Assessing the Impact of Peat Bog Restoration in Mitigating Carbon Loss by Upland Erosion

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Declaration

I confirm that no part of the material presented in this thesis has previously been submitted by me or any other person for a degree in this or any other university. In all cases, where it is relevant, material from the work of others has been acknowledged.

Abstract

Demonstrating the impact of peat bog restoration in mitigating carbon loss by upland erosion requires careful field monitoring. This thesis presents results of a year-long field monitoring project at Flow Moss, a 7 ha area of eroding upland blanket bog in the North Pennines, UK. The aim of the project was to estimate the size of the carbon store at Flow Moss, identify the main drivers and pathways through which peat and carbon were leaving the site, and investigate the effectiveness of restoration methods in reducing peat loss. Three main approaches were used:

1) A subsurface Ground Penetrating Radar (GPR) survey to quantify peat depths. This was coupled with results from peat core analysis (loss on ignition, bulk density, total organic carbon and heavy metal analysis of 298 peat samples) to estimate the amount of peat and carbon stored at Flow Moss.

2) Surface erosion monitoring using sediment traps, fixed pole transects, erosion pins and Terrestrial Laser Scanning (TLS) to establish through which mechanisms peat is eroded and transported. A dGPS survey was implemented and the results compared to historic satellite and UAV imagery to monitor changes in surface vegetation cover.

3) Environmental monitoring of rainfall, wind speed / direction, temperature and water table height. The results are compared with those collected during surface monitoring to identify the drivers of erosion at Flow Moss.

Results show that currently there are 4004 (±0.03) tonnes of carbon stored at Flow Moss, which equates to 572 tonnes per ha. At present this is relatively

stable, but the site is a slight net source of carbon emitting approximately two tonnes or 0.05% of the stored carbon each year. The bare peat flats are actively eroding with 35 tonnes of sediment being transported by wind-related processes annually. High wind speed and high intensity rainfall are the main drivers of erosion at Flow Moss and their effectiveness increases when they occur concurrently. Sediment and carbon loss from the channel system, although small, has significantly decreased (a reduction of 98%) since the start of restoration. This is most likely due to vegetation encroachment from the margins of the bare peat with a reduction in the bare peat area of 21% occurring since 2007 and a reduction of 997 m^2 or 12% occurring since restoration began in 2010. This suggests that that restoration attempts have shown some limited success, however for Flow Moss to become a net carbon store, full re-vegetation of the bare peat is necessary.

Acknowledgements

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1. Introduction and rationale

The observed increase in the temperature of the Earth has been attributed to anthropogenic activities resulting in an increase in the amount of greenhouse gases in the Earth's atmosphere (Petit *et al*., 1999). In the decade preceding 2002, the Earth's climate has warmed by approximately 0.6°C and the 2007 report by the Intergovernmental Panel on Climate Change (IPCC) suggests that temperatures will continue to increase within the range of 1.8 – 4°C by the end of the current century. As a response, the United Nations Framework Convention on Climate Change (UNFCCC, 2004) calls for the "stabilisation of greenhouse gas concentrations". This requires a drastic reduction of global CO_2 emissions (Le Quere *et al*., 2009).

The carbon cycle acts to mitigate climate change through absorbing CO_2 into the oceans and terrestrial biosphere (Archer, 2010). In the terrestrial biosphere, peatlands are one of the largest stores of soil carbon and form a significant component of the carbon cycle (Gorham, 1991), containing almost one third of the global soil carbon store in only 3% of the global land surface (Joosten *et al*., 2012). In addition to this vital function, peatlands deliver a range of ecosystem services that are essential to human well-being, including water purification, climate regulation and recreational opportunities (Kimmel and Mander, 2013). However, it is predicted that under future climate change scenarios, many terrestrial stores of carbon, including peatlands, may cease to act as a carbon sink and switch to become a source of CO_2 to the atmosphere, further exacerbating climate change (Cox *et al*., 2000). Climate change is perhaps the most pressing issue to be faced by mankind (Cantell, 2008) and there is still a large amount of uncertainty about the magnitude of soil carbon feedbacks to the climate system (Lal, 2003). Effective monitoring of peatlands is therefore essential to gain more information about how much carbon is contained within these ecosystems, the loss of carbon from peatlands through erosion and how peatlands can be successfully managed to increase the amount of carbon stored within them (Fyfe *et al*., 2014).

Peatlands are fragile and sensitive landscapes, and this sensitivity may manifest itself through vegetation change or peatland erosion. Bragg and Tallis (2001) suggest that erosion of peatlands may be a response of blanket mire systems to environmental perturbation. In many upland areas where rainfall is the dominant source of water, projected increases in winter precipitation could result in increased erosion (Heathwaite, 1993). Alongside this, climate change may indirectly affect the water table level of the peatland, which in turn will impact on the stability of the peatland ecosystem (Frolking *et al.,* 2011). Peatland vegetation composition may change in response to changes in water table level and the underlying peat may undergo degradation or drying, which could be further exacerbated due to inappropriate land management practices (Bragg and Tallis, 2001).

In recent years, there has been a drive to restore peatlands within the UK (Dixon *et al.,* 2014). One of the motivating factors for peatland restoration is their ability to act as a carbon sink. The UK has one of the most ambitious climate polices, with the 2008 climate change act legally committing the UK government to cut national greenhouse gas emissions by 80% by 2050 in relation to 1990 levels (Lorenzoni and Benson, 2014). The sequestration of carbon through the restoration of peatlands can be used as a way to offset emissions and assist with meeting these targets and the mitigation of anthropogenic driven climate change (Couwenburg, 2011).

1.1 Project aim and objectives

The aim of this project is to assess the role of peat bog restoration in mitigating carbon loss by erosion. Specifically the project aims to estimate carbon storage at Flow Moss, an upland blanket peat bog located in the North Pennines (UK) and investigate the effectiveness of restoration measures in reducing erosion and carbon loss.

The project addresses three key research questions:

1. How can peatland carbon stores be accurately assessed, and what is the local carbon store at Flow Moss?
2. What are the dominant processes driving erosion at Flow Moss?
3. How have restoration methods at Flow Moss impacted on sediment yield and carbon loss from erosion?

These questions will be answered by fulfilling the following five objectives:

1. Sample the local peat to establish the variation in bulk density of the peat samples and assess the carbon content and organic matter content using Loss on Ignition (LOI) and Total Organic Carbon (TOC).
2. Carry out a Ground Penetrating Radar (GPR) survey to assess the spatial variation in the depth of peat. Data from the GPR survey can be used alongside bulk density; Loss on Ignition (LOI) and

Total Organic Carbon (TOC) content to estimate the amount of peat at Flow Moss and the size of the carbon reservoir.

3. Undertake a survey (dGPS) of the vegetation cover at the site and compare results with data collected in April 2011 (UAV survey) and historical imagery (2007) to assess changes in surface vegetation cover.

4. Record meteorological conditions using an Automated Weather Station (AWS): air temperature, rainfall, wind speed / direction, water table height and a time lapse imagery of the surface peat condition. Environmental factors, such as wind and rain, are key drivers of erosion and metrological data will be compared with sediment yield data collected from traps and erosion and deposition data.

5. Construct a sediment budget for Flow Moss and quantify peat transfer at the site using wind erosion traps, erosion pins, fixed pole monitoring, sediment traps and Terrestrial Laser Scanning (TLS). Use sediment budget data alongside carbon content and LOI data to estimate the amount of carbon lost from the peat bog through erosion.

Figure 1.1 outlines the objectives of this project, the methods that will be used to fulfil these objectives and how the methods link to the research questions selected to fulfil the overall aim of the project, which is to assess the impact of peat bog restoration in mitigating carbon loss by erosion.

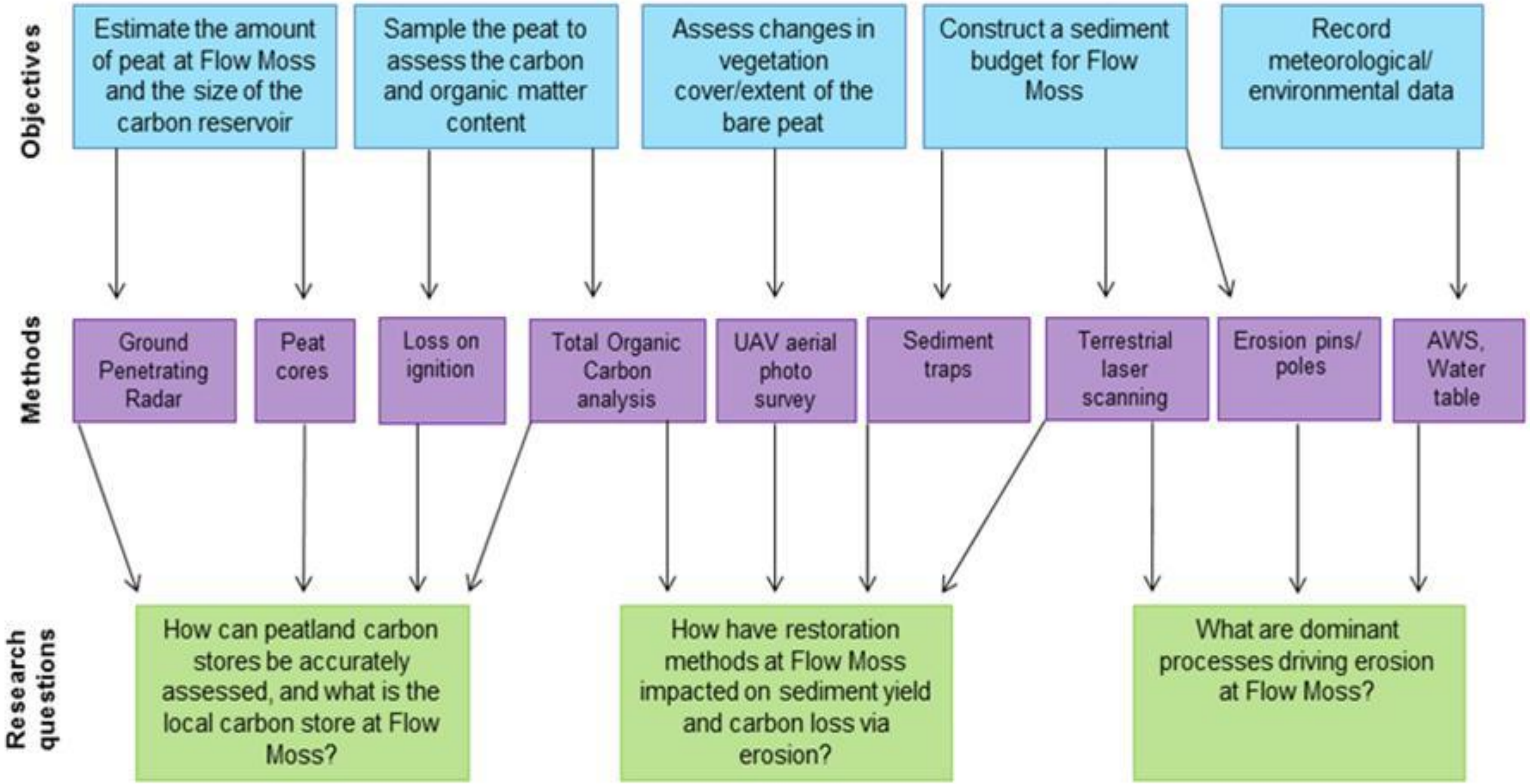


Figure 1.1 - *Project Framework showing how the research objectives, data sources and research questions are linked.*

1.2 Thesis structure

This thesis begins with a critical review of the existing literature (Chapter Two). This discusses peatlands as carbon stores, some of the threats to peatland carbon storage, the use of sediment and carbon budgets to monitor peatlands, and restoration efforts used to mitigate against peatland degradation. Chapter Three describes the location, geology and climate of the field site and Chapter Four evaluates the methods used in this study.

