased ($p < 0.0001$; $\omega^2 = 13.85\%$), followed by variation between wet and dry locality ($p < 0.0001$; $\omega^2 = 7.09\%$). Variation between treatment sites within wet or dry locality ($p < 0.0001$; $\omega^2 = 1.58\%$) was more important than variation between locality ($p = 0.004$; $\omega^2 = 0.53\%$). Project year was a significant factor ($p < 0.0001$; $\omega^2 = 0.41$ %) of low importance similar to that of an interaction between locality and project year ($p = 0.005$ $\omega^2 = 0.76\%$). The most important covariate was peat depth ($p < 0.0001$; $\omega^2 = 5.09\%$), followed by the natural log of water table depth ($p = 0.002$; $\omega^2 = 4.36\%$). Vegetation mass ($p = 0.002$; $\omega^2 = 0.01\%$) was significant, but not important compared to plot heather cover ($p = 0.003$; $\omega^2 < 2.59\%$). The sum of sampling day rainfall was a significant factor, but of low importance ($p < 0.0001$; $\omega^2 < 0.24\%$). Post hoc analysis revealed the dry locality (103.00 ± 0.10 mg C/L) soil pore water concentration remained higher than the wet locality (73.63 ± 0.11 mg C/L). It was also revealed that when covariates accounted for (Figure 5.14.a), the wet heather control (58.78 ± 0.50 mg C/L) had lower soil pore water DOC concentrations than the wet old burn (98.98 ± 0.50 mg C/L). The wet control differed to the dry control (88.81 ± 0.43 mg C/L), but not the wet new burn (66.06 ± 0.42 mg C/L). The dry new burn (116.5 ± 0.19 mg C/L) had similar DOC to the dry old burn (104.7 ± 0.33 mg C/L). The sites with the highest DOC concentrations were therefore the new burns at the dry locality and the old burn at the wet locality.

5.5.4.3 Runoff water DOC – managed cut vs. burn (dry locality)

Analysis of square root normalised surface runoff water DOC data at the dry locality heather cut vs. burn study using an ANOVA showed the variation between plots nested within treatment site ($p < 0.0001$; $\omega^2 = 5.80\%$) was more important than variation between treatment sites ($p = 0.001$; $\omega^2 = 2.07\%$). Variation between month nested within project year was also an important factor ($p < 0.0001$; $\omega^2 = 35.59\%$), however project year was a less important factor ($p < 0.0001$; $\omega^2 = 2.07$ %). Post hoc analysis revealed the old burn (134.95 ± 0.42 mg C/L) had significantly greatest mean runoff water DOC concentration; it did not differ to the locality heather control (114.92 ± 0.54 mg C/L). Both the new cut (88.12 ± 0.20 mg C/L) and old cut (87.29 ± 0.17 mg C/L) did not differ to the heather control. The new burn (74.84 ± 0.266 mg C/L) however had lower DOC concentrations than both the heather control and the old burn. No difference in DOC was found between project years.

The addition of covariates slightly increased the R^2 from 41.03 to 42.85%. The ANCOVA found that the most important factor remained difference between plots nested within treatment site ($p < 0.0001$; $\omega^2 = 18.04\%$) and it was more important than variation between treatment sites ($p < 0.0001$; $\omega^2 = 7.55\%$). Variation between the factor month within year ($p < 0.0001$; $\omega^2 = 15.84\%$) was more important than project year, which was an insignificant factor ($p = 0.145$; $\omega^2 < 0.000$ %). The covariate mean monthly surface soil temperature was significant ($p = 0.046$; $\omega^2 = 1.72\%$). A positive correlation was found between DOC concentrations and mean soil temperatures (Pearson's correlation = 0.136; $p = 0.003$). Residuals of the ANCOVA were normal, and it was revealed through post hoc (Figure 5.14.a), analysis that the addition of covariates to the old burn (102.96 ± 0.46 mg C/L) had a significantly higher surface runoff DOC concentration than the other treatment sites. The heather control (79.35 ± 0.72 mg C/L) did not differ to the new cut (63.46 ± 0.47 mg C/L), old cut (61.61 ± 0.44 mg C/L) or new burn (52.11 ± 0.55 mg C/L). Additionally, no difference in DOC concentrations was found between project years.

5.5.4.4 Soil pore water DOC – managed cut vs. burn (dry locality)

Analysis of square root normalised soil pore water DOC data for cut vs. burn sites was conducted using ANOVA. The variation between month nested within project year was the most important factor ($p < 0.0001$; $\omega^2 = 26.75\%$), followed by variation between plots nested within treatment site ($p < 0.0001$; ω^2 5.09%) which was more important than the inter sites variation ($p < 0.0001$; $\omega^2 = 0.96\%$). Variation between project years ($p < 0.0001$; $\omega^2 = 1.73\%$) was significant and slightly more important in the model than variation between sites. Residuals were normal. Post hoc analysis revealed soil pore water DOC concentrations at the new burn site (127.08 ± 0.28 mg C/L) were greater than the aged treatment sites, both the old burn (100.18 ± 0.26 mg C/L) and the old cut (104.33 ± 0.27 mg C/L). The new burn also did not differ to the new cut (121.13 ± 0.27 mg C/L) (100.18 ± 0.26 mg C/L) or heather control (120.78 ± 0.27 mg C/L). The heather control had higher soil pore water DOC concentrations than the old burn. DOC concentrations were significantly highest in the first project year five (138.89 ± 0.26 mg C/L) than they were during project year two (89.72 ± 0.26 mg C/L) and four (104.73 ± 0.19 mg C/L).

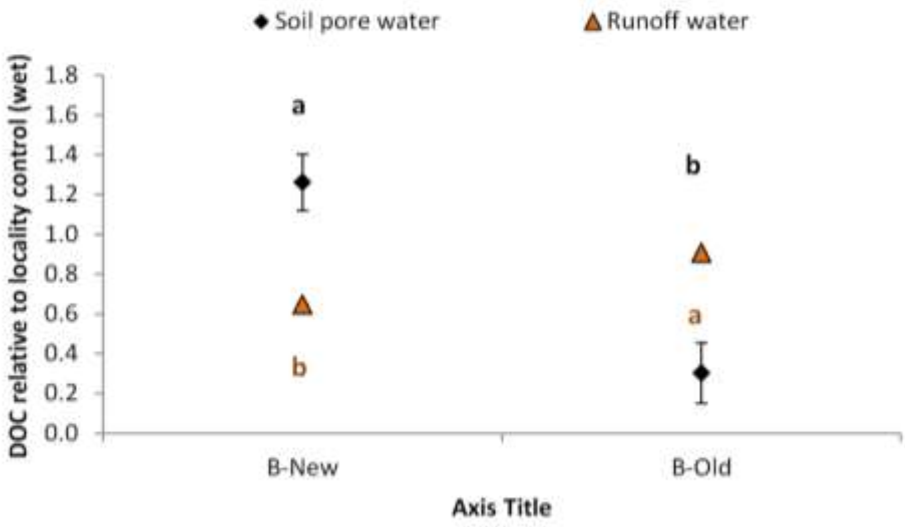
The addition of covariates improved the R^2 from 34.56% to 35.38%. The ANCOVA found that variation between month nested with project year was increased in importance ($p < 0.0001$; $\omega^2 = 24.03\%$); however project year remained a factor of low importance ($p < 0.0001$; $\omega^2 = 1.22$ %). Variation between treatment was significant in the model ($p = 0.001$; $\omega^2 = 2.81\%$), however less important than variation between plots nested within the treatment site ($p < 0.0001$; $\omega^2 = 3.94\%$). The covariate WTD ($p = 0.014$; $\omega^2 = 2.82$ %) was of similar importance in the model as site treatment was. Soil pore water DOC concentrations positively correlated to WTD (Pearson's correlation 0.170; $p = 0.003$). Post hoc analysis revealed that when covariates were accounted for in the model, the soil pore water DOC concentrations at the heather control (112.62 ± 0.27 mg C/L) did not differ to any of the treatment sites. However, the old burn (102.29 ± 0.27 mg C/L) and old cut (102.11 ± 0.27 mg C/L) had lower concentrations than the new burn (125.4 ± 0.30 mg C/L) and new cut (133.4 ± 0.14 mg C/L) treatment sites (Figure 5.12.a). Soil pore DOC concentration fluctuated between project years. Project year one (141.94 ± 0.31 mg C/L) had higher DOC concentrations than project year two (90.17 ± 0.86 mg C/L) and four (62.85 ± 0.72 mg C/L) (Figure 5.12.b)

5.5.4.5 Runoff water DOC – managed burn sites relative to wet locality control

Using an ANOVA it was found that, over five project years, the relative to the heather control runoff water DOC concentration square root of data at the wet sites locality was most influenced by the factor month nested within project year ($p < 0.0001$, $\omega^2 = 15.13\%$). Variation between project year was significant ($p < 0.0001$, $\omega^2 = 7.44\%$) as was variation between the two sites ($p < 0.0001$, $\omega^2 = 2.38\%$). Additionally there was a significant interaction between site and project year ($p = 0.002$, $\omega^2 = 5.82\%$). Residuals were normal. Post hoc analysis revealed that both sites had relatively lower DOC than the heather control. The old burn (0.906 ± 0.05) had more similar DOC to the heather control than the new burn (0.647 ± 0.04). No significant difference was revealed between project years one, three, four and five. It was not possible to find difference in DOC relative to the heather control in year two due to lack of data in every level of the factors month and site.

The addition of covariates in the ANCOVA increased the R^2 from 30.91 to 31.08 %. The most important factor remained month nested within project year ($p = 0.023$, $\omega^2 = 10.21\%$). Variation between project year was insignificant ($p = 0.130$, $\omega^2 = 2.80\%$). However variation between the two sites was a significant factor ($p = 0.002$, $\omega^2 = 2.10\%$) and there was a significant interaction between site and project year ($p = 0.023$, $\omega^2 = 7.02\%$). The covariate E_4/E_6 ($p = 0.013$, $\omega^2 = 8.73\%$) was of greater importance than both treatment and project year. Residuals were normal. Post hoc analysis showed (Figure 5.20.a) that the old burn (1.076 ± 0.05) had more similar DOC to the heather control than the new burn (0.558 ± 0.04). No significant difference was revealed between project years (Figure 5.20.b).

a)



b)

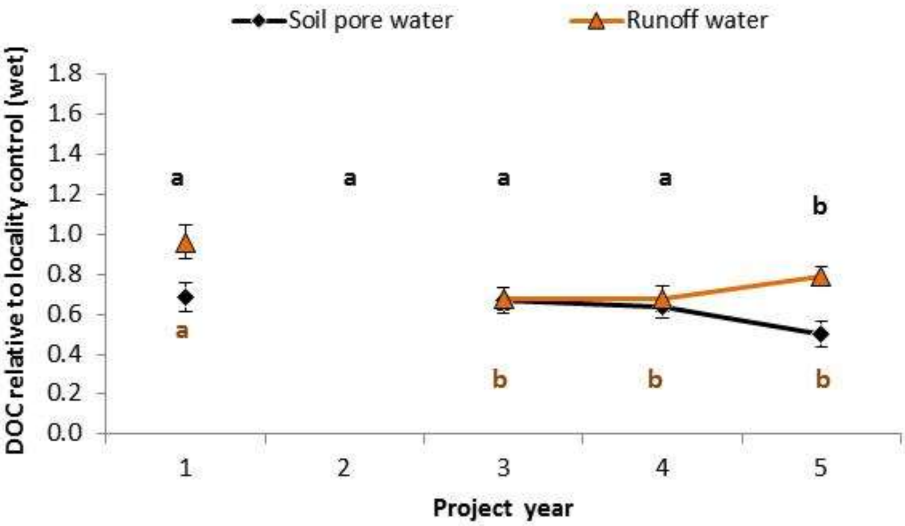


Figure 5.20: Wet locality burn sites (old and new) relative to control water DOC concentrations (soil pore black diamonds, black letters; surface runoff - yellow triangles, brown letters) ANOVA main effects plots: a) by site, b) by project year. Relative DOC values >1 indicate site sample DOC (soil pore or runoff) is greater than the locality control. The letters represent tukey post hoc test; shared letters indicate no significant between factor levels.

An ANOVA was used to further investigate the E_4/E_6 (R^2). It was found that the factor month nested within project year ($p < 0.0001$, $\omega^2 = 13.00\%$) was of greater importance than variation between project year ($p < 0.0001$, $\omega^2 = 5.29\%$). Variation between treatment site nested within locality was significant ($p < 0.0001$, $\omega^2 = 2.40\%$), and of greater importance than locality ($p = 0.264$, $\omega^2 = 0.205\%$). Post hoc analysis revealed (Figure 5.21) that the dry old burn (5.67 ± 1.06) and the wet new burn (5.64 ± 1.06) had higher runoff E_4/E_6 ratio than the dry site new burn (4.34 ± 1.06). Furthermore, project years one (6.1 ± 1.06) and five (5.95 ± 1.05) had higher E_4/E_6 ratio than project years three (4.29 ± 1.06) and four (4.47± 1.06).

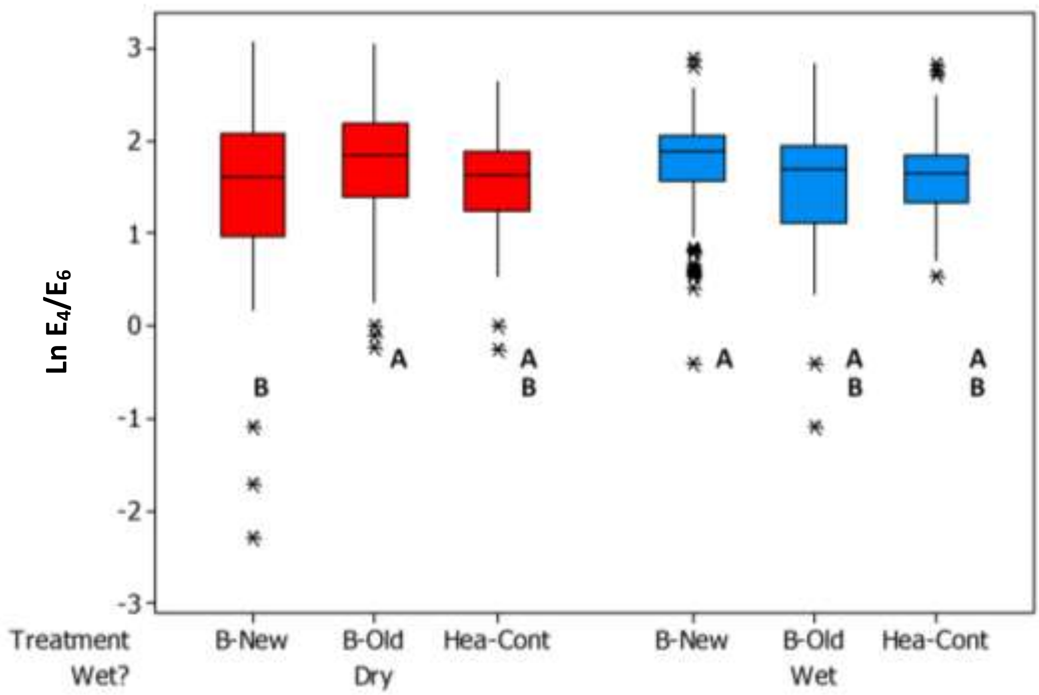


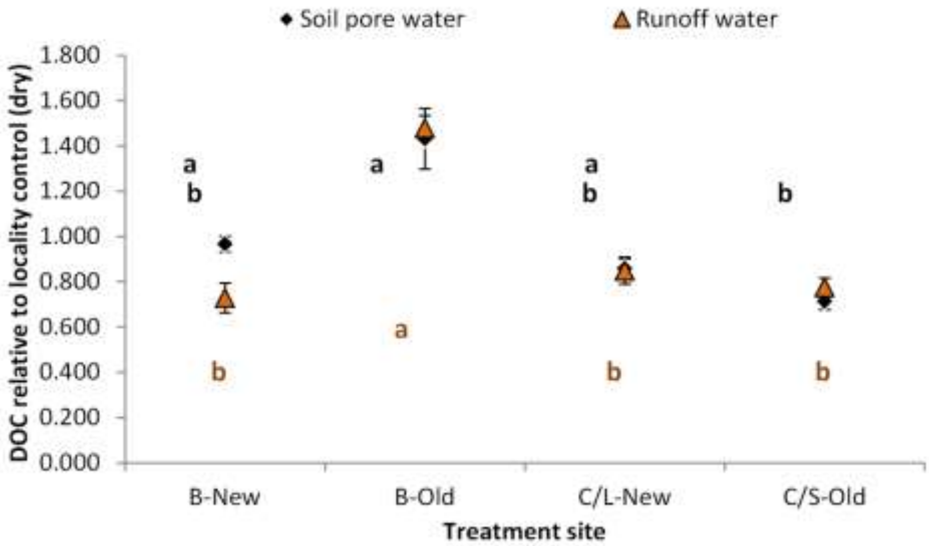
Figure 5.21: Box plot of E_4/E_6 ratio, by locality (Red bars = dry, Blue bars = wet) and treatment type. Post hoc analysis results are represented with letters. Shared letters indicate no significant difference; the opposite is true for different letters.

5.5.4.6 Runoff water DOC – managed cut vs. burn sites relative to dry locality control

Analysis of relative to the heather control runoff water DOC concentration data at the dry sites locality was conducted using an ANOVA. Over five project years, the data were most influenced by the factor month nested within project year ($p < 0.0001$, $\omega^2 = 32.49\%$), which was more important than project years ($p < 0.0001$, $\omega^2 = 1.05\%$). Variation between the factor plots nested within site ($p < 0.0001$, $\omega^2 = 5.89\%$) was more important than site ($p < 0.0001$, $\omega^2 = 1.03\%$). Residuals were normal. Post hoc analysis revealed that both cut sites had lower DOC concentrations that the heather control although more similar to the control than the burn sites.

The addition of covariates only slightly improved the R^2 from 40.49 to 40.57 %. Over five project years, the data was most influenced by the factor month nested within project year ($p < 0.0001$, $\omega^2 = 27.75\%$), which was more important than project years ($p < 0.0001$, $\omega^2 = 2.76\%$). Variation between the factor plots nested within site ($p < 0.0001$, $\omega^2 = 3.96\%$) was more important than site ($p < 0.0001$, $\omega^2 = 0.74\%$). The covariate natural log transformed conductivity ($p = 0.029$, $\omega^2 = 2.96\%$) and water table height ($p = 0.029$, $\omega^2 = 2.35\%$) were both significant and of similar importance in the ANCOVA. The residuals were normal. The post hoc test revealed (Figure 5.22.a) that there was a significant difference between the sites water runoff DOC variation relative the heather control at the dry locality. The old burn had significantly higher relative DOC, hence its DOC higher than the heather control (1.43 ± 0.13). The new burn (0.983 ± 0.04), new cut (0.857 ± 0.04), and old cut (0.714 ± 0.04) had lower relative DOC than the old burn and did not significantly differ to each other. The first project year (Figure 5.22.b) (1.496 ± 0.07) had the higher runoff water DOC concentrations relative to the heather control than project years three, four and five (0.832 ± 0.07, 0.802 ± 0.06, 0.703 ± 0.06).

a)



b)

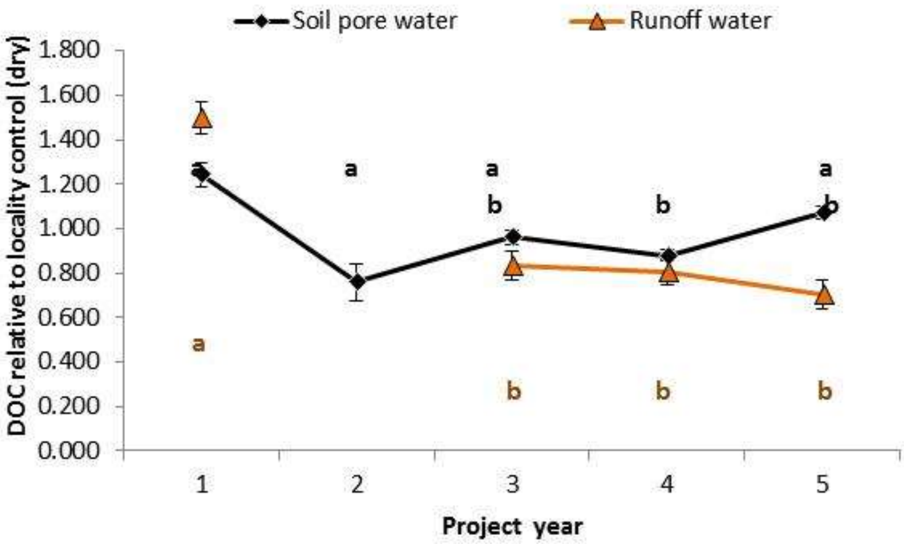


Figure 5.22: Dry locality burn and cut sites (old and new) relative to control water DOC concentrations (soil pore black diamonds, black letters; surface runoff - yellow triangles, brown letters) ANOVA main effects plots: a) by site, b) by project year. Relative DOC values >1 indicate site sample DOC (soil pore or runoff) is greater than the locality control. The letters represent tukey post hoc test; shared letters indicate no significant between factor levels.

5.5.4.7 Soil pore water DOC – managed burn sites relative to wet locality control

Soil pore water log normal DOC concentrations relative to the heather control at the wet sites locality were analysed using ANOVA. It was found that month nested within year was the most important factor ($p < 0.0001$, $\omega^2 = 23.51\%$). Annual variation was important ($p = 0.002$, $\omega^2 = 4.40\%$). The variation between sites was significant ($p = 0.061$, $\omega^2 = 2.64\%$), however less important than the variation of plots nested within site ($p < 0.0001$, $\omega^2 = 6.68\%$). Residuals were normal. Post hoc analysis revealed that the old burn (0.679 ± 1.04) had significantly higher DOC concentrations relative to the heather control. A significant difference was revealed between project years. The fifth project year (0.508 ± 1.07) had significantly higher relative DOC than project year one (0.709 ± 1.08), three (0.688 ± 1.07) and four (0.648 ± 1.05). Lack of data during project year two resulted in its exclusion from the model.

