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A study into the impacts of water table and nitrogen pulse on cellulose decomposition rates and carbon release in active peatland

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Abstract

Since the 1980s overstocking of sheep has become an increasing problem on UK uplands. This study examined the effect of livestock urination, and also water table depth, on active peatland on cellulose decomposition and fluvial carbon release. Urination was simulated using synthetic sheep urine (SSU) and applied to quadrats of active peatland at Redlake Mire on South Dartmoor and compared to quadrats without SSU treatment. Cellulose decomposition was measured using the cotton strip assay (CSA) method as a measure of microbial activity by examining a loss in tensile strength of cotton strips buried for 45 days from September 20th to November 3rd 2012. Carbon release was measured via dissolved and particulate organic carbon (DOC and POC) concentration analyses. Application of SSU caused a biologically significant ($P < 0.1$) reduction in cotton tensile strength (CTSL) of approximately 8% compared to cotton strips buried with SSU, which is less than previous studies. Despite this, no statistically significant effects of SSU were found, which was most likely due to wet weather conditions during the fieldwork causing a high water table, meaning an anaerobic environment was maintained, making oxygen the limiting factor. This study suggests livestock urination does have some detrimental impact on peat so stocking patterns should be carefully considered; however peatland restoration and preservation projects will be most beneficial if they focus on hydrology making rewetting and maintenance of a high water table a priority.

Introduction
In the UK 80% of the 2.3 million hectares of blanket bog are damaged. The sources of degradation include burning, exploitation (for fertilisers and fuel), drainage (for forestry and agriculture), over-stocking and grazing. In 2008, carbon dioxide release as a result of peatland degradation, was estimated to be over 3 x 10^9 t y^{-1}; equivalent to at least one tenth of global fossil fuel emissions (Parish *et al.*, 2008). If degradation becomes severe, peatland can become a significant source of carbon, rather than a sink, into the atmosphere and the aquatic system. In 2011, the IUCN stated that if 5% of the carbon stored in UK peatlands were to be released, it would be equal to the UK's annual greenhouse gas emissions (IUCN, 2011).

The main reason for upland peat management in the UK is for farming (including sheep grazing) or animal breeding for hunting (Worrall *et al.*, 2007a). It has been recognised that the presence of livestock on peatlands may have detrimental effects, yet a number of previous studies have simply examined the effects of having sheep present, and not specifically sheep urination, especially on carbon and greenhouse house (GHGs) (Worrall *et al.*, 2012).

70% of water in the UK comes from uplands (Labadz *et al.*, 2010), and with the frequency of extreme flooding events increasing the need for peatland protection is evident. Failure to do so may have environmental, social and economic impacts.

Aims and Objectives of this Study
This study aims to examine the impacts of livestock urination and water table fluctuations on cellulose decomposition and organic carbon release in upland blanket bog. Synthetic sheep urine (SSU) will be used as a substitute for sheep urine, and the water table will be measured using piezometers. Cotton strips will be buried to act as an organic substitute for peat, and the degradation due to microbial activity measured, representing cellulose decomposition. Dissolved organic carbon (DOC) and particulate organic carbon (POC) in peat waters will be analysed in the labs as an indication of degradation.

Literature Review
There are 4.0 - 4.16 x 10^6 km^2 of peatlands worldwide (Parish *et al*., 2008; Holden *et al.*, 2011). They are of vital importance in relation to carbon as they store on average 1,375 t ha^{-1}; the largest amount of all vegetation types. This makes up almost 30% of carbon stored terrestrially (>550 Gt). On a global scale the soil-carbon content of peatlands is 75 times more than annual CO_2 emissions from anthropogenic sources (Bridgham *et al.*, 2008). Despite covering just 3% of the total land surface peatlands equate to as much as all other terrestrial biomass combined. 87% of UK peatland is blanket peat (Holden *et al.*, 2011) containing a minimum of 3,121 Mt of carbon (Lindsay, 2010). This accounts for approximately 13% of the global area covered by blanket bog (Ratcliffe and Oswald, 1988). Estimates of the amount of carbon stored in UK peatlands are based on four factors: the area of peat, its density and depth, and its carbon content (Worrall *et al.*, 2010). Other large stores of carbon in peat occur in Western and Northern Europe, Canada and Siberia (Worrall *et al.*, 2003).

The main condition which enables this large carbon sequestration is the anaerobic environment created by sustained waterlogging (Parish *et al*., 2008), as 85% of blanket bog is water (DNPA, 2004). Initially, photosynthesis fixes carbon which is then stored in the aerobic peat surface layer called the acrotelm. Of the carbon fixed by vegetation, approximately 90-97% is lost in the acrotelm due to decomposition. The remaining carbon is stored in the deeper, anaerobic catotelm (Ohlson and Økland, 1998). The acrotelm is the partially waterlogged aerobic surface layer of peat which is young in relation to the deeper, anaerobic catotelm. The lowest water table depth therefore is the bottom of the acrotelm (Holden and Burt, 2003). Hydraulic conductivity in the catotelm is 3-5 orders of magnitude less than the acrotelm, due to the small pore spaces in the dense peat (Daniels *et al.*, 2008). It is in the catotelm that slow decomposition rates result in peat accumulation (Lindsay, 2010). The volume of carbon stored in peat varies between these two layers; in the upper parts of the acrotelm 0.03 g cm^{-3} is stored, whereas further down the catotelm contains 0.12 g cm^{-3} (Lindsay, 2010).

As well as carbon sequestration, peatlands provide a number of other important ecosystem services. They are an important habitat for many plant and animal species, including endangered species such as the Golden Plover and Dunlin (Natural England, 2010). Pristine peat is a vital regulator in terms of catchment flood protection. The slow rate of decomposition has an archaeological significance as it allows conditions for preservation of organic remains (Freeman *et al.*, 2004). Peatlands have great recreational value for hikers and also hunting of grouse. Dartmoor, where this study took place, is used for military training, however artillery shelling was ceased in 1988 due to the heavy damage it caused (DNPA, 2004). In the past peat has been for use as a fuel source, however this was extremely detrimental. 2 kg of dried peat contains the same amount of energy as 1 kg of decent coal (DNPA, 2004), but a far greater weight would need to be cut and dried to produce the 2 kg due to the high water content of peat. This made it an extremely damaging activity and hence it has now been phased out.

Peat is made up of partially decomposed vegetation remains that accumulate at the top of the soil profile over long periods of time under waterlogged conditions. It builds up when organic material inputs are greater than the decomposition rate (JNCC, 2011). The major process that occurs during decomposition is the loss of polyphenols and polysaccharides (Kalbitz & Geyer, 2002). This is due to increased phenol oxidase activity when oxygen is present, and Freeman *et al.* (2004) found activity to increase 700% in response to the addition of oxygen, resulting in a 27% reduction in phenolic material concentrations from 1.985 to 1.444 mg l^{-1}. Therefore, maintaining a waterlogged, anaerobic condition is vital for peat preservation to keep high levels of phenolics as they inhibit enzyme activity, hence reducing decomposition.

The European Commission Habitats Directive defines “active” blanket bog as “still supporting a significant area of vegetation that is normally peat forming” (Maddock, 2008). A typical peat profile consists of a 5 cm upper layer of dark brown peat that is poorly humified (H2-H3 on the von Post scale). Below this is a slightly humified brown layer to 10-15 cm depth which may contain layers of lighter brown *Sphagnum* peat above darker brown *Eriophorum-Calluna-Sphagnum* peat (H3-H4). Deeper still the peat becomes more humified and decomposition is virtually complete by 1.5 m depth (H9) (Holden and Burt, 2003).

Peat degradation is major issue. Over 99% of deep peat (peat over 40 cm deep) in the UK has been mapped as damaged and over 70% of shallow peat (10 – 40 cm deep) (Natural England, 2010). It is thought approximately 3 million tonnes of CO_2 are emitted from UK peatland annually, equivalent to over 300,000 households. One of the most detrimental effects on peatland is burning, and it is estimated approximately 40% of English moorland has undergone some form of burn management (Worrall *et al.*, 2007a), and 20% over these uplands have had grips (drains) implemented. To try and mitigate the harmful effects the Department of Environment, Food and Rural Affairs (DEFRA) has developed restrictions that ensure burning only takes place in strips less than 9,000 m^2 in area with a minimum of 150 m between areas. Also, burning should not occur between 15th April and 1st October when the ground and vegetation is likely to be dried out. This “cool burn” aims to minimise the damage done to the underlying soil and peat. Burned areas should then be left for at least 12 years before further burning (DEFRA, 2005). Many bird species such as grouse prefer habitats where burning has taken place, which is one of the main drivers.

Burning has been linked with improving grazing, as it promotes the growth of grasses and sedges, rejuvenates heather, and removes undesired litter and woody growth (Lance, 1983). In the UK sheep are the primary animal used for grazing, although cows and deer are also used in some areas (Worrall *et al.*, 2009).

The degrading effects of livestock are of growing concern, as in England stocking rates in general have increased since 1980 (Evans *et al.*, 2005). Livestock have an impact on soils in a number of ways. Physical impacts include plant biomass consumption, trampling (above and below ground compaction), and seed introduction and dispersal. Increased grazing has also been linked with a reduction in soil infiltration rates and soil structure (Worrall *et al.*, 2007a). Chemical and biological impacts include input of nutrients and bacterial contamination from excretion (Reeves and Champion, 2004). There are also knock-on effects such as reducing vegetation cover and heterogeneity, and therefore habitat diversity, and trampling causes damage to reproductive habitats. When considering the use of grazing farmers should consider a number of factors: type and quantity of livestock, potential disturbance caused, time since previously stocked, and duration of grazing (Reeves and Champion, 2004). In order for keeping livestock to be sustainable it must be seasonal and not intensive due to low vegetation productivity, allowing time for regrowth and recovery (JNCC, 2011). The recommended stocking density is 0.1 sheep ha^{-1} (or 0.2 sheep ha^{-1} for the summer 6 months) (Lunt *et al.*, 2010).

Tanentzap *et al*. (2012) looked at the effect of herbivores on a range of terrestrial ecosystems, and found that removing herbivores had a positive mean effect on carbon stocks per unit area. The findings varied greatly, with wetlands, including peatlands, showing a mean global increase of 80,000 t yr^{-1} (standard deviation 100,000 t yr^{-1}) where herbivores had been removed. The study highlights the importance for studies to consider duration where herbivory effects are non-linear. Some studies that have examined the impacts on carbon stock over long periods of time by looking at averages for years, or even decades, may underestimate the effects of herbivory if all herbaceous biomass is removed in a short initial period at the start of the time of study. Herbivory reduces plant litter quality and quantity and changes the rate of soil respiration (Tanentzap et al., 2012).

Urination is a significant way that livestock affect below-ground carbon. Peat in temperate uplands is typically acidic and low in nutrients, and this low pH in the soils limits important nitrogen cycle process such as nitrification (Rooney, 2006). However soil pH increases directly below areas of urination, suggesting a potential for increased productivity. The major nitrogenous component of urine is urea, which is quickly converted to ammonium (NH_4^+) by the process of hydrolysis (Thomas et al., 1988). Sheep urine typically has an N concentration with the equivalent rate of application of 500 kg N ha^{-1}. This is more than plants can utilize; for example pasture plants only take up approximately 20-30% (Limmer and Steele, 1983). Thomas et al. (1988) investigated the effects of urine-N on Scottish upland soil and found soil N levels remained higher than usual for 40-90 days after application. It is worth noting this study used genuine sheep urine rather than synthetic sheep urine (SSU). Rooney *et al.* (2006) examined the effects of SSU with a microcosm study and found an increase in soil pH, microbial activity and microbial biomass positively correlated to addition of SSU, as shown in Table 1.

Table 1: Effect of SSU addition on soil and microbial biomass measurements (Rooney *et al.*, 2006). *TPF (triphenylformazan) measurement is used to determine dehydrogenase activity, which itself is used to represent total microbial activity (Partoazar *et al.*, 2011)

SSU treatments	pH	Microbial activity (mg TPF* g^{-1} dry soil)	Microbial biomass (mg biomass C g^{-1} dry soil)
Zero SSU	5.379	25.65	2.395
Low SSU	6.021	22.05	2.485
Medium SSU	6.021	41.68	2.532
High SSU	6.404	41.88	2.57

This study investigated microbial activity by looking at cotton strip tensile strength loss (CTSL). There is no naturally occurring “standard form” of cellulose; therefore cellulosic substrates, such as cotton, are used as proxies for assessing decomposition. CTSL is a good indication of cellulose decomposition, which itself can be used as an index for the processes of organic matter decomposition in general. Due to these two ideas cotton strip assay is used in ecological studies (French, 1988). Howard (1988) argues the cotton strip assay (CSA) may not be used as an accurate indicator of litter decomposition or “general biological activity”. One reason for this is that natural plant cellulose contains lignin and hemicelluloses, and therefore is not pure when it enters the soil like the cotton strips used in this method. Vegetation in soils usually contains a fraction of nitrogen also. It is suggested a method is required that that considers cellulose decomposition and also other parts of plant litter (Howard, 1988). However, to date, the CSA is still the widely accepted method used.