Chapters Five, Six and Seven present the results of this research. Chapter Five provides information relating to subsurface properties of the peat at Flow Moss including bulk density, carbon content and metals concentrations of the peat profile. These are examined to establish how they change with depth of the peat. Results from GPR survey are presented and the effect of sampling design is critically evaluated.

Chapter Six focuses on the results of surface monitoring and identifies the main mechanisms of erosion. Chapter Six includes a comparison of peat surface height changes monitored using two different methods: fixed pole transects and Terrestrial Laser Scanning. The chapter concludes with an assessment of changes in the extent of bare peat at Flow Moss between 2007 and 2014.

The meteorological and environmental conditions during the monitoring period are discussed in Chapter Seven. Data are linked with the information from Chapter Six to identify which processes are the dominant drivers of erosion at Flow Moss.

Chapter Eight collates the results from Chapters Five, Six and Seven and discusses how these results address the main research questions and fulfil the research objectives outlined in Chapter One. Data from chapter Five, Six and Seven are used to construct an annual sediment budget and an annual carbon budget for Flow Moss. Data are used to assess whether restoration measures implemented by the North Pennines AONB peatlands programme are impacting on yields of sediment and carbon lost through erosion; and future threats to the Flow Moss carbon store are discussed.

Finally, Chapter Nine provides a synthesis of the results, conclusions of this study and suggestions for further research.

2. Context: a critical review of peatlands as carbon stores and current restoration efforts

This chapter reviews the literature associated with two key themes. Firstly, research into erosion and carbon budgets, including peatlands as carbon stores, some of the threats to peatlands and examples of how sediment budgets are implemented to identify the main mechanisms of erosion. Secondly, peatland management and restoration are considered, including case studies of UK peatland restoration projects. Finally, there is a discussion of some of the limitations of current restoration efforts.

2.1 Peatlands as a carbon store

Currently, around half of anthropogenic CO_2 emissions are being absorbed by the ocean and terrestrial ecosystems (Archer, 2010). Significantly more carbon is stored within soils, including wetlands and peatlands, than is currently present in the atmosphere (Davidson and Janssens, 2006). Furthermore, soils globally contain about twice the amount of carbon found in the atmosphere and three times the amount found within vegetation (Salazar *et al*., 2011). Peatlands are a particularly important part of the terrestrial biosphere and are estimated to contain almost a third of global soil carbon (Worrall *et al*., 2010). Estimates of the amount of Carbon (*C*) stored within peatlands differ. Gorham (1991) states that this is between 350 and 545 Gt C. Moore (2002) suggests a similar figure of 455 Gt C, whereas Turumen *et al*. (2002) estimate that the total carbon store of all boreal and subarctic mires lies between 270 and 370 Gt C (Table 2.1).

Table 2.1 - *Estimates of carbon stored within global peatlands, ranked by size.*

Authors	Estimate of C stored (GtC)
IUCN (2011)	320
Bridgham *et al*. (2008)	329 – 525
Gorham (1991)	350 – 545
Freeman *et al*., (2004)	390 – 455
Moore (2002)	455
Strack and Zuback (2013)	469-486
JNCC (2011) (http://jncc.defra.gov.uk/page-5905)	~500

It has therefore been suggested that the conservation and restoration of peatlands can contribute significantly towards managing the carbon budget and potentially mitigating climate change (Couwenburg, 2011). This is significant because the terrestrial biosphere and climate system are closely linked (Berher *et al*., 2007) and Lal (2007) states that a direct link exists between soil carbon and atmospheric CO_2 concentrations. Utilising peatlands to sequester carbon could provide a way for countries to reduce emissions and meet targets outlined by the UNFCCC (Worrall *et al*., 2003).

2.2 Peatland carbon budgets

Globally, northern peatlands are the most important terrestrial store of carbon (Worrall *et al*., 2009) and within the UK; peatlands are the largest terrestrial carbon store (Cannell *et al*. 1993) with more than 50% of the UK's soil carbon stored within peatland habitats (Defra, 2009).

Figure 2.1 provides a framework summarising the key components of a peatland carbon budget. The main pathways of carbon loss from peatlands include: dissolved organic carbon (DOC), dissolved inorganic carbon (DIC), particulate organic carbon (POC) and dissolved CO_2 (Worrall and Evans, 2009). The present study focuses mainly on the Particulate Organic Carbon aspect of the carbon budget, but other elements are discussed.

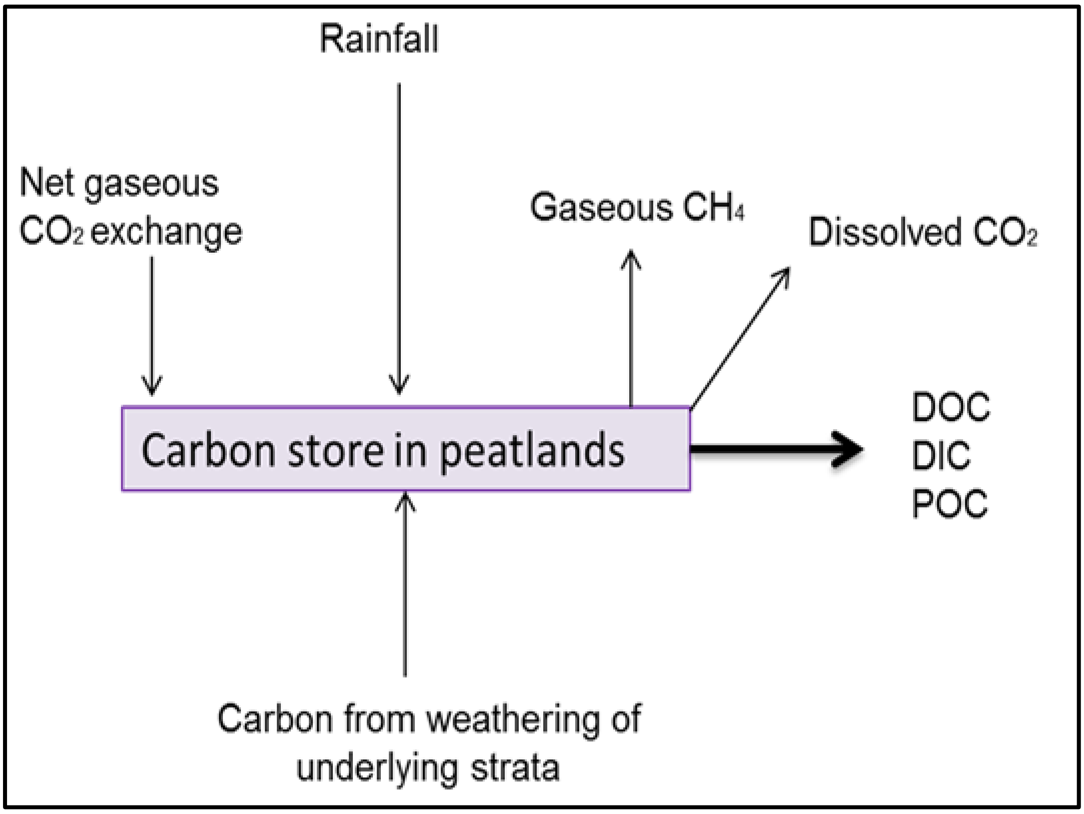


Figure 2.1 - *Schematic summarising the carbon uptake and release pathways for upland peat (adapted from Worrall and Evans, 2009).*

Studies often use two approaches to estimate the carbon budgets of peatlands. These are the dating of peat accumulation using radiocarbon methods to provide a rate of carbon accumulation (RCA); and estimating fluxes of carbon at the surface of peatlands (Worrall *et al*., 2009). However, the first method using carbon dating is criticised by Worrall *et al*. (2010) who state that these techniques are often only capable of measuring the accumulation of peat and cannot be easily used to estimate carbon loss where there is a hiatus in peat accumulation. The impacts of restoration or peatland management are unlikely to be instantaneous and it may be several years before changes in carbon loss occur. Therefore, the RCA method could take decades to provide results that demonstrate any impacts of management or intervention on the carbon balance. The second method of measuring carbon fluxes is also criticised by Worrall *et al*. (2010) as the carbon budget of peatlands has many more factors than just the exchange of CO_2 at the soil surface. Worrall *et al*. (2010) state that currently, only a limited number of studies attempt to measure complete carbon budgets for a peatland and many of the studies that do exist are for intact or pristine peats rather than damaged or restored peats.

Worrall *et al*. (2011) develop methods that combined studies of managed, damaged and restored peat soils in order to assess whether certain management interventions resulted in a benefit for the carbon budget. This work used eight study sites (four re-vegetated and four bare peat) which were located across Bleaklow Plateau in the Peak District. The results identified that there are several management practices that could benefit the carbon budget of peatlands including drain blocking; cessation of managed burning and removing grazing from peatlands. However, although the authors found that peatland management may increase carbon storage, they may also lead to the release of other greenhouse gases. Couwenberg (2011) suggests that while the rewetting of peatlands can cause a decrease of CO_2 an increase of CH_4 may occur. Currently, atmospheric concentrations of CH_4 are far lower than concentrations of atmospheric CO_2 (1840 ppb CH_4 compared with 399 ppm CO_2 (NOAA, 2014), however, CH_4 has a far higher global warming potential than CO_2 and is far more effective at absorbing and re-emitting infrared radiation, thus resulting in additional radiative forcing and positive feedback systems (Reay *et al.,* 2010). Nonetheless, Grand-Clement *et al*. (2013) state that despite the fact the creation of open water pools behind ditch blocks is likely to increase CH_4 emissions in the short-term, methane production is only a temporary stage of peatland restoration that will be mitigated by *Sphagnum* growth within 5-10 years. They argue that overall, CH_4 emissions will be largely compensated by the long-term benefits of restoration including decreased CO_2 emissions and increased C accumulation.