The addition of covariates in the ANCOVA did not change the R^2 much, from 36.14 to 35.97 %. The factor month nested within year remained the most important factor ($p < 0.0001$, $\omega^2 = 23.43\%$), followed by annual variation ($p = 0.002$, $\omega^2 = 4.50\%$). The variation between sites became more important with the addition of the covariates ($p < 0.0001$, $\omega^2 = 4.73\%$). The covariate plot vegetation (at 20 cm^2) ($p = 0.001$, $\omega^2 = 3.39$ %) was significant and more important than the covariate soil bulk density ($p = 0.010$, $\omega^2 < 0.0001\%$), which was unimportant in the model. These covariates accounted for the differences between the plots within in each treatment site. Residuals were normal. There was a positive correlation between relative to heather control soil pore water DOC concentration and plot vegetation mass (Pearson correlation 0.153; $p = 0.004$). The post hoc test revealed that (Figure 5.20.a) the new burn had significantly higher (1.261 ± 1.15) than both the old burn (0.303 ± 1.16) and the control, and the old burn had relatively lower DOC concentrations than the old burn.

5.5.4.8 Soil pore water DOC – managed cut *vs.* burn sites relative to dry locality control

Analysis of the soil pore water DOC concentration, relative to the heather control, at the dry site locality was conducted using an ANOVA. Over the five project years, the most important factor was month nested within project year ($p < 0.0001$, $\omega^2 = 32.48\%$). Variation between plots, as plots nested within treatment site ($p < 0.0001$, $\omega^2 = 5.89\%$), was more important that variation between the factor site ($p < 0.0001$, $\omega^2 = 1.05\%$). Variation between sites was of similar importance to variation between project year ($p < 0.0001$, $\omega^2 = 1.03\%$). Residuals were normal. Post hoc analysis revealed that all the treatment sites had lower DOC relative to the control site. The newly treated sites (both burn and cut) had more similar DOC concentrations to the control than the old treated sites. The new burn (0.991± 0.00) did not differ to the new cut (0.956 ± 0.00). The old burn (0.781± 0.001) also did not differ to the new cut (0.789 ± 0.001). The treatment sites relative DOC concentrations were highest during the first project year (1.022 ± 0.00) and final project year (0.985 ± 0.0) and at its lowest in project year four (0.830 ± 0.000).

The addition of covariates in the ANCOVA slightly increased the R^2 from 40.49 to 40.57 %. The factor month nested within year remained the most important factor ($p < 0.0001$, $\omega^2 = 27.75\%$). The factor plots nested within treatment site ($p < 0.0001$, $\omega^2 = 3.96\%$) was of similar importance to project year ($p < 0.0001$, $\omega^2 = 2.76\%$). The significance of variation between treatment sites as a factor remained of low importance ($p = 0.002$, $\omega^2 < 0.74\%$). The covariate normalised natural log of sample conductivity ($p = 0.029$, $\omega^2 = 2.96\%$) was of similar importance as the covariate water table high (up from the mineral layer) ($p = 0.029$, $\omega^2 = 2.35\%$). There was a positive correlation between DOC relative to control soil pore water DOC concentration and log transformed soil sample conductivity (Pearson correlation 0.166; $p < 0.0001$), in addition to a negative correlation between soil pore water DOC and WTH (Pearson correlation -0.176 ; $p < 0.0001$). Post hoc analysis revealed (Figure 5.22.a) the addition of the covariates removed the differences between the treatment sites. All the treatments had relatively higher DOC concentrations than the control. The old cut (0.741 ± 0.04) and new cuts (0.926 ± 0.04) became insignificantly different to the old burn (1.432 ± 0.13). The new burn (0.966 ± 0.04) had higher relative DOC than the old cut site. Despite the addition of covariates, the relative DOC remained highest in project years one (Figure 5.22.b) (1.243 ± 0.05) and five (1.071 ± 0.03) and lowest in project year four (0.879 ± 0.03).

Data set	Predictor	Factor (F)/ covariate (C)/ interaction (I)/ nested factor (N)	Factor	P value	ω^2	R^2
ANOVA	Goyt study fainfall (mm)	F	Locality	0.057	4.5	82.71%
	Project year 5	F	Month	< 0.0001	77.9	
	SCaMP rainfall (mm)	F	Year	< 0.0001	7.7	33.57%
		F	Month	< 0.0001	25.9	
	Peat depth	F	Locality	< 0.0001	46.9	86.30%
		N	Site (Locality)	< 0.0001	39.2	
	Bulk density	F	Locality	0.001	24.4	48.24%
		N	Site (Locality)	0.027	23.9	
	Slope	F	Locality	< 0.0001	-	75.98%
		F	Locality	0.214	0.1	61.19%
	Heather hieght (mm)	N	Site (Locality)	< 0.0001	60.8	
	Vegetation cover (%)	F	Locality	< 0.0001	6.6	37.94%
		N	Site (Locality)	< 0.0001	31.3	
ANCOVA	Vegetation cover (%)	F	Locality	0.478	<0.001	40.51%
		F	Site	0.849	<0.001	
		C	Cover type	< 0.0001	28.1	
		I	Site*Cover type	< 0.0001	5.3	
		I	Locality*Cover type	< 0.0001	7.5	

Table 5.3: Output for Goyt study catchment predictors ANOVA/ANCOVA.

Predictor	Test	Study	Factor (F)/ covariate (C)/ interaction (I)/ nested factor (N)	Factor	P value	ω^2	R^2
WTD	ANOVA	Cut *vs.* burn (Dry locality)	F	Site	< 0.0001	38.16	66.03%
			F	Project year	< 0.0001	0.30	
			N	Plot (Site)	< 0.0001	25.61	
			N	Month (Project year)	< 0.0001	4.69	
			I	Site*Project year	< 0.0001	0.63	
		Burn in wet *vs.* dry locality	F	Locality	< 0.0001	35.77	55.89%
			F	Project year	0.666	<0.00	
			N	Site (Locality)	< 0.0001	15.52	
			N	Month (Project year)	< 0.0001	4.45	
			I	Locality* Project year	0.018	0.20	
	ANCOVA	Burn in wet *vs.* dry peat locality	F	Locality	< 0.0001	3.26	66.56%
			F	Project year	0.217	0.03	
			N	Site (Locality)	< 0.0001	4.86	
			N	Month (Project year)	< 0.0001	3.82	
			I	Locality*Project year	0.045	0.13	
			C	Altitude ASL (m)	< 0.0001	19.07	
			C	Sampling day rainfall sum	0.039	0.94	
			C	C. vulgaris hieght (mm)	< 0.0001	3.23	
			C	Peat depth/ Bulk	< 0.0001	25.21	

Table 5.4: Output for Goyt study WTD and DOC ANOVA.

Predictor	Test	Water sample type	Study	Factor (F)/ covariate (C)/ interaction (I)/ nested factor (N)	Factor	P value	ω2	R^2
DOC	ANOVA	All site samples	Runoff and soil pore water DOC	F	Locality	< 0.0001	3.00	18.74%
				N	Site (Locality)	< 0.0001	3.25	
				F	Project year	0.017	0.24	
				N	Month (Project year)	< 0.0001	11.05	
				F	Sample type	0.002	0.54	
				I	Locality*Sample type	< 0.0001	0.38	
				I	Locality*Project year	0.021	0.27	
		Runoff	Burn in wet *vs.* dry locality	F	Locality	< 0.0001	1.36	32.18%
				N	Site (Locality)	< 0.0001	8.15	
				F	Project year	0.201	< 0.001	
				N	Month (Project year)	< 0.0001	8.86	
				N	Plot (Site)	< 0.0001	13.82	
			Cut vs.burn (Dry locality)	F	Site	0.001	1.11	41.03%
				N	Plot (Site)	< 0.0001	5.80	
				F	Project year	< 0.0001	2.07	
				N	Month (Project year)	< 0.0001	32.59	
		Soil pore water	Burn in wet *vs.* dry locality	F	Locality	< 0.0001	1.36	34.56%
				N	Site (Locality)	< 0.0001	8.15	
				F	Project year	0.201	<0.00	
				N	Month (Project year)	< 0.0001	8.86	
				N	Plot (Site)	< 0.0001	13.82	
			Cut vs.burn (Dry locality)	F	Site	0.001	0.96	34.56%
				N	Plot(Site)	< 0.0001	5.09	
				F	Project year	< 0.0001	1.73	
				N	Month (Project year)	< 0.0001	26.75	
	ANCOVA	All site samples	Runoff and soil pore water DOC	N	Site (Locality)	< 0.0001	5.46	21.21%
				F	Locality	0.726	< 0.001	
				F	Project year	0.021	0.25	
				N	Month (Project year)	< 0.0001	11.35	
				F	Sample type	< 0.0001	0.90	
				I	Locality*Sample type	< 0.0001	0.69	
				I	Locality*Project year	0.031	0.24	
				C	Bulk_density	0.001	2.19	
				C	Peat_depth_(Jun13)	< 0.0001	0.68	
				C	Sampling day rainfall sum (mm)	< 0.0001	< 0.001	
		Runoff	Burn in wet *vs.* dry locality	F	Locality	0.172	0.03	38.09%
				N	Site (Locality)	< 0.0001	6.58	
				F	Project ur	0.63	0.34	
				N	Month (Project year)	< 0.0001	13.45	
				N	Plot (Site Locality)	< 0.0001	14.01	
				C	pH	0.008	3.60	
			Cut vs.burn (Dry locality)	F	Site	< 0.0001	7.55	42.85%
				N	Plot (Site)	< 0.0001	18.04	
				F	Project year	0.145	< 0.001	
				N	Month (Project year)	< 0.0001	15.84	
				C	Month C	0.008	1.72	
		Soil pore water	Burn in wet *vs.* dry locality	F	Locality	< 0.0001	1.36	41.03%
				N	Site (Locality)	< 0.0001	8.15	
				F	Project year	0.201	< 0.001	
				N	Month (Project year)	< 0.0001	8.86	
				N	plot (Site)	< 0.0001	13.82	
			Cut vs.burn (Dry locality)	F	Site	0.001	2.814	35.38%
				N	Plot(Site)	< 0.0001	3.942	
				F	Project_year	< 0.0001	1.744	
				N	Month (Project_year)	< 0.0001	24.029	
				C	Ln WTD	0.014	2.822	

Table 5.5: Output for the Goyt valley studies (wet vs. dry burn; cutting vs. burning in dry locality) DOC ANCOVA.

Predictor	Test	Water sample type	Study locality	Factor (F)/ covariate (C)/ interaction (I)/ nested factor (N)	Factor	P value	ω^2	R^2
Relative to locality control DOC	ANOVA	Runoff	Wet locality	F	Site	< 0.0001	2.38	30.91%
				F	Project year	0.005	7.44	
				N	Month (Project year)	0.001	15.13	
				I	Site*Project year	0.002	5.82	
			Dry locality	F	Site	< 0.0001	8.03	49.11%
				F	Project year	< 0.0001	1.70	
				N	Month (Project year)	< 0.0001	19.76	
				N	Plot (Site)	< 0.0001	19.56	
		Soil pore water	Wet locality	F	Site	0.024	1.48	36.14%
				F	Project year	0.002	4.40	
				N	Plot (Site)	< 0.0001	6.68	
				N	Month (Project year)	< 0.0001	23.51	
			Dry locality	F	Site	< 0.0001	1.05	40.49%
				F	Project year	0.002	1.02	
				N	Month (Project year)	< 0.0001	32.48	
				N	Plot (Site)	< 0.0001	5.89	
	ANCOVA	Runoff	Wet locality	F	Project year	0.130	2.80	31.08%
				F	Site	0.002	2.10	
				N	Month (Project year)	0.023	10.21	
				I	Site*Project year	0.006	7.02	
				C	E_4/E_6	0.013	8.73	
			Dry locality		N/A			
		Soil pore water	Wet locality	F	Site	0.000	4.73	35.97%
				F	Project year	0.002	4.50	
				N	Month (Project year)	0.000	23.43	
				C	Veg cover (g/10 cm^3)	0.001	3.38	
				C	Bulk density	0.010	<0.00	
			Dry locality	F	Site	0.002	0.74	40.57%
				F	Project year	0.002	2.76	
				N	Month (Project year)	0.000	27.75	
				N	Plot (Site)	0.000	3.96	
				C	Ln Conductivity	0.029	2.96	
				C	WTH	0.029	2.35	

Table 5.6: Output for Goyt study relative to locality control DOC ANOVA and ANCOVA.

5.6 Discussion

5.6.1 Variation between Goyt Valley sites and localities

Analysis of the five project years of data was conducted to contribute towards the debate on upland vegetation management effects on water quality. Weather, site specific and catchment related variables were also considered in order to constrain the effects of heather treatment on water quality in the Goyt. In the previous chapters (on Bleaklow), it was determined that effects of site, specifically associated with bare restoration treatment method, on DOC and WTD were overridden by the effects of air temperature.

The variation between sites and localities (wet and dry) were characterised to better explain potential variation in DOC concentration. Site and locality factors were investigated in terms of peat (depth and bulk density) and heather (height and cover). Analysis of peat depth demonstrated the importance of locality over site, indicating that large scale features and topography had greater consequence on peat depth than site treatment. This finding in the Goyt study sites is supported by the findings of Holden *et al.* (2007) which link drainage and erosion to peat depth in addition to Holden and Connolly (2011), which indicated that peat depth is significantly related to slope. Slope influences drainage through the force of gravity; it can have strong localised effects on the water table (Shepherd *et al.* 2013). Ravenslow (the wet locality), positioned in a topographic depression, had significantly gentler slopes than Big Moss (the dry locality). The locality, site altitude and slope position as well as peat depth are important in explaining water table depth variation between sites over the five project years. Increase in slope influences water flow and was linked to increased water table depth (Boothroyd 2014). The differences in water table by locality can be attributable to slope as no significant difference in rainfall was found between the two localities. The study site altitude position and peat depth were linked to WTD variation.

There was a significant variation in site water table depth over the five project years, attributable to differences in heather management. Both managed heather cutting and burning in a dry locality promoted a shallower water table depth than the untreated heather control sites (last burnt >15 years ago at the start of the cut versus burn study in 2008). Notably in the dry locality, the more recently cut site had the shallowest water table (Worrall *et al.* 2012). This indicated that cutting heather promoted a shallower water table depth than burning did. The aged managed heather cut site had significantly deeper water tables than the new cut. Furthermore the findings at the dry locality indicated that, as a managed cut site ages (over time) the water table becomes deeper and indistinguishable to the water table depth of the dry burn site (both old and new). Hydrological, the findings therefore support cutting as an alternative management method to burning (in a dry locality).

At the wet locality, the burnt treatment sites (both old and new) had significantly shallower water table depth than the burn sites (old and new) in the dry locality. It was found that although managed heather burns did influence water table depth (raise water table closer to the surface); the effect of locality was of greater importance to water table depth at sites managed through burning. Interestingly the water table depth at the control site in both the wet and dry locality did not significantly differ, which could indicate that treatment of degenerate heather could encourage the water table to rise closer to the surface in both dry and wet localities. Water table depth was correlated to the peat depth, with shallower water tables linked to deeper peat. This result is likely due to the conditions (high water table depth and anaerobic) which promote the formation of peat. Leaf litter is deposition accompanied by shallower water tables (as present at the wet locality) and anaerobic conditions, reduce the rates of aerobic microbial activity and decomposition (Mezbahuddin *et al.* 2014); thus promoting peat deposition. The significant difference in peat depth, linked to wet or previously wet environmental conditions, indicate the deep water table at the dry locality did not promote peat deposition at the same rates as the wet locality. Peat depth is a good indication of the history of peat deposition and erosion, although less useful in explaining recent vegetation management effects on peat, as peat depth pre-management depths were not recorded. The measurement of the surface soil erosion rates using a grid erosion pins method (referred to in Chapter 4), would have been useful in explaining short term changes in vegetation as a consequence of treatment. Overall the finding on water table depth and peat depth have drawn attention to the importance of locality and topographical condition which should be accounted for within a model explaining the effects of management on DOC.

Peat bulk density (at peat surface top 10 cm) was linked to site peat depth. Both locality and site were of high importance when explaining surface peat bulk density and peat depth variation. At the wet locality, peat depth was found to be higher at the old dry treatment sites than at the treatments sites. Bulk density, unlike peat depth, was less strongly influenced by localities than site variation. This evidence would suggest that treatment had a slightly greater influence on bulk density than on peat depth in the short term (five project years), although no pre-treatment samples were collected. Bulk density was measured as it is an indicator of a peatlands health; higher densities are associated with dry peat, as found in the dry locality in which there is also significantly shallower peat depths and greater slope. Bulk density can be used as a measure of decomposition (Boelter 1986). Increasing decomposition is correlated to an increase in bulk density (Nichols and Boelter 1984). The lower bulk density in wet localities is likely due to the higher water table occupying pore space; a higher water retention capacity retained in the peat due to the presence of *Sphagnum spp*. on Ravenslow, which was absent from all sites on Big Moss. Therefore the difference in Goyt Valley study site bulk density indicated that there are greater peat decomposition rates at Big Moss than Ravenslow. Interestingly the treatment site bulk density did not differ to the heather control site. This can potentially be explained through the difference in wet sample and dry sample weight, as the difference between the wet sample weight and dry sample weight at the dry control site is smaller than the burn and cut treatment sites. Sample water storage capacity could have been measured to clarify this point by saturating the sample prior to weighting. The difference otherwise can be more indicative of the WTD at site, which would have contributed to the volume of water within the sample, and would in turn be a greater reflection of the differing site morphologies (e.g. gully edge draw-down effects, Allott *et al.* 2009, Daniels *et al.* 2008) instead of differences between treatment sites. In theory, bulk density was used to give an indication of the effect of treatments on the health of the peat. The evidence in Goyt Valley suggests that benefit of heather management through burning (on peat heath) is greater in localities with shallow water table depth, as found in both the old and new burn sites in Ravenslow (the wet locality).

This Goyt Valley study is about the impact of vegetation development, rather than loss (i.e. Bleaklow bare peat study). Burning of heather is a common practice which some research has shown to have significant effects on biodiversity, influencing flora species composition and growth. Managed burning can be a benefit or disadvantage to different species (McFerran *et al.* 1995, Tharme *et al.* 2001). The slow heather regeneration results in a year lag post a cut (Muñoz *et al.* 2012). This is due to heather being a relatively slow-growing species and the length of heather left behind post a burn is higher than that after a cut. In addition to the direct effect of management on vegetation, water table fluctuation, according Breeuwer *et al.* (2009), is an important factor in controlling vegetation type. Emphasis was placed on heather, as that was both the target and treated vegetation. *Eriophorum spp.* was found to be the most dominant vegetation type across the Goyt study plots. Heather is a relatively slow-growing shrub. When cut, the length of heather left behind is shorter than that post a burn, and there is a year lag in heather regeneration post a cut (Muñoz *et al.* 2012). It was found on Big Moss that two of its sites had higher heather cover than on Ravenslow. As locality explained around

10% of the variation, the difference in cover can partially be explained by the lower water tables, which are associated with higher vascular communities (Urbanova *et al.* 2012). Vascular plants and shrubs are relatively unaffected by water table drawdown because plant roots can compensate for reduced water uptake at near-surface by increasing root distribution and hence water uptake deeper in the soil (Dimitrov *et al.* 2010). The old cut site was most similar, in terms of both heather height and cover, to the new burn sites at the wet locality, with lower heather height than its locality control. This could be due to the delayed in heather regeneration, post cut, which resulted in a lower heather plot cover dominance and allowing for dominance of faster growing vegetation (Mitchell *et al.* 2008), such as herbaceous species (Calvo *et al.* 2005). The regeneration of heather on a new burn has the greatest success. The change in the vegetation cover is probably due to the nature of heather stands, as they age the heather canopy 'opens up' would allow light to reach the understory (McFerran *et al.* 1995), thereby increasing cover percentage of sedge functional group plants such as *Eriophorum spp.* and grasses such as *Molinia* (Ross *et al.*, 2003). The new cut site had the highest *Eriophorum spp.* cover, due to the lag in heather regeneration (Mitchell *et al.* 2008). However, as the vegetation grew, the percentage cover of *Eriophorum spp.* was reduced to that of an old burn. Ravenslow had greater *Sphagnum spp*. establishment success, which is evident particularly in the old wet burn and as mats in blocked grips. The absence of *Sphagnum spp*. on Big Moss on the other hand is likely due to deep water table. There is an important relationship between water table depth and *Sphagnum spp*. (Clymo and Duckett 1986), as *Sphagnum spp.* growth is reduced in dry conditions (Rydin 1993), such as on Big Moss the dry locality. Because unlike vascular species with far reaching roots, *Sphagnum spp*. rely on passive water transport through an external capillary network (Thomsen *et al.* 2002).

5.6.2 Goyt DOC

This study found that over the five project year in the Goyt valley, soil pore water DOC concentrations were ~100 mg C/L, exceeding runoff water DOC concentrations by ~13%. This finding is supported by Clay *et al.* (2009) who found lower site runoff DOC concentration than soil pore water on burnt and grazed sites in Northern England. Surface Runoff water in an ombrotrophic bog, such as that on the Goyt, would be expected to have similar DOC concentrations to rainwater, which typically has lower DOC than that soil pore water. The difference in DOC can be explained by the lower water residence time of runoff water at sites compared to soil pore water. Soil pore water would have greater humic and fulvic DOC components associated peat humification (Whitbread 1994) and lignin derived and root exudates (Carlson *et al.* 2000). Although Goyt DOC concentrations significantly differed between surface runoff and soil pore water samples, it was found that in fact seasonal variation in was of greater importance in influencing DOC concentrations. This finding in the Goyt followed Clay *et al.* (2009); where monthly variation was more important than burning on the DOC concentration. There was also some annual DOC fluctuation. Within pooled data, generally concentrations of DOC were greater at old managed burn sites (dry locality), and smaller at new cuts (also in dry locality). Additionally the old burns (wet locality) had higher concentrations than the dry new burn (dry locality). Furthermore site morphology, specifically peat depth and the bulk density had greater relation to the DOC variation than sample type. An increase in peat depth was linked to reduced DOC concentrations (more so in soil pore water than surface runoff). This is possibly due a locality effect in which sites with greater peat depth are associated with shallower water table depth. This is evident as water table height up from the mineral soil was more important than water table depth in influencing soil pore water DOC concentrations. In the managed burn study in the wet vs. dry locality, the variation in water DOC (both runoff and soil pore water) at managed burns, relative to the heather control, was greatly influenced by variation between sites; differences between site plots (within a site) were not important. This finding is likely due to the homogeneity of the plots. The importance of conductivity This supports the finding of Yallop and Clutterbuck (2009) who highlighted the importance of catchment on DOC concentrations, particularly in the case of deep or blanket peat covered catchments.