Up until the last decade or so, estimates of carbon losses from peatlands were primarily based on gaseous C loss. Worrall *et al.* (2003) stated that considering carbon in particulate and dissolved state also is equally important in order to fully understanding the carbon budget of peatlands. Carbon losses to streams and rivers include particulate and dissolved, as well as dissolved gaseous carbon (Worrall *et al.*, 2010). Dissolved organic matter (DOM) is the most mobile part of soil organic carbon, even though it accounts for just 0.04-0.22%. Despite this, it is still seen as an effective indicator of soil processes that influence cycles of C and N (Zsolnay, 1996). DOC is defined as the fraction of organic compounds which passes through a 0.45 µm filter (Roulet and Moore, 2006). Concentrations in natural waters range from less than 1 to more than 50 mg l^{-1}, with the higher concentrations found in waters from organic soils or draining from peatlands (Evans *et al.*, 2005). Less is known about DOC release than carbon in gaseous forms such CO_2, yet DOC is a vital part of peat ecosystems and may be the difference between a peatland being a sink or a source of carbon (Clark *et al.*, 2009). Figure 1 shows the main imports and exports of carbon in organic soils.

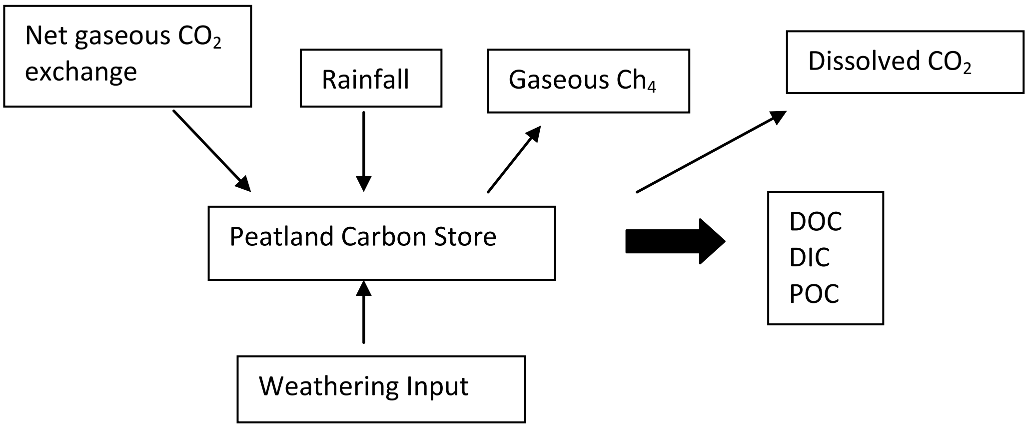

Figure 1: Major carbon fluxes from organic soils (adapted from Worrall *et al.*, 2010).

Factors influencing DOC include temperature (Clark *et al.*, 2005), presence of vegetation (Gogo *et al.*, 2012), stabilising the water table depth (Worrall *et al.*, 2007a) and nitrate deposition, the latter of which is effected by livestock urination. However Worrall *et al.* (2007a) found no significant difference in DOC content where grazing animals had been present. Water table dynamics can function differently in peatland that has been damaged by drainage compared to undisturbed peat (Holden *et al.*, 2011). Fluctuations in the water table occur in the acrotelm where there are larger pore spaces, but these are relatively limited in undisturbed peat. When peat is no longer waterlogged the aerated state allows decomposition of the peat and hence DOC is released (JNCC, 2011). Despite temperature being known to influence concentrations, Preston et al. (2011) states it is not a good predictor of annual DOC variation, based on long-term data analysis. Water flow also has a strong influence on DOC concentrations. 50% of all DOC release occurs during the top 10% of flow values which occur, for example, during storm flows (Schiff *et al.*, 1998). As a result many studies may underestimate total DOC by a considerable amount if they fail to include times of highest flow.

DOC is sometimes referred to as total organic carbon (TOC) where amounts of particulate organic carbon (POC) are negligible (Charman, 2002). POC is organic compounds ranging from 0.45 to 1.0 µm (Dawson *et al.*, 2002), as high concentrations are typically associated with damaged peatland. One way in which it differs from DOC, as well as particle size, is that POC can be removed by wind as well as water (Lindsay, 2010). Concentrations of POC tend to very low in natural, intact peatlands. Whereas DOC is an issue for water companies due to discolouration, POC is an issue because sediments can build up in reservoirs and reduce their capacity. In some cases in Britain capacity is thought to have been reduced by at least one third as a result of POC sediment accumulation (Lindsay, 2010). The impacts of POC are not yet extensively understood in comparison to DOC (Pawson *et al.*, 2006). This study recognises the importance the importance of considering POC as a significant source of carbon loss, as Pawson *et al.* (2008) found 80% of fluvial carbon loss in degraded peatland was in the form of POC.

Kalbitz & Geyer (2002) found high levels of peat degradation resulted in a lower DOC:DON (dissolved organic nitrogen) ratio than the solid phase C:N ratio. This suggests that a loss of soil organic matter leads to release of DON.

Sphagnum species are commonly chosen as an indicator of activity in peatlands as they account for much of the productivity in the system and have a significant effect on the dynamics and nutrient cycling (Rice, 2000). This study takes *Sphagnum* cover into consideration when assessing the spatial variation in activity. The holocellulose in *Sphagnum* has a high cation exchange capacity that binds the cations required for microbial activity and therefore prevents the decay of peat (Gogo and Pearce, 2009). Ohlson and Økland (1998) found the carbon exchange capacity of peatlands with the atmosphere to vary greatly spatially. This is strongly linked with variations in peat forming processes which can also be large in a single area of peatland.

One restoration technique is re-establishment of peatland vegetation such as *Sphagnum*. Other methods used are stabilisation of eroding surfaces and raising the water table. Despite preventing the loss of carbon from peatlands, restoration may, in the short term, lead to an increase in methane (CH_4) emissions which has a global warming potential (GWP) 23 times that of CO_2. This is because it is produced when organic matter in soil undergoes anoxic decay. However, CH_4 is often not included in carbon budgets as it usually represents less than 5% of the total budget (Worrall *et al.*, 2010).

Location

Redlake was chosen as a suitable location as it is well recognised for its peat deposits. The area used to be a china clay (kaolin) works up until 80 years ago, and peat accumulates where the underlying granite has completely decomposed to kaolin, or partially decomposed to growan. Much of the soil on Dartmoor is peaty, although true peat is found in areas where there is a combination of wet ground, moist air and cool temperatures (Harris, 1986), which results in high plant productivity and slow decomposition (Parish *et al*., 2008). Therefore the majority is found on higher levels, such as Redlake at an altitude of approximately 450m. Figure 2 shows the location of Redlake and Figure 3 shows the Redlake site and the location of the quadrats used.

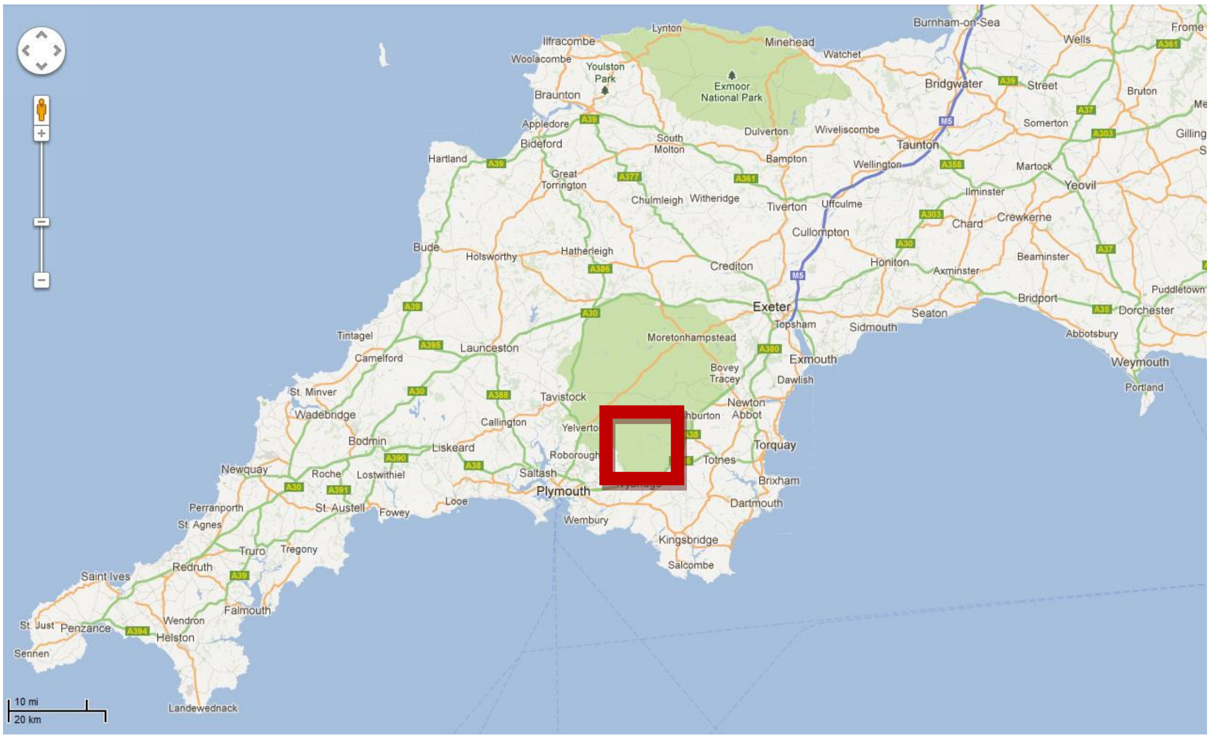

Figure 2: Redlake location in Dartmoor National Park, South Devon.

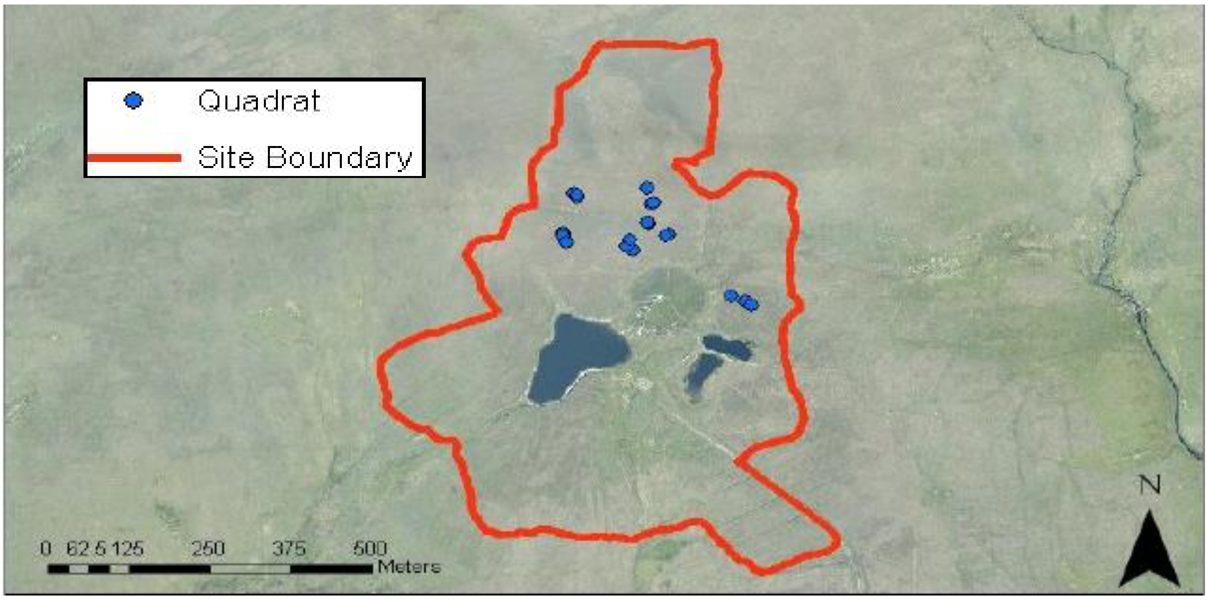

Figure 3: Extent of Redlake site and quadrat location (adapted from Hicks, 2012).

It is a 40 ha ombrotrophic blanket bog (Selwyn, 2012), meaning it receives all the required water and minerals from precipitation (Bridgham *et al.*, 2008). Redlake is situated approximately 8 km southeast of Princetown where the average annual rainfall from 1971 to 2000 was 1974.2 mm from 181 days of rainfall greater than 1 mm (Met Office, 2007). The average minimum and maximum temperatures over the same period ranged were 5.4°C and 11.2°C respectively. The majority of this study was carried out over October which typically has 17 days of rainfall resulting in 215.3 mm of rain, with average air temperature ranging from 6.4-11.6°C (Met Office, 2007).

Peat was cut for use as a fuel source until WWII (DNPA, 2004), and from the summit of the spoil heap at Redlake (from the china clay works) it is apparent that the area has been drained for cutting, as the grips (gullies) are still clearly visible across the majority of the site (Figure 4). This is concerning as drainage is a significant cause of peat fires which are one of the largest sources of carbon release in relation to land management (Parish *et al*., 2008). This scene is not uncommon, as approximately 50% of UK blanket bog (~75,000 ha) has been drained (Milne and Brown, 1997).

Figure 4: Evidence of peat drainage by grips visible at Redlake.