There are several studies which have examined components of the carbon budget from managed or restored peatland sites, but few holistic studies that bring together all the components which may influence the carbon budget of peatlands and this is a key knowledge gap (Worrall *et al*. 2010). The potential impact of carbon losses from the terrestrial biosphere and the impact this will have on CO_2 emissions remains one of the greatest uncertainties in knowledge about global climate change (Quinton *et al*., 2010).There are many future threats that could impact peatlands as carbon stores and quantifying the carbon flux from peatland systems is vital in understanding the global carbon cycle (Evans and Warburton, 2007).

2.3 Threats to peatlands as a carbon store

Current land management practices may have negative or positive impacts on peatlands, affecting their capacity to store carbon (Worrall *et al*. 2010). In the UK, over 80% of peatlands have been impacted by drainage, fire, grazing or peat extraction (Moxey and Moran, 2014). For example, Worrall *et al*. (2009) observed that peat drainage, which lowers the average water table depth allowing greater oxygen ingress, could lead to increased CO_2 respiration. Couwerberg (2011) corroborates this and states that the drainage of peat leads to aeration and decomposition resulting in substantial losses of greenhouse gases to the atmosphere while Joosten (2011) estimates that drainage of peatlands could result in emissions of more than 2 Gt of CO_2 per year globally. Fire also impacts on the peatland carbon cycle. During the last three decades, the use of prescribed moorland burning as a land management tool has increased significantly, and worldwide the controlled use of fire is an essential management tool (Yallop *et al.,* 2006). Nonetheless, fire can have both positive and negative impacts on the peatland carbon cycle. The burning of peatlands and removal of vegetation can initially lead to increased erosion and carbon loss as the surface of the peat becomes more susceptible to erosion from elements such as wind and rain (Holden *et al.,* 2007). However, over a longer time scale, peatland carbon sequestration may increase as older, less productive vegetation is burnt and replaced by new vegetation with a higher NPP.

However, inappropriate land management practices alone are not the only factor influencing the carbon storage capacity of peatlands. Dise (2009) highlights that while anthropogenic activities provide the greatest and most visible threat to peatlands, a far less obvious, but potentially as damaging threat to peatland ecosystems is long term environmental change.

Carbon is sequestered into peatlands through positive net primary productivity (NPP) which occurs when photosynthesis is greater than plant respiration (Charman, 2002). The accumulation of carbon within peatlands is determined by a balance of inputs (plant growth and litter) and outputs (organic matter decomposition) and both of these factors will be influenced by a changing climate (Yu *et al*. 2009). Currently, within the UK, peatlands are thought to be acting as a slight net carbon sink (Holden *et al*., 2007) with more carbon being sequestered than lost. Nevertheless, climate projections (UKCP09) imply that under future climate scenarios, a decrease in summer precipitation and increase in summer temperatures may occur resulting in drought conditions and lower water tables. This could be significant for carbon loss because Bridgham *et al*. (2008) observed a positive linear relationship between carbon accumulation and water-table depth. This is supported by Blodau *et al*. (2004) who state that a reduction in the depth of the water table will lead to an increase in CO_2 emissions as O_2 can penetrate further into the peat column leading to a loss of carbon through oxidation; consequently converting peatlands from a carbon sink to a carbon source (Alm *et al*., 1999). Together with the increased possibility of drought, the 2007 IPCC report suggests that under future climate scenarios, there is likely to be an increase in the number of extreme rainfall events; a factor that can result in increased erosion. Kløve (1998) suggests that one of the main causes of erosion of peat surfaces is intense rainfall. During an extreme rainfall event, the energy from a raindrop falling is transferred to the surface (Morgan, 2005). This leads to detachment by water which occurs through the processes of splash from raindrop impact and scour from surface runoff (Abu-Hamdeh *et al.,* 2006).

Rowson *et al*. (2013) state that net ecosystem respiration (NER) is the largest flux of carbon from peatland ecosystems to the atmosphere and is closely controlled by temperature and water table depth. Increasing temperatures may lead to increased respiration and accelerated decomposition rates which could lead to a positive feedback caused by loss of carbon from peatlands resulting in more atmospheric CO_2 and further accentuating rising temperatures (Luke and Cox, 2011). There is some debate as to the extent of accelerated decomposition and the timescale over which it will operate (Knorr *et al*., 2005), however, Cox *et al*. (2000) propose that, if anthropogenic emissions remain as they are now, the terrestrial biosphere will act as a carbon sink until around 2050, but thereafter will switch to a carbon source.

In addition to atmospheric fluxes, carbon can be lost from peatlands through fluvial systems which drain these areas. The fluvial flux of carbon from a peatland can occur through several carbon pathways (Worrall *et al*., 2003). The main pathways are summarised in Figure 2.1 and include dissolved organic carbon (DOC), dissolved inorganic carbon (DIC), particulate organic carbon (POC) and dissolved CO_2 (Worrall and Evans, 2009).

Freeman *et al.* (2001) note that within the UK, a 65% increase in DOC concentration in freshwater draining from upland catchments has been observed in the 12 years preceding 2001 and Worrall *et al.,* (2014) estimate the total flux of carbon to UK Rivers from the terrestrial biosphere to be 5020 ktonnes C per year or 21.8 tonnes C per km^2 per yr. These increasing concentrations have led to concerns that carbon stores within peatlands are beginning to destabilise. Bardgett (2005) states there are several hypotheses that have been proposed to explain the cause of increased DOC export. One explanation is that increased temperatures over a long time frame have caused increased rates of peat decomposition due to the increased activity of decomposer organisms (Freeman *et al*., 2001, Worrall *et al*., 2004). Another proposed explanation that rising trends in DOC are linked to atmospheric deposition chemistry, with DOC concentrations increasing as the rate of atmospherically deposited sulphur has declined (Monteith et al., 2007). A further possible explanation is that an increase in summer droughts may be causing destabilisation of peatlands (Tipping *et al*., 1999). However, Freeman *et al*. (2004) suggest that an increase in summer droughts is unable to fully account for the observed increased DOC. These authors propose that rather than fluctuating temperatures, it may be an increase in CO_2 that is leading to CO_2 mediated stimulation of primary productivity and this could be the mechanism responsible for observed increases in DOC. Freeman *et al*. (2004) carried out an experiment which demonstrated that when subjected to elevated CO_2 levels, the proportion of DOC from the peat was 10 times that of the control samples. Nonetheless, DOC is not the only export of carbon from fluvial systems and in recent years, there has been an increased interest in the mobilisation of POC. Shuttleworth *et al*. (2014) state that the majority of work examining fluvial carbon exports focus on DOC, with Particulate Organic Carbon (POC) given far less attention. The present study will focus on the POC component of the carbon budget and attempt to establish how restoration methods implemented at Flow Moss have impacted on yields of POC lost though erosion. In a study in the Peak District, Shuttleworth *et al.* (2014) found that POC fluxes are greatly reduced following restoration, reaching levels comparable to intact sites. In order to evaluate the pathways through which POC is leaving a site, a sediment budget approach can be implemented to quantify these mechanisms.

2.4 Sediment budget approach

A sediment budget is defined by Slaymaker (2003) as the accounting of sources, sinks and redistribution of sediments in a unit region over a unit time. The construction of a sediment budget quantifies erosion agents, sediment storage elements and the processes linking these. In order to make informed land management decisions, there is a need to predict how land management practices will change erosion and sedimentation rates (Reid and Dunne, 1996). This information can be used to assign priorities for erosion control and to create effective management plans aimed at limiting impacts associated with erosion (Walling and Collins, 2008). One of the major advantages of the sediment budget approach is that it attempts to provide an integrated overview of a range of processes acting in a catchment or landscape unit rather than just a single factor. This allows an assessment to be made of the linkages between terrestrial and hydrological environments (Walling *et al*. 2002).

Hinderer (2012) suggests that many of the previous studies, attempting to establish sediment loss through erosion, have been completed for mineral soils, but far fewer have attempted to establish soil loss from peatlands. Peatlands are often located in upland areas and in addition to erosion leading to a significant carbon loss (Evans *et al*. 2006); the degradation of peatlands can also impact on patterns of stream flow and erosion in the headwaters of most major UK Rivers (Labadz *et al*. 1991). Furthermore, the erosion of peatlands contaminated with heavy metals may lead to the pollution of rivers and other surface water (Nelson and Booth, 2002, Rothwell *et al.,* 2008a). A sediment budget approach can be used in an attempt to quantify the pathways of sediment transfer and loss; this can address important components of the carbon budget (Figure 2.1), as it can provide information on the storage and movement of peat particulate matter and sediment within peatlands.

2.5 Peatland sediment budgets and the carbon cycle

Peatland erosion has the potential to deliver large sediment yields and can dramatically increase drainage density and the efficiency of slope-channel linkages within a sediment cascade (Evans and Warburton, 2007, Burt and Allison, 2010). Funds assigned to peatland restoration projects are often limited and therefore information is needed on how to spatially prioritise restoration efforts to maximise effectiveness (Glenk *et al.,* 2014). Peatland sediment budgets can be implemented to gain a holistic understanding of the balance of erosion process acting on a particular peatland site and in this respect contribute to management decisions (Evans and Warburton, 2005).

Evans and Warburton (2005) provide one of the best examples of a sediment budget for a peat catchment. A sediment budget was constructed for the Rough Sike peat catchment in Northern England using information relating to sediment transfer on slopes, sediment flux on the floodplain and through the main stream channel and sediment yield at the catchment outlet. This study demonstrated that fluvial suspended sediment flux is controlled by channel processes and that a significant correlation exists between reduction in fluvial suspended sediment yield and re-vegetation of adjacent headwater gullies, suggesting that re-vegetation can be used as a management tool to reduce sediment loss from peat catchments.