5.6.3 Managed heather burning effects on DOC in wet vs. dry locality

The effect of managed burning in wet vs. dry locality was investigated further by water sample type (runoff and soil pore water). In the case of runoff water, it was found that the variation of sample DOC concentrations within a site, between plots, was greater than between treatment sites and locality. Interestingly, the treatment sites runoff concentrations did not differ to that of the control. However, the dry old burn had higher DOC concentrations than both the wet and the dry new burn sites. Runoff sample DOC concentrations were positively correlated to, and increased with pH. Site variation was important, although runoff DOC concentration did not significantly vary between control burns and the control sites. The pH at control sites was higher than at old burns (wet and dry), and both the new burns and control sites had higher pH than the new burn site. The pH is linked with locality water table depth, and there was insufficient evidence to suggest an interaction between the mineral layer and soil pore water significantly influences pH. The relationship between pH and DOC on the Goyt is supported by finding by Scott *et al.* (1998) who highlighted the link between pH and DOC. Clark *et al.* (2005) explained the link through the effect of pH on the solubility of DOC. The solubility of humic compounds in the soil are particularly sensitive to pH (Weishaar *et al.* 2003). It is evident in this study that DOC variation occurs in a complex manner.

Analysis also found that the DOC concentration in runoff water at the new burns had ~40% lower DOC concentrations than the control sites. However, as the burn sites age they become more similar to the control sites with ~20% lower runoff DOC concentrations than the control. This result is expected, since the control sites would have been burnt at some stage (more than 15 years prior the study). Variation at the treatment site relative to the control was influenced by month as a function of seasonality. Thus, the burn sites variation differed to the control throughout the year in addition to between years. Variation occurring at treatment sites varied with project year. The importance of sample E_4/E_6 ratio in relation to runoff DOC concentrations reduced the relative importance of seasonal variation. Thus, concentrations at the burn sites were related to DOC speciation. Armstrong *et al.* (2012) found that higher soil pore water DOC concentrations were more associated with heather than with sedges and *Sphagnum spp.*, which would suggest the vegetation type influenced the E_4/E_6 ratio (i.e. the lower E_4/E_6 ration is associated with humic components of lower molecular weight (Carlsen *et al.* 2000). At the dry locality, the E_4/E_6 ratio of soil pore water of burn sites increased with age, as old burns had higher E_4/E_6 ratio. The wet new burn had a higher ratio than the dry new burn. A negative correlation of E_4/E_6 ratio to runoff DOC concentrations a burn treatment sites indicated that DOC concentration in runoff samples were less associated with fulvic derived components from vegetation than humic acids.

Clay *et al.* (2009) and Worrall *et al.* (2007) found that differences in DOC can be explained by month as a function of seasonal variation, the authors also found that burn explain a small portion of DOC concentration variation. This study also finds that soil pore water DOC concentrations were linked to burning, specifically soil pore water DOC concentrations were ~ 26% higher at the new burns than the control, whereas aged plots have 70% lower soil pore water DOC concentrations than the control. Thus, there was a decrease in soil pore water DOC as the burn sites age.

The effect of vegetation management on DOC concentration at burn sites by locality is important as the differences between the sites increased when surface vegetation mass and soil bulk density were accounted for. Yallop and Clutterbuck (2009) found that the use of management burning on blanket peat increased productivity of DOC and its release in an upland environment, such as that on the Goyt. The changes to surface vegetation were found by Chen *et al.* (2008) to influence microbial community diversity and abundance. Thus, the above ground vegetation treatment had a significant effect on microbial activities which influence DOC concentrations (Caporn *et al.* 2007). As previously mentioned in the chapter on Bleaklow, newly established vegetation such as heather can increase DOC fluvial outputs in relation to enhanced exudation and decomposition of litter and peat (hence the significant of bulk density) (Fenner *et al.* 2007). Thus old burn plots may have lower DOC production due to the more mature vegetation present at the surface. The wet site burn differs in DOC to the heather control more than the dry sites due to the understory vegetation. The dry surface and lack of understory in the dry site resulted in lower DOC production and the surface slope results in shorter retention time which resulted in the higher concentrations of DOC not be intercepted.

5.6.4 Managed heather burning and cutting effects on DOC (dry locality)

As mentioned previously, management of upland peatland vegetation through cutting had a significantly different effect on both water table depth and vegetation to that of burning. Analysis of DOC concentrations found that variation between plots was greater than variation between treatment sites. Furthermore, monthly variation as a function of seasonality was again important, as found in the burns in wet vs. dry peatland condition study and on Bleaklow (Chapter 3).

In the case of cut vs. dry vegetation treatments runoff water DOC, it was found that a portion of the seasonal variation can be explained by near surface soil temperatures, which increased runoff DOC concentrations. Treatment site, both cut and burnt, did not differ to the control sites. The control site had higher runoff water DOC concentrations than the new cut and the old burn. Both of the sites had the highest *Eriophorum spp.* plot dominance of the sites present at the dry locality.

The soil pore water DOC concentrations, like runoff, were again influenced by seasonal variation. Variation between treatment sites was less important than between plots. Unlike cut and burn site runoff DOC concentrations, there were no significant differences between the control site and the cut and burn treatment sites. However, the age of the treatment plots was important. Specifically, the new treatment sites (both cut and burnt) had higher soil pore water DOC concentration than their aged comparators. This variation was further explained through WTD as shallower WTD was linked to reduced DOC concentration. It is evident that as a treatment site ages, on Big Moss, there is a decrease in soil pore DOC. This relationship of DOC and vegetation age is likely due to newly established vegetation performing a higher rate of photosynthesis (as found in Chapter 4), thereby producing greater amounts of rout exudate than is usually associated with old degenerate heather, which becomes dry and woody (Hobbs and Gimingham 1987). Armstrong *et al.* (2012) found higher soil pore water DOC concentrations were more associated with heather dominance than with sedges and *Sphagnum spp.* dominance. Furthermore, *Eriophorum spp.* dominated newly cut treatment sites and their DOC concentrations did not significantly differ to that of the heather dominated control and new burn sites. Additionally mature heather, associated with deeper water tables (Urbanova *et al.* 2012) as found on at the control site, can compensate for reduced water uptake at near-surface by increasing root distribution at greater depth within the soil (Dimitrov *et al.* 2010).

Variation in runoff DOC concentrations occurring at cut and burn treatment sites were varied between each other seasonally but also to the dry heather control. The ability to explain the data variation occurring at sites was improved when explained in relation to the controls. The old burn had ~50% higher DOC concentration than the control. The burn differed to the three other treatments (new burn and both old and new cuts) which had between 20-30% DOC concentrations than the control. In the first project year after treatment, the treatment sites had ~ 50% higher runoff DOC concentrations than the control. This levelled out over the last three project years to 20 -30 % lower than the control.

Comparatively, soil pore water DOC concentrations variation at the same cut and burn treatment site was linked to sample conductivity, and water table height. There were significant deviations in soil pore water conductivity from the heather control. Changes in managed burn sites soil pore conductivity is important, as it influences coagulation of DOC (Monteith *et al.* 2007), increases in ionic strength can suppressed the release of DOC (Clark *et al.* 2005). Therefore it is reasonable to infer that a mechanism of a change in surface vegetation cover combined with a raise water table depth can reduce soil water conductivity thereby increasing DOC solubility and concentrations relative to the locality control. Water table height up from the mineral layer (as a function of WTD and peat depth), was of greater importance to soil pore water DOC concentrations than the management type. This evidence points to the importance of a catchment scale effect on soil pore water DOC. However, old burns had ~43% higher soil pore DOC than the control site, and had higher DOC than the aged cut site which had ~30% less soil pore water DOC than the heather control. These two treatment (old cut and burn) sites did not differ to the new bun and new cut. This result at the dry locality could be related to that fact that peat depth at the control site was lower than some of the old burn plots. The importance of peat depth in influencing site water table depth and DOC may simply reflect the differing site morphologies (Daniels *et al.* 2008). Therefore it is important to measure and account for peat depth during site selection and data analysis, as the link between DOC and peat depth could have masked the effects of treatment.

5.7 Conclusion

Over a period of five years, runoff and soil pore water were sampled and analysed from managed vegetation cut and burnt sites, in the Goyt valley of the peak district: the following conclusions were made:

- In the Goyt valley peatland runoff water DOC concentrations are lower than concentrations of soil pore water.
- Runoff and soil pore water DOC concentrations are primarily determined by seasonal variation and are have a complex relationship with upland vegetation management.
- Peat depth and bulk density are linked to water table depth.
- Sites with deeper peat are associated with lower DOC concentrations than site with shallow peat.
- Topographical linked variables, such as altitude, slope and peat depth are linked to DOC and WTD. The effects of topography should therefore be accounted for DOC models in order to constrain the impacts of surface vegetation management on water quality.
- In the dry locality, site managed through burning have increased DOC concentrations with age. The management of heather using either cutting or burning would therefore be preferential to unmanaged, untreated mature/degenerate in dry conditions.
- At wet localities, when burn sites age, their runoff water DOC concentrations decrease while their soil pore water DOC concentration increase.
- Catchment scale effects on DOC concentrations such as peat depth, site slope and peat bulk density, can mask the effects of vegetation treatment and should be accounted for during site selection and WTD and DOC concentration analysis.

6 *The effects of vegetation management on DOC, at multiple scales, from headwaters to reservoir*

6.1 Rationale

The production of DOC in an upland peat have been largely influenced by seasonal temperature (Clay *et al.* (2009a). This was also the finding on the Goyt sites (Chapter 5). Above ground vegetation treatment has a significant effect on microbial activities which influence DOC concentrations (Caporn *et al.* 2007). Management of vegetation through cutting and burning give rise to dormancies by different vegetation types (Calvo *et al.* 2002). Root exudates represents significant carbon loss from vegetation within the surface root zones, occupying the top 10cm (Lindsay 2010). Change in vegetation type would be expected to influence microbe communities (Chen *et al.* 2008), and DOC production associated with root exudates which are exerted from the plant roots along with ions, free oxygen, water, enzymes, mucilage, Metabolites (Mitchell *et al.* 2008, Wallage and Holden 2010). Worrall *et al.* (2003b) found a rise of DOC concentrations within blanket peat dominated catchments. There is great interest in the relationship between vegetation and peatland carbon fluvial export on reservoir water quality. Management is considered a key factor influencing the concentration of DOC within surface water (Yallop and Clutterbuck 2009). The potential to manipulate surface water (river and reservoir) quality and DOC concentrations through vegetation management has come into question.

Managed burning of peatlands vegetation can influences peatland biota and abiota , both latterly and vertically at multiple scales from plot to catchment. Managed burns change the thermal dynamics of surface of peat soil and have effects reaching up to 20cm peat depth (Brown *et al.* 2015). It was also demonstrated by Brown *et al.* (2014) that in untreated peatland sites, there is an increased percentage of soil organic matter with depth. Clark *et al.* (2008) identified a strong correlation in DOC concentrations between soil-pore water at the top 10 cm of peat to stream water (Clark *et al.* 2008). A study by Holden and Burt (2003) on runoff DOC also emphasised the importance of the top 1-5 cm of the peat, the depth at which runoff is generated and would transport most soil pore water DOC into streams. These findings by Clark *et al.* (2008) and Holden *et al.* (2003) emphasise the importance of the top 1 – 10 cm depth, to catchment (rivers and reservoirs). This link between the peat surface and stream DOC concentrations explains how managed burns have had significant influences on river aquatic communities (Holden *et al.* 2012).

Given the importance of water flow at the peat surface, it is important to consider surface DOC concentrations, and the connectivity of soil pore water between experiment treatment site and catchment surface waters. In Chapter 5, it was found that catchment related variable (i.e. peat depth) had a significant impact on water table depth and DOC concentrations at a plot scale (soil pore water across 1m dipwell). The importance of temporal scales was emphasised throughout the thesis Chapters 3- 4. The study in this chapter was designed to further constrain the effect of surface vegetation treatment on soil pore water DOC concentrations and related plot scale variation to that of variation in reservoir DOC concentrations.

6.2 Aim

The aim of this study is to investigate:

- *C. vulgaris* burning vs. cutting at multiple peat depths.

The null hypotheses of this study area are:

- The locality *C. vulgaris* controls, runoff and through-flow water sample presence will not differ between localities.
- Water table depth and will not differ between treatment sites.
- There is no difference in water runoff between sites with differing water table depth.
- There is no significant difference in DOC concentrations between rainwater, runoff water, soil water (from 1m cross section dipwells or through-flow intercept at intermediate 10 soil cm depth), stream water and reservoir water.
- There is no difference in site DOC concentrations between the untreated *C. vulgaris* control and those where the vegetation was burnt or cut; at wet vs. dry peat conditions.

The null hypothesis will be tested through analysis of a multi annual data set of soil water table, soil pore water sample DOC concentration, UV-Vis absorbance, pH, conductivity. Analysis of seasonal and annual data trend will be conducted using graphs, comparative analysis of the data, ANCOVA GLM models and post hoc tests to find where differences lie.

6.3 Methodology

6.3.1 Treatment sites hydrology

The experimental sites and plots used in this study are the same as those used in Chapter 5. In addition to the pre-existing dipwell (Figure 6.1.a) and crest fall surface runoff traps and (Figure 6.1.b) Equipment was installed for the study including: soil pore water 10cm depth flow intercept traps, rainwater water gauges and a stream auto sampler. This stream was selected as it was a first order stream, downslope from the Goyt Valley study sites; it was therefore in theory representative of water drained from the Goyt Valley. The stream feed water from the Goyt Valley, into Errwood reservoir and then onto Fernilee reservoir (Chapter 2). Samples of water were collected of rainfall, surface runoff, through-flow, soil pores (at water table depth), stream and reservoir; all water samples collected were analysed as per the previous methodology stated in the thesis methodology (Chapter 2) (i.e. for DOC, pH, Conductivity, abs 400, 465 and 665).

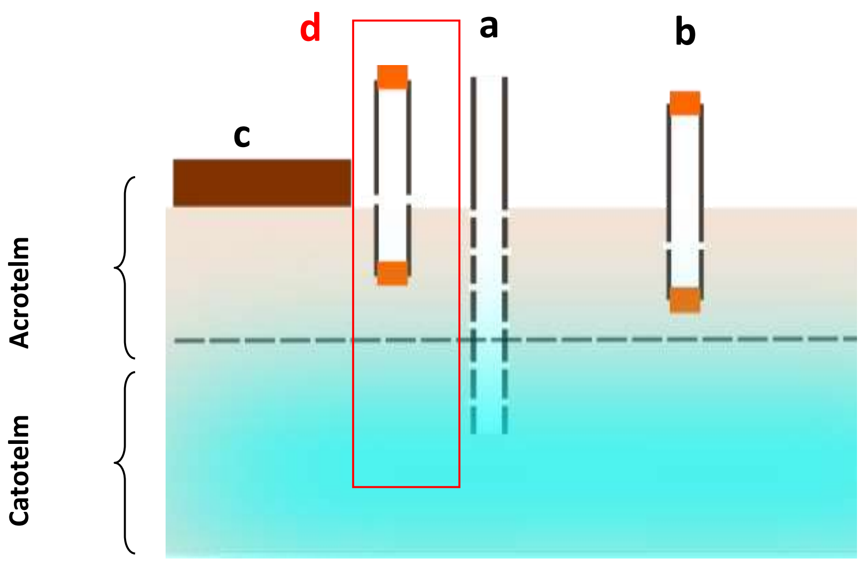


Figure 6.1: Experimental plot design. a) dipwell, b) surface runoff trap, c) gas collar, d) intermediate depth soil pore water sample trap.

Monitoring and sampling in the Goyt Valley commenced in May 2008 on two studies into heather management: a) managed burn (wet versus dry locality); b) managed burn versus cut (dry locality). Sampling and monitoring was conducted monthly, usually within the last week of every month. Samples were collected from the ~1 m deep dipwells and runoff traps from the original plot design (Figure 6.1). A new soil pore sample trap was installed named '10DD' (Figure 6.1.d; Figure 6.2.b), in order to relate sites soil pore water DOC concentrations to the surface treatment, and remove the influence of the catchment related variables effect on water quality (i.e. peat depth). The 10DD water traps restricted water sampling to a depth of 10 cm, allowing the through-flow water to enter the trap horizontally at a set depths of 10 cm below the peat soil surface.

Figure 6.2: An example of a Goyt Valley site triplicate plot (set) (Ravenslow – dry locality, 2013). Visible in the figure are three dipwells (labelled a.1 – a.3) and a 10DD though flow trap (labelled b). Also present in the photo are: three runoff traps, three collars (each plot) and an additional through-flow 10DD trap (between dipwell a.2 and a3), however they are not visible due to the high vegetation.

The 10DDs were installed (in April 2012) in sets of four per treatment, at both Ravenslow and Big Moss sites (Table 6.1). Within each set of triplicate plots two 10DD traps were installed, each equidistant between two site plots (Figure 6.2). The 10DD intermediate flow traps were made of ~40 cm long UPVC tubing (the same tubing used for the dipwell and runoff trap), with 4 symmetrical holes drilled at 20 cm down the tube, on opposite sides. These inlet holes allowed water to enter from different directions. A bung was inserted into the bottom and at the top of the tube in order to: a) exclude the entrance of ground water or rainwater, and b) prevent environmental degradation (photo degradation) and airborne contamination of the sample. The trap was inserted into the ground with the inlet holes at 10cm depth below the ground surface. This depth allowed the top peat layer, root zone, and soil pore water to be sampled at 10cm depth. Once installed, the ground level was then marked on the soil pore water trap, in order to ensure the trap was at the appropriate depth upon future visits. On the monthly site visits, sample presence was noted, after which the samples were collected. To prevent disturbance of the soil and vegetation, the bung would be removed from the top of the 10DD trap and the samples were collected using a pump (designated for 10DD sample collection only). The water sample was then poured into a sample bottle and the bung replaced, while ensuring the trap was at the correct level (marked during installation). Sample presence was noted in order to investigate water flow pathways.

Due to the increased number of water samples to be analysed in this head water to reservoir study, only four of the six dipwells present at each treatment site were sampled, although water table depth was measured at every dipwell. Additionally, to maximise the potential to explain the data variation, all 6 runoff traps at each treatment site were sampled, as runoff samples are often not successfully intercepted.

In addition to the through-flow samples collected at the site plots (Figure 6.1), a rainfall gauge (detailed in Chapter 5 methodology) was used to measure monthly rainfall (mm) and collect duplicate rainfall samples at four of the Goyt Valley sites, two in the wet locality and two in the dry (Table 6.1). Duplicate samples were also collected from the northern edge of Errwood reservoir (accessed via Sandy lane), and from the southern edge of Fernilee reservoir (accessed via a pathway that connects to the Goyt lane). Reservoir samples were collected monthly, at the end of a day, during visits to Big Moss. The samples were collected while wearing disposable safety gloves, for health and safety reasons as reservoirs can experience toxic algal blooms. A dip stick was used to reach the water. The reservoir water sample was poured into a small bottle and the gloves were safely disposed of. Reservoir water quality data was also obtained from the Sustainable Catchment Management Programme SCaMP (Reservoir water level, total organic carbon (TOC), pH, conductivity and hazen unit for water colour).

Location	Site	Treatment	Easting	Northing	Time since treatment, prior instalation	Installation date	2008 Equipment	Number of Collars	2012 April equipment A	No. of peizometers installed 2012	2012 April equipment B
Big moss (dry locality)	GS1	C.L-New	402140	373707	< 1 yr	May-08	Gas collars, open dip wells and runnoff traps	6	10 cm peizos	4	1 Rain water sampler
	GS3	C.L-Old	402052	373795	1 yr	May-08		3	10 cm peizos	2	
	Ben	C.L-Old	402013	374076	1 yr	Jun-08		6	10 cm peizos	2	
	BN	B-Old	402027	374012	1 yr	Jun-08		3	10 cm peizos	2	1 Rain water sampler
	BS	B-Old	402088	373969	1 yr	Jun-08		3	10 cm peizos	2	
	Pat	Cont	402052	373818	N/A	May-08		6	10 cm peizos	4	
	OB	B-New	402164	372600	< 1 yr	May-08		6	10 cm peizos	4	
Ravenslow (wet locality)	Pos 1	B-New	402075	371985	1 month	May-08		3	10 cm peizos	2	
	Pos 2	B-New	402165	372155	1 month	May-08		3	10 cm peizos	2	
	Nep	B-Old	402125	371850	1 < age < 5 yrs	May-08		6	10 cm peizos	4	
	Kra	Cont	402020	372170	N/A	May-08		6	10 cm peizos	4	1 Rain water sampler

Table 6.1: Goyt Valley sites equipment installed between 2008 and 2012.

6.3.2 Statistical methods

Statistical analytical methods are presented from plot to catchment scale in the following order: sample presence; DOC concentration by site and sample type (10DD vs. 1m depth dipwell), DOC concentration from head to reservoir, sample absorbance at 400nm. Refer to the thesis methodology (Chapter 2) for details on data quality check and GLM ANCOVA.

Site hydrology was investigated using water sample presence. Sample presence in the runoff and 10DD through-flow was analysed using Chi squared test as used by Clay *et al.* (2009b). There were four 10DD through-flow sample traps installed, and six runoff samples installed. The counts were calculated as a monthly percentage by samples type and month, and plotted along with WTD. In order to indicate the pathways in which water travels in each of the study sited, a bar chart of runoff and 10DD sample presence was used alongside WTD.

A factorial approach was implemented in this study to understand the impact of vegetation cutting and burning on peatlands water sample DOC concentrations, at site plot scale verses catchment scale. The study was designed to include a number of factors, including treatment site. The treatment sites used for this study were the same as those used in Chapter 5. To assess differences between sites, 1m depth soil pore water, 1DD through-flow, and runoff water DOC concentrations were analysed in addition to sample absorbance at 400 nm (abs_{400}). Analysis was conducted by wet verses dry burn sites, and on dry cuts verses burns, using ANOVA. Factors included in the ANOVA were 1) site locality with two levels, which represented the difference between the wet (Ravenslow) and dry heath sites (Big Moss); 2) site treatment, with eight levels including: new managed burn, old managed burn, fresh cut and leave, old cut and leave, old cut and lift, and a *C. vulgaris* control; 3) project month, which had 12 levels, January till December. Finally, plot was a factor nested within site.