Methodology

In this study a total of 20 quadrats were used; synthetic sheep urine was applied to 10, and the remaining 10 remained as controls. Quadrats were randomly selected across the active area of the Redlake site. The location of each quadrat was marked using a Trimble GeoHX handheld GPS to aid relocation. Percentage coverage of *Sphagnum* was identified by using a 50 cm x 50 cm quadrat with 5 cm^2 divides (100 squares) on each quadrat and the results were used as a proxy for identifying the more active areas of peat. Of the 10 quadrats with a higher *Sphagnum* percentage coverage 5 had SSU applied, and 5 did not, and likewise for the 10 quadrats with the least *Sphagnum* present. This was to try and ensure the quadrats with and without SSU had an even divide of more and less active peat.

Cotton Strip Assay (CSA)

Preparation

A total of 65 strips of organic cotton were cut measuring 12 cm by 7 cm. Batches of ten were wrapped in aluminium foil and heated for 20 minutes at 115°C. This eliminates any foreign organisms that may affect the cotton before burial.

Burying

Three strips were buried in each of the 20 quadrats; one in the centre and the remaining two 10cm either side, at a depth of 10-15 cm in the acrotelm. These were left in the soil for a period of 45 days from the 20th September to the 3rd November 2012. When digging up strips extra care was taken to ensure no tearing/damage was caused. Strips were kept in sealed plastic bags and labelled for returning them to the laboratory for treatment and testing. The 5 remaining strips were not buried, but kept under controlled conditions of 19°C and 65% relative humidity. These were to be used as control strips.

Washing

Strips were washed in de-ionised water for 15 minutes and soaked in a 70% ethanol solution for 15 minutes on returning to the laboratory after retrieval from the field. This was to prevent soil particles having a binding effect and potentially increasing the strength of the cotton, and also to reduce the continuation of soil microbial activity. If the tensile strength were going to be assessed immediately washing was not required, although still advised for the sake of keeping the tensometer and jaws clean. The control strips were also washed to ensure all strips received the same treatment. All strips were dried at room temperature for at least 12 hours, and then kept in the fridge for storage. Before testing for tensile strength, strips were dried for 4 hours at 50°C then stored at a relative humidity of 65% like the control strips.

Tensile Strength Testing

Tensile strength was examined as it is a more sensitive measure of decomposition compared to measuring weight loss. The effect of soil adhering to the strips is also less (Latter *et al.*, 1988). Edge effects and fraying were eliminated by removing the outer threads of each strip down to a uniform number of 120 strands. An Instron Tensometer with a 5 kN load cell and a separation speed of 20 mm min^{-1} was used to determine the pressure at which the cotton strips snapped, measured in Newtons (N). Control strips were tested to determine the actual strength of the cotton. Buried strips were then measured and compared to control strips, allowing a percentage loss in tensile strength to be calculated.

Synthetic Sheep Urine (SSU)

Preparation
The SSU consisted of two solutions which were only mixed immediately prior to application in the field to prevent hydrolysis (Caine, 2011). The first solution consisted of urea (21.4 g) dissolved in 500 ml of de-ionised water. The second was made up of potassium bicarbonate $KHCO_3$ (23.1 g), glycine $C_2H_5NO_2$ (10.7 g), potassium chloride KCl (3.8 g) and potassium sulphate K_2SO_4 (1.9 g) dissolved in 500 ml of de-ionised water.

Application
SSU was applied to the 50 cm x 50 cm quadrat after the burial of the three cotton strips. 125 ml of each of the two solutions was combined and sprayed evenly over the quadrat using a sterilised spray bottle. A bottle with a flexible plastic tube was chosen to ensure all of the solution was drawn from the bottom of the reservoir. The urea content within 250 ml of SSU is the equivalent to the average amount recorded in sheep urine patches of 500 kg N ha^{-1} (Clough *et al.*, 1996). Plastic sheeting approximately 30cm high was used around the edge of the quadrat whilst applying SSU to minimise any loss via wind spreading. 1L of surface water from the surrounding was applied immediately afterwards to ensure no SSU remained on vegetation. The plastic sheeting was also rinsed to ensure any SSU that had landed on it was applied to the soil.

Depth to Water Table
Piezometers were used for measuring the water table depth. They were buried in the SE corner of each quadrat and, along with the Trimble GeoHX, acted as an aid for relocation of quadrats. Strips of cardboard 50-60 x 1 cm were cut and dipped into the piezometers and the point where the moisture came up to on the strips represented the water table depth.

Water Sampling

Cleaning of Equipment
20 glass 50 ml sample vials were placed in a 2% decon bath for 24 hours. Then, after rinsing in deionised water, they were placed in a 10% HCl bath for another 24 hours, followed by further rinsing in milli-Q water. Once dry, vials were wrapped in aluminium foil and baked at 450°C in a muffle furnace for 6 hours. Vials were then ready for use.

Sample Collection
Approximately 15 ml of water was collected from the piezometers using a vacuum pump and stored in sample vials.

Treatment of Samples
Samples were individually filtered through Whatman GF/F glass microfibre 0.45μm filters. Approximately 50 ml of each water sample was collected, however only 10 ml was required for DOC analyses. The full samples were still filtered, as the sample size needed to be known when calculating the POC and PON concentrations.

The filter papers had also been previously baked at 450°C in a muffle furnace for 6 hours to reduce organic residues. Filters were weighed before and after filtration to calculate the weight of the particulate content of samples. The filtered samples and filter papers were both frozen until ready for laboratory analysis.

Particulate Organic Carbon (POC) and Particulate Organic Nitrogen (PON)
Filters were removed from the freezer and baked at 40°C, followed by fumigation for 24 hours using 12M hydrochloric acid, then returned to the 40°C oven to dry overnight. This removes any remaining inorganic carbon (Hicks, 2012). Eight 5 mm disk cores were taken from random locations on each filter paper and placed in high purity 8 x 5 mm tin cups which were folded into rough spherical parcels ready for analyses. It must be noted that taking cores assumes that particles are spread evenly over the filter paper. POC and PON were determined using a CHN Elemental Analyser EA1110.

Cyclohexanone-2,4-dinitrophenylhydrazone (CYC) and PACS calibration standards were run at the same time as the samples to check the validity of the results. Carbon content was given as a percentage, so to quantify the results into concentrations filter paper weight was recorded and used.

Dissolved Organic Carbon (DOC)
Filtered samples were acidified ready for analysis using a Shimadzu TOC 5000a analyser coupled to an ASI 5000A autosampler and a Sievers NCD 255 Detector. The addition of HCl ensures there is no inorganic carbon (namely CO_2 from the atmosphere) present in the samples. Calibration standards were prepared to produce an equation for calculation of the DOC concentrations of the field water. The 6 calibration samples used were 0, 121, 242, 484, 726 and 968 µM l^{-1}. Blank samples were run at the beginning analysing, after the calibration standards, at one stage during field sample analysing, and after all analysing. This was done to monitoring the cleanliness of the equipment throughout testing.

Temperature
EasyLog EL-USB-2^+ temperature loggers were buried at the same depth as the cotton strips in quadrats 8, 16 and 18 to monitor the subsurface temperature for the duration of the fieldwork taking readings every 30 minutes. These quadrats were chosen as they were well spread out in relation to all 20 quadrats.

Statistical Analysis
Data from lab analysis was output in Microsoft Excel, and copied into Minitab for analysis. A Grubbs test was applied to all data sets to identify any outliers. Boxplots were used to visually display the comparison between the SSU application and no SSU and a Mann-Whitney U test used to determine significance. Scatterplots with trend regression lines were produced to show the relationship between measured variables. Data points were ranked and correlation calculated.

Results

The mean subsurface temperature over the duration of the fieldwork was 10.1 °C. No outliers were identified having run a Grubbs test. No results were obtained for site 4 due to the site being unable to be relocated.

Synthetic Sheep Urine

Cotton Tensile Strength Loss

Testing the tensile strength of the five control cotton strips found the initial maximum load to be 564 N (σ = 27 N). The strength of the buried strips was subtracted from this initial strength to give the loss in strength. The strength of cotton strips that had been buried with SSU applied ranged from 432 N to 565 N. The mean max load was 516 N (σ = 61 N) equating to an average loss in tensile strength of 8.53% compared to the control strips. For buried strips without SSU applied max load ranged from 467 N to 660 N. Mean max load was 558 N (σ = 74 N), meaning CTSL was 0.88%. Some strips that had been buried showed an increase in strength from the control strips. The difference between cotton strips with and without SSU application had a Mann-Whitney significance value of 0.0831. As this is less than 0.1 the difference may be classed as biologically significant, despite not being statistically significant. Cotton strips were not retrieved from quadrats 11, 12 and 17 as on the day of retrieval the peat was frozen at 15 cm depth. Also, at quadrats 1, 2, 3, 7 and 8 only two of the three buried strips were retrieved for the same reason. Therefore CTSL results were based on 19 individual strips for sites with synthetic sheep urine application and 24 strips for sites without SSU. This was still an adequate sample size to run statistical tests on. Figure 5 shows the comparison of the three sets of strips.

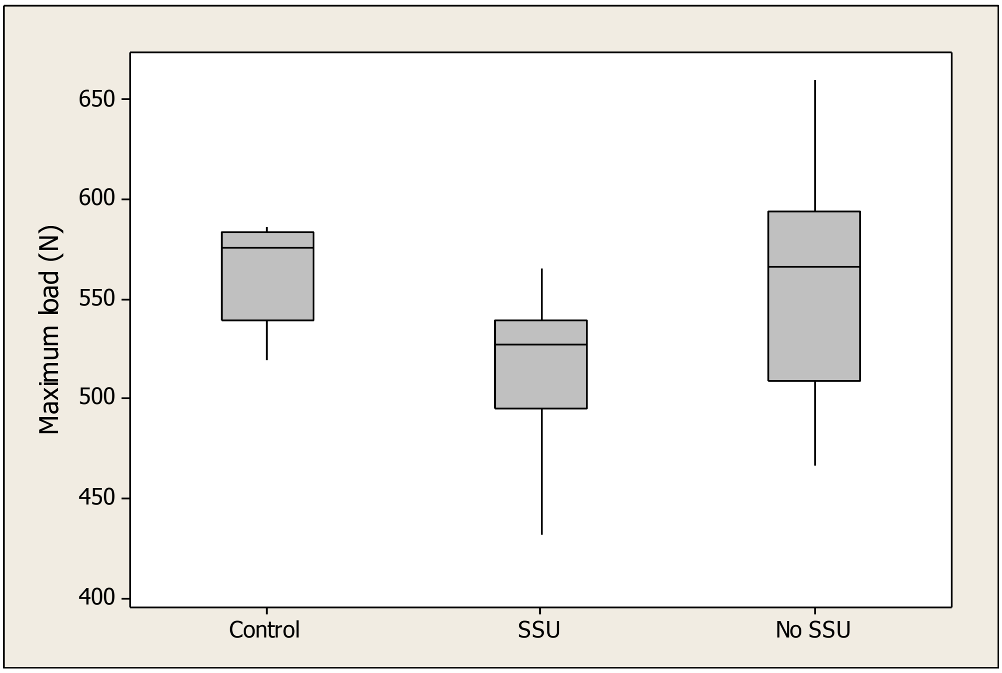

Figure 5: Tensile strength of control cotton strips, and buried strips with and without SSU. 5 strips were used for the control. Boxes show lower quartile, median and upper quartile values and whiskers represent lowest and highest values.

Dissolved Organic Carbon in Water Samples

The highest calibration standard had a concentration of 968 µM l^{-1} (34.85 mg l^{-1}). At first all field sample concentrations were far higher than expected, so a 3 times dilution was used to make samples within the range of the calibration standards. A number of samples had concentrations above this value, even after dilution. Having plotted all of the calibration standards and a line of best fit, the equation of the line was used to extrapolate above 968 µM l^{-1}, however this was only reliable to do so as high as 1175 µM l^{-1} (42.31 mg l^{-1}) (Tappin, 2013). Three samples still had DOC concentrations above this value (quadrat 6 = 42.73 mg l^{-1}, quadrat 18 = 54.35 mg l^{-1}, and quadrat 20 = 46.24 mg l^{-1}), so for statistical analysis these were treated as 42.31 mg l^{-1}.

Figure 6 compares DOC at sites with and without SSU applied. The mean DOC concentration for sites with SSU application was 27.95 mg l^{-1} (σ = 13.52 mg l^{-1}), and for sites without SSU was 35.96 mg l^{-1} (σ = 4.70 mg l^{-1}). The difference in DOC concentrations at sites with and without SSU application had a Mann-Whitney significance value of P = 0.3387.