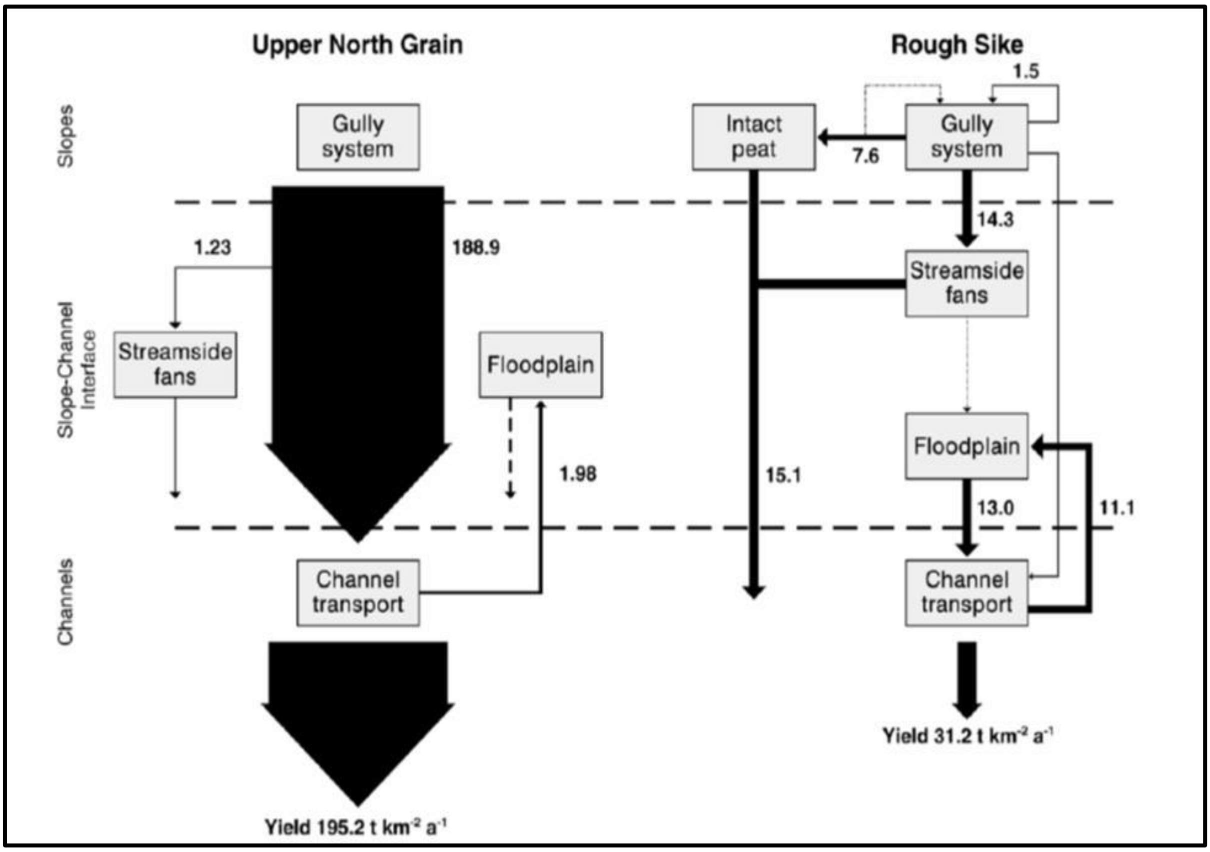


Figure 2.2 - *Schematic displaying the sediment budget for two sites in the North and South Pennines. All units in tonnes km^{-2}a^{-1} (Evans et al., 2006).*

Evans *et al*. (2006) compared the Rough Sike sediment budget for the blanket peat catchment in the North Pennines and with the Upper North Grain catchment in the South Pennines (Figure 2.2). Figure 2.2 shows a clear contrast between the magnitude of the two sediment budgets, a higher yield of sediment is lost from the gully system at Upper North Grain, with relatively little storage of sediment occurring and most eroded sediment entering the channel system and being lost from the site. In contrast, the

Rough Sike budget indicates a far lower magnitude of sediment lost from the gully system, and almost all the eroded sediment is stored in other areas.

Using similar methods to those outlined above, Baynes (2012) produced an annual sediment budget (Figure 2.3) for the field site at Flow Moss in the North Pennines.

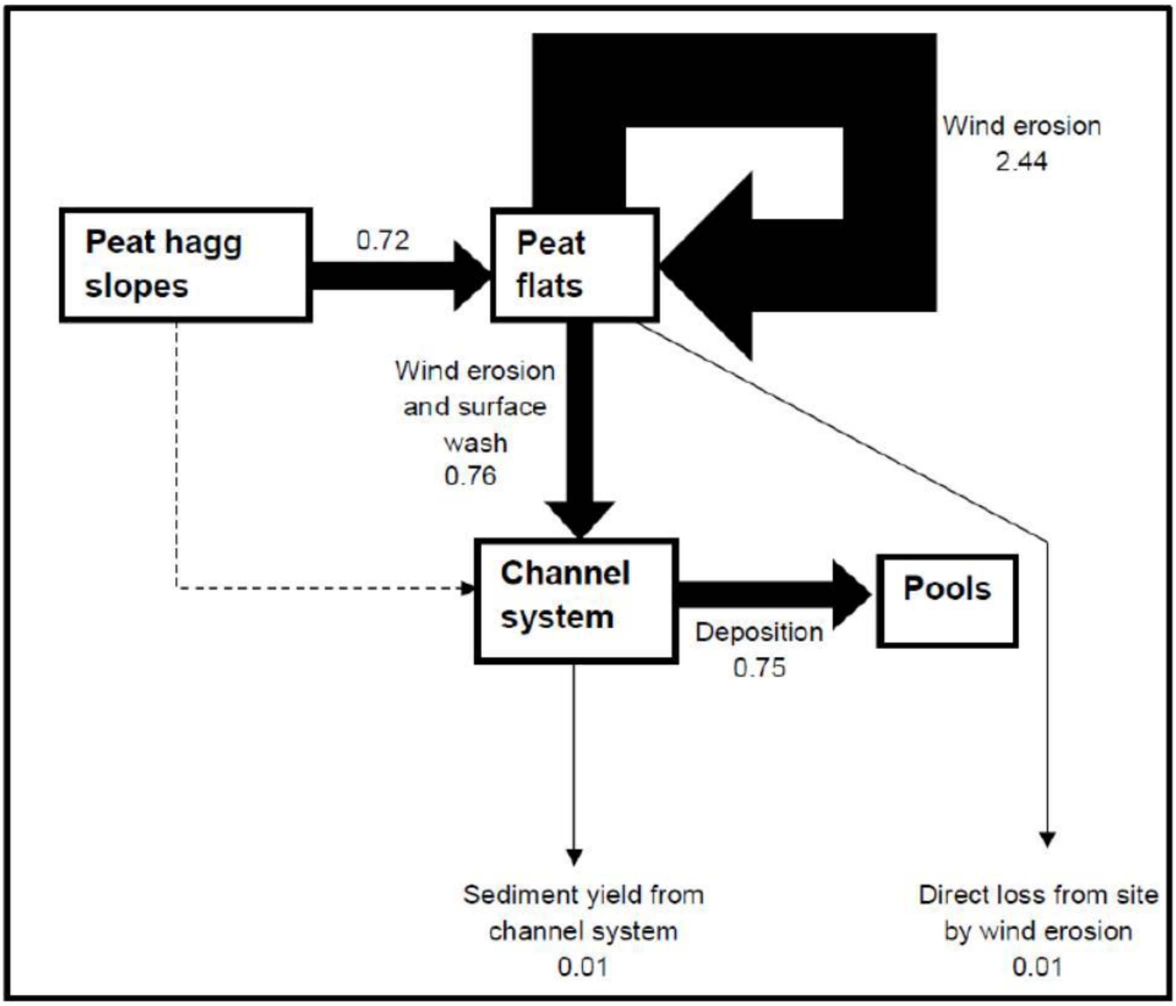


Figure 2.3 – *Diagram representing the annual Flow Moss Sediment budget. All values are in tonnes (Baynes, 2012).*

At Flow Moss (Figure 2.3), the dominant flux of eroded peat is transferred across the bare peat flats through wind erosion, but most of this sediment does not enter the channel system. The eroded peat is instead reworked or deposited elsewhere on the peat flats. Baynes (2012) concluded that the amount, type and timing of erosion of the bare peat flats at Flow Moss were closely controlled by environmental conditions occurring both before and during an erosion event. The study demonstrated that the fluvial export of peat from Flow Moss is very low due to the active deposition of transported peat in the pools and the ephemeral nature of the channel system. It was concluded that the total terrestrial carbon store at Flow Moss was relatively stable, but in the future could potentially become more unstable due to increased fluvial export of peat (POC) if the bare peat surfaces remain exposed. Restoration of vegetation could reduce erosion rates and mitigate against potential impacts of enhanced fluvial erosion (Baynes, 2012).

There is now considerable evidence to suggest that the erosion of peatlands leads to carbon being released to the atmosphere and in the future this may be further exacerbated by climate change. Evans and Lindsay (2010) suggest that the onset of erosion in peatlands has turned these vast carbon stores into, at best, a carbon neutral status, but more than likely a carbon source. Nonetheless, Van Oost *et al*. (2007) suggest there is an inherent difficulty in quantifying a net carbon flux controlled by interacting processes that are most often studied in isolation. For effective land management it is essential to consider the sediment system as a holistic system rather than a collection of single entities (Walling and Collins, 2008). Monitoring is required to quantify sediment loss from peatlands through erosion and the construction of a sediment budget will allow the identification of the main pathways of sediment transfer. Furthermore, the sediment budget approach combined with estimates of carbon storage can be used to assist in quantifying the amounts of carbon lost from peatlands through erosion

(Worrall *et al*., 2009). This information can be used to quantify erosion induced carbon emissions, which Lal (2003) described as “a globally significant, but misunderstood and poorly quantified component of the global carbon cycle”.

2.6 Peatland restoration

Once a peatland sediment budget has quantified the main drivers and pathways of erosion, priorities can be assigned for restoration projects. In response to threats from erosion, peatland restoration has become an increasingly common land-management practice (Kimmel and Mander, 2013). Over the last few decades, the number of peatland restoration projects has increased significantly as a greater variety of restoration techniques have become available (Cris *et al*., 2011). Vasander *et al*. (2003) state the fundamental goal of peatland restoration is to revitalise a self-sustaining, naturally functioning peatland ecosystem which accumulates carbon and retains nutrients from through-flowing waters. A range of methods exist that have been used in an attempt to restore peatlands from a degraded state (Anderson *et al*., 2009). These methods include drain blocking, gully blocking and profiling and bare peat stabilisation (Parry *et al*., 2014). Further examples of methods used in peatland restoration discussed by Anderson *et al*., (2009) are highlighted in Table 2.2.