Data were firstly analysed as absolute across sample type (soil pore water and runoff water samples), and were tested across all sites (in wet and dry sites), treatments (all eight levels) and months. Secondly, the 10DD through-flow water sample DOC concentration data were analysed by locality, relative to the *C. vulgaris* control.

Analysis of variance was conducted similar to that included in Chapters 3 - 5. The quality checked untransformed data were analysed first, after which logged and square root values of the data were investigated. The Anderson-Darling test was used to select which transformation if any was most appropriate. In addition to the use of the factors mentioned, covariates were also used to produce best fit data models. Covariates used in the soil ANCOVA included: water pH and conductivity, WTD, WTH, sample presence, surface soil temperature, rainfall, *C. vulgaris* height, *C. vulgaris* percentage plot cover, *Eriophorum* percentage plot cover, plots vegetation mass at 20cm^2, peat depth, bulk density, DOC and E_4/E_6. Pearson's correlation coefficient was used to find the direct relationship between the predictor and significant covariates; some were investigated further using regression plots.

The seasonal trends of relative difference in DOC concentration between 1m soil to 10DD through-flow water samples were analysed. This was conducted by calculating the mean monthly 10DD DOC sample concentration divided over the mean monthly 1m soil pore water DOC concentrations, then analysed using ANOVA by location and sampling month and year.

As in the previous chapters' statistical analysis methods (Chapter 3 and 4), models in which all inputs were significant (all significant differences are assessed at the 95% probability of not being zero) were accepted on the condition the residuals were 'normal'. These normal residuals were also prioritised over a model with the highest adjusted R^2 (referred to as R^2). The magnitude of the differences between the factors and covariates used in the ANOVA were calculated using ω^2. The differences between the levels of factors found to be significant were compared using the post hoc Tukey test. Results were expressed as least squares means (mean standard error is included) as these provided better estimates of the factor levels mean, as they took account of other factors, interactions and covariates included in the analysis.

6.4 Results

6.4.1 Treatment sites hydrology

6.4.1.1 Water table depth

Analysis of the burn wet vs. dry WTD April 2012- May 2013 data (Figure 6.4) (natural log transformed) using an ANOVA found that treatment site within locality was the most important ($p < 0.0001$; $\omega^2 = 29.95\%$) followed by between localities ($p < 0.0001$; $\omega^2 = 16.60\%$). Variation between the factor month was also significant ($p < 0.0001$; $\omega^2 = 5.82\%$). Residuals were normal. Post hoc analysis revealed that the wet locality had shallower WTD than the dry locality (as found across the five project years; see Chapter 5). Additionally, the sites with the shallowest WTD were the wet old and new burn in addition to the new cut.

The addition of covariates improved the R^2 from 51.62% to 54.67%, and found that the importance of treatment site within locality was reduced although it was still the most important factor ($p < 0.0001$; $\omega^2 = 23.94\%$). The importance of the factor localities was reduced ($p < 0.0001$; $\omega^2 = 5.61\%$) as was the factor month ($p < 0.0001$; $\omega^2 = 3.57\%$). The most important covariate was peat depth ($p < 0.0001$; $\omega^2 = 12.95\%$) followed by percentage of plot covered by heather ($p < 0.0001$; $\omega^2 = 7.28\%$). Monthly rainfall measured at the study sites was significant although of low importance ($p \leq 0.000$; $\omega^2 = 1.28\%$). Post hoc analysis found that the deepest WTD were found in the dry control (-44.19 ± 7.77 mm), and the wet control (-41.03 ± 7.79 mm). These two controls did not differ to the dry old burn (-36.75 ± 7.77 mm). The old cut (-35.49 ± 7.77mm), dry new burn (-33.01 ± 7.77 mm), and wet old burn (-36.82 ± 1.07 mm) all had shallower WTD than the dry heather control, but deeper WTD than the wet new burn (-17.98 ± 7.78 mm). The old cut had deeper WTD than the new cut (-21.05 ± 7.76 mm).

6.4.1.2 Sample presence

Sample presence in the runoff and 10DD traps were converted to a percentage of the traps present (Figure 6.3). Samples presence in the runoff and 10DD traps was variable throughout the 14 month monitoring period (Figure 6.4), notably fewer samples were intercepted at the dry locality control sites. Using a Chi squared test, it was found that there was a significant difference in sample presence between the runoff and 10DD traps in the dry locality ($p < 0.0001$) and in the wet locality ($p < 0.0001$). It is worth noting that of all the treatment sites the new burn site generally had the greatest counts of runoff in both the dry (10DD = 10% < R) and wet locality (10DD = 1% > R) (Figure 6.3). The control sites had lower runoff in comparison to the other sites both in the wet locality and particularly at the dry locality. However, they had higher counts of 10DD samples than runoff and on the wet locality in particular they had the highest counts of 10DD.

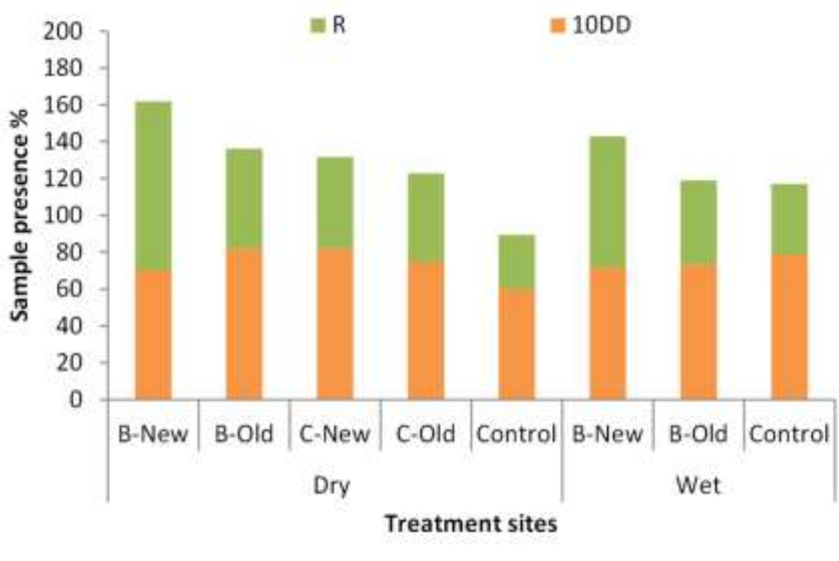


Figure 6.3: Goyt Valley intercepted sample presence by sample type (green bars, runoff; orange bars, 10DD) and treatment sites in both the wet and dry localities (April 2012 – May 2013).

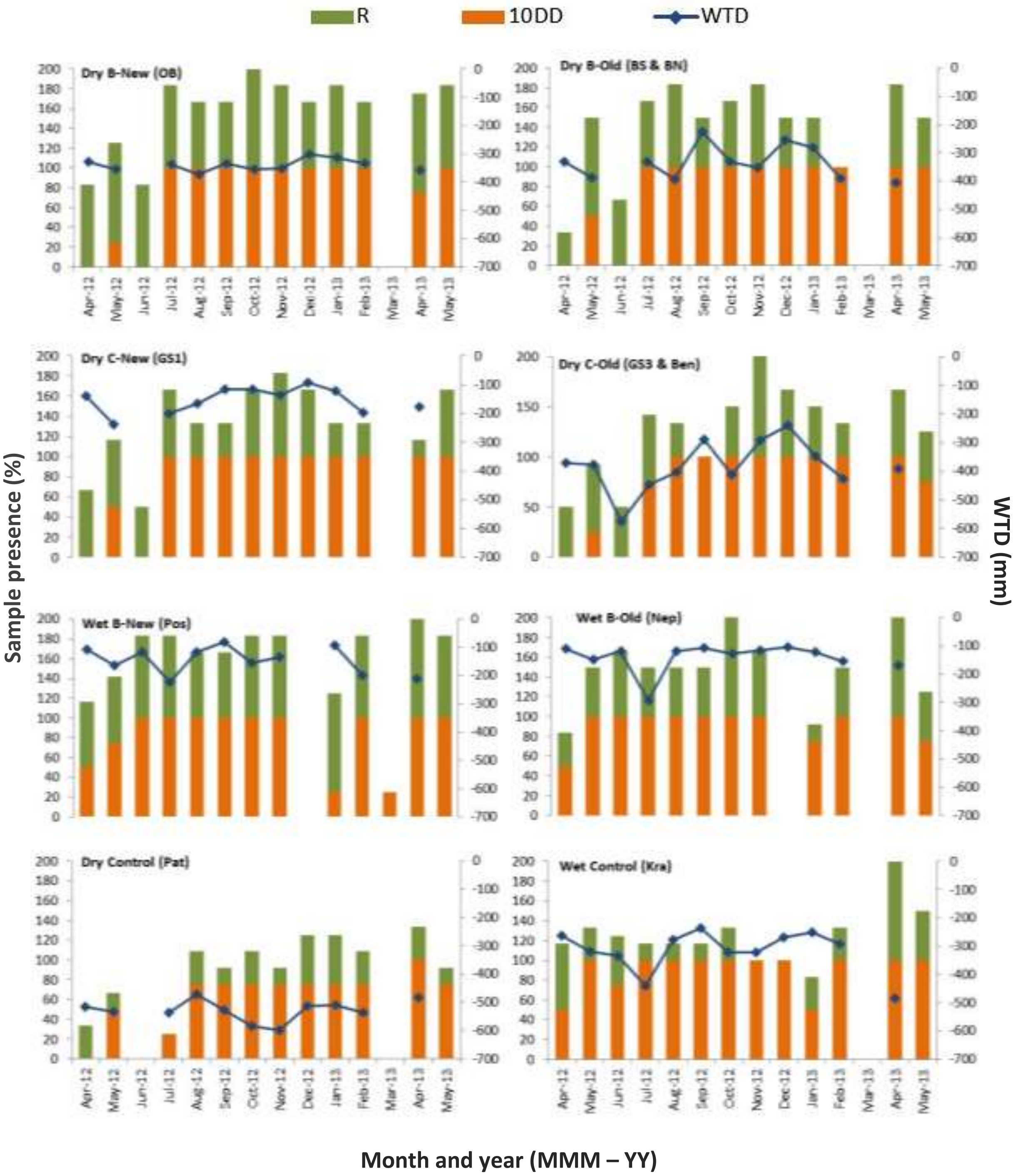


Figure 6.4: Water table (blue diamond) measured from dipwells and presence of sample as a percentage for both runoff sample (green bar) and 10DD through-flow (orange bar). Gaps in the data in December 2012 and March 2013 were due to heavy snow.

6.4.2 Treatment sites water sample DOC concentrations (Runoff, 10DD through-flow & soil pore water)

Analysis of DOC concentrations using ANOVA found that treatment site nested within locality was the most important factor ($p < 0.0001$; $\omega^2 = 5.43\%$), which was of similar importance to variation between months ($p < 0.0001$; $\omega^2 = 5.15\%$). Variation between water treatment by water samples was significant ($p < 0.0001$; $\omega^2 = 3.58\%$). Variation between locality was significant although of lesser importance ($p < 0.0001$; $\omega^2 = 1.31\%$). Post hoc analysis revealed that the 10DD sample water DOC concentration were greater than the 1 m soil pore water and the runoff samples.

The addition of covariates increased the ANOVA from R^2 15.48% to 24.31%. The addition increased the importance of DOC concentration variation between DOC sample type ($p < 0.0001$; $\omega^2 = 6.38\%$). Variation between month was significant ($p < 0.0001$; $\omega^2 = 5.15\%$) and more important than the treatment site nested within locality ($p \leq 0.003$; $\omega^2 = 3.08\%$). The log transformed sample conductivity was significant ($p < 0.0001$; $\omega^2 = 4.40\%$). The plot vegetation mass was significant ($p \leq 0.008$; $\omega^2 = 2.95\%$) as was *Eriophorum* plot cover dominance ($p \leq 0.002$; $\omega^2 = 0.80\%$). The DOC concentrations across the treatment sites positively correlated to both vegetation (Pearson's correlation 0.136, $p < 0.0001$) and natural log transformed conductivity (Pearson's correlation 0.213, $p < 0.0001$). It also negatively correlated to *Eriophorum* plot cover (Pearson's correlation -0.123, $p < 0.0001$). Post hoc analysis revealed that the addition of covariates explained greater difference in DOC concentrations between water sample types. The 10DD samples had the greatest concentrations (117.98 ± 0.25 mg C/l), followed by 1 m soil pore water (99.72 ± 0.25 mg C/l) (Figure 6.9). The lowest concentrations were for the runoff samples (68.77 ± 0.25 mg C/l). The control sites did not differ to the treatment sites. The wet new burn (84.75 ± 0.36 mg C/l) and old burn (82.97 ± 0.30 mg C/l) had lowest DOC concentration as did the dry new burn (81.83 ± 0.33 mg C/l).

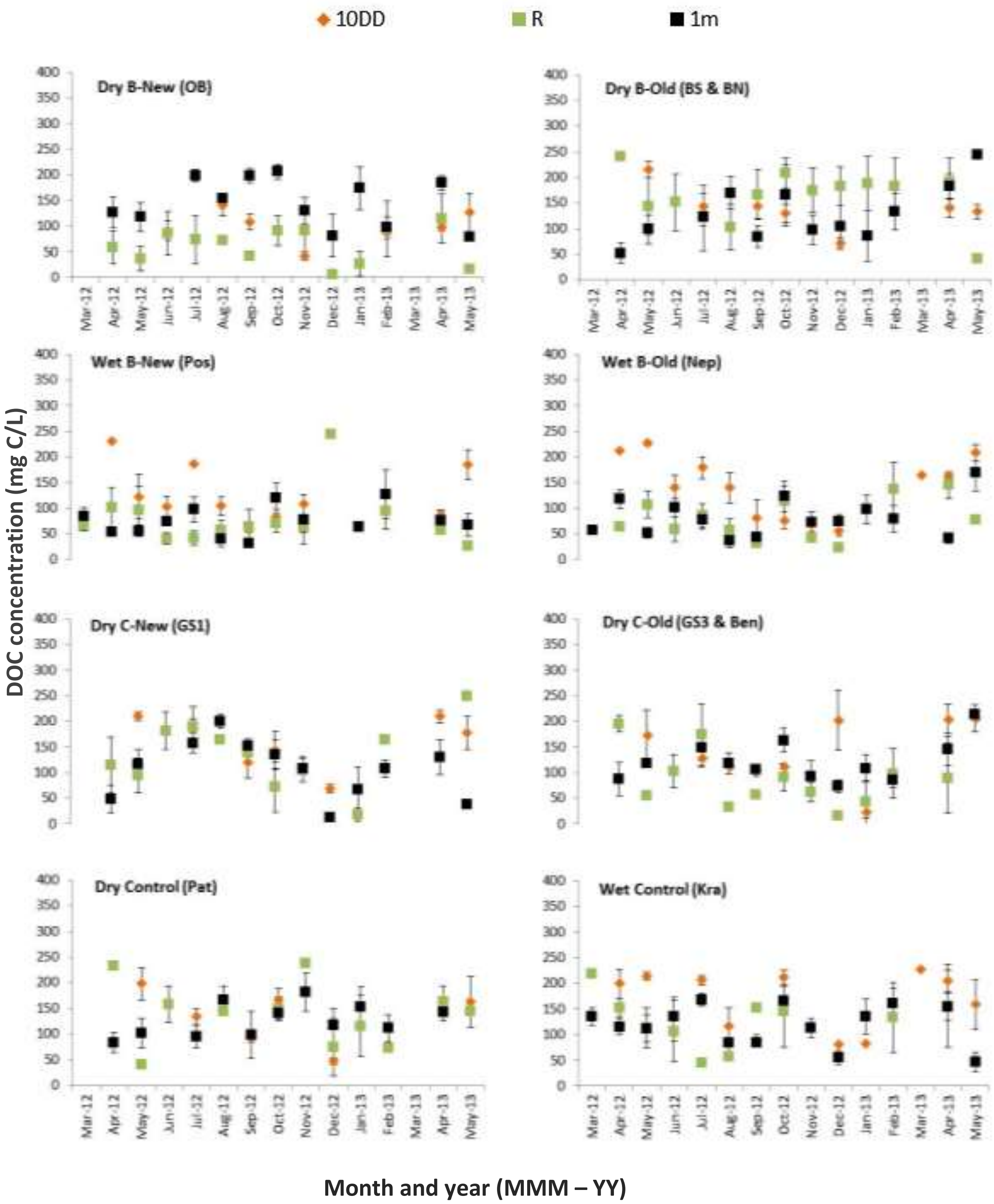


Figure 6.5: Plots of Goyt Valley sites mean DOC concentrations and SE mean (error bars), in both the wet and dry locality (April 2012 – May 2013). The data are plotted by site, water sample type (orange diamonds, 10DD; green square, runoff; black square, 1m soil pore water) and month and year.

The concentrations in DOC fluctuated in between month (Figure 6.5), notably high during warmer months. The concentration of DOC monthly fluctuation at the newly cut site had similar trends within 1m soil pore water as in the 10DD through-flow and runoff water. The ratio of mean 10DD to 1m DOC concentration were calculated and analysed using an ANOVA (R^2 31.10%). It was found that there was variation between month ($p \leq 0.005$; $\omega^2 = 22.20\%$) as was between locality ($p \leq 0.004$; $\omega^2 = 8.58\%$). Post hoc analysis revealed that the mean ratio of 10DD sample DOC concentrations to 1m soil pore water was greater at the wet locality (1.44 ± 0.11) than at the dry (0.94 ± 0.11) (Figure 6.6). Furthermore, there was a greater difference between 10DD and 1m soil pore water DOC concentrations during the month of May (2012 - 2.21 ± 0.20 and 2013 - 1.77 ± 0.20) than in January 2013 (0.36 ± 0.38). January was the month where the concentration of DOC within 1m soil pore water samples were higher than 10DD through-flow samples (Figure 6.7).

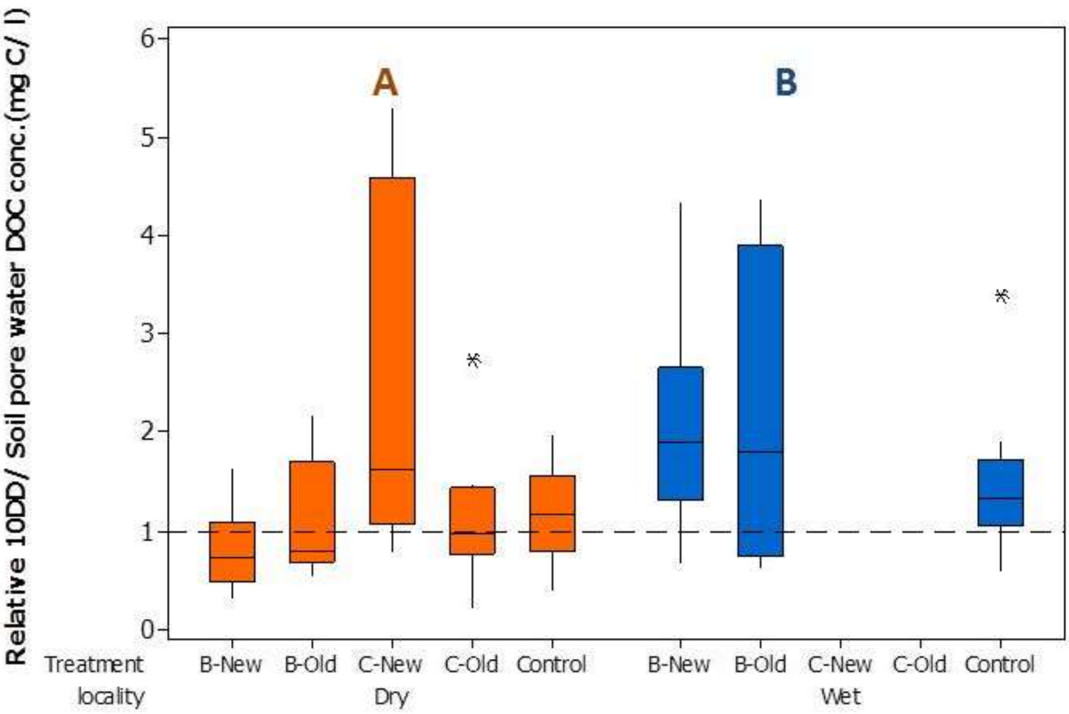


Figure 6.6: Boxplot of difference relative DOC concentration 10DD/1m. The letters signify post hoc results for difference between the factor locality (wet or dry). The dashed line represents a 1:1 ratio of DOC concentration in 10DD and soil pore water samples. Note: there are only three wet site (blue) box whiskers as there were no cut sites in the wet locality

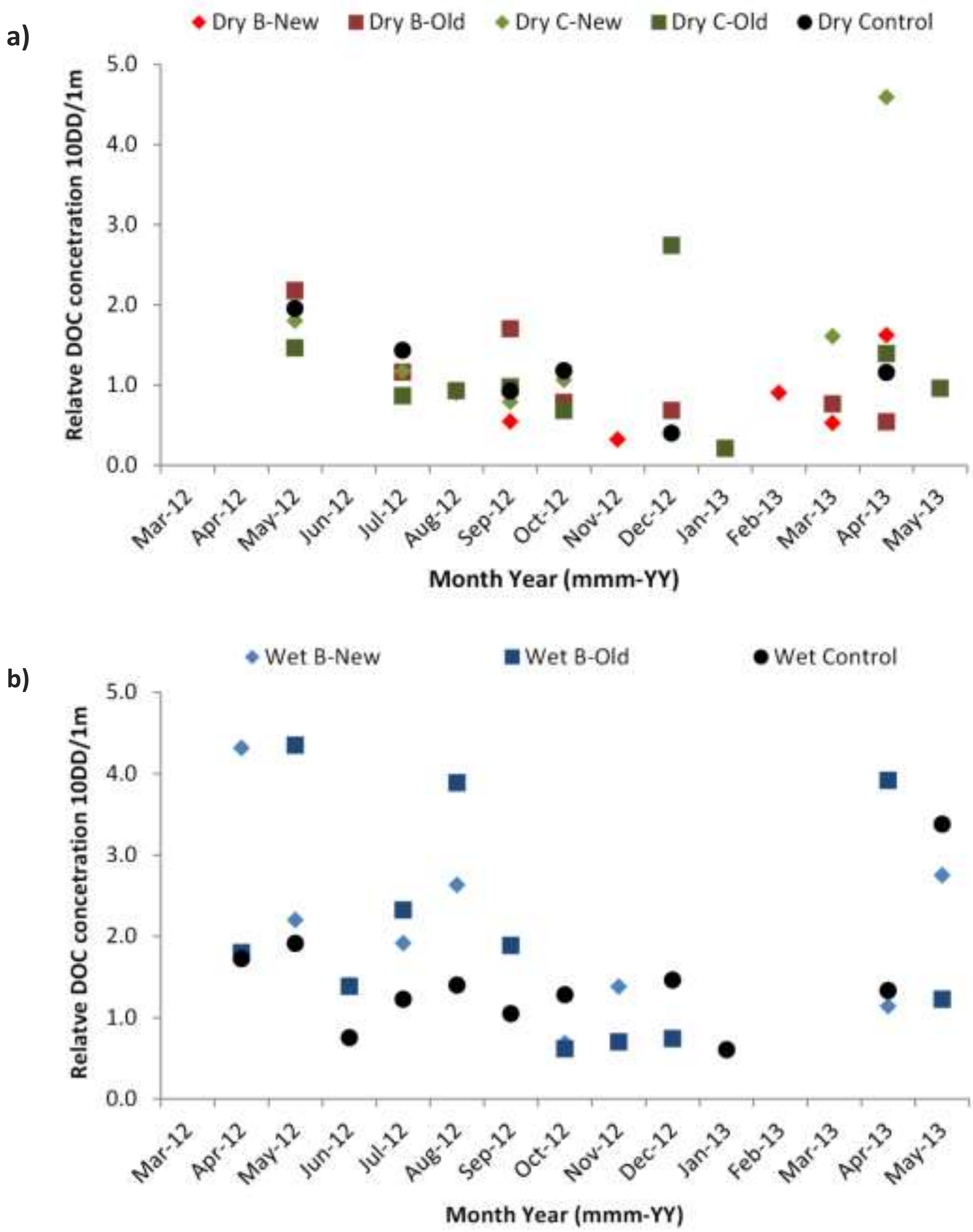


Figure 6.7: Plot of monthly DOC concentration in 10DD through-flow water vs. across 1m dip well soil pore water samples. a) Burn vs. cut sites (red diamond, new burn; brown square, old burn; green diamond, new cut, green square, old cut; and black circle, control). b) Burn wet locality (blue diamond, new bur; blue square, old burn, and black circle, control).