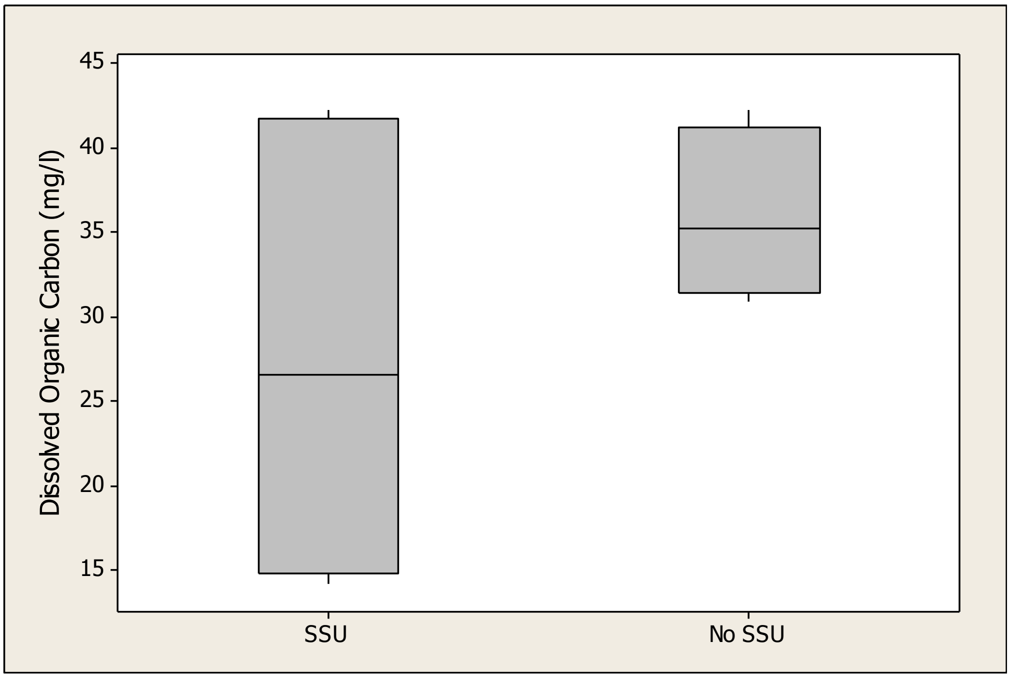

Figure 6: Dissolved organic carbon concentrations at sites with and without SSU.

Results were not obtained at sites 5, 7, 8, 9, 14 and 15 due to equipment damage during storage of water samples. To check the quality of the DOC results two sets of certified reference material were analysed. 14 separate analyses were carried out (50 separate determinations). The mean concentrations of the two sets were 39.1 µM (σ = 4.9 µM) and 44.5 (σ = 2.3 µM). These are close to the consensus range of 41-44 µM and therefore the concentrations found are reliable.

Particulate Organic Carbon in Water Samples

POC concentrations ranged from 4.31 to 78.02 mg l^{-1}. Sites with SSU application had a mean concentration of 22.23 mg l^{-1} (σ = 22.47 mg l^{-1}), and at sites without SSU the mean was 36.86 mg l^{-1} (σ = 24.60 mg l^{-1}). No significant difference between sites with and without SSU was found (Mann-Whitney significance value of P = 0.4379) (see Figure 7).

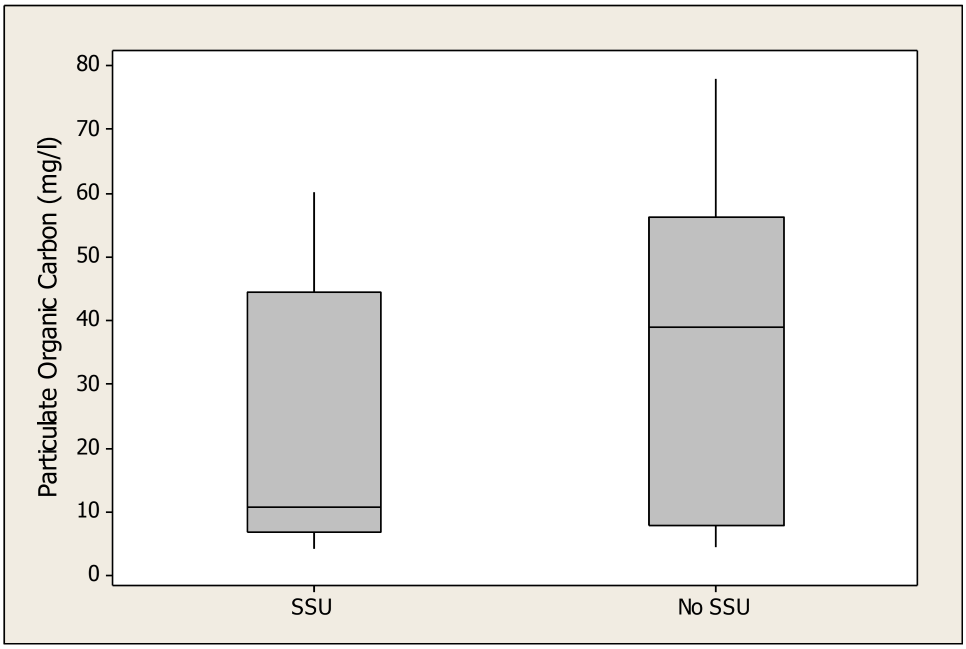

Figure 7: Particulate organic carbon concentrations at sites with and without SSU.

Particulate Organic Nitrogen in Water Samples

Concentrations of PON ranged from 4.31 to 78.02 mg l^{-1}. Sites with SSU application had a mean concentration of 2.86 mg l^{-1} (σ = 3.74 mg l^{-1}), and at sites without SSU the mean was 5.10 mg l^{-1} (σ = 4.43 mg l^{-1}). The difference between sites with and without SSU had a Mann-Whitney significance value of P = 0.5636 and is therefore not significant (see Figure 8).

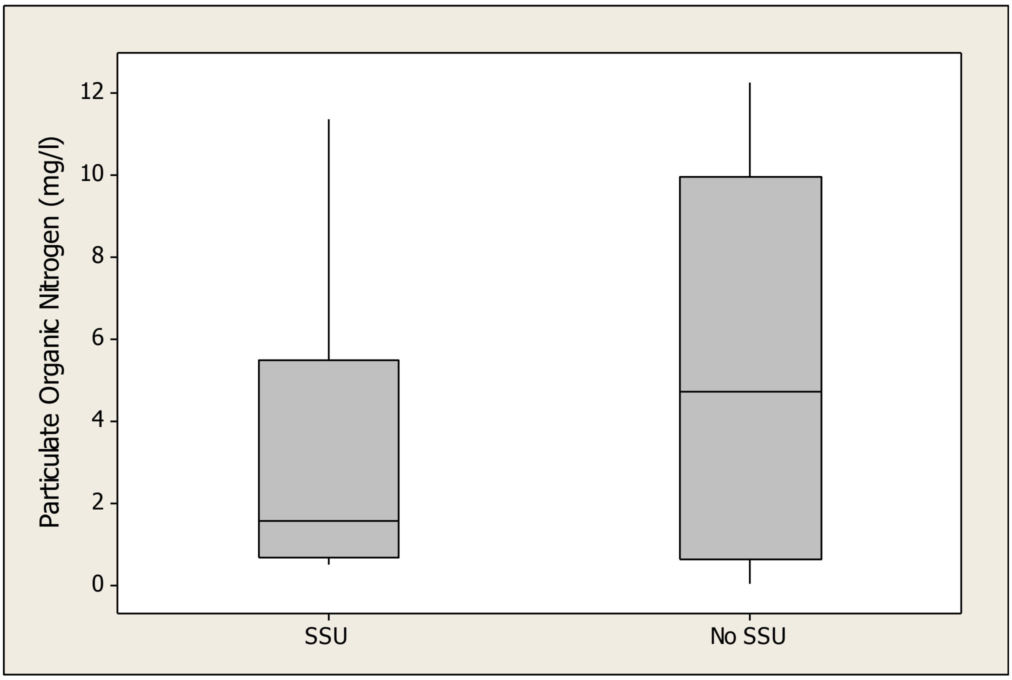

Figure 8: Particulate organic nitrogen concentrations at sites with and without SSU.

Water Table (from 20th September to 3rd November 2012)
Initially water samples were going to be taken from 30 µm porous pots. Due to the age of equipment and length of time left in the field many of the pots buried were damaged so all water samples were taken from the piezometers. Water table depth ranged from 0-18 cm below ground level as shown in Figure 9. At all sites except site 12 the height of the water table increased over the course of the fieldwork, and for sites 1, 2 and 9 the ground was supersaturated by the end of the 45 day period. On 20th September the average depth was 8.16 cm below ground level, which dropped to 5.50 cm by 3rd November.

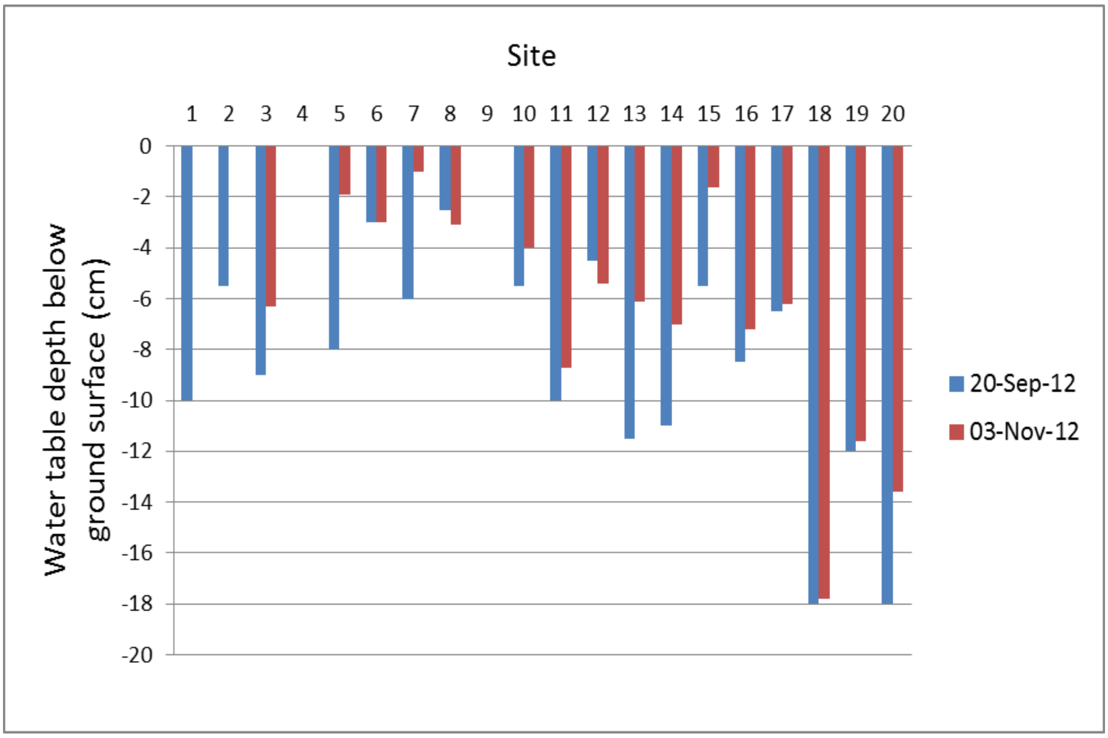

Figure 9: Water table depth below ground surface at start of the fieldwork on 10th September and at the end of fieldwork on 3rd November (blue and red bars respectively).

Cotton Tensile Strength Loss
Figure 10 shows a weak negative correlation between CTSL and water table depth, with the higher the water table the greater the tensile strength loss. The relationship was not found to be statistically significant (P = 0.344).

Dissolved Organic Carbon in Water Samples
A slight relationship was found between water table and DOC with the lower the water table the higher the DOC concentration (Figure 11). However this relationship was not found to be statistically significant (P = 0.085).

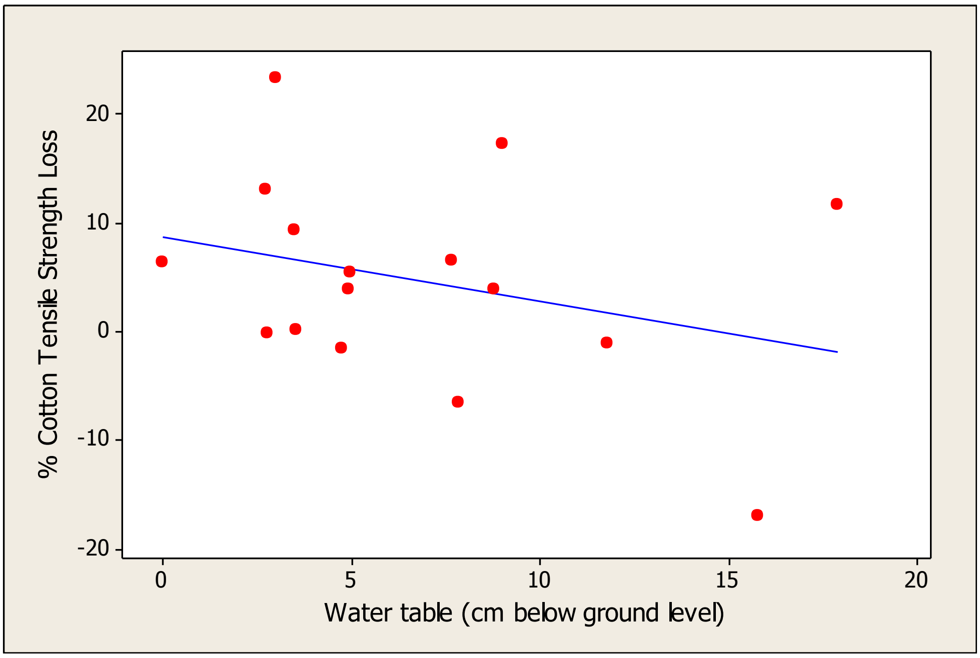

Figure 10: Relationship between water table depth and percentage loss in cotton strip tensile strength. Cotton strips retrieved 03/11/12.