Table 2.2 - *Examples from Anderson et al., (2009) of methods used to restore peatlands*

Problem	Impacts	Possible driver	Restoration method/ mitigation
Overgrazing	Loss of biodiversity, increased erosion	Inappropriate land management, i.e. too much stock	Fence off area to reduce stock numbers
Loss of native species	Loss of biodiversity, reduction in ecosystem function.	Inappropriate burning, excessive grazing	Introduction of desired species, alteration of burning management, remove non-native species.
Reduction of water table	Loss of ecosystem function, including carbon storage, increased erosion.	Climate, drainage for land management.	Gully blocking, coir rolls to reduce surface runoff.
Fire	Increased loss of DOC, loss of vegetation, increase in susceptibility to erosion.	Wildfires – possible increase due to climate change, managed burns.	Remove old dry heather which is susceptible to fire, alter managed burning.

The restoration methods in Table 2.2 focus on reducing erosive agents (such as sheep), managing vegetation and raising the water table, which Vasander *et al.* (2003) believe should be the first step in a restoration project.

2.6.1 International Peatland restoration

Globally, many peatland ecosystems are threatened by anthropogenic activities including agriculture, forestry and peat mining (Joosten, 2009). Furthermore, peatlands located downwind of heavy industry are impacted by the deposition of pollutants such as SO_2, nitrogen and heavy metals (Marsden and Ebmeier, 2012).

While some countries have very little of their peatlands in productive use, others have drained almost their entire peatland resource (Dommain *et al*., 2012). Although only approximately 15% of the world's peatlands have been drained, peatland drainage results in substantial emissions of CO_2, accounting for almost 6% of global CO_2 emissions (Joosten, 2009). Inappropriate land management practices can result in a reduction of long term peatland carbon stores, however land management practices can be reversed (Worrall *et al*., 2009) and peatland conservation is one of the most cost effective methods to stop increases in global CO_2 emissions (Schumann and Joosten, 2008, Ritzema *et al*., 2014).

2.6.2 UK peatland restoration

Drainage is a major threat to UK peatlands with, over 80% of peatlands in a degraded state due mainly to past drainage programmes. It is estimated that almost half of the country's 2.9 million ha of peatlands have been drained (Worrall *et al*., 2009). Drainage of peatlands for agricultural reclamation during the 19th and 20th century was responsible for alterations to ecological and hydrological functioning (Grand-Clement *et al*., 2013). Additionally, in the late 20th century, the EU provided subsidies for sheep farming, a practice which resulted in a 30% increase in sheep farming on UK moorlands and added to peatland degradation (Holden *et al*., 2007). However, over recent years, attempts have been made to reduce some of the damage inflicted on peatlands from land management activities. In 1996, Defra launched the Moorland Scheme, which attempted to reduce livestock numbers on moorland. This scheme provided a headage payment on the basis of each breeding ewe removed from moorland (Condliffe, 2009). More recently, appeals have been made for further peatland restoration, with the IUCN peatland programme calling for 1 million hectares of peatlands to be restored to good condition, or be under restorative measures by 2020 (Cris *et al*., 2011).

The majority of England's peatlands are distributed across the uplands of the Pennines, with other upland areas such as Dartmoor, Exmoor and the North York Moors also supporting significant areas of upland peat (JNCC, 2011). Estimates suggest that that within England, upland blanket bog and upland valley mire cover an area of 3553 km^2, whilst the area covered by lowland fens is estimated to be 2880 km^2 (Natural England, 2010). There are several excellent examples of projects implemented to restore degraded peatlands in England. In Yorkshire, the Yorkshire Peat Partnership (YPP) is aiming to restore 50% of Yorkshire's blanket bog by March 2017 (Cris *et al*., 2011). Restoration methods by the YPP have been two stage. Firstly, an extensive survey was implemented to identify target areas for restoration including areas of bare peat, grips and gullies. Once these had been identified, attempts have been made to re-vegetate bare peat and block grips through the addition of peat dams. This has been done with the aim of reducing the amount of sediment eroded from the bare peat and raising the water table of the peat bog. Similar initiatives have been implemented in the Peak District, with restoration projects dating back as far as the Moorland erosion study of 1981 (Philips *et al.,* 1981).

More recently, the Moors for the Future partnership, established in 2003, is aiming to restore over 800 ha of the South Pennines moors by 2015 (Bonn *et al*., 2009). The three main objectives of the Moors for the future project are to: raise awareness of the value of the moors and encourage responsible use of the landscape, to restore and conserve moorland resources and to develop expertise on how to protect and manage the moors sustainably (Moors for the future, 2014) To achieve these objectives, several methods have been used. Areas of bare peat which are most susceptible to erosion have been identified and heather brash has been spread in these areas to try and reduce erosion and to provide a protective layer for grass seed that has previously been spread, allowing vegetation to establish. Lime has also been added to the bare peat areas to try and adjust the pH of some of the very acidic soil, again this was an attempt to allow vegetation to recolonize. Fencing has been used to protect fragile peatland areas from erosion caused by grazing stock, and similar to restoration projects previously discussed, attempts have been made to block gullies using stone dams, heather bales and timber in an attempt to raise the water table (Cris *et al*., 2011).

The North Pennines AONB peatland programme provides another example of a project undertaken to restore and preserve peatlands. The peatland project was established in 2006 and has several objectives. These are to support restoration and management work; to support research into peatland ecology, process and management; to promote best practice in the management of peatlands and to raise the level of understanding and appreciation of the significance of the peatland resource (Cris *et al*., 2011). The primary methods of restoration implemented by the North Pennines AONB peatland programme are the use of peat dams to block grips in an attempt to raise the water table of the peatlands; however the programme also aims to restore actively eroding peat flats in an attempt to reduce the amount of erosion. Flow Moss is one small area (7 ha) of peatland within the North Pennines where a trial of restoration work is being undertaken.

2.6.3 Limitations of Peatland restoration

Over recent years, many attempts have been made to restore degraded peatlands. However, restoration efforts are not always straight forward. For example, frequent burning has resulted in the loss of carbon from peatlands and although burning could be stopped, there are several caveats to this. Worrall and Evans (2009) state that some studies have shown DOC can increase under non-burn sites in comparison to burnt sites. A lack of burning could result in build-up of mature heather, potentially resulting in areas becoming more prone to wildfires, which could cause greater carbon losses than managed burns. Mature heather may also add little to carbon fixation from NPP in comparison to that provided by young heather. Arguments above suggest that some degree of burning may be desirable, however this must be done on a scale and at a frequency which maximise carbon storage (Worrall and Evans, 2009).

Management practices can be effective in restoring peatland functionality and although restoration attempts are becoming increasingly common, Strack and Zuback (2013) have commented that few studies have investigated the longer term impact of restoration upon the peatland carbon balance. Kimmel and Mander (2013) support this, stating that there is a need for long-term research on restoration impact of the C and N balance in restored peatlands. Parry *et al*. (2014) highlight that the success of restoration attempts will vary depending on the condition of the peat and the stage and type of degradation, both of which can differ within and between sites. Due to this, restoration methods should be tailored to suit the specific peatland (Parry *et al*., 2014). Results from the current study will help identify which restoration methods implemented at Flow Moss have been most successful, allowing future restoration strategies at the site to be tailored to maximise efficiency.

2.7 Chapter summary

Peatlands play a vital role in the global carbon cycle. One of the main motivations behind peatland restoration is increasing the sequestration of carbon and decreasing the amounts of peat lost through erosion. Vigorous attempts are now being made to preserve and restore peatlands both internationally and in the UK (Yorkshire Peat Partnership, North Pennines Peatland programme, Moors for the Future). However, for restoration efforts to be successful, the amounts of sediment lost from peatlands, and the main mechanisms through which this occurs needs to be clearly established. Sediment budget approaches provide a valuable tool for quantifying peat loss; this can be combined with carbon storage data to help establish the main mechanisms of sediment/carbon lost from peatlands. Using this information, restoration strategies can be targeted in areas susceptible to erosion. If the UK is to meet emissions targets and offset carbon emissions, the amounts of carbon within terrestrial stores needs to be known and effectively communicated to policy makers.

Flow Moss is the first site of degraded peatland to be restored by the North Pennines Peat Programme. Flow Moss is fairly typical in terms of location and climate of most UK blanket bogs. Restoration methods implemented at Flow Moss include the exclusion of grazing sheep since April 2010 and the spreading of heather brash over the bare peat in December 2010. Both of these methods have previously been used by the Moors for the Future partnership in the Peak District. However, Flow Moss is the first site in the North Pennines where these methods have been implemented.

3. Site description

Flow Moss is a 7 hectare area of upland blanket bog located in the North Pennines Area of Outstanding Natural Beauty (AONB) at NY 806 537, 450 m.a.s.l (Figure 3.1). The North Pennines region is an upland area generally above 400 m.a.s.l with the highest point being Cross Fell at 890 m.a.s.l. Flow Moss is located in Northumberland between the River West Allen and the River East Allen. The land owner is Allendale Estates and the site is managed by the North Pennines AONB Peatlands Programme. This programme aims to enhance and conserve the internationally important peatland resource located within the North Pennines. Flow Moss is characterised by an extensive area of bare peat (Figure 3.2) and for this reason it was selected as one of the first peatbogs in the North Pennines to be targeted for restoration work beginning in April 2010. The Flow Moss area has been subjected to a range of pressures including grazing, fire and, historically, significant disturbance from 19th century metal smelting. Peat profiles shown in the peat cores indicate erosional hiatuses typical of multiple phases of erosion. Nonetheless, the cause and timing of the erosion remains uncertain and requires further investigation through palynology, macrofossil analysis and detailed dating of the peat cores.