6.4.3 DOC concentrations (10DD through-flow)

6.4.3.1 Cut vs. burn (10DD through-flow)

For the analysis of the DOC concentrations in 10DD water for the dry localities cut vs. burn sites the data were square root transformed. It was found using an ANOVA (R^2 37.27%) that variation between the factor month was more important ($p < 0.0001$; $\omega^2 = 30.46\%$) than variation between treatments sites ($p \leq 0.002$; $\omega^2 = 6.63\%$). There were no significant covariates. Post hoc analysis revealed that the control sites (107.04 ± 0.52 mg C/l) did not differ between the treatment sites. Furthermore the cut sites, both new (126.2 ± 0.47 mg C/l) and old (120.2 ± 0.47 mg C/l), had higher DOC concentrations than the new burn (77.3 ± 0.52 mg C/l), the old burn did not significantly differ to the other sites (107.89 ± 0.52 mg C/l) (

Figure 6.8.a). The winter months December and January had lower DOC concentrations than April, May, July, September and November. December and January did not differ from March, June or August (Figure 6.9).

6.4.3.2 Burn on wet *vs.* dry peat (10DD through-flow)

Analysis of square root transformed DOC concentration of the 10DD water samples at the burns in the wet vs. dry locality was conducted using ANOVA (R^2 34.96%). Variation between the factor month remained the most important factor ($p < 0.0001$; $\omega^2 = 28.13\%$). Variation between treatments sites within the locality was significant ($p \leq 0.001$; $\omega^2 = 5.41\%$) and more important than variation between locality ($p \leq 0.000$; $\omega^2 = 1.30\%$). Post hoc analysis revealed that the wet control sites (132.4 ± 0.49 mg C/l) did not differ to the dry control. However, the wet control 10DD DOC concentrations were higher than both the wet new burn (94.6 ± 0.48 mg C/l) and dry new burn (84.14 ± 0.53 mg C/l) (Figure 6.8.b).The wet old burn (120.8 ± 0.45 mg C/l) did not differ from the dry old burn (117.8 ± 0.52 mg C/l). However, it did have higher DOC concentrations than the dry new burn. There were no significant covariates.

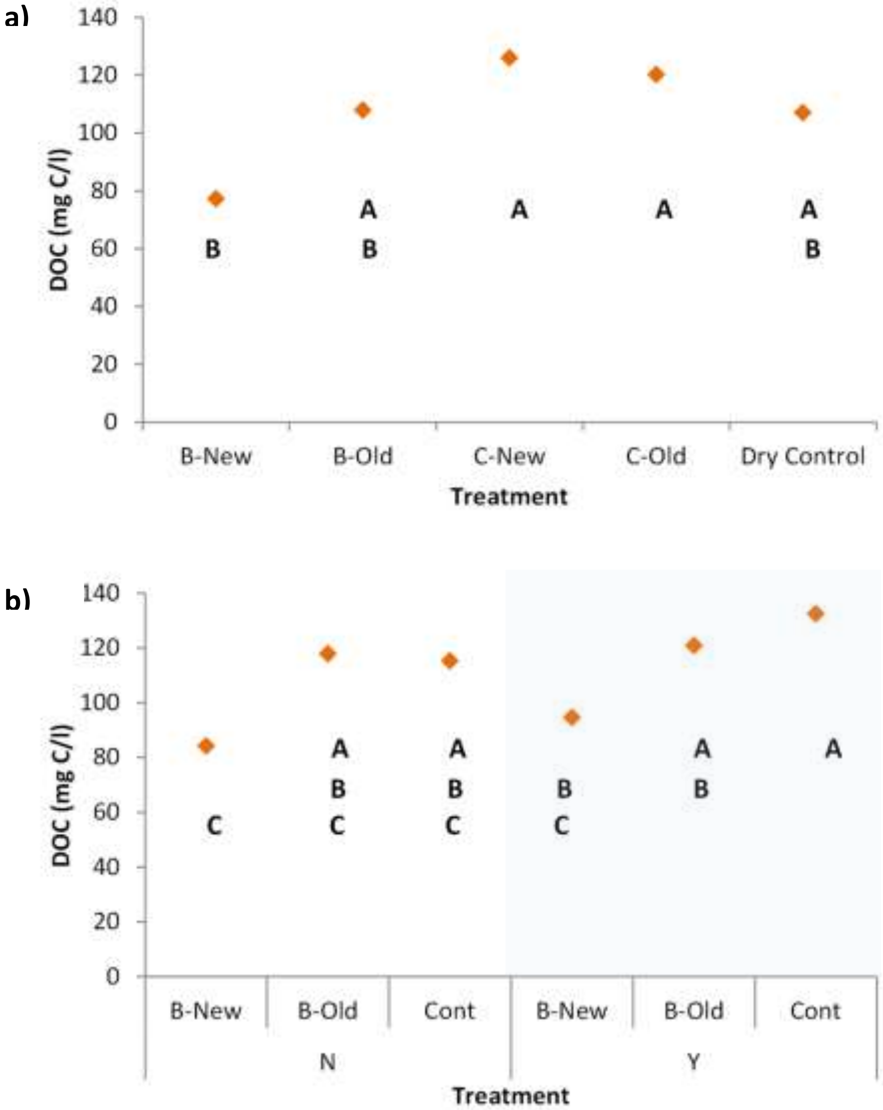


Figure 6.8: Main effects plots of 10DD through-flow DOC concentrations ANOVA by the factor site. a) Burn vs. cut study, b) Burn in wet vs. dry study. Letters on the plots represent post hoc analysis results. The different letters indicate where significant differences between the levels lie.

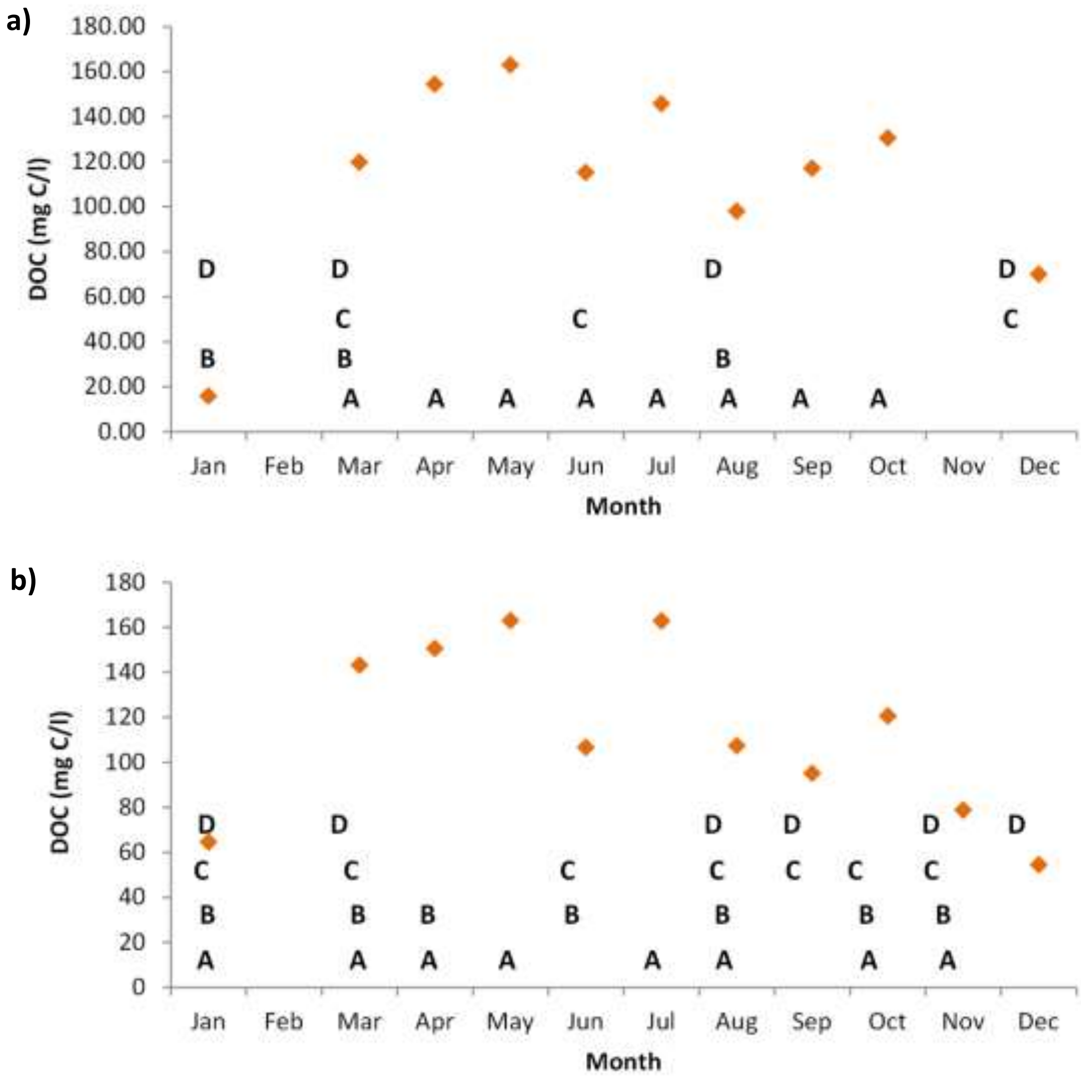


Figure 6.9: Main effects plot of 10DD through-flow DOC concentrations ANOVA by the factor month. a) Burn vs. cut sites. b) Burn sites in wet vs. dry localities. Letters on the plots represent post hoc analysis results. The different letters indicate where significant differences between the levels lie.

6.4.4 10DD through-flow DOC concentrations (relative to locality control)

6.4.4.1 Cut vs. burn - DOC relative to dry locality control (10DD through-flow)

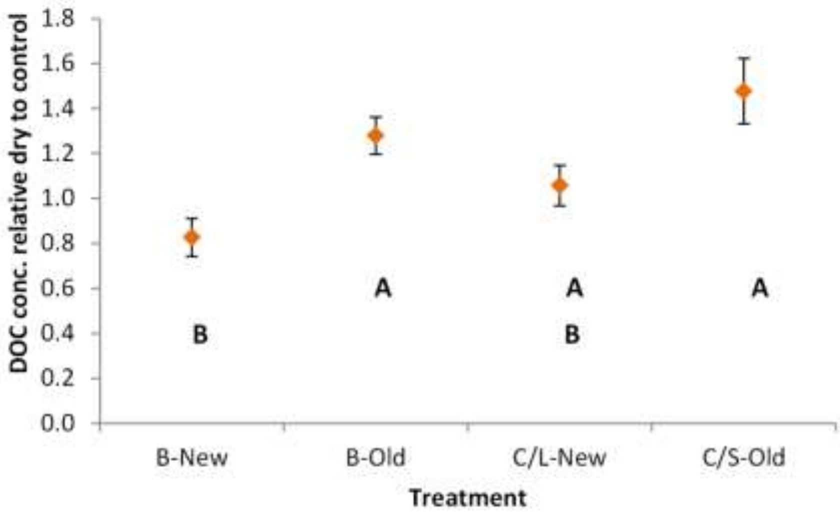
Analysis of the 10DD water (square root transformed DOC concentration) in the dry localities cut vs. burn sites were conducted relative to the dry control. It was found that month was the most important factor ($p < 0.0001$; $\omega^2 = 16.72\%$). Treatment was a significant factor ($p < 0.0001$; $\omega^2 = 4.45\%$) and there was also a significant interaction between treatment and month ($p < 0.0001$; $\omega^2 = 13.68\%$). Post hoc analysis indicated that only the new burn site had lower 10DD through-flow water sample DOC concentrations than the dry locality control (0.83 ± 0.08 mg C/l) (Figure 6.10.a). The new cut (1.06 ± 0.08 mg C/l) DOC concentration did not differ to new burn. However, the new cut, old cut (1.48 ± 0.08 mg C/l) and old burn (1.28 ± 0.08 mg C/l) had relatively higher DOC 10DD through-flow water concentration than the control site. No covariates were significant.

6.4.4.2 Burn DOC - relative to wet locality control (10DD through-flow)

Analysis of the 10DD water (square root transformed DOC concentration) at burn treatment sites relative to the wet localities. It was found using an ANOVA that variation between the factor month was more important ($p < 0.0001$; $\omega^2 = 41.39\%$) than variation between treatments sites ($p \leq 0.002$; $\omega^2 = 5.04\%$). Post hoc analysis revealed that the DOC concentration within site 10DD through-flow at the old burn (1.13 ± 0.17 mg C/l) had higher DOC concentration and were higher than the wet locality, while the new burn had lower DOC concentrations than the control (0.80± 0.14 mg C/l). There were no significant covariates.

The addition of covariates improved the R^2 from 46.43% to 57.81%. Variation between month increased in importance, however the importance of treatment site was reduced. The remaining variation was explained by plot vegetation mass ($p \leq 0.011$; $\omega^2 = 2.51\%$). Log transformed E_4/E_6 was also significant although relatively unimportant in the model ($p \leq 0.003$; $\omega^2 < 0.00\%$). A positive correlation was found between vegetation and square root natural DOC concentrations (Pearson correlation = 0.108, $p \leq 0.004$). Post hoc analysis revealed greater differences between the sites. The DOC concentrations within site 10DD through-flow at the old burn (1.48 ± 0.17 mg C/l) were greater than the site control, and the new burn (0.45 ± 0.14 mg C/l) was lower than the control (Figure 6.10.b). There were no significant covariates.

a)



b)

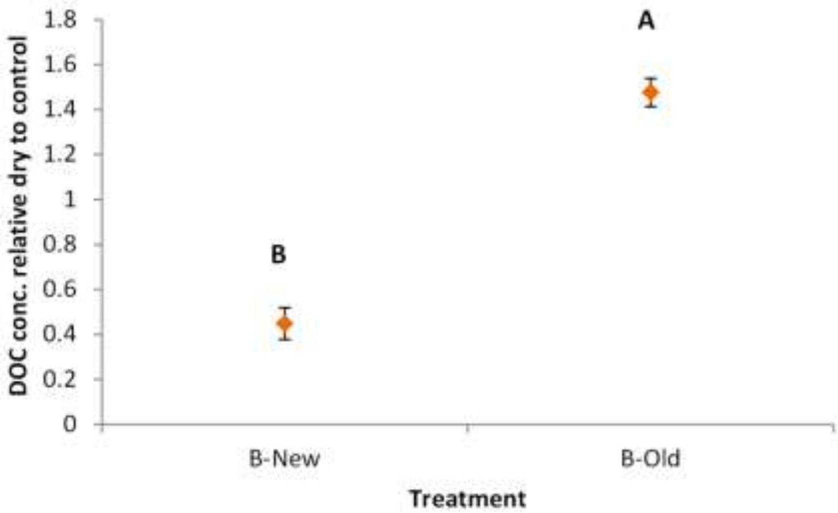


Figure 6.10: Main effects plot of 10DD through-flow DOC concentrations, relative to the locality control ANOVA by the factor site. a) Burn vs. cut sites. b) Burn sites in wet vs. dry localities. Letters on the plots represent post hoc analysis results. The different letters indicate where significant differences between the levels.

6.4.5 Goyt valley water sample DOC concentrations (head to reservoir)

Goyt valley water samples (from head to reservoir) DOC concentrations (square root transformed) between April 2012 and May 2013 were analysed using ANOVA. Sample type was found to be more important ($p < 0.0001$; $\omega^2 = 30.99\%$) than the factor month ($p < 0.0001$; $\omega^2 = 3.53\%$) in explain DOC variation. Post hoc analysis determined that sample DOC concentration collected at plots (runoff water, though flow and soil pore water) were higher than those from rainwater, stream water and reservoir.

The addition of covariates improved the R^2 from 34.54% to 41.61%. The importance of variation sample type increased ($p < 0.0001$; $\omega^2 = 34.60\%$) as did monthly variation ($p < 0.0001$; $\omega^2 = 4.49\%$). The covariate water sample conductivity was significant ($p < 0.004$; $\omega^2 = 2.44\%$) and more important than mean monthly temperature ($p \leq 0.004$; ω2 = 0.05%). Post hoc analysis revealed (Figure 6.11) that the samples sourced for the treatment sites had higher DOC concentrations than the reservoir, stream and rainwater. The DOC concentrations were greatest in 10DD samples (121.75 ± 0.27 mg C/L), followed by soil pore water (98.82 ± 0.17mg C/L) and runoff soil pore (61.72 ± 0.25 mg C/L). Rainwater DOC concentration were lowest (14.88 ± 0.25 mg C/L), but did not significantly differ to the reservoir (16.23 ± 0.58 mg C/L). Reservoir sample DOC concentrations did not differ to the stream water (30.43 ± 0.40 mg C/L). Observations of sample DOC concentration fluctuation (Figure 6.12), indicate that there are similarities between stream and reservoir water DOC concentrations (as found in the ANOVA). Furthermore in 2012 both stream and reservoir water samples DOC concentrations peaked during early summer (June), late autumn (October) and again in June 2013. This monthly DOC variation followed a similar peaks trend as 10DD samples, although there was a lag in the peak concentrations, whereby peak DOC productions at 10DD occurred during June (Figure 6.9.a and b), months earlier than at the reservoir observed in October and (Figure 6.12).

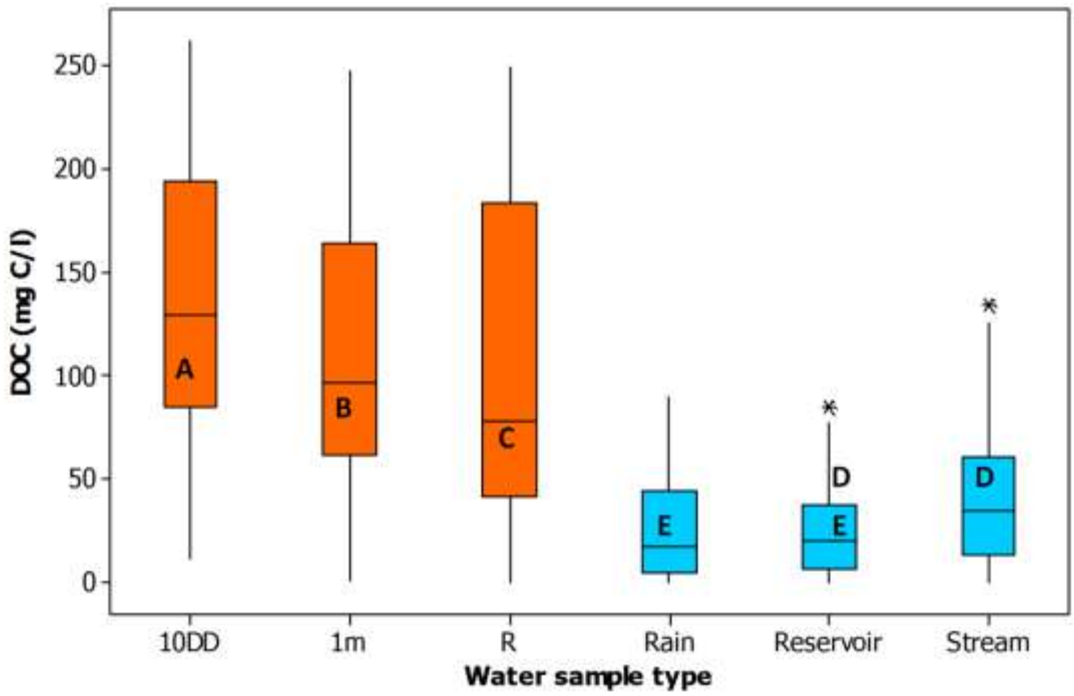


Figure 6.11: Box plot of sample DOC concentration by water sample source type. The letters signify the post hoc test results. Different letters indicate significant differences between sample type factor levels.