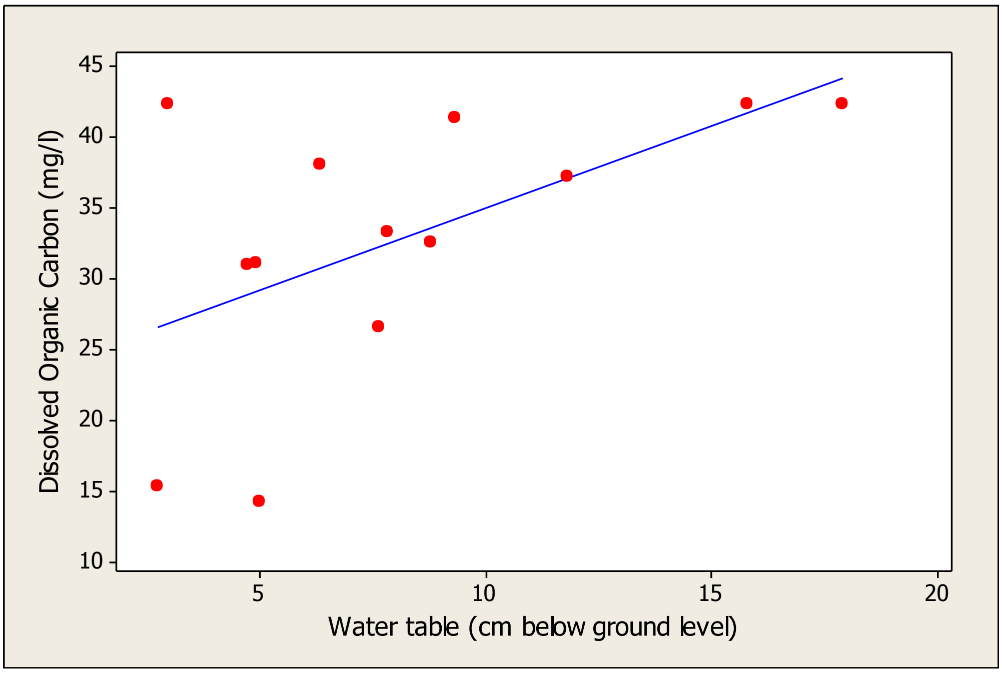

Figure11: Relationship between water table depth and dissolved organic carbon concentration. Samples collected 03/11/12.

Particulate Organic Carbon in Water Samples

The correlation in Figure 12 shows the lower the water table the higher the POC concentration, yet the relationship was not statistically significant (P = 0.092).

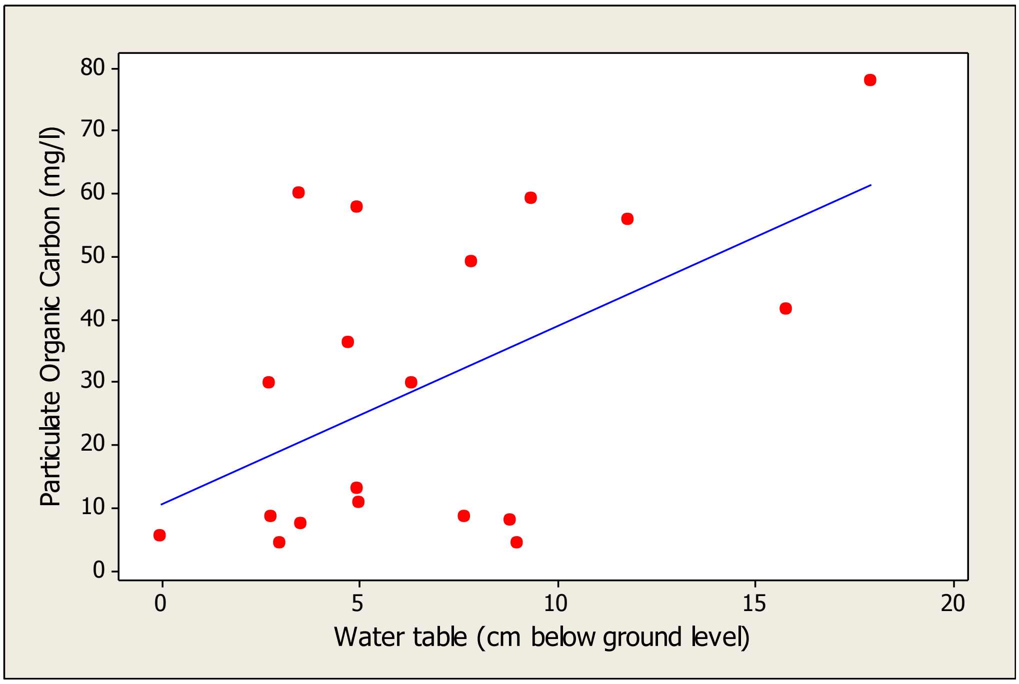

Figure 12: Relationship between water table depth and particulate organic carbon concentration. Samples collected 03/11/12.

Particulate Organic Nitrogen in Water Samples
A positive correlation between water table depth and PON concentration is shown in Figure 13, but the trend is not statistically significant (P = 0.102). As the water table depth increase so does the PON concentration.

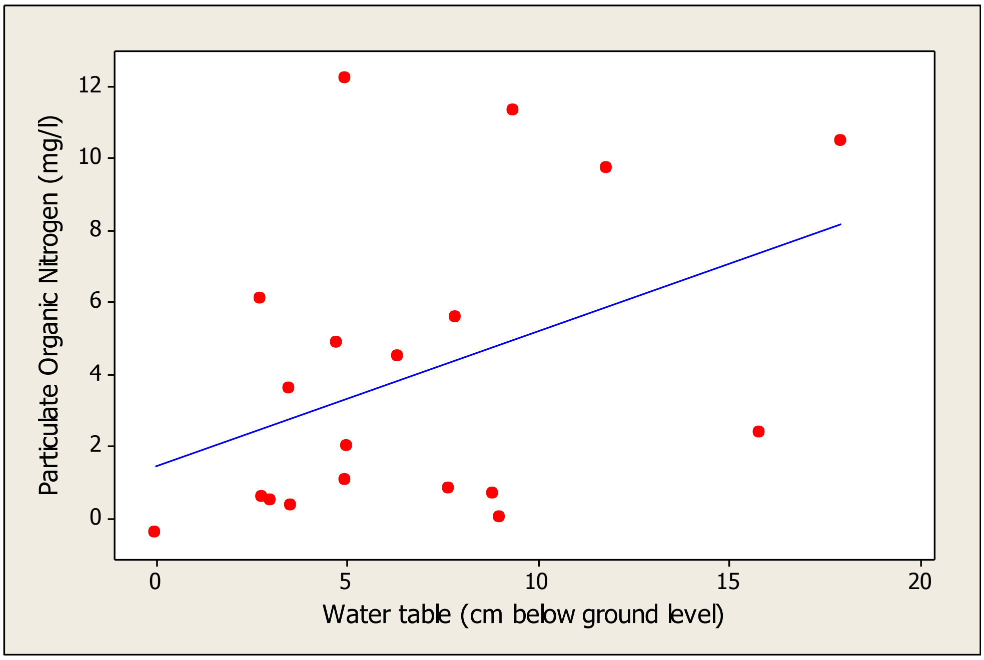

Figure 13: Relationship between water table depth and particulate organic nitrogen concentration. Samples collected 03/11/12.

Other Relationships

Figure 14 shows a weak negative correlation between CTSL and DOC and POC respectively. In both cases the higher the DOC/POC concentration the lower the percentage cotton tensile strength loss. Neither relationship was found to be statistically significant, with CTSL and DOC having a *P* value of 0.629, and CTSL and POC having a *P* value of 0.280.

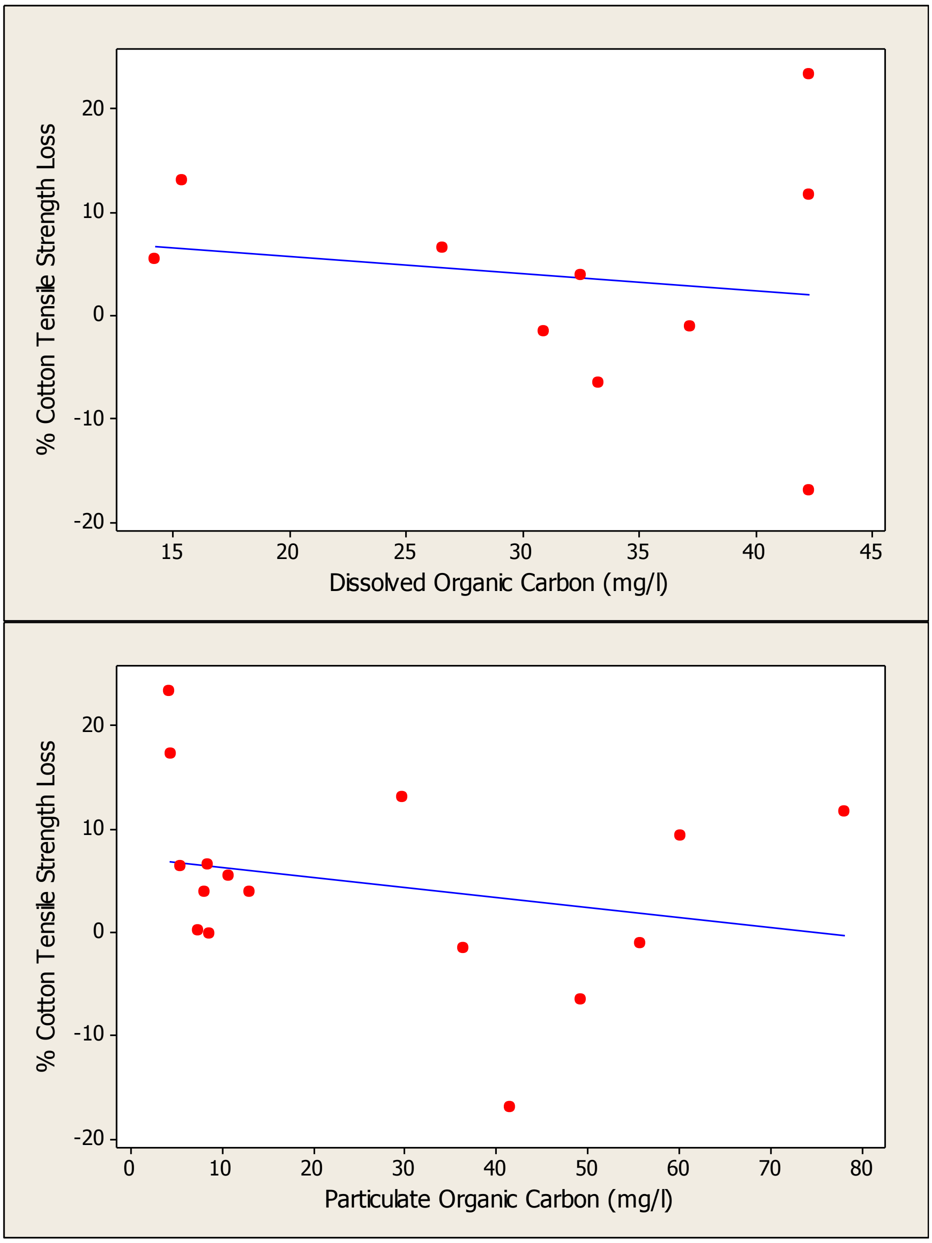

Figure 14: Relationship between % cotton tensile strength loss and **a)** dissolved organic carbon concentration and **b)** particulate organic carbon concentration.

A positive correlation was found between DOC and POC (Figure 15); as the concentrations of one increased as did the other, however the trend is not statistically significant (*P* = 0.489).

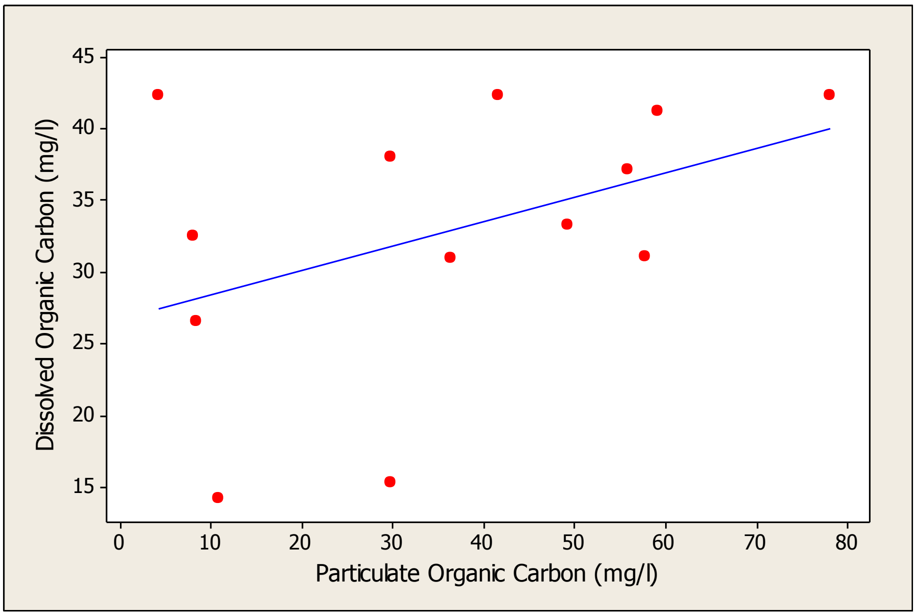

Figure 15: Relationship between dissolved organic carbon and particulate organic carbon.

As POC concentrations increased so did the concentrations of PON and the relationship was found to be statistically significant (*P* = 0.000), as shown in Figure 16.