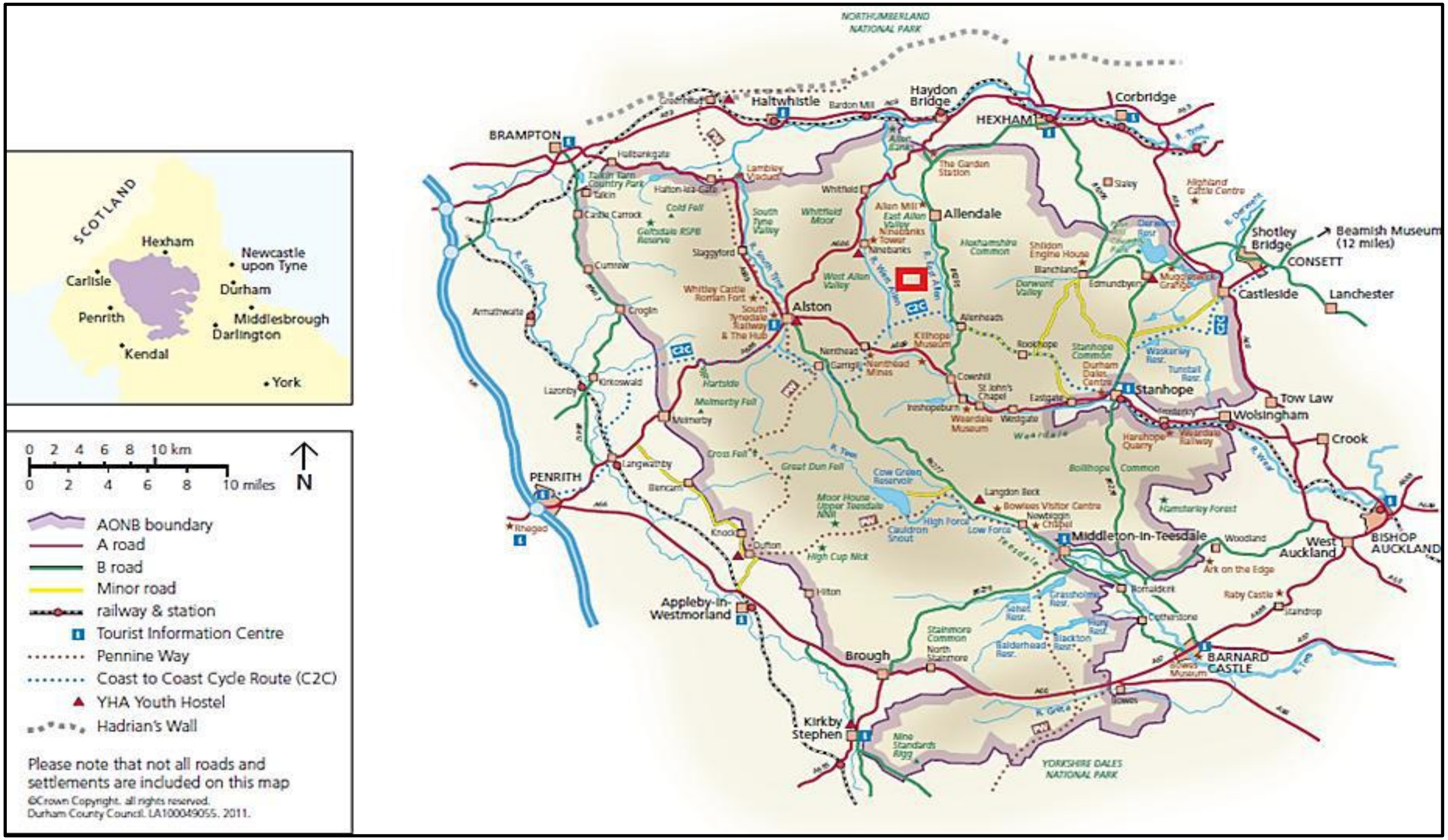


Figure 3.1 *- Map of North Pennines AONB, the location of Flow Moss is marked with a red box, Source:* *http://www.northpennines.org.uk/Lists/DocumentLibrary/Attachments/90/NPAP-map.pdf*

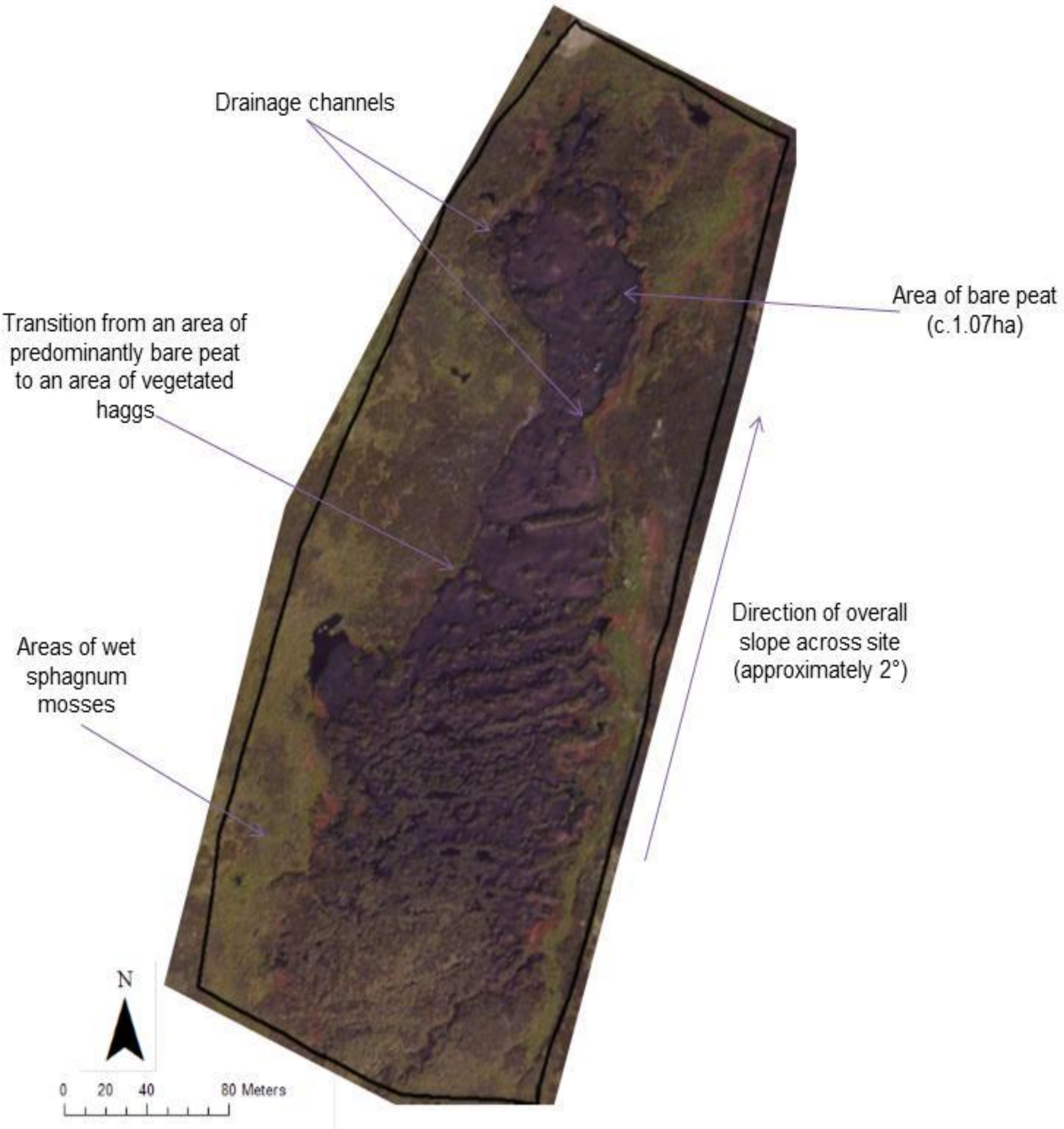


Figure 3.2 *–An aerial view of Flow Moss with geomorphological features marked on (Updated from Baynes, 2012),*

3.1 Flow Moss Climate

The climate of the North Pennines was classified by Manley (1943) as 'ocean subarctic' and described as being dominated by cool, wet and cloudy weather. Pigott (1956) adds that the North Pennines experiences frequent heavy storms. Although there is no long-term record of climate conditions available for Flow Moss, Moor House National Nature Reserve (NNR) is located approximately 25 km from Flow Moss and holds the longest record for climate monitoring at any upland site in the UK (Holden and Adamson, 2001). Recording of climate data at Moor House has occurred at around 550 m.a.s.l. since 1931. Analysis of the Moor House climate data until the year 2000 showed the average temperature at Moor House to be 5.3°C and average precipitation of 1982 mm per year (Holden and Adamson, 2001). Manley (1943) identified that the prevailing wind direction at Moor House was from the south west, while the mean number of frost days was calculated to be 105 per year, with lying snow on the ground for an average of 55 days (Archer and Stewart, 1995). As Flow Moss is approximately 100 m lower in altitude than Moor House and further west of the main Pennine divide, it is likely that temperature will be higher, rainfall will be less and there will be less snow covered days per year, however, over all, the climate will be broadly similar. Table 3.1 summarises the climate data recorded by Baynes (2012).

Table 3.1 – *A summary of the environmental conditions recorded by Baynes (2012)*

Condition	
Total Rainfall	756
Average Temperature	1.8
Mean wind direction	224

3.2 Geology of the North Pennines

Flow Moss is located within the Alston Block which Johnson and Dunham (1963) describe as consisting of sedimentary rocks forming a series of sandstone, limestone and shale formed during the Upper Carboniferous series (c. 300 Ma) and the Lower Carboniferous series (c. 350 Ma) (Figure 3.3). Bouch et al. (2008) state that the North Pennine Orefield is best known for its vein-style mineralisation, which shows strong concentric zonation consisting of a central fluoride zone. Mineralisation in the central fluoride zone has formed from hot (120–200 ºC) metal-rich saline brines. Substantial ore bodies are thought to have resulted from metasomatic replacement adjacent to the veins producing lead, zinc, fluorite, barite, witherite and iron ore (Bouch et al. 2008). Baynes (2012) describes the local bedrock geology of Flow Moss as being composed mainly of sandstone, millstone grit and limestone.

Extensive areas of upland UK are covered by peatlands of which there are three types: fens, raised bogs and blanket bogs. Blanket bogs are the most extensive within the UK covering 7.5% of the landmass of the British Isles (Tallis, 1998). Within the North Pennines, blanket peat covering approximately 900 km^2 (North Pennines AONB Peatland programme) has formed during the Holocene up to a thickness of 2 to 3 m. Beneath the peat are deposits of glacial till and boulder clay. The clay rich nature of the deposits results in restricted drainage of the area, leading to waterlogging and the formation of peat even on limestone bedrock (Evans and Warburton, 2005). A GPR survey of Flow Moss has identified that the deepest areas of peat are c. 4.5 m, and located at the South West area of the site. The peat does not have a uniform depth across the whole 7 ha site and in some areas, particularly along the course of the channels, the peat has been eroded completely.