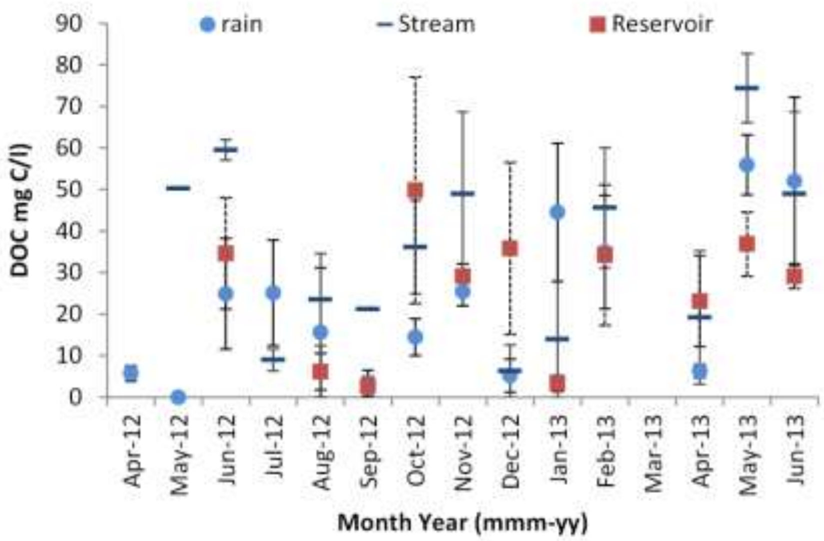


Figure 6.12: Plot of mean DOC concentration in rainwater (blue circles), stream water (blue dash) and reservoir (brown square). Error bars represent standard error of the mean.

6.4.6 Water sample absorbance at abs_{400} (head to reservoir)

Goyt valley water sample absorbance (natural log transformed) between April 2012 and May 2013 were analysed using ANOVA. Sample type was found to be more important ($p < 0.0001$; $\omega^2 = 50.93$ %) than the factor month ($p < 0.0001$; $\omega^2 = 5.74\%$). Post hoc analysis determined that sample DOC concentration collected at plots (runoff water, though-flow and soil pore water) were higher than those from rainwater, stream water and reservoir although the runoff samples did not significantly differ to the rainfall.

Addition of covariates improved the R^2 from 56.69% to 60.80%. The variation of abs_{400} by the factor sample type remained of similar importance ($p < 0.0001$; $\omega^2 = 34.60\%$) as did the month ($p < 0.0001$; $\omega^2 = 6.02\%$). The covariate monthly total rainfall was significant ($p \leq 0.031$; $\omega^2 = 3.83\%$) and more important than sample pH, which was significant but of little importance ($p \leq 0.004$; $\omega^2 = 0.05\%$). A negative correlation was found between abs_{400} and sample pH (Pearson correlation = -0.115, $p < 0.0001$), while there was a small positive correlation with total rainfall (Pearson correlation = 0.084, $p < 0.0001$). Post hoc analysis (Figure 6.13) indicated that the highest mean abs_{400} was measured at through-flow 10DD sample (0.169 ± 0.07) and soil pore water sample across 1m depth (0.182 ± 0.07). They were both higher than the runoff water sample (0.169 ± 0.07), which did not differ to the stream water (0.119 ± 0.07) and reservoir (0.142 ± 0.08). Rainfall had the lowest abs_{400} (0.106 ± 0.07).

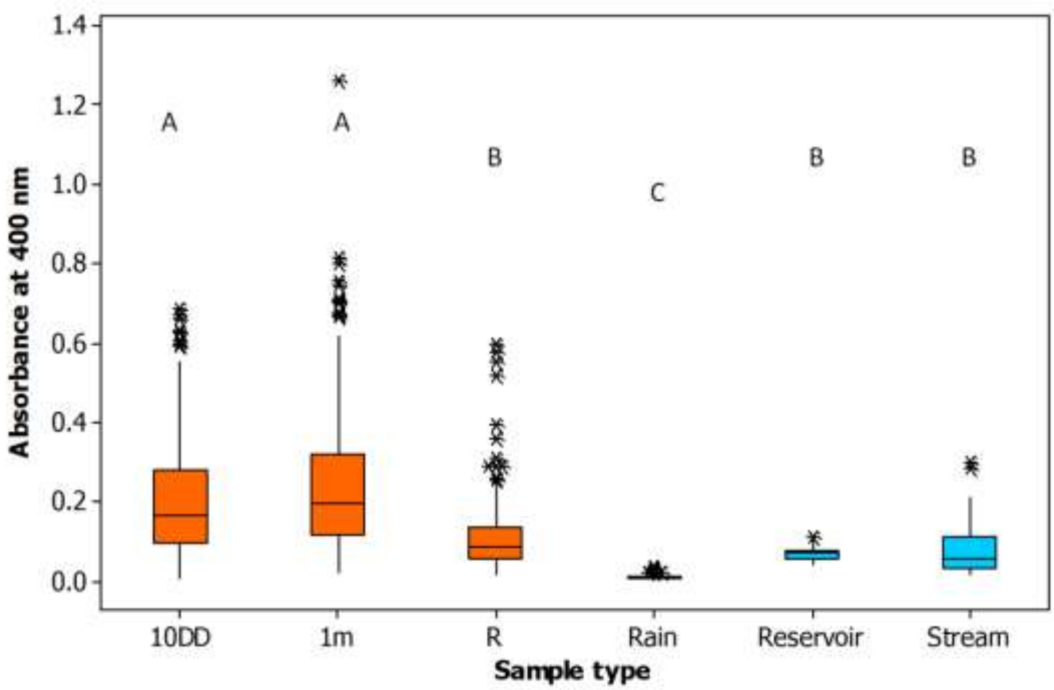


Figure 6.13: Box plot of sample absorbance at 400 nm by water sample source type (orange box plot, samples from treatment sites; blue box plot, rainfall, reservoir and stream water). The letters signify the post hoc test results. Different letters indicate significant differences between sample type factor levels.

6.4.7 Water sample absorbance at abs_{400} (10DD through-flow)

Treatment sites though-flow at 10DD water sample absorbance data (natural log transformed) were analysed using ANOVA. Variation between month was found to be the most important factor ($p < 0.0001$; $\omega^2 = 27.38$ %). The treatment site within locality ($p < 0.0001$; $\omega^2 = 14.81\%$) was more important than locality ($p \leq 0.000$; $\omega^2 = 1.31\%$). Post hoc analysis determined that the dry old burn and new cut had higher mean 10DD abs_{400} than the dry heather control, dry new burn, and wet new and old burn. Only the dry new burn had lower mean 10DD abs_{400} than the wet and dry heather controls.

The addition of covariates improved the R^2 from 43.58% to 50.90%. Treatment sites though-flow at 10DD water sample absorbance data (natural log transformed) were analysed using ANOVA. Variation between month was found to be the most important factor ($p < 0.0001$; $\omega^2 = 27.87$ %). The treatment site within locality ($p < 0.0001$; $\omega^2 = 16.29\%$) was more important than locality ($p \leq 0.722$; $\omega^2 = 2.35\%$). Heather plot cover percentage was a significant covariate ($p \leq 0.002$; $\omega^2 = 0.46\%$) as was *the Eriophorum* plot cover, which was of little importance ($p \leq 0.046$; $\omega^2 < 0.00\%$). Post hoc analysis determined that the wet heather control mean abs_{400} (0.161 ± 0.14) did not differ to the heather control (0.092 ± 0.14) (Figure 6.14). The dry new burn had the lowest abs_{400} (0.041 ± 0.17), lower than the wet new burn (0.092 ± 0.14). The old dry burn (0.248 ± 0.12) had higher absorbance than the wet old burn (0.108 ± 0.12). The new cut (0.294 ± 0.13) did not differ to the old cut (0.184 ± 0.13) or old burn. Furthermore, abs_{400} was significantly higher during June and July (0.216 ± 0.13 and 0.249 ± 0.14) than in November (0.116 ± 0.14) and December (0.028 ± 0.15) (Figure 6.15). December had the lowest abs_{400} than every other month. No difference was found in mean abs_{400} between the wet and dry locality.

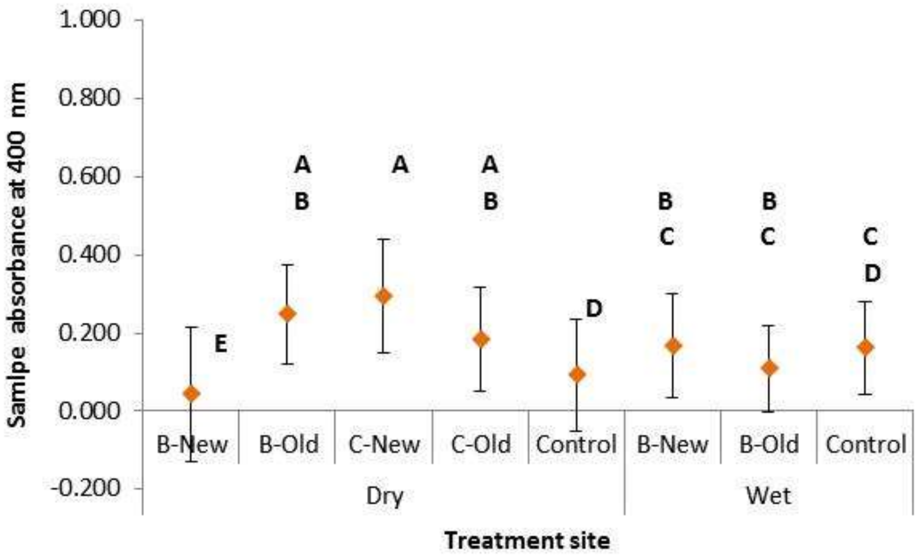


Figure 6.14: Water sample abs400 main effects plot by site within locality. The letters signify the post hoc test results. Different letters indicate significant differences between factor levels.

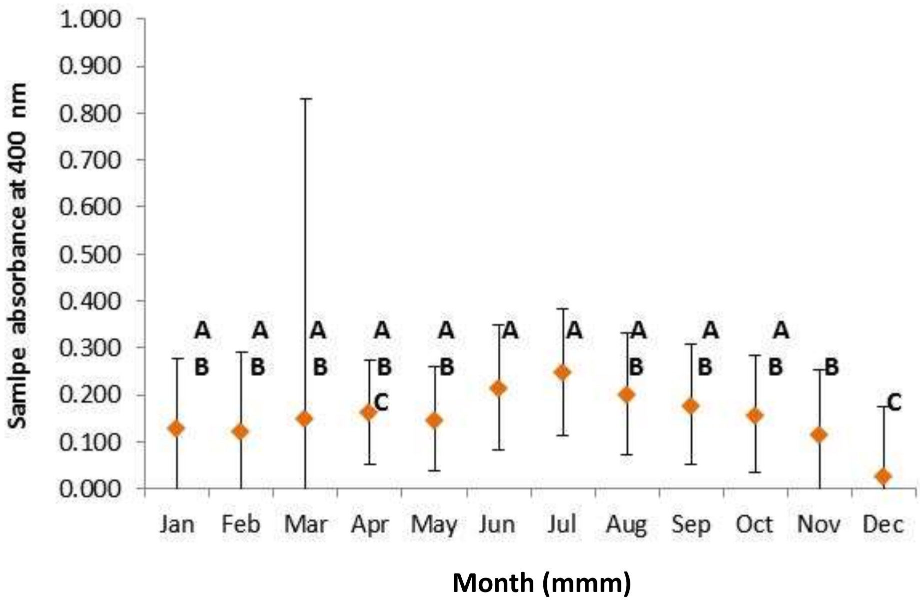
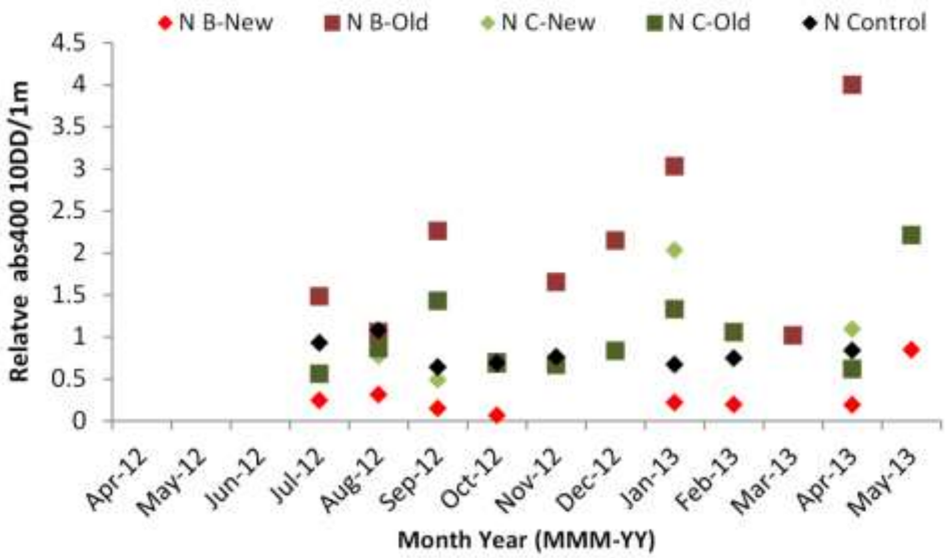


Figure 6.15: water sample abs_{400} main effects plot by treatment site within locality. The letters signify the post hoc test results. Different letters indicate significant differences between factor levels. Note: large error bars in March are due to it being the first month of the study and is indicative of site disturbance.

a)



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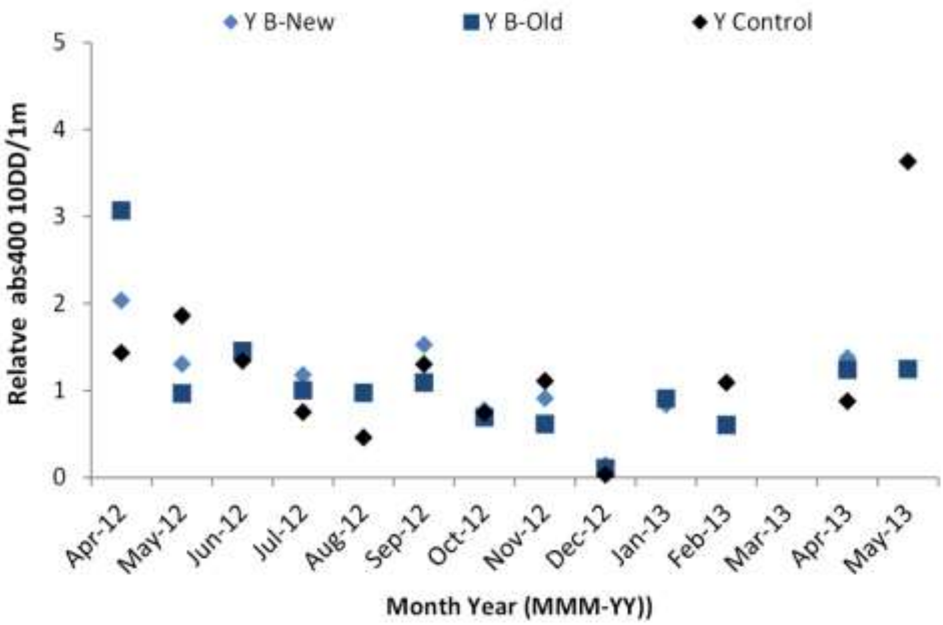

Figure 6.16: Plot of monthly abs_{400} *concentration in 10DD through-flow water vs. across 1m soil pore water samples. a) Burn vs. cut sites (red diamond, new burn; brown square, old burn; green diamond, new cut; green square, old cut; and black circle, control). b) Burn wet locality (blue diamond, new burn; blue square, old burn; and black circle, control).*

Analysis of the ratio between 10DD and 1m soil pore water abs_{400} using (Figure 6.16) an ANOVA (R^2 59.52%) found that monthly variation was the most important factor ($p < 0.0001$; $\omega^2 < 33.83$ %). This was followed by treatment sites within locality ($p < 0.0001$; $\omega^2 < 25.38$ %), although locality was not a significant factor ($p \leq 0.837$; $\omega^2 < 0.00$%). Post hoc analysis determined that abs_{400} at 10DD was higher than at the 1m soil pore water samples (Figure 6.17) at every treatment site except for the dry new burn (0.323 ± 0.18), where the ratio of absorbance at 10DD samples was lower than a 1m. The wet new burn did not differ to the new cut (1.153 ± 0.23) or dry control (1.034 ± 0.14). Both the dry and wet control sites (1.231 ± 0.14) had higher abs at 10DD than at 1m soil pore water. The site in which the ratio was greatest was the old burn (2.061 ± 0.17). Furthermore, this ratio was greater during May 2013 and April 2012 than during summer/autumn 2012 (July, August, November, October, December) and April 2013.

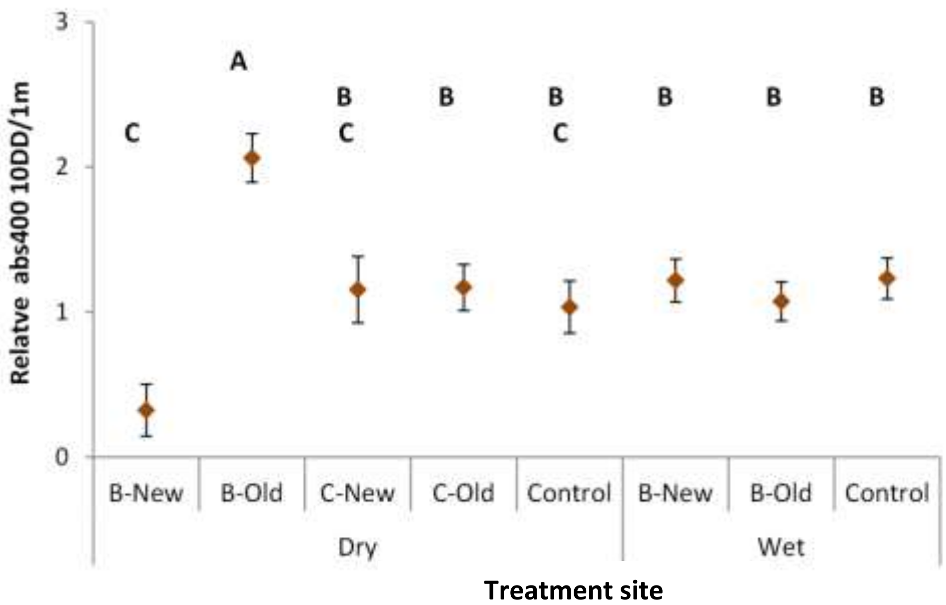

Figure 6.17: Plot of monthly abs_{400} concentration in 10DD through-flow water vs. across 1m soil pore water samples. a) Burn vs. cut sites (red diamond, new burn; brown square, old burn; green diamond, new cut; green square, old cut; and black circle = control). b) Burn wet locality (blue diamond, new burn; blue square, old burn; and black circle, control).

Response variable	Test	Factor (F)/ covariate (C)/ interaction (I)/ nested factor (N)	Variable	P	ω^2	adjR^2
WTD	ANOVA	F	Wet?	< 0.0001	16.60	51.62%
		F	Month	< 0.0001	4.31	
		N	Treatment site (Wet?)	< 0.0001	63.42	
	ANCOVA	F	Wet?	< 0.0001	16.60	54.67%
		N	Treatment site (Wet?)	< 0.0001	23.29	
		F	Month	< 0.0001	3.57	
		C	Peat depth	< 0.0001	12.95	
		C	Plot-Heather cover %	< 0.0001	7.85	
		C	Monthly rainfall (mm)	< 0.0001	1.28	
Head to reservoir DOC concentrations	ANOVA	F	Sample type	< 0.0001	30.99	34.54%
		F	Month	< 0.0001	3.53	
	ANCOVA	F	Sample type	< 0.0001	34.60	41.61%
		F	Month	< 0.0001	4.49	
		C	Month C	0.004	0.05	
		C	Conductivity	0.001	2.44	
Treatment site DOCconcentrations	ANOVA	F	Wet?	< 0.0001	1.31	15.48%
		N	Treatment site (Wet?)	< 0.0001	5.43	
		F	Month	< 0.0001	5.15	
		F	Sample type	< 0.0001	3.58	
	ANCOVA	F	Month	< 0.0001	5.47	24.31%
		F	Wet?	0.003	1.19	
		N	Treatment site (Wet?)	0.003	3.08	
		F	Water sample type	< 0.0001	6.38	
		C	ln Conductivity	< 0.0001	4.40	
		C	Plot vegetation mass at 20cm²	0.008	2.95	
		C	Plot-CG	0.002	0.80	

Table 6.2: Results summary table ANOVA and ANCOVA for: site WTD; head to reservoir DOC concentrations in the Goyt Valley (rainfall Runoff, 10DD through-flow, soil pore water, stream, reservoir) and site DOC concentrations (Runoff, 10DD through-flow, soil pore water) (April 2011 – May 2013).

Response variable	Test	Factor (F)/ covariate (C)/ interaction (I)/ nested factor (N)	Variable	P	ω^2	$adjR^2$
10DD DOC concentrations	ANOVA	F	Treatment site	0.002	6.63	37.27%
		F	Month	< 0.0001	30.46	
	ANCOVA	F	Wet?	0.172	1.30	34.96%
		F	Month	< 0.0001	28.13	
		N	Treatment (Wet?)	0.001	5.41	
10DD Dry locality - DOC relative to control	ANOVA	F	Treatment	< 0.0001	4.45	34.93%
		F	Month	< 0.0001	16.72	
		I	Treatment site*Month	< 0.0001	13.68	
	ANCOVA		N/A			
10DD wet locality - DOC relative to control	ANOVA	f	Month	< 0.0001	41.39	46.43%
		F	Treatment site	0.024	5.04	
	ANCOVA	f	Treatment site	0.001	0.86	57.81%
		F	Month	< 0.0001	54.73	
		V	ln E_4/E_6	0.003	< 0.001	
		V	Plot vegetation mass at 20cm²	0.011	2.51	
10DD/1m DOC		F	Wet?	0.004	8.58	31.10%
		F	Month	0.005	22.20	
	ANCOVA		N/A			
Head to reservoir abs_{400}	ANOVA	F	Sample type	< 0.0001	50.93	56.69%
		F	Month	< 0.0001	5.74	
	ANCOVA	F	Sample type	< 0.0001	50.94	60.80%
		F	Month	< 0.0001	6.02	
		C	Month(mm)	0.031	3.83	
		C	pH	0.010	< 0.001	
10DD abs_{400}	ANOVA	F	Wet?	0.300	1.31	43.58%
		N	Treatment site (Wet?)	< 0.0001	14.81	
		F	Month	< 0.0001	27.38	
	ANCOVA	F	Wet?	0.722	2.35	46.99%
		N	Treatment site (Wet?)	< 0.0001	16.29	
		F	Month	< 0.0001	27.87	
		C	Plot-Heather cover %	0.046	0.46	
		C	Plot-CG cover %	< 0.0001	< 0.001	

Table 6.3: Results summary table ANOVA and ANCOVA for: 10DD through-flow DOC concentration by locality relative to locality control; ratio of DOC concertation in 10DD through-flow vs. soil pore water; water sample absorbance at 400nm from head to reservoir in the Goyt Valley (rainfall Runoff, 10DD through-flow, soil pore water, stream, reservoir); and absorbance at 400nm in 10DD through-flow samples.