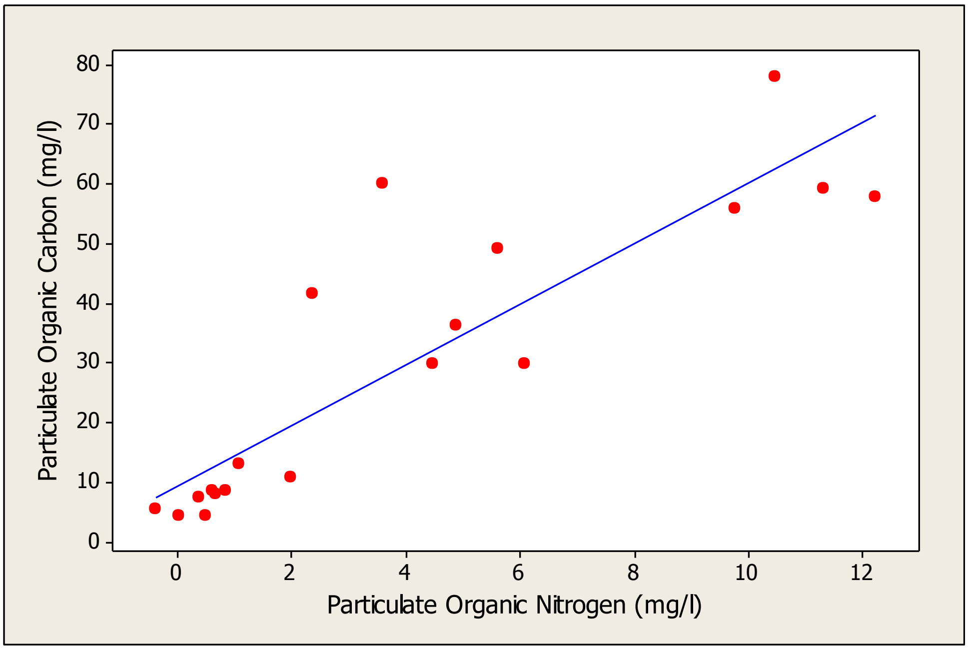

Figure 16: Relationship between POC and PON concentrations.

Discussion

Application of SSU had a biologically significant detrimental effect ($P < 0.1$) on cotton strip tensile strength loss; despite this, the difference was not statistically significant ($P > 0.05$). It was expected that the effect of SSU application would be greater, as previous studies (Rooney *et al.*, 2006; Caine, 2011) found the application of SSU to have far more significant effects upon microbial activity ($P < 0.001$ and $P < 0.005$ respectively). Caine used the cotton strip assay method to examine peat at Foxtor Mire, also on Dartmoor, and found the tensile strength of the control strips was far higher than the control in this study (1005.7 N compared to 564.2 N). Caine found that strips with and without SSU treatment both showed significant reductions in tensile strength. Application of SSU caused a reduction of 54%, while strips that had simply been buried but not treated were still 28% weaker.

In comparison, cotton strips in this study with SSU showed a reduction in tensile strength of 8.53% ± 7.15%, whilst the strips without SSU were just 0.88% ± 10.54%. Some of the maximum tensile strength loads recorded for the retrieved cotton strips were higher than the 5 control cotton strips that had been kept under controlled lab conditions. This can be seen in Figure 5, yet the reason for this is unclear.

The most likely reason for the less significant impact in this study is due to the weather conditions witnessed over the course of the fieldwork, with a sustained high water table resulting in a primarily waterlogged environment, making oxygen the limiting factor. During the field work period of this investigation levels of rainfall were exceptionally high; average rainfall in October 2012 was 115mm, 11% above normal, with the wettest recorded location in the country was at East Okement Farm in Devon that received 212mm (Flitton, 2012). As a result, by the end of the fieldwork the water table at all but three quadrats was less than 10 cm from the surface (Figure 9). The South West of England is prone to rare periods of severe rainfall lasting for an average of 5 to 15 hours (Met Office, 2007). In Caine's study, burial of strips took place between June 13th and July 28th 2011. Weather conditions at this of year tend to be warmer and drier than September 20th to November 3rd when this current study took place, so rainfall was likely to be less of an issue.

It is not exactly clear whether microbial activity increases as a result of increased soil pH (from hydrolisation of urea), or greater nitrogen availability, or both. The extent of the impact of SSU depends on the plant species affected, the quantity of SSU and the time after application (Rooney *et al.*, 2006). The effects of SSU were examined in a more limited manner in this study, as an equal amount of SSU was applied to each quadrat (250 ml), and all cotton strips were retrieved at the same time, as were the water samples. Rooney *et al.* (2006) also found SSU to cause a significant increase in pH ($P < 0.005$) from pH 5.4 to pH 6.4. If the current study were to be continued pH would be an additional variable to consider.

The SSU also had no statistically significant effect on any of the other variables measured (DOC, POC and PON). DOC concentrations at Redlake ranged from 14.25 to 42.31 mg l^{-1}, well within recorded literature values of 3 - 400 mg l^{-1} in natural peatlands (Strack *et al.*, 2008). Unexpectedly, the mean DOC concentration was higher for quadrats that had not had SSU applied (35.96 ± 4.70 mg l^{-1}) than those that did (27.95 ± 13.52 mg l^{-1}), and the interquartile ranges in Figure 6 show that many of the DOC concentrations for quadrats without SSU than quadrats with SSU. Periods of severe rainfall, such as was witnessed in October 2012, can flush DOC-rich waters out of peatland systems (Strack *et al.*, 2008), which may explain why little difference was seen due to SSU application.

The increase in microbial activity caused by application of SSU has previously been found to cause an increase in DOC concentration (Shand *et al.*, 2000). Williams *et al.* (2000) found microbial activity to also result in the export of CO_2 to the atmosphere. When organic matter is broken down due to microbial activity a proportion of the organic carbon is converted to CO_2 and released into the soil. This causes soil CO_2 concentrations to be high in comparison to the atmosphere; hence CO_2 is emitted to the atmosphere by diffusion (Simmons, 2009). Therefore, if urination increases soil microbial activity it is indirectly contributing to climate change by increasing soil CO_2 concentrations, leading to increased diffusion of CO_2 to the atmosphere.

POC concentrations were also within literature values of 0 - 250 mg l^{-1} (Pawson *et al.*, 2008), ranging from 4.31 – 78.02 mg l^{-1}. Application of SSU was expected to result in substantially higher concentrations of POC. However, as with DOC, concentrations of POC were not significantly different (Figure 7) and quadrats with SSU applied (22.23 ± 22.47 mg l^{-1}) were lower than in quadrats without SSU (36.86 ± 24.60 mg l^{-1}). As POC is a characteristic of degraded peat (Lindsay, 2010), the fact that quadrats with SSU had a lower mean concentration suggests that the SSU did not have detrimental effect in this study.

It was surprising to find that when DOC and POC concentrations were higher, percentage CTSL was less (Figure 14). It was expected that as one variable increased the other would also, as both are regarded as measures of peat degradation.

Concentrations of PON were higher for quadrats which had not had SSU applied (5.10 ± 4.43 mg l^{-1}) compared to quadrats which had (2.86 ± 3.74 mg l^{-1}). This was not expected as it was thought the urea in the SSU would have resulted in higher N concentrations where it had been applied, as total urine-N uptake by plants has been found to only be 11-35% (Clough *et al.*, 1996), meaning the remainder stays in the soil. The same study found plant uptake of N was greater where there was a water table, and the results from this study agree, as PON concentrations were lower when the water table was higher. PON and POC concentrations showed a statistically significant positive correlation. It was visibly clear that some water samples were darker and cloudier than others, so in general the amount of particulate matter in those samples was higher (Figure 16).

It was thought a higher water table would show a reduction in CTSL due to a lack of oxygen limiting microbial activity. However, Figure 10 shows a weak trend (P = 0.344) showing the reverse.

As the depth of the water table below ground level increased higher concentrations of the other three variables (DOC, POC and PON) were found (Figures 11, 12 and 13 respectively). This is most likely due to the anaerobic environment caused by waterlogging preventing degradation of the peat. However, none of the correlations observed were statistically significant. At quadrat 6 the water table was only 3 cm from the surface, and yet the DOC concentration was equally as high as quadrats 18 and 20 where the water was 17.9 and 15.8 cm from the surface respectively. As expected, when DOC and POC were plotted against each other there was a positive correlation (Figure 15), as they are both comprised of the suspended and colloidal particles, the main difference simply being the size of filter used to define them. Given this, it was thought the relationship shown may have been more significant than was found (P = 0.489).

Worral *et al.* (2007a) suggests that DOC release may potentially be controlled by stabilising the water table depth, and possibly increasing it, and the results from this study agree with the statement. It was found the lowering of the water table resulted in increased DOC concentrations due to increased phenol oxidase activity in aerobic conditions. Surprisingly, this does not agree with Pastor *et al.* (2003) who found similar DOC concentrations when the water table was at the surface (77.3 ± 13.8 mg l^{-1}) and when it was at 20 cm below ground level (79.9 ± 18.5 mg l^{-1}). The same study found where water output from peatland is above 150 l m^{-2} $year^{-1}$ the DOC budget changed from a net import to a net export. So, larger catchments with a higher water output are likely to have greater losses of DOC, with higher concentrations found in river waters (Aitkenhead *et al.*, 1999).

Recently increasing levels of DOC concentrations have been observed in water draining from peatlands across the UK (Wallage *et al.*, 2006; Lindsay, 2010), with the amount doubling in the last 30 years (Labadz *et al.*, 2010). Evans *et al.* (2005) found mean DOC concentrations had risen by 91% between 1988 and 2003. This is of concern with regard to human health, for when DOC is chlorinated during water treatment by-products can be produced (carcinogenic trihalomethane (THM) compounds) which may be harmful (Chow *et al.*, 2003). DOC also mobilises metals and pollutants, as well as reducing light penetration in streams (Wallage *et al.*, 2006). The EU Directive 80/778/EEC has set a drinking water standard, requiring water colour to be 20 Hazen Units (Turner *et al.*, 2013). There is an economic cost to higher DOC concentrations; treatment costs will rise due to an increase in coagulant usage, filter backwashing and the amount of sludge produced (Peacock *et al.*, 2013). In terms of the carbon budget, higher DOC concentrations suggest a decrease in uptake and storage of carbon. This increase has been put down to a number of possible reasons: increasing temperatures, enzyme activity triggered by drought, increasing atmospheric CO_2, interactions between CO_2 and temperature, and deposition of atmospheric nitrogen and sulphur (Clark *et al.*, 2009). Changes in water discharge may affect DOC export as well (Pastor *et al.*, 2003).

Despite this general trend, Worrall *et al.* (2007b) found a decrease in DOC concentration over a period of at least 10 years in 14% of 315 catchments, most of which were in the South West of England. This is positive news for areas such as Dartmoor. There is still much uncertainty over the main driver of DOC concentrations, so further studies into the issue are required. Consideration must also be given to the origin of fluvial DOC, as it may come from a number of sources such as deep peat decomposition, sewage, industrial discharges and early plant decomposition (Yallop *et al.*, 2010).

All peatlands are vulnerable to the impacts of climate change, with ombrotrophic bogs being the most threatened (Essl *et al.*, 2012), such as Redlake. However, in recent years rainfall has become increasingly unpredictable, and if the frequency and severity of intense precipitation increases this may benefit peatlands by providing recovery periods which help maintain the water table. Currently the majority of predictions about the future of peatlands are negative, and in general the effect is predicted to be a reduction in locations suitable for peatlands in the UK. As the climate warms, peatlands in the south such as on Exmoor, Dartmoor and the Somerset Levels are expected to become increasingly at risk. The primary climatic variable that affects distribution of bog ecosystems is summer temperatures; this causes prolonged periods when the water table is low and evapotranspiration is greater, leading to greater risk of natural fires and erosion (Essl *et al.*, 2012). Intense rainfall events may help buffer against this by restoring the water table, as peatlands that are hydrologically intact are the most resilient. If at different times of the year the ground is too wet for livestock then farmers will need to start re-thinking stocking patterns, possibly reducing the length of time livestock are present which will also benefit peatlands.

The Intergovernmental Panel on Climate Change (IPCC) climate scenarios predict minimal changes to annual precipitation across central Europe, with Northern Europe experiencing a 1-2% increase per decade and decreases of no more than 1% in Southern Europe (IPCC, 2013). A study by Essl *et al.* (2012) examined the potential impacts of climate change on various mire ecosystem types in Austria based on four of the IPCC's climate scenarios (A1F1, A2, B1 and B2). Under the B1 scenario bog ecosystems would suffer a loss of 59% by 2051-2060, and as much as 84% by 2081-2090 according to the A1 scenario. Table 2 shows that no matter which IPCC scenario is considered, the projected losses for the coming decades are severe.

Table 2: Projected loss of mire ecosystem based on four IPCC climate change scenarios (Essl *et al.*, 2012).