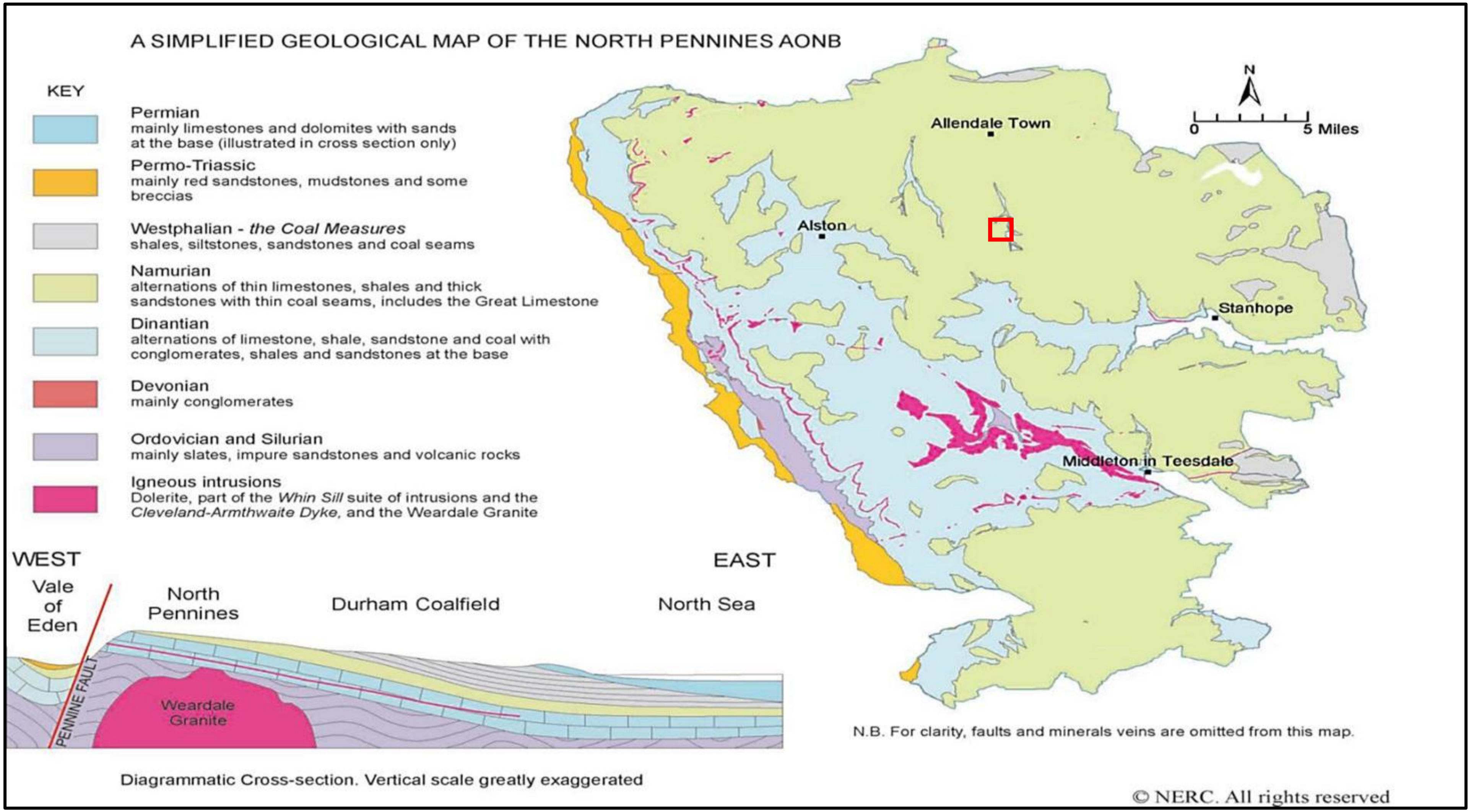


Figure 3.3 *- A simplified map of the geology of the North Pennines (North Pennines AONB Geodiveristy Audit 2010). The approximate location of Flow Moss is indicated by the red square.*

3.3 Historic and contemporary land use

UK upland blanket peatlands are some of the most heavily managed environments (Ramchunder et al., 2009). Historically, much of the North Pennines has been subjected to mining activities, and the Northern Pennines Orefield was once the most productive Lead and Zinc mining area in Britain (Dunham, 1944). The mining landscape located in the North Pennines is described by Mighall et al. (2004) as unique and containing a relatively large concentration of mines dating back possibly as far as the roman period. The mining industry in the Pennines has left a legacy of contamination and many floodplains located in the mining catchments remain highly polluted (Macklin et al. 1994). The area surrounding the Flow Moss field site has a rich mining heritage, and evidence of historic mining activity can be identified. Located in close proximity (approximately 100m away) on a small ridge to the east of the site are two large chimneys which were used for dry condensation of fumes emitted from lead smelting which operated 3.5 km away at Catton in the valley of Allendale until 1894.

Much of the upland area of the Allendale catchment is covered by heather moorland which is actively managed for grouse shooting. Flow Moss is located on one of the grouse moors of the Allendale Estates, which is periodically burnt to maintain dwarf shrub habitats. This provides the optimum conditions for the breeding and growth of Red Grouse (Yallop *et al.*, 2006).The other major historical land use of Flow Moss is low density sheep grazing (0.33 sheep per hectare; Rawes and Hobbs, 1979); however fencing currently prevents sheep from entering the study site.

3.4 Management of the site

Flow Moss is currently undergoing restoration as part of the North Pennine's Peatland Programme (established in 2006). Three main restoration techniques have been implemented at Flow Moss. These are:

1. Grazing exclusion by fencing – in April 2010 the site was fenced to prevent access from grazing animals and in November 2011, secondary rabbit fencing was added to further secure the site (Figure 3.4a).

2. Re-vegetation - cutting and spreading of *Calluna Vulgaris* (heather brash) (Figure 3.4b) – Initial brash spreading occurred in November and December 2010, with the final 20% being spread in April 2011.

3. Surface water erosion control - Coir rolls were installed in February 2013 (Figure 3.4c) to manage surface water flows on the bare peat margins.

Figure 3.4 - *(**a**) Sheep and rabbit fencing added in 2010, (**b**) heather brash spread to reduce bare peat erosion in November and December 2010, (**c**) Coir rolls to reduce surface erosion, added February 2013.*

3.5 Geomorphological features at Flow Moss

Within the 7 ha fenced area at Flow Moss, there are several geomorphological features which could be considered typical of an eroding area of upland blanket bog. The significant features are highlighted in Figure 3.2. The site slopes gently at a gradient of approximately 2° from the southwest to the northeast. One of the largest features within the study site is an extensive area of bare peat, which Baynes (2012) estimated to be around 1.75 ha. This bare area of peat is the focus of current restoration measures as this surface is assumed to be actively eroding. From the UAV photography captured in April 2011, it was clear that this area of bare peat had no significant vegetation cover apart from small vegetated haggs. Alongside the main area of bare peat, there are two small channels and several peat pools. Two drainage channels flow either side of the bare peat from the southwest, draining through a series of channels and small pools into the larger pool at the northern end of the site (Figure 3.2). To the south of the bare peat, there are several larger peat haggs which Baynes (2012) described as dissecting the site producing a series of low ridges and having an approximately southeast-northwest orientation.

3.6 Peat stratigraphy

Baynes (2012) collected two short cores (106 cm and 161 cm in length), from the bare peat area to identify the characteristics of the peat and the degree of decomposition. The results from the Troels-Smith and von Post classification of the cores indicate that there were three key sections identified in the cores. Firstly, just beneath the surface there is dark peat which was found to contain herbaceous plant material and show very little stratification. The majority of the peat was classified, using the von Post scale for assessing peat decomposition, as ‘H7’ indicating that the samples were ‘strongly decomposed’. The second section (43-93 cm in core one and 52 – 79 cm in core 2) of the cores is described by Baynes (2012) as slightly lighter in colour and containing root and stem material from mosses, however it is still classified on the von Post scale as ‘H7’. At the base of both cores there was a section of darker peat that was similar to the peat at the top of the core; again samples in this section were classified using the von Post scale as ‘H7’. Both peat cores were underlain by a sandy gravel layer and there was very little variability in the degree of decomposition within the cores. From the peat cores, 23 peat subsamples were analysed by Baynes (2012) to establish TOC values. The recorded values ranged from 25.4% to 68.4% with an average value of 51.8% and a standard deviation of 8.94%.

3.7 Chapter summary

Data collection for this study began on 8th March 2013, almost three years since the site perimeter fence was added to the site, and two years since the final 20% of brash was spread in April 2011. This should allow the assessment of the success of these restoration methods, as they have had time to impact on erosion dynamics at the site. The monitoring of Flow Moss provides an opportunity to observe the effects of restoration methods several years after implementation. Furthermore, combining data collected in the present study with those collected with Baynes (2012) will provide an indication of the effectiveness of restoration measures over a four year time scale.

4. Methods

This chapter provides a description of the methods used in this study. The methods were selected to answer the research questions outlined in Chapter 1 and are divided between field monitoring and laboratory techniques. The main field monitoring methods are shown schematically in Figure 4.1. The methods can broadly be divided into environmental monitoring, measurement of subsurface properties and sediment budget monitoring including direct measurements of sediment flux and terrestrial laser scanning (TLS). The main period of monitoring for this study occurred between 8th March 2013 and 4th March 2014.