6.5 Discussion

In Chapter 5, it was found that catchment topography had a significant impact upon the soil pore water DOC concentrations across a 1m deep dipwell. The effect on site hydrology and DOC concentration in the root zone was investigated in order to constrain the effects of heather management on water quality in the Goyt. This chapter used a factorial designed experiment to link the findings on water quality at several sampling levels (runoff, 10DD through-flow and 1m soil pore water) at experimental treatment sites (cut, burn, control) in the Goyt, to findings on a catchment scale (rainfall and surface waters - stream and reservoir).

At the start of this short term study, the sites had already been monitored for around four project years. The new burn and new cuts were about four years of age, the old burn and old cuts were around five to eight years of age, while it would have been over 19 years since the control sites were burnt. Thus the findings and conclusions drawn from the results must keep the site treatment ages in mind when referring to new and old sites and considering the short term of the study. Analysis of water table depth found that in addition to new burn sites having shallower WTD among the treated sites after four years of treatment. The new burn sites also had a higher count of runoff water and 10DD through-flow samples intercepted, which is indicative of water flowing at the surface and through the 10cm of peat at the new burns. The control sites had the lowest water table depth in addition to fewest samples intercepted. Monthly rainfall volume measurements in the previous chapter found there was no significant difference in rainfall across the Goyt sites. It would therefore be expected that although the sites had low water table depth, there may be a similar amount of intercepted through-flow samples. The small sample size of intercepted runoff and through-flow indicated that the control site (particularly in the dry locality) was indeed drier than the treatment sites. According to Holden *et al.* (2014), hydraulic conductivity was significantly reduced in recent burns compared to unburnt sites. Thus, potentially higher hydraulic conductivity facilitates water movement within a site. However, this was not confirmed as hydraulic conductivity was not measured on the Goyt this. The lack of samples can be attributed to deeper water tables and reduced volume of through-flow. The deeper water table depth at the heather control is linked to the dominance of degenerative heather typical of older burn sites (McFerran *et al.* 1995). As explained in Chapter 5, sites may have had greater water uptake due to the greater dominance of heather which and explanatory variable

6.5.1 Water sample DOC concentrations (runoff, 10DD through-flow, soil pore water)

Concentration of DOC within surface runoff water, and 1m soil pore water were lower than 10DD through-flow. This follows the finding of Fraser et al. (2001) who documented a significant difference in DOC concentration at different peat depths. Treatment site DOC variation, as a mean of all site sample types, was linked to surface vegetation. Greater vegetation mass was associated with increased DOC concentrations, while increased *Eriophorum* plot cover dominance was associated with reduced DOC concentrations. This finding follows that of Armstrong *et al.* (2012). Water sample (runoff, 10DD through-flow, soil pore water across a 1m dipwell) conductivity (indicative of ionic strength) was found to correlate to sample DOC concentration. This is likely due to ionic strength influencing DOC solubility, as increased ionic strength correlate with a decrease in DOC concentrations (Scott *et al.* 1998). Month as a function of seasonality was found to be important in explaining DOC variation across all water sample types in the Goyt (from head to reservoir). Following the findings of the previous chapter on DOC (Chapter 3 and 5), temperature was an important covariate; higher soil temperatures increased exudation of DOC from roots (Uselman *et al.* 2000).

In contrast to the findings on 1m soil pore water DOC concentrations in Chapter 5, the 10DD DOC concentrations were not significantly influenced by peat depth. Variation in the

DOC was related to surface vegetation, particularly in comparison to the control site. Thus, the variation in 10DD DOC concentration had a greater dependence on surface vegetation treatment than the 1m soil pore water DOC concentrations. Therefore, the findings on 10DD through-flow DOC concetrations, were less masked by the variation of peat depth which was a locality dependent variable. When comparing site mean DOC concentrations, it was found that with the exception of the new burn there was little difference between the treatment site DOC concentrations.

When comparing the 10DD samples within sites to each other it was found that the most important factor was monthly variation. Treatment site was also important, although to a lesser extent. Analysis indicated the peak DOC production in the top 10cm of peat was during the summer months of April until July. There was much overlap in the mean DOC concentration between sites. In both the dry and wet localities, the new burn had lower DOC concentrations than the control, thus new burns are preferential to the control in both wet and dry peat conditions. The new cut DOC concentration did not differ to the old cut 10DD concentrations. This is likely a result of *Eriopherium* dominance at the old cut sites, wet new burn and dry old burn.

After treatment of a control site through heather management, it was found that changes in DOC occurring at the treatment site relative to the control were largely effected by monthly variation as it was also found that there was an interaction between sites and month. The deviation of monthly mean sites DOC concentrations away from the control mean DOC concentration, varied by monthly and by site. The new burn site had ~17% less 10DD DOC than the control, the new cut had ~4% lower DOC than the control, and both old sites had higher DOC than the control (48% and 28% for the old cut and old burn, respectively). This follows the finding of Worrall et al. (2007) and Helliwell et al. (2010) who demonstrated a significant decrease in DOC concentration in soil water on burnt sites. Thus, when considering which treatment would be preferred to produce lower 10DD DOC concentration, the case would be put forward for burn sites over cut site. However, as the control site are actually very old burn treatment sites and peat deposition occurs over a long time period, it is difficult to say without monitoring if a site where heather is cut would have an overall greater export in DOC than burn sites over a 20 year period. So it could be of interest to go back to the sites in 10 years to investigate the status of the DOC concentration.

Research by Clark *et al.* (2008), Holden and Burt (2003) found greater connectivity of the surface peat layer to surface running water than deeper peat. This finding was the motivation for investigating through-flow at 10DD. However, the 1m depth soil pol water DOC concentrations are of great importance particularly at the sites within the dry locality which is topographically elevated on Big Moss. It was found that the dry locality had a lower DOC concentration at the 10DD samples than at the 1m soil pore water samples, and the wet locality DOC concentration were greater at the 10DD. Monthly variation in DOC found that production of DOC at 10DD was more sensitive to monthly variation than 1m soil pore water depth. The difference between DOC at 10DD and 1m soil pore water is greater during the summer month and reduced during the winter period. In January the concentration of DOC at the 1m depth soil pore water was in fact greater than at the 10DD. This indicates that during the winter period, when the vegetation and microbial activity is reduced, the main source of DOC production is in the lower peat layer. The finding of DOC at 10DD being more variable than at soil pore water (across a 1m depth) is supported the findings of Fraser et al. (2001) on shallower peat depth being more variable.

Sample abs_{400} were more strongly influenced by site treatment type than locality. The burn treatment sites (at the wet locality) had ~8-29% higher 10DD abs than 1m. At the dry locality burns had a greater range of difference between the 10DD and 1m (~67-206%). Furthermore, differences found between the cut and burn site 10DD sample abs_{400}, indicated that vegetation treatment can influence water colour quality at the dry locality. The dry new burn had lowest absorbance, lower than the locality control. However, the burn site abs_{400} became greater than the control as it aged. Interestingly, the difference between 10DD and 1m is greatest between the old and new burns. The new burn had lower 10DD absorbance and the old burn had greater absorbance at 10DD than at the 1m depth. This correlation of increasing sample water absorbance in burn sites with site age is supported by Clay *et al.* (2012), who found changes in sample water colour up to four years after a plot was burnt.

6.5.2 Catchment scale DOC concentrations (rain, stream, reservoir)

Having considered the effects of vegetation treatment at the plot scale, the mobility of the waters at runoff, 10DD and 1m must again be considered in context of the catchment. Worrall *et al.* (2003a) found a widespread increase in surface and soil pore-water, colour and DOC concentration over a 29 year period. Through a one year intensive study of 50 British catchments, concerns were raise by Yallop and Clutterbuck (2009), about the release of carbon (in the form of DOC) into headwaters and their link o burning as a management practice. This finding was in contrast to Chapman *et al.* (2010) who found that over a 10 year period there was no link between surface water colour and the area size of a catchment 15 sub-catchments being burned.

Mean DOC concentrations at site plot scale was significantly greater than that of catchment scale concentrations (i.e. rainwater, stream water and reservoir). The reservoir DOC concentrations did not significantly differ to the stream water or the rainwater concentrations. This is indicative of possible mixing of the stream and rainwater within the reservoir. The lower DOC at reservoirs compared to plots scale DOC can be explained by photo and aphotic degradation which occurs in surface water. In accordance with Moody *et al.* (2013) who found that the in-stream DOC degradation results in losses of carbon as CO_2 emissions.

Using absorbance a measure relating to water colour, where by an increase in absorbance related to higher water colour, is an important consideration for water treatment. It is evident that waters interacting with peat soil had higher abs_{400}. As the water moved downstream through the catchment, water colour is reduced. The reduced water colour is likely due to adsorption of DOC by the mineral horizon (Chapman *et al.* 2010). The summer months had greater absorbance than winter months. This is likely due increased microbial activity during drier periods, which produce dissolve organic material. These are then flushed out during rainfall events (Clark *et al.* 2007, Worrall *et al.* 2002).

As the study sites are positioned within ombrotrophic peatland, the measurement of rainwater DOC concentration were intended to be used as a baseline for DOC concentrations input into sites to explain the portion of the variation associated with the treatments. Unfortunately, there was a slight upward trend in rainwater DOC concentrations which was potentially due to sample contamination during the late spring and summer months of 2013, potentially due to algal build on the flask.

6.6 Conclusion

The study was conducted in the Goyt Valley (April 2012 – May 2013); to constrain the effects of cutting and burning vegetation management methods, on soil pore water quality related predictors of WTD, DOC and absorbance. This study put the small spatial site variations (at plot scale) in context of the catchment and larger site temporal changes observed in Chapter 5.

This study determined that:

- The control sites had both lower water tables and fewer runoff and through-flow samples intercepted than the treated sites.
- Through-flow at 10cm depth within the peat had higher DOC concentrations than both runoff water and dipwell soil pore water.
- Wet localities 10cm depth DOC sample concentrations were relatively higher than the soil pore water (1m dipwell), more so at the wet locality than the dry.
- Concentrations of DOC in through-flow water samples intercepted at 10cm depth were more sensitive to seasonal variation than soil pore water sample across a 1m peat cross section.
- Winter DOC production is lower at the peat surface top 10cm at soil pore waters across a 1m cross section of peat.
- The new burn sites had both lower DOC and absorbance than the control and old burn site.
- At 10cm depth, the more recently burned sites are preferential to cutting and to not untreated heather, in terms of reduced DOC production.
- At a dry locality: as a treatment site ages there is greater variation in DOC concentration over time at the burned sites than at the cut sites.

7 Thesis conclusion

7.1 Introduction

The overall aim of this research was to investigate peatland vegetation management fluvial carbon exports; contribute toward the ongoing debates on the benefits of peatland management, and provide best practice recommendations. The emphasis of the research was based upon a few components of the carbon cycle.

The majority of peatlands within the UK are in poor condition, with only 1% of English peatlands classed as pristine according to a Natural England report (2010). Mismanagement was considered a main contributing factor for peatland degradation. Two important management themes were investigated in this thesis research; specifically bare peat revegetation and heather management through cutting and burning. There have also been few multi annual scale studies with as many treatment types as in the Bleaklow bare peat restoration study (Chapter 3 and 4) or to the effect of cutting *vs.* burning study (Chapter 5 and 6). Research on the effects of cutting on water quality was especially lacking.

This conclusion chapter is a critical assessment of the research. It is used to discuss the key objectives, limitation of the studies, the implications of the research conclusions and recommendations for future peatland management and research.

7.2 Review of key objectives

- **Chapter 3**: The effect of bare peat restoration over a multi annual scale. The objective was to investigate three bare peat treatments compared to untreated and undisturbed control sites in order to explain the effects of these treatments on water table depth, soil pore water dissolved organic carbon (DOC) concentration and water quality. Samples were analysed for DOC concentration, as well as sample absorbance at 465/665 nm (E4/E6) ratio indicative of molecular mass and put in context of previous literature.
- **Chapter 4:** The effect bare peat restoration on CO_2 carbon fluxes over the 5 year timescale. The study was conducted at the sites used in Chapter 3. The goal was to investigated the CO_2 fluxes (R_{eco}, P_g and NEE) from the peat, and assess which bare soil site treatment technique would give rise to reduction of carbon losses and improve potential for a site to become a net sink.
- **Chapter 5:** The effect of *C. vulgaris* management burning and cutting/burning wet vs. dry studies on DOC. Burning and cutting were compared between locations with contrasting water tables in order to explain changes in runoff water and soil pore water at plot scale in relation to the site treatment.
- **Chapter 6:** The finding in Chapter 5 indicated that there were significant differences between sites independent to the treatment types. An investigation was carried out over a short times scale and multiple scales from head source to reservoir, with emphasis on comparing finding at 10cm deep through-flow and soil pore water across a dipwell up to 1m deep in order to constrain the finding to the effect of surface vegetation.

7.3 Findings and management recommendation

7.3.1 Bare peat restoration study (Bleaklow plateau - Chapters 3 and 4)

The aim of this study was to establish if bare peat restoration was a significant factor in influencing soil DOC concentrations. Data were used from eight monitoring sites. Four of the sites received no treatment (controls), and four had intervention. There were two bare controls (a bare gully and bare flat) and two vegetated controls (a least disturbed vegetated control and naturally revegetated sites). Four sites received combinations of one or several of the following treatments techniques: seed and lime (NPK) application, gully block installation, geojute and heather brash installation. The bare peat control sites were considered the starting point for each site; as the sites were bare prior to the intervention efforts. The least disturbed vegetated control was considered the ideal site scenario at Bleaklow. Sites were monitored monthly.

The key findings of the Bleaklow bare peat restoration project are represented in Figure 7.1 in a schematic diagram (study sites monitored over a five year period). The primary objective of the restoration techniques employed were to re-establish vegetation and prevent loss of peat through erosions, the effects of the restoration on the dominant cover types were investigated first. It was found that the bare site did not achieve a reduction in bare cover and the most effective method in reducing bare peat cover (within a short three year period) was the gully blocked and geojute site. Seed germination is not possible at the bare sites due to the low pH <4. To enhance the benefit of reduced acidification the use of seeding and liming, stabilisation techniques should be employed. In order to restore a degraded ombrotrophic peat bog to a peat forming system restoration of the hydrological regime was identified as vital. The use of gully blocking or heather brash is recommended to raise the water table up toward the soil surface within a three year period.

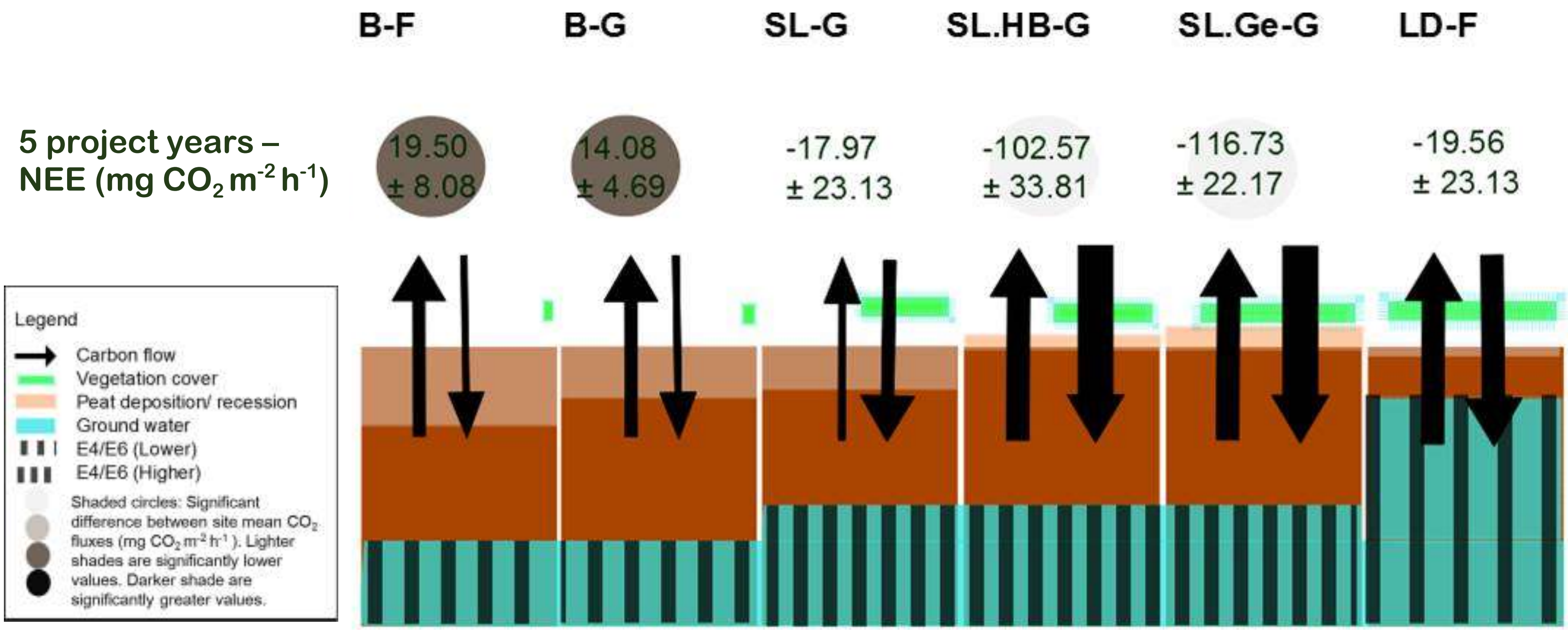

Figure 7.1: Conceptual diagram of Bleaklow bare peat restoration study (2007-2011) key result (ANCOVA and post hoc results). The figure includes six schematics/ peat cross sections representing two bare controls (B-F and B-G) three treatment sites (SL-G, SL.HB-G, SL.Ge-G) and vegetated control (LD-F) The figure includes: mean NEE values; shaded circles, significant differences between the sites NEE; black arrows, carbon flow to and from the site (pointing up, $R_{eco;}$ pointing down, P_g); green units, vegetation cover, light brown unit, peat surface (above the is peat surface) and deposition (below peat surface); blue unit, ground water and water table; black stripes, E_4/E_6 ratio (fewer stripers indicate lower E_4/E_6 and more stripes indicate greater E_4/E_6 ratio). Site details are available in thesis methodology (Chapter 2).

Seasonal variation was the most important factor influencing site DOC over the five year period. No significant difference in soil pore DOC was found between the control sites and sites with treatment. However in 2008 the site with naturally significantly raised water table depth had higher DOC than the least disturbed vegetated control. Furthermore, It was found that DOC concentrations did not significantly differ between vegetation functional groups, however plots dominated by sedge had lower E_4/E_6 ratios (within a humic range) than bare and moss dominated plots (within a fulvic range). Based on the findings, fulvic components are linked to: a) vegetating and microbial activity, b) restoration as a form of site disruption and c) site acidification at a locality scale on Bleaklow. Sites with treatment had increased soil pore water fulvic components (Figure 7.1). It is predicted that there would be a gradual shift towards a greater dominance of humic components; if a site with treatment reach a stable ecosystem with a shallow water table within the range of the vegetated control.

Sites with higher NEE (function of both CO_2 influx and outflux) had greater P_g. (CO_2 influx). All the treated sites had relatively greater P_g than the bare sites (Figure 7.1). The restoration sites with stabilisation methods used had greatest P_g (higher than the bare controls, vegetated control and seeded limed site) due to the vegetative (newly established) success. The heather brash site had ~5 times greater mean P_g flux and the geojute site had a ~8 times greater mean P_g flux than the bare controls. Both R_{eco} (CO_2 outflux) and P_g was at smallest the dominantly bare sites, which explains their small NEE. The benefit of revegetation of bare peat, specifically where stabilisation techniques were used, were not to bring about immediate reduction in DOC but to change the DOC composition; reduce peat erosion and promote litter deposition; peat formation; raise the water table depth; and increase pH to a level in which vegetation can more easily establish. To give the best combination of benefits, it would be recommended to seed and lime as well as use heather brash, or gully blocking in very dry gullies.

7.3.2 Management of C. vulgaris cut vs. burn study (The Goyt Valley - Chapters 5 and 6)

The aim of this study was to establish if there was as significant difference in soil pore water quality and DOC between sites where vegetation is managed through cutting and those that are burned. Seven monitoring sites were installed for the study. These were installed across two localities, one wet and one dry. On the wet locality there were three sites, one untreated C .vulgaris control, one newly burned site and one old burn. At the dry locality these treatment times were replicated and there were also two cut sites, one new and one old. The sites were monitored monthly over the period of five project years. The key findings of the Goyt Valley heather management study are represented in two schematic diagrams in Figures 7.2 (study sites monitored over a five project year period May 2008 - April 2013) and Figure 7.3 (April 2012 – May 2013)

Peat depth varied between the sites (Figure 7.2) but that was explained by locality slope and water table variation. Sites at the dry locality had higher slope than sites at the wet locality and water table than the sites located in the wet locality with low slope. Bulk density was lower at the wet locality than the dry unlike peat depth. At the dry locality the cut and older burn sites had lower C. vulgaris dominance and higher *Eriophorum* cover, than the new burns and control. At the wet locality the new burn also had lower C. vulgaris dominance than the control.