	Projected loss in % (2051–60)				Projected loss in % (2081–90)			
IPCC Scenario	A1	A2	B1	B2	A1	A2	B1	B2
Bog	56	48	59	48	84	60	72	51

Continuous work is being carried out on the restoration and preservation of peatlands to protect them from degradation from climate change, as well as the current anthropogenic threats. On Dartmoor and Exmoor the Mires on the Moors project has a strong emphasis on peatland hydrology. Part of this is the Dartmoor Mires Project which is co-ordinated by Dartmoor National Park Authority (DNPA), and involves a partnership of South West Water (SWW), the Environment Agency (EA), Natural England (NE), Dartmoor Commoners' Council and the Duchy of Cornwall. The Dartmoor Mires Project focusses specifically on blanket bog, such as Redlake, which is a UK priority Biodiversity Action Plan habitat with global importance. The project's restoration focusses on drain blocking, which can be done by various methods. Complete infilling of drains is an effective method, using peat from elsewhere. The more common technique is the implementation of dams at regular intervals; this still allows surface water flow, but it traps sediment and allows the peat to build up gradually (Holden *et al.*, 2011). This has proved to be an effective method in reducing DOC concentrations in soil waters (Wallage *et al.*, 2006), and Worrall *et al.* (2013) found statistically significant reductions in DOC export of 2.2 to 9.2% as a result of drain blocking. Prevention of peat drainage is essential as it is one of the highest CO_2 emitting forms of land use (Morrison *et al.*, 2013). Figure 17 shows an example of drain blocking preventing water loss.

Figure 17: An example of drain blocking (Natural England, 2010).

Drain blocking is also of great importance in the Peak District National Park (PDNP). Here gullies are as much as 8 m deep, mostly caused by overgrazing, fire and pollution from industry (Holden *et al.*, 2005). Research into effective drain blocking techniques found wood and stone to be the materials most effective in aiding sedimentation, providing they were built in close proximity. Dams were further strengthened by sowing cotton grass seed to help bind trapped sediments. Economic efficiency must also be considered when designing restoration programmes. Blocking every gully would be expensive, so detailed mapping of the area allows gullies and surface water pathways to be cross-examined and the gullies identified where blocking would be most beneficial (Holden *et al.*, 2005).

Other peat protection projects include the Peatscapes and Moors for the Future projects. These focussed on restoration techniques, communication of knowledge, and establishing relationships with stakeholders. If the private sector begin to invest in management projects with the intention of meeting their own carbon targets it would also have positive impacts on the national targets (Natural England, 2010).

Until now, most peat management schemes are aimed at habitat preservation and not carbon stewardship. With a growing low carbon economy and carbon trading schemes now in place there is now a financial incentive to maintain peatland. There is now the Rural Economy and Land Use programme (RELU) for sustainable uplands which has looked at land use policies in the PDNP and has created a carbon flux model which considers all inputs and outputs of carbon in fluvial and gaseous forms (Worrall *et al.*, 2009). Further research is still required before the emissions benefits of peat preservation can be given an accurate value. New research by Fry (2013) estimated the 40 ha of blanket bog at Redlake to have a CO2 sequestration rate of 2.26 ± 1.31 t ha^{-1} $year^{-1}$. Given this, Redlake has the capacity to sequester 90.4 t ha^{-1} $year^{-1}$ and, based on a carbon credit market value of £2.63 (April 2013), gives a monetary value to the site of £237 in terms of carbon credit value. However, another study at Redlake by Coombe (2012) states a CO_2 sequestration rate of 10.5 ± 1.24 t ha^{-1} $year^{-1}$, equating to a far greater monetary value of £1105.

Despite many peatlands having a CO_2 sequestration potential, where degradation is severe peat can release significant amounts of CO_2 into the atmosphere as mentioned previously. At one heavily cultivated peatland in East Anglia CO_2 release was found to be 20 - 25 t ha^{-1} $year^{-1}$ (Morrison *et al.*, 2013), so losses here are significantly greater than the estimated quantity sequestered at Redlake. The damage was caused by not only draining of the land, but also fertilisation for use for arable farming.

In the PDNP it is predicted that in the areas studied that in 2036 the peat will change from a sink to a source of carbon and begin emitting CO_2; Figure 18 shows how mixed management schemes could prevent this occurring for an additional 55 years until 2091 (Worrall *et al.*, 2010). Of the other individual actions that may be taken to prevent peat degradation, stopping grazing was the most efficient, showing the significance of the presence of livestock.

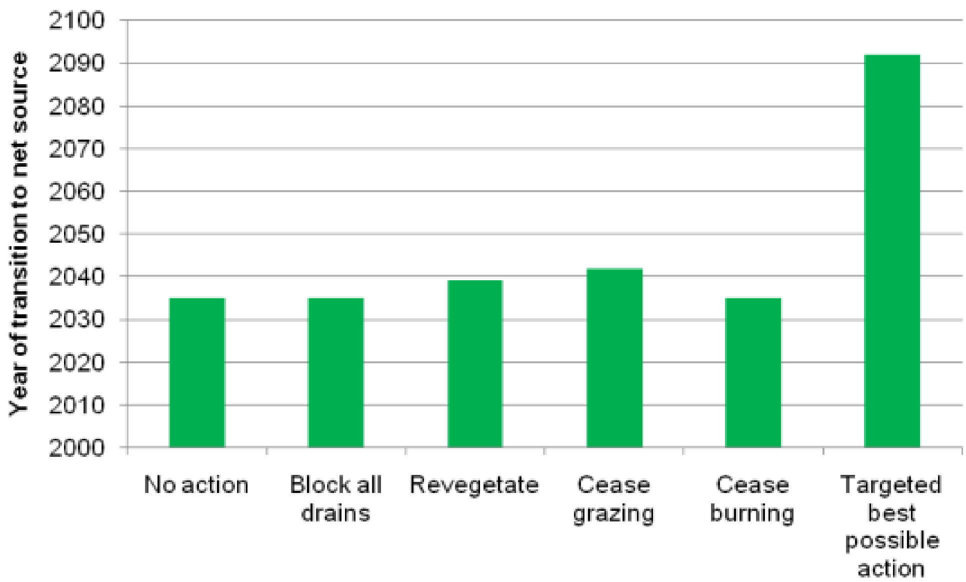

Figure 18: Estimated date that studied peat in Peak District will become a carbon source, rather than a sink, under different management (Worrall *et al.*, 2010).

The general awareness of peatland preservation has been highlighted by recent amendments to the Kyoto Protocol from the Cancun Conference in 2010 which recognise the carbon sequestration potential of peat. During the second commitment period (2013 to 2020) Annex 1 Parties, of which the UK is one, can now account for GHG removal by land management programmes such as wetland rewetting, following changes to the land use, land-use change and forestry section of the protocol (UNFCCC, 2011).

Future Work and Improvements

This study was carried out for a predetermined 45 day period, with only one set of water samples taken on the 3rd of November. Therefore extrapolating the results to draw general conclusions about peatlands must be done with caution. Ideally, cotton strips would be buried at different times of the year, with water samples taken on a regular basis to monitor surface waters taking into account annual variations. Some of the maximum tensile strength results for retrieved strips were higher than the 5 control strips, so in future a larger set of control strips would help discount any error in the control strips.

To study further the relative significance of urination and water table depth on peat degradation a similar investigation could be carried out where SSU is applied to peat mesocosms with controlled known water table heights.

Despite measuring the change in the water table depth from the start to the end of the fieldwork, the study could have been improved by examining the hydrology of Redlake further. Standard and tipping bucket rain gauges would have provided useful information about the precipitation at the site, and more regular recording of water table depth would show any fluctuations over the fieldwork. It is expected that if more extensive fieldwork were carried out the correlation between water table depth and the variables measured would become more apparent. Water samples could be taken a number of times during the study period for replication, and more than one sample taken from each quadrat. This would improve the accuracy of results, and also provide backup samples in case of sample loss due to damaged equipment, as occurred during laboratory analysis in this study.

This study only looked at DOC concentrations in water samples, and the results suggest that if a peat system is a net importer/sink of DOC it is good for carbon budget. However, gaseous C was not investigated, and Pastor *et al.* (2003) found that gaseous C release in the form of CO_2–C and CH_4–C increased *only* when the DOC budget changed from net export to net import. Therefore in future to fully understand the export of carbon more, if not all, forms of carbon release should be studied.

Conclusion

The cotton strip assay method used in this study was an effective technique in measuring cellulose decomposition. However, when it came to retrieving the buried cotton strips the weather conditions meant that some strips were lost and therefore a full analysis of quadrats at Redlake was not possible.

Livestock urination can potentially have damaging effects on active peatlands by increasing microbial activity, and subsequently may contribute towards climate change via diffusion of soil CO_2 to the atmosphere (Simmons *et al.*, 2009). In this study the application of synthetic sheep urine resulted in a slight increase in microbial activity and hence cellulose decomposition was greater. However, the negative effects of urination may be reduced if a high water table is maintained. The resulting anaerobic environment means oxygen is a limiting factor restricting phenol oxidase activity and reducing degradation. Waterlogged conditions also restrict carbon release from peatlands; DOC and POC concentrations were lower in areas where the water table was closer to the surface. The severe wet weather during the fieldwork meant the impact of urination on carbon release was not significant.

Peatland that is hydrologically intact is the most stable (Essl *et al.*, 2012), and in this study was the reason for the limited detrimental effects of SSU. Consideration must be given to peatlands where the water table is lower, for natural or anthropogenic reasons, as it is in these areas where urination will have a greater effect. Further studies are required at different times of the year to assess annual variations in the effects of urination under different weather conditions. If long term climate change predictions are correct and peatlands in the south of the UK experience warming in the coming decades, prolonged dry periods may reduce the water table and increase the vulnerability of peat to the detrimental effects of livestock urination. However, heavy rainfall events are common in the South West and if the unpredictability of the weather over the last few years continues this may provide recovery periods allowing the water table to recharge.

The findings agree with previous studies (Worral *et al.*, 2007a; Strack *et al.*, 2008) which have found water table height to be a limiting factor with regards to DOC release. If the water table in peatlands can be maintained, if not improved, it has the potential to enhance the role of peat as a carbon sink, or at least ensure peatlands remain a carbon sink for as long as possible. This emphasises the importance of peatland hydrology in management programmes; current and future projects should prioritise the preservation of high water tables by methods such as drain blocking and grip infilling, and also carefully monitor water export, to ensure drainage does not occur wherever possible.

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References

Aitkenhead, J.A., Hope, D., Billet, M.F. (1999) The relationship between dissolved organic carbon in stream water and soil organic carbon pools at different spatial scales. *Hydrological Processes,* 13: 1289–1302.

Bridgham, S.D., Pastor, J., Dewey, B., Weltzin, J.F., Updegraff, K. (2008) Rapid carbon response of peatlands to climate change. *Ecology.* 89: 3041-3048.

Caine, S. (2011) The effect of simulated sheep grazing upon microbial activity in active upland peat soil. University of Plymouth: MSc Environmental Consultancy Thesis.

Charman, D. (2002) *Peatlands and Environmental Change*. John Wiley & Sons Ltd, Chichester.

Chow, A.T., Tanji, K.K., Gao, K.K.T. (2003) Production of dissolved organic carbon (DOC) and trihalomethane (THM) precursor from peat soils. *Water Research,* 37: 4475–4485.

Clark, J.M., Chapman, P.J., Adamson, J.K., Lane, S.N. (2005) Influence of drought-induced acidification on the mobility of dissolved organic carbon in peat soils. *Global Change Biology,* 11 (5): 791–809.

Clark, J.M., Ashley, D., Wagner, M., Chapman, P.J., Lane, S.N., Evans, C.D., Heathwaite, A.L. (2009) Increased temperature sensitivity of net DOC production from ombrotrophic peat due to water table draw-down. *Global Change Biology,* 15: 794-807.

Clay, G.D., Worrall, F., Fraser, E.D.G. (2009) Effects of managed burning upon dissolved organic carbon (DOC) in soil water and runoff water following a managed burn of a UK blanket bog. *Journal of Hydrology,* 367: 41-51.

Clough, T.J., Sherlock, R.R., Cameron, K.C., Ledgard, S.F. (1996) Fate of urine nitrogen on mineral and peat soils in New Zealand. *Plant and Soil,* 178: 141-152.

Coombe, R. (2012) Methodological analysis of peat core dating to establish rates of peat growth and carbon accumulation: Red Lake mire, Dartmoor, UK. *Journal of Environmental Management*: 1-55.

Correll, R.L., Harch, B.D., Kirkby, C.A., O'Brien, K., Pankhurst, C.E. (1997) Statistical analysis of reduction in tensile strength of cotton strips as a measure of soil microbial activity. *Journal of Microbial Methods,* 31: 9-17.

Daniels, S.M., Agnew, C.T., Allott, T.E.H., Evans, M.G. (2009) Water table variability and runoff generation in an eroded peatland, South Pennines, UK. *Journal of Hydrology,* 361: 214-226.

Dawson, J.J.C., Billett, M.F., Neal, C., Hill, S. (2002) A comparison of particulate, dissolved and gaseous carbon in two contrasting upland streams in the UK. *Journal of Hydrology,* 257 (1-4): 226-246.

DEFRA (Department for Environment, Food and Rural Affairs) (2005) *Review of the Heather and Grass etc. (Burning) Regulations 1986 and the Heather and Grass Burning Code 1994 in England*. Department of Environment, Food & Rural Affairs. London.

DNPA (Dartmoor National Park Authority) (2004) *Bogs and Wetlands Factsheet*. [Online] Available at: http://www.dartmoor-npa.gov.uk/__data/assets/pdf_file/0015/41244/lab-bogs.pdf (Accessed 22 December 2012).

Essl, F., Dullinger, S., Moser, D., Rabitsch, W., Kleinbauer, I. (2012) Vulnerability of mires under climate change: implications for nature conservation and climate change adaption. *Biodiversity and Conservation,* 21 (3): 655-669.

Evans, C.D., Monteith, D.T., Cooper, D.M. (2005) Long-term increases in surface water dissolved organic carbon: observations, possible causes and environmental impacts. *Environmental Pollution,* 137: 55–71.

Flitton, R. (2012) *The Weather in October. The Guardian Sunday 4 November 2012*. [Online] Available at: http://www.guardian.co.uk/uk/2012/nov/04/weather-temperature-rainfall-sunshine (Accessed 1 February 2013).

Freeman, C., Ostle, N.J., Fenner, N., Kang, H. (2004) A regulatory role for phenol oxidase during decomposition in peatlands. *Soil Biology and Biochemistry,* 36 (10): 1663-1667.

French, D.D. (1988) Patterns of decomposition assessed by the use of litter bags and cotton strip assay on fertilized and unfertilized heather moor in Scotland. In: Harrison, A. F., Latter, P.M., Walton, D.W.H., (eds.) *Cotton strip assay: an index of decomposition in soils*. Grange-over-Sands, NERC/ITE, 100-108. (ITE Symposium, 24).

Fry, C. (2013) An Investigation into the Productivity and Carbon Content of *Sphagna* on Red Lake, Dartmoor. University of Plymouth. BSc Environmental Science Dissertation.

Gogo. S., Pearce, D.M.E. (2009) Carbon, cations and CEC: Interactions and effects on microbial activity in peat. *Geoderma,* 153: 76-86.

Gogo, S., Albéric, P., Laggoun-Défarge, F., Binet, S., Aurouet, A. (2012) Spatial and temporal variations of dissolved organic carbon and inorganic carbon concentrations and $\delta^{13}C$ in a peatland-stream continuum: implications of peatland invasion by vascular plants. *Biogeosciences Discuss,* 9: 3515-3544.

Harris, H. (1986) *Industrial Archaeology of Dartmoor*. 3rd Edition. David & Charles Inc. North Pomfret (Vt).

Hicks, S. (2012) An Investigation into the Relationship between Vegetation and Decomposition in Peatlands. University of Plymouth: MSc Environmental Consultancy Thesis.

Holden, J., Burt, T. P. (2003) Hydrological studies on blanket peat: the significance of the acrotelm-catotelm model. *Journal of Ecology,* 91: 86-102.

Holden, J., Trotter, S., Hodson, S., Lindup, S., Milner, S., McHale, S., Worman, C., Flitcroft, C., Buckler, M., Bonn, A., Evans, M., Allot, T., Crowe, S., Liddaman, L., Hobson, G., Irvine, B., Maxfield, E., James, T. (2005) Understanding gully blocking in deep peat. *Geophysical Research Abstracts,* 7.

Holden, J., Wallage, Z.E., Lane, S.N., McDonald, A.T. (2011) Water table dynamics in undisturbed, drained and restored blanket peat. *Journal of Hydrology,* 402 (1-2): 103-114.

Howard, P.J.A. (1988) A critical evaluation of the cotton strip assay. In: Harrison, A.F., Latter, P.M., Walton, D.W.H., (eds.) *Cotton strip assay: an index of decomposition in soils*. Grange-over-Sands, NERC/ITE, 34-42. (ITE Symposium, 24).

IPCC (Intergovernmental Panel on Climate Change) (2013) *Working Group II: Impacts, Adaptation and Vulnerability: 13.1.5. Climate Scenarios for the Future*. [Online] Available at: http://www.ipcc.ch/ipccreports/tar/wg2/index.php?idp=495 (Accessed 25 March 2013).

IUCN (International Union for Conservation of Nature) (2011) *Hidden benefits of peatlands revealed by new inquiry. IUCN UK Committee: Peatland Programme*. [Online] Available at: http://www.iucn-uk-peatlandprogramme.org/news/189 (Accessed 6 October 2012).

JNCC (Joint Nature Conservation Committee) (2011) *Towards an assessment of the state of UK Peatlands. JNCC Report* 445 [Online] Available at: http://jncc.defra.gov.uk/pdf/jncc445_web.pdf (Accessed 1 October 2012).

Kalbitz, K., Geyer, S. (2002) Different effects of peat degradation on dissolved organic carbon and nitrogen. *Organic Geochemistry,* 33 (3): 319-326.

Labadz, J., Allott, T., Evans, M., Butcher, D., Billett, M., Stainer, S., Yallop, A., Jones, P., Innerdale, M., Harmon, N., Kieron, M., Bradbury, R., Mount, D., O'Brien, H., Hart, R. (2010) *Peatland Hydrology: Draft Scientific Review*. IUCN UK Peatland Programme.

Lance, A.N. (1983) Performance of sheep on unburned and serially burned blanket bog in Western Ireland. *Journal of Applied Ecology,* 20: 767–775.

Latter, P.M., Bancroft, G., Gillespie, J. (1988) Technical aspects of the cotton strip assay in soils. *International Biodeterioration,* 24: 25-47.

Limmer, A.W., Steele, K.W. (1983) Effect of cow urine upon denitrification. *Soil biology & biochemistry,* 15 (4): 409-412.

Lindsay, R. (2010) *Peatbogs and Carbon: A Critical Synthesis. RSPB Scotland*. [Online] Available at: http://www.rspb.org.uk/Images/Peatbogs_and_carbon_tcm9-255200.pdf (Accessed 13 November 2012).

Lunt, P., Allott, T., Anderson, P., Buckler, M., Coupar, A., Jones, P., Labadz, J., Worrall, P. (2010) *Peatland Restoration: Scientific Review*. IUCN UK Peatland Programme.

Maddock, A. (2008) *UK Biodiversity Action Plan; Priority Habitat Descriptions*. JNCC.

Met Office (2007) *South West England: climate*. [Online] Available at: http://www.metoffice.gov.uk/climate/uk/sw/print.html (Accessed 13 November 2012).

Milne, R., Brown, T.A. (1997) Carbon in the vegetation and soils of Great Britain. *Journal of Environmental Management,* 49: 413-433.

Morrison, R., Cumming, A.M.J., Taft, H.E., Kaduk, J., Page, S.E., Jones, D.L., Harding, R.J., Balzter, H. (2013) Carbon dioxide fluxes at an intensively cultivated temperate lowland peatland in the East Anglian Fens, UK. *Biogeosciences Discuss,* 10: 4193–4223.

Natural England (2010) England's peatlands: Carbon storage and greenhouse gases. *Natural England Publications and Products*. NE257.

Ohlson, M., Økland, R.H. (1998) Spatial variation in rates of carbon and nitrogen accumulation in a boreal bog. *Ecology,* 79 (8): 2745-2758.

Parish, F., Sirin, A., Charman, D., Joosten, H., Minayeva, T., Silvius, M. (2008) Assessment of Peatlands, Biodiversity, and Climate Change. *National Wetlands Newsletter,* 30 (1): 13-17.

Partoazar, M., Hoodaji, M., Tahmourespour, A. (2011) The effect of glyphosate application on soil microbial activities in agricultural land. *African Journal of Biotechnology,* 11 (83): 19419-19424.

Pastor, J., Solin, J., Bridgham, S.D., Updegraff, K., Harth, C., Weishampel, P., Dewey, B. (2003) Global warming and the export of dissolved organic carbon from boreal peatlands. *Oikos.* 100 (2): 380-386.

Pawson, R.R., Evans, M.G., Allott, T.E. (2006) The role of particulate organic carbon (POC) in the carbon cycle of degrading upland peat systems. *Geophysical Research Abstracts,* 8.

Pawson, R.R., Lord, D.R., Evans, M.G., Allot, T.E.H. (2008) Fluvial organic carbon flux from an eroding peatland catchment, southern Pennines, UK. *Hydrology and Earth System Sciences,* 12: 625–634.

Peacock, M., Burden, A., Cooper, M., Dunn, C., Evans, C.D., Fenner, N., Freeman, C., Gough, R., Hughes, S., Jones, T., Lebron, I., West, M., Zieliński, P. (2013) Quantifying dissolved organic carbon concentrations in upland catchments using phenolic proxy measurements. *Journal of Hydrology,* 477: 251-260.

Preston, M.D., Eimers, C.M., Watmough, S.A. (2011) Effect of moisture and temperature variation on DOC release from a peatland: Conflicting results from laboratory, field and historical data analysis. *Science of the Total Environment,* 409: 1235-1242.

Ratcliffe, D.A., Oswald, P.H. (1988) In: Clay, G.D., Worrall, F., Fraser, E.D.G., 2009. Effects of managed burning upon dissolved organic carbon (DOC) in soil water and runoff water following a managed burn of a UK blanket bog. *Journal of Hydrology,* 367: 41-51.

Reeves, P.N., Champion, P.D. (2004) *Effects of Livestock Grazing on Wetlands: Literature Review*. National Institute of Water & Atmospheric Research Ltd. Client Report HAM2004-059.

Rice, S.K. (2000) Variation in carbon isotope discrimination within and among *Sphagnum* species in a temperate wetland. *Oecologia,* 123: 1-8.

Rooney, D., Kennedy, N., Deering, L., Gleeson, D., Clipson, N. (2006) Effect of sheep urine deposition on the bacterial community structure in an acidic upland grassland soil. *Applied and Environmental Microbiology,* 72 (11): 7231-7237.

Roulet, N., Moore, T.R. (2006) Environmental chemistry: browning the waters. *Nature,* 444: 283–284.

Schiff, S., Aravena, R., Mewhinney, E., Elgood, R., Warner, B., Dillon, P., Trumbore, S. (1998) Precambrian shield wetlands: Hydrologic control of the sources and export of dissolved organic matter. *Climate Change,* 40 (2): 167-188.

Selwyn, E. (2012) Mapping 'active' peatland by vegetation composition on Red Lake mire and identifying the vegetational attributes of active peatlands. University of Plymouth: MSc Environmental Consultancy Thesis.

Shand, C.A., Williams, B.L., Smith, S., Young, M.E. (2000) Temporal changes in C, P and N concentrations in soil solution following application of synthetic sheep urine to a soil under grass. *Plant and Soil,* 222: 1-13.

Simmons, J.A. (2009) Decomposition and Soil CO_2 Emission. *Teaching Issues and Experiments in Ecology,* 6.

Strack, M., Waddington, J.M., Bourbonniere, R.A., Buckton, E.L., Shaw, K., Whittington, P., Price, J.S. (2008) Effect of water table drawdown on peatland dissolved organic carbon export and dynamics. *Hydrological Processes* John Wiley & Sons DOI: 10.1002/hyp6931.

Tanentzap, A.J., Coomes, D.A. (2012) Carbon storage in terrestrial ecosystems: do browsing and grazing herbivores matter? *Biological Reviews,* 87: 72-94.

Thomas, R.J., Logan, K.A.B., Ironside, A.D., Bolton, G.R. (1988) Transformations and fate of sheep urine-N applied to an upland U.K. pasture at different times during the growing season. *Plant and Soil,* 107 (2): 173–181.

Turner, E.K., Worrall, F., Burt, T.P. (2013) The effect of drain blocking on the dissolved organic carbon (DOC) budget of an upland peat catchment in the UK. *Journal of Hydrology,* 479: 169-179.

UNFCCC (United Nations Framework Convention on Climate Change) (2011) *Consideration of further commitments for Annex I Parties under the Kyoto Protocol*.

[Online] Available at: http://unfccc.int/resource/docs/2011/awg16/eng/l03a02.pdf (Accessed 20 March 2013).

Wallage, Z.E., Holden, J., McDonald, A.T. (2006) Drain blocking: An effective treatment for reducing dissolved organic carbon loss and water discolouration in a drained peatland. *Science of the Total Environment,* 367: 811-821.

Williams, B.L., Grayston, S.J., Reid, E.J. (2000) Influence of synthetic sheep urine on the microbial biomass, activity and community structure in two pastures in the Scottish uplands. *Plant and Soil,* 225: 175-185.

Worrall, F., Reed, M., Warburton, J., Burt, T.P. (2003) Carbon budget for a British upland peat catchment, *Science of the Total Environment,* 312: 133-146.

Worrall, F., Armstrong, A., Adamson, J.K. (2007a) The effects of burning and sheep-grazing on water table depth and soil water quality in an upland peat. *Journal of Hydrology,* 339: 1-14.

Worrall, F., Guilbert, T., Besien., T. (2007b) The flux of carbon from rivers: the case for flux from England and Wales. *Biogeochemistry.* 86: 63-75.

Worrall, F., Evans, M.G., Bonn, A., Reed, M.S., Chapman, D., Holden, J. (2009) Can carbon offsetting pay for upland ecological restoration? *Science of the Total Environment.* 408: 26-36.

Worrall, F., Chapman, P., Holden, J., Evans, C., Artz, R., Smith, P., Grayson, R. (2010) *Peatlands and Climate Change: Scientific Review.* IUCN UK Peatland Programme.

Worrall, F., Clay, G.D. (2012) The impact of sheep grazing on the carbon balance of a peatland. *Science of the Total Environment,* 438: 426-434.

Yallop, A.R., Clutterbuck, B., Thacker, J. (2010) Increases in humic dissolved organic carbon export from upland peat catchments: the role of temperature, declining sulphur deposition and changes in land management. *Climate Research,* 45: 43-56.

Zsolnay, A., 1996. In: Kalbitz, K., Geyer, S. (2002) Different effects of peat degradation on dissolved organic carbon and nitrogen. *Organic Geochemistry,* 33 (3): 319-326.