The hydrological pathway of peatland water significantly influenced their DOC concentrations. Water that travelled over the surface of the peat as runoff had lower DOC concentrations than soil pore water. The sites had different hydrology, with the control sites having the deepest water tables. At the dry locality soil pore water was significantly higher at the new cut and burn than the old cut and burn, and runoff concentration were lower at the new burn and old cut.

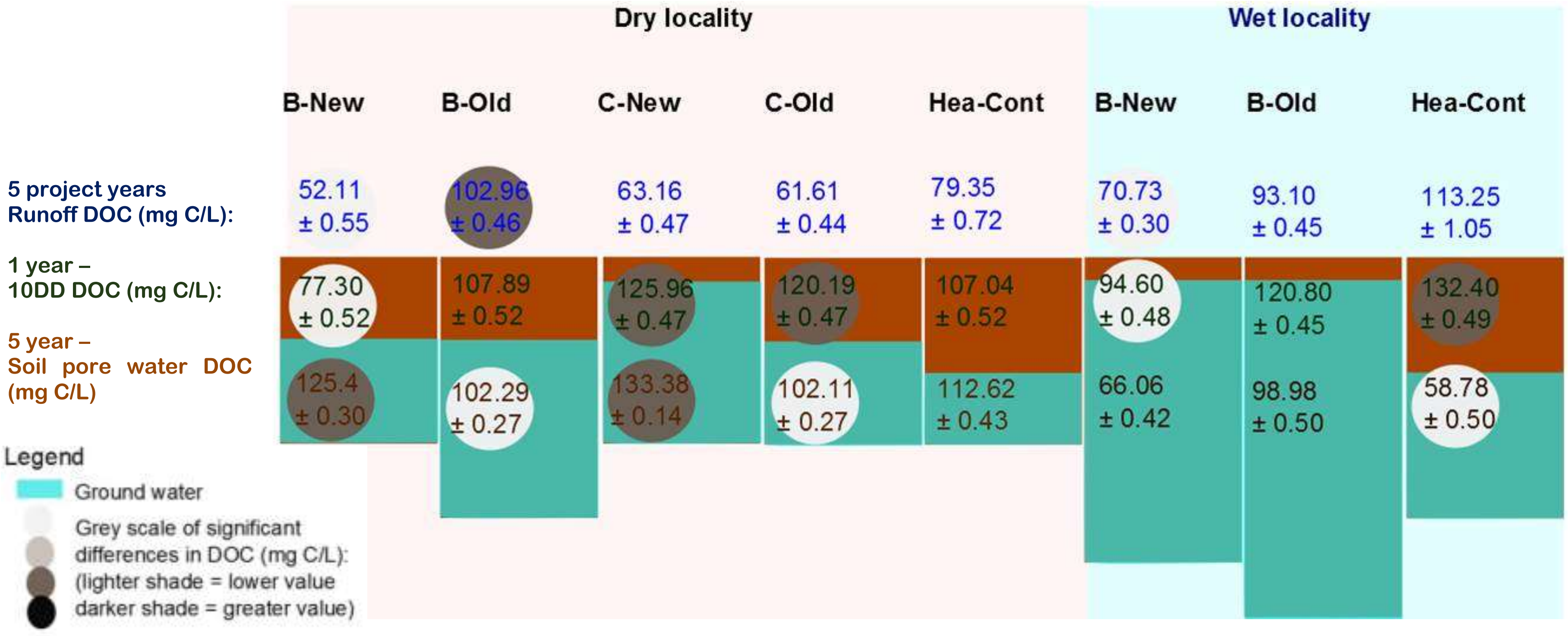

Figure 7.2: Conceptual diagram of the Goyt Valley heather management study (2008 – 2013) key result (ANCOVA and post hoc results). The figure includes eight schematics/ peat cross sections representing managed heather sites and controls at: a dry on the left (heather control, new burn, old burn, new cut, old cut) and wet locality on the right (heather control, new burn and old burn). Significant differences are read by locality and horizontally by sample type (not between water samples types). The figure includes: mean DOC concentration values; shaded circles, significant differences between the sites DOC (by sample type); blue unit, ground water and water table. Site details are available in thesis methodology (Chapter 2).

At the dry locality burn sites did not significantly differ to the control site. There was little difference between the burn ages, and locality was the most important factor. The difference between the control and the treatment sites was greater at the wet locality than the dry locality. Runoff and soil pore water DOC concentrations of cutting *C. vulgaris* are lower than the control and have little variation over time as the site aged. The dry burn had lower DOC concentration than the control; however concentrations increase as the burn site ages. Hence, considering runoff and DOC concentration, cut sites (new and old) and new burns are preferential to no treatment.

Analysis of shallow through-flow water DOC concentration at 10cm depth (10DD) was conducted. It was found that the 10DD sample mean DOC concentrations were: a) lower than runoff samples and b) greater than soil pore water across a 1m deep dipwell (Figure 7.3). This is important as previous research has found water at the top layer of peat (root layer) has greater connectivity with surface waters (streams and reservoirs) (Clark *et al.* 2008, Holden and Burt 2003).

There was no significant difference in annual rainfall volumes between the localities. It was not always possible to intercept runoff and 10DD samples at the drier sites, particularly the controls. Lack of sample intercept was attributed to differences between sites. The variation of 10DD was more sensitive to seasonal variation than 1m soil pore water was, and therefore more representative of vegetation and the treatment type than 1m depth spoil pore water samples. The runoff, 10DD and 1m soil pore water samples at the cut sites suggest greater connectivity within the peat than in other sites. Concentration of DOC within 10DD at the dry site found that the new burn had lower concentrations than the control. As the waters moved downstream through the catchment towards the reservoir (Figure 7.3); both mean DOC concentration water colour was reduced. More research would be required into the water flow paths and connectivity of water at different depths from site to stream to determine the potential impact of the treatment types on reservoir DOC concentration. If the water at 10cm depth has greater connectivity to the reservoirs than water at 1m depth, then at a dry locality both new burns and new cuts are preferred to old burns. Also in a wet locality a new burn is preferential to an untreated C. vulgaris dominated site. However, the old burn had both higher runoff and soil pore water concentration than the old cut. It is recommended to cut *C. vulgaris* instead of burning them to reduce peatland DOC production in the long term.

7.3.3 General conclusion

The Bleaklow and Goyt Valley studies emphasised three important reoccurring themes relating to upland peatland management: a) temporal and spatial scale, b) peatland hydrology and c) site morphology. Site treatment or management was not always the most important explanatory factor during analysis of Bleaklow and Goyt Valley data. Variation in site CO_2 and water sample DOC concentrations were strongly linked to variation between months (which accounted for between 9 – 24% of data variation). Promoting ecological change in an upland peatland can be a slow process depending on the purpose and the management methods: bare peat restoration, methods such as gully blocking, geojute or heather brashing can significantly reduce bare peat dominance within 3 years of treatment. Heather management, through burning has a 1 year lag in vegetation regrow compared to a managed cut which has a ~2 year lag. These time scales mentioned are important when considering the monitoring time scale. Long term site monitoring is vital, as it enables the analysis of data trends and identifies changes in the data attributable to inter-annual variation and weather variables (e.g. temperature, PAR and rainfall).

Variation between sites (bio- chemical conditions) explained a greater portion of CO_2 data variation than site could in the case of DOC concentrations (monthly variation was most important). Furthermore, a sites natural morphology is important to consider such as: peat depth, slope and if the site positioned along a gullies. Locality depended factors were all important and linked to a sites hydrology.

The locality of a study site was an important factor as it influenced site WTD in addition to CO_2 flux and DOC, at a greater magnitude than surface vegetation management did. For example on Bleaklow, at a naturally revegetating gully water tables naturally rose and bare peat was reduced without intervention. In the Goyt Valley there were significant differences between WTD and DOC in managed burn sites at wet and dry localities. As the changes in DOC are more readily controlled by weather variables than management, this increases peatlands (degraded or mismanaged) susceptibility to climate change. Evidence on Bleaklow and the Goyt Valley indicate vegetation management had a small impact on peatland carbon cycles. There is therefore a potential to use management as a tool to 'buffer' against peatland carbon losses related to the effects of increasing temperature (Freeman *et al.* 2001), reduced rainfall and draught (Clark *et al.* 2009, Neff and Hooper 2002, Worrall and Burt 2004), PAR and changes in soil pH (due to acid deposition) (Clark *et al.* 2008, Scott *et al.* 1998) and erosion (Gallego-Sala *et al.* 2010) potentially associated with climate change (Dinsmore *et al.* 2013, Frolking *et al.* 2006, Worrall *et al.* 2003).

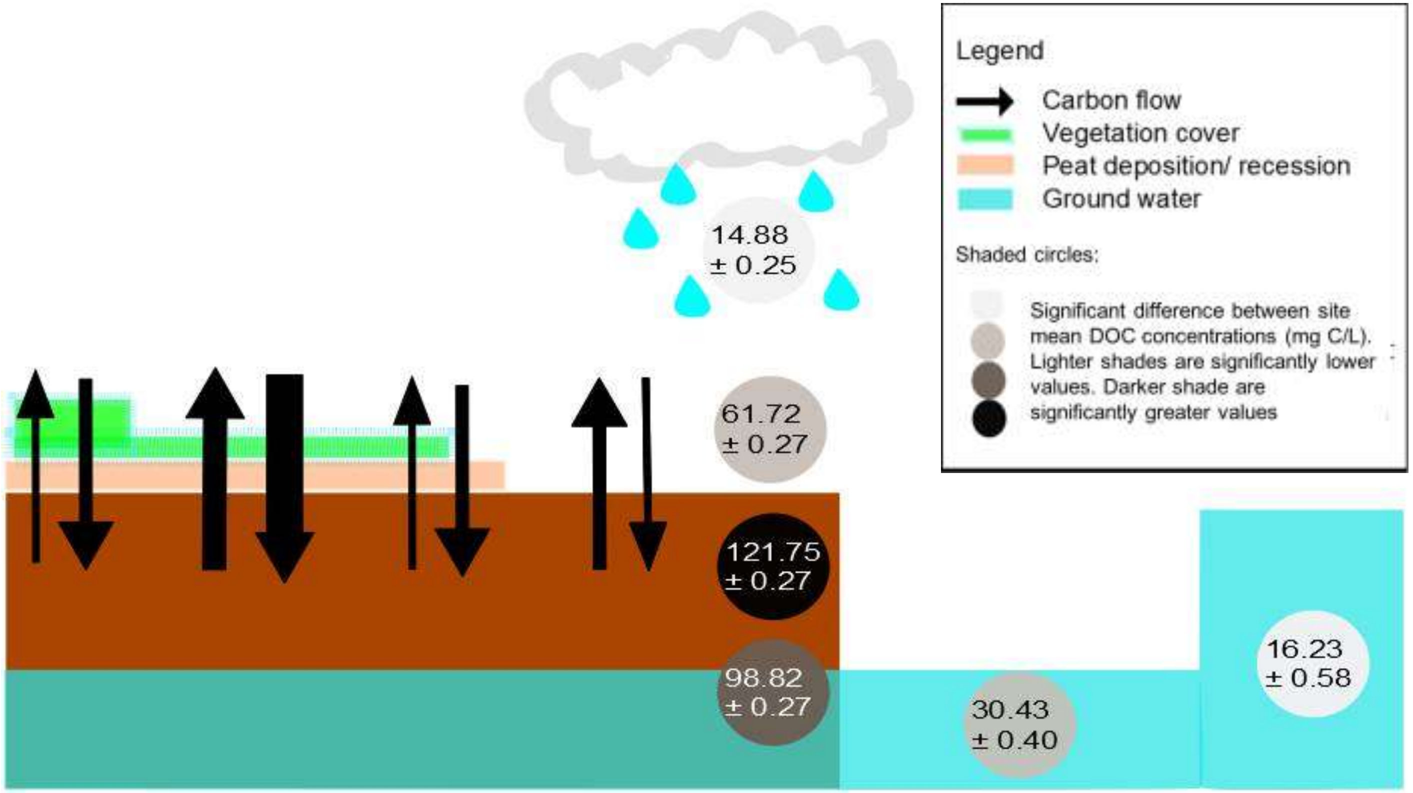

Figure 7.3: Conceptual diagram of the Goyt Valley heather management study (April 2012 – May 2013) key result (ANCOVA and post hoc results). The figure represents the carbon cycle along peatland cross section. It incorporates a) Bleaklow findings (on gas fluxes) with b) Goyt Valley head waters to reservoir study. The figure includes: values of the water sample mean DOC concentrations within: rainfall, surface runoff, 10DD through-flow, soil pore water, stream water and reservoir; shaded circles, significant differences between the sample DOC; black arrows, carbon flow to and from the site (pointing up, Reco; pointing down, Pg); green units, vegetation cover, light brown unit, peat surface (above th e is peat surface); blue unit, ground water and water table; Site details are available in thesis methodology (Chapter 2).

7.4 Limitations

Every experimental study can be improved with a greater number of replicate readings or longer time series etc. This section aims to cover limitations of the datasets used in this thesis that should be noted when considering the conclusions drawn from these results.

- **Site Disturbance**: Installation of monitoring equipment (i.e. dipwells, runoff traps, 10DD through-flow traps and gas collars) into the peat cause unavoidable disturbances as they damage vegetation roots and can compact the surrounding peat. Additionally, the monitoring process causes disruption through frequent visits which can damage the surface vegetation, impede their growth and result in areas of bare peat. To minimise the disruption of the site and potential impact of the carbon fluxes and water quality being measured some steps were taken. Firstly, installed equipment was allowed to settle for at least one month after initial installation. Secondly, disruption was minimised when navigating across the sites particularly around the monitoring plots. Finally, although the data were potentially impacted by the disturbance, all the sites including the control were monitored in the same way and changes were all relative to the control.
- **Site selection**: To compare the effect of vegetation treatment and attribute the results to the management method and not the site variation was a challenge. In order to select representative sites for the treatment it was difficult to find sites with the exact same morphology. The use of six plots replicated for each treatment type allowed for sampling of multiple conditions. There were significant differences between peat depths between the Goyt sites. Therefore, to account for this variation and prevent the site variation masking the effects of surface vegetation treatment, site characteristics (such as peat depth, bulk density and vegetation dominance) were included in analysis of covariance (ANCOVA).

- **Data gaps:** Weather condition did result in data gaps particularly during winter months in which heavy snow prevented safe sight access. This could have skewed the data annual mean, particularly as greater DOC production occurs during the summer months than the winter. The use ANOVA/ANCOVA of month nested within year as a factor in addition to weather variable (e.g. temperature, rainfall) allowed for better explanation of the variation in context of environmental conditions.
- **Lack of pre-treatment measurements:** It was not possible to monitor sites pre-treatment, particularly in the case of wild fire restoration method (Chapter 3 and 4). The use of an untreated control was therefore essential. In the case of the Goyt dry vs wet burn study (Chapter 5 and 6) it was also important to account for wet and dry control to relate the data variation occurring at the sites in which no pre-treatment monitoring had been conducted.

7.5 Further work

After 5 years of monitoring the Bleaklow sites it is evident that revegetation of the bare peat occurred at different rates. Changes to a peatland are relatively slow, thus the long term (> 10 year) effect of restoration is an important consideration. After three years of monitoring, it was found that gully blocking was the most successful method at restoring a bare sites vegetation cover and hydrological regime, to a regime one more similar to the vegetated control. By the end of five year study, the geojute and heather brash sites were the most successful. The study demonstrated that bare peat revegetation is followed by an increase in DOC production and increased P_g and peat deposition. It would be of interest to follow the progress of the sites over a longer period of time. In order to find out: at what point in time does each of the treatment sites (seeded and limed, heather brash site and geojute) site reach the point at which the site are totally revegetated? What is the climax ecosystem? How long does it take for: a) the hydrological system to be restored, and for b) changes in the pH, DOC concentration and ratio of E4/E6 to stabilised? The findings would be analysed relative to the vegetated control as it is likely that the bare soil control sites may receive much needed restoration treatment.

Previous research has established the link between peat and surface water quality (Worrall *et al.* 2002). The question of water connectivity between plot scale DOC concentrations and surface water is an important one for both the Bleaklow and the Goyt study. On Bleaklow, the subject of fluvial carbon mobility and connectivity to surface water can be investigated. This investigation could conduct existing data specific to the sites in chapters 3 and 4. Soil pore water and stream water (from geojute site, least disturbed vegetated control and the bare soil control) water quality data (water anion, pH conductivity and DOC concentration) could be along site Defra sourced air quality data. The data could be used to also investigate the relationship between site pH and acid deposition; as pH influences DOC solubility and mobility as well as the ability for vegetation to establish.

Waters connectivity would be especially interesting on the Goyt valley. Previous studies on lowland peat have demonstrated the link between DOC concentrations of the top 10 cm to that of surface water (Clark *et al.* 2008). To establish the potential for vegetation management through cutting and burning to have a significant impact upon reservoir water DOC concentration, it is important to the flow paths which the mobile waters a and labile fluvial carbon take. This can be done using existing anion data from the study to give detail on waters mixing as done by Clay *et al.* (2010) who found runoff waters have a closer ionic compassion to rainwater than soil pore water. Principle component analysis could be employed to address the point on connectivity. The other option would be to use a tracer study, as done by Boothroyd (2014) who used it to trace flow paths down a hillslope.

The final point of interest would be to investigate the effect of C. vulgaris cutting on water quality DOC concentrations and peat compaction in wet vs dry condition. This is an important question as there is little existing research on the effect of C. vulgaris cutting on peatland geochemistry. To cut peatland vegetation tractors are employed, these tractors disturbs the peat surface. It was found in chapter 5 that peat in the dry locality had lower bulk density than that at the wet site. According to Brown *et al.* (2014) burning of vegetation increase peat bulk density. A great sample size is required to ascertain how cutting would affect peat bulk density. The research conducted on C. vulgaris cutting on the Goyt (chapter 5 and 6) was carried out at a dry locality. If vegetation cutting is established as the preferred method for upland C. vulgaris management, the effect of cutting in wet peat condition should also be considered.

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Appendices

Appendix A (electronic):

i. **Qassim et al. 2014 (PDF):**

Contributers: Qassim S. M., Dixon S. D., Rowson J. G., Worrall F., Evans M. G., Bonn A.

Published paper title: 'A 5-year study of the impact of peatland revegetation upon DOC concentrations'

ii. **Thesis data (Excel files)**

Chapter 3 and 4 –

Data used for the study on bare peat revegetation effect of DOC and water quality. The file contains a collation of the data used (Predictors *e.g* DOC and CO_2; covariates *e.g*. vegetation cover, temperature and PAR).

Chapter 5 –

Data used for the study on the effects of C. vulgaris management of OC and water quality

This file contains the collation (Predictors *e.g.* DOC and WTD; covariates *e.g*. temperature and Rainfall)

Chapter 6 – head to reservoir DOC concentration and water quality

This file contains three tabs. The first tab 'Chap 6 Head to stream' is a collation of all the site details (eg. Slope and vegetation cover) and water quality data on rainfall, surface runoff, through-flow, stream water and reservoir. The second 'Chap 6 treatment site' is a collation of data for peat soil runoff, through-flow and soil pore water. The final tab 'Chap 6 reservoir data' is a complication of the data gathered as part of this thesis research and data obtained from the United Utilises SCaMP project.

Appendix B

Bleaklow study site plates

a) Bare gully control (B-G)

b) Bare flat control (B-F)

c) Least disturbed vegetated control (LD-F)

d) Naturally revegetated gully control (NRv-G)

e) *Gully blocked, seeded and limed (SL.B-G)*

f) *Geojuted, seeded and limed (Sl.Ge-G)*

g) *Heather brashed, seeded and limed gully (SL.HB-G) part 1*

h) *Heather brashed, seeded and limed gully (SL.HB-G) part 2*

i) Seeded and limed flat (SL.F)

Appendix C

Important terminology:

Context	Abbreviations and acronyms	Definition
Commonly used abbreviations	10DD	Soil pore water trap/ sample specific to 10cm peat depth
	Abs	Absorbance - Abs400, Abs465 and Abs665 absorbance at 400, 465 and 665 nm
	ANCOVA	Analysis of covariance
	ANOVA	Analysis of variance
	CH_4	Methane
	CO_2	Carbon dioxide
	DOC	Dissolved organic carbon
	DOM	Dissolved organic matter
	E_4/E_6	Ratio (ratio of Abs465 to Abs665)
	IRGA	Infra-red gas analyser
	K	Air temperature kelvin (0 degrees Celsius = −273.15)
	MW	Molecular weight
	NEE	Net Ecosystem Exchange
	NER	Net ecosystem respiration
	OC	Organic carbon
	OM	Organic matter
	PAR	Photosynthetically active radiation ower molecular weight
	P_g	Pg gross primary productivity
	POC	Particulate organic carbon
	UV-VIS	Ultraviolet-visible spectrophotometry
	WTD	Water table depth
	WTH	Water table high in peat up from mineral soil
Site abbreviations:	Cont	Field control (on Bleaklow and in the Goyt Valley)
	LD-F	Least disturbed flat site on Bleaklow
	B-F	Bare flat site on Bleaklow
	B-G	Bare gully site on Bleaklow
	SL-F	Seeded and limed flat site on Bleaklow
	SL.Ge-G	Seeded, limed and geojute flat site on Bleaklow
	SL.HB	Seeded, limed and heather brashed flat site on Bleaklow
	SL.B-F	Seeded, limed and gully blocked flat site on Bleaklow
	Nrv-G	Naturally revegetating gully on Bleaklow
	B-New	New burn in the Goyt Valley
	B-Old	Old burn in the Goyt Valley
	C/L-New	New cut and lift in the Goyt Valley
	C/S-Old	Old cut and scatter in the Goyt Valley

Table 1: Terms referring to important commonly used abbreviations thesis.

Spatial scale	Term	Definition
Large	Area	Is the general region a study is conducted within (i.e. Bleaklow Plateau and Goyt valley).
	Locality	The sub region (ie. Bleaklow on the Bleaklow Plateau; Big Moss and Ravenslow in the Goyt Valley
	Site	Is the study site selected based on a treatment type (e.g. restored, control etc). Each site is composed of replicate study plots.
Small	Plot	A plot is a monitoring unit installed on a site. All plots have a dipwell and gas collar installed.

Table 2: Terms referring to the study spatial scale (large to small).