4.1 Environmental Monitoring

4.1.1 Automatic Weather Station

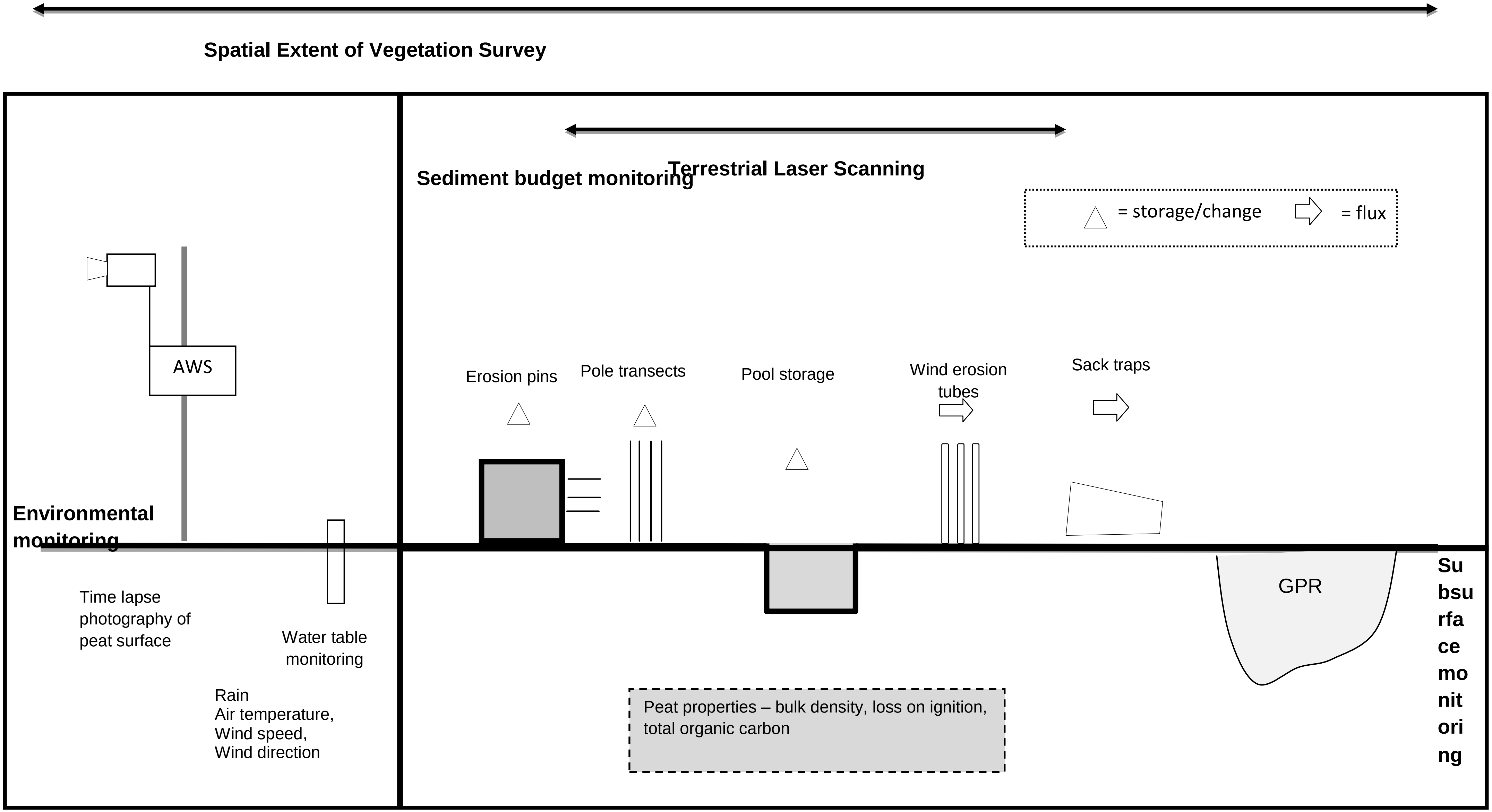
Environmental processes were monitored using an Automatic Weather Station (AWS). The AWS recorded: air temperature, wind speed and direction, rainfall, local water table and time lapse imagery of the peat surface condition. All meteorological measurements have been concatenated and are reported at one hour intervals. The time lapse photography of surface conditions was documented at 0900 and 1500 GMT daily. Rainfall was monitored using a tipping-bucket rain gauge and the AWS records the number of times the bucket is filled and tipped within a 30 minute measurement period (tip increment 0.202 mm). Water table measurements were collected from an established dipwell which was installed in an area of bare peat in 2010. The dipwell was intentionally positioned in the bare peat so that the hydrological response of the bare peat (areas of erosion) could be quantified. The technical details of the AWS are summarised in Table 4.1.

***Table 4.1** – technical details of AWS equipment*

Item	**Instrument / sensor**	**Sampling Frequency**	**Accuracy / Error**
Data Acquisition System	Campbell CR10X logger, PS100E-LA 12V Power supply and SOP10/X 10 W solar panel	30 min	
Comprising			
Air temperature	Campbell 107 temperature probe and T351-RS radiation shield	30 min	± 0.4 ˚C
Rain gauge	ARG100 Tipping Bucket Rain gauge (0.2 mm / tip)	30 min	+ 1% to -5 %
Wind speed	Environmental Measurements Ltd WSD1 Wind speed and direction sensor	30 min	2 %
Wind direction	Environmental Measurements Ltd WSD1 Wind speed and direction sensor	30 min	± 2 ˚
Water table depth	PDCR1830 Pressure Transducer 350 mB	15 min	± 0.06%
Time Lapse Camera	JE Teknik RDC365 ver.2 (Kodak CX6200, 2.0 m pixels)	0900 and 1500 h	

Key drivers of erosion (such as periods of intense wind and rain) can be identified and compared to sediment yields collected from sediment traps. Wind direction recorded by the AWS can be compared with results from the wind flux tubes, as a correlation between wind direction and traps containing the largest amount of sediment should occur (Warburton, 2003). As discussed in Chapter 2, a reduction in the height of the water table of a peatland may result in increased erosion and carbon loss. Therefore, monitoring of water table height is important. To monitor water table height, a pressure transducer, connected to a Campbell CR10X data-logger was installed in 2011.This transducer records the height of the water table at 30 minute intervals. The water table depth data can be compared to rainfall data collected by the AWS to establish if a lag exists between precipitation and hydrological response.

__Figure 4.1__ - Framework of the main methods used in the project



4.2 Sediment Budget Monitoring

This study aims to produce a sediment budget focused on monitoring surface change and assessing peat losses at Flow Moss. The elements of the sediment budget measured during this study are summarised in Figure 4.2. The mass balance equation (Eq. 4.1) can be used to establish the amounts of sediment lost if the following are known: ΔS = change in storage, I = sediment inputs and O = yield of sediment output.

$$\Delta S = I - O \qquad \text{Eq. 4. 1}$$

Once an estimate of the volume or mass of erosion has been calculated based on the specific field measurements, this can be combined with peat carbon content data to establish a carbon budget which estimates how much carbon is lost from the site via erosion. The priority is to assess the effectiveness of the restoration strategy so the sediment budget is constructed to provide key estimates of erosion / deposition processes on the bare peat area and the amount of peat lost in the ephemeral stream flow at the catchment outlet.

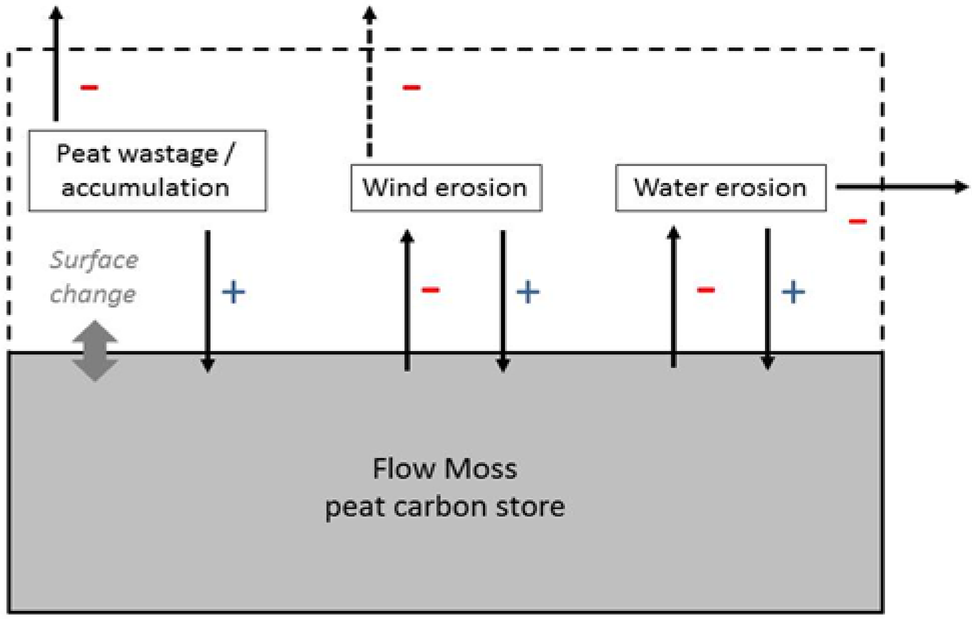


Figure 4.2 – *Elements of the sediment budget measured during this study (+ are the inputs, - are the outputs)*

4.2.1 Terrestrial Laser Scanning (TLS)

Terrestrial Laser Scanning (TLS) systems utilise LiDAR (Light Detection and Ranging) to provide laser based measurements of the distance between a sensor (the scanner) and the target surface being scanned. A “point cloud” is produced where each point recorded by the scanner is represented by a coordinate (*X, Y, Z*) in 3D space (Slob and Hack, 2004). The measurements resulting from this method of data capture can be processed to produce a Digital Elevation Model (DEM). A DEM of Difference (DoD) can then be created to detect changes in peat surface that may have occurred between two dates. A previous example of the application of TLS to peatland monitoring is provided by Grayson *et al.,* (2012), who implemented a pilot study to test if TLS was an appropriate method to monitor erosion. This was done by recording changes in the height of the peat surface using TLS. They concluded that TLS is superior to traditional methods for the monitoring of peatland surfaces; however, some improvements to their methods were required. Data collection using TLS only occurred twice during their study period and it was not possible to establish whether the observed net increase in the surface height of the peat had occurred due to the temporary effects of mire-breathing, or whether the surface height had actually increased.

In this study, repeat scanning collecting data relating to the surface height of the peat occurred nine times over a 10 month period. The aim was to complete a TLS survey approximately every two weeks, however due to inclement weather and equipment availability this was not always possible. Data were collected using a Riegl VZ-1000 time of flight laser scanner (accuracy 8 mm, precision 5 mm). During each TLS survey, a network of ground control markers was used to fix the scanner and survey target positions. The control network was configured to ensure good geometric coverage of the site so that gross changes could be immediately detected. Great care was exercised when positioning the scanner tripod to ensure a firm, secure base which was correctly positioned above the ground control marker. Captured data were used to create DEMs of the peat surface (Figure 4.3 **(b)**) and the DEM consisting of data captured at an earlier date, subtracted from a later one to create a DoD and identify if changes in the surface elevation of the peat have occurred. A subsection of the Flow Moss site was scanned from six scan locations (Figure 4.3 **(a)**), chosen to incorporate a range of surface conditions at the site including bare peat; vegetated peat and part of a channel which runs through the site. Figure 4.3a shows the six scan locations, represented by the blue stars, from which data were collected and an example of the TLS DEM data. The information gained from the TLS survey was compared with erosion pin data and used to quantify elements of the Flow Moss sediment budget relating to surface change (Figure 4.2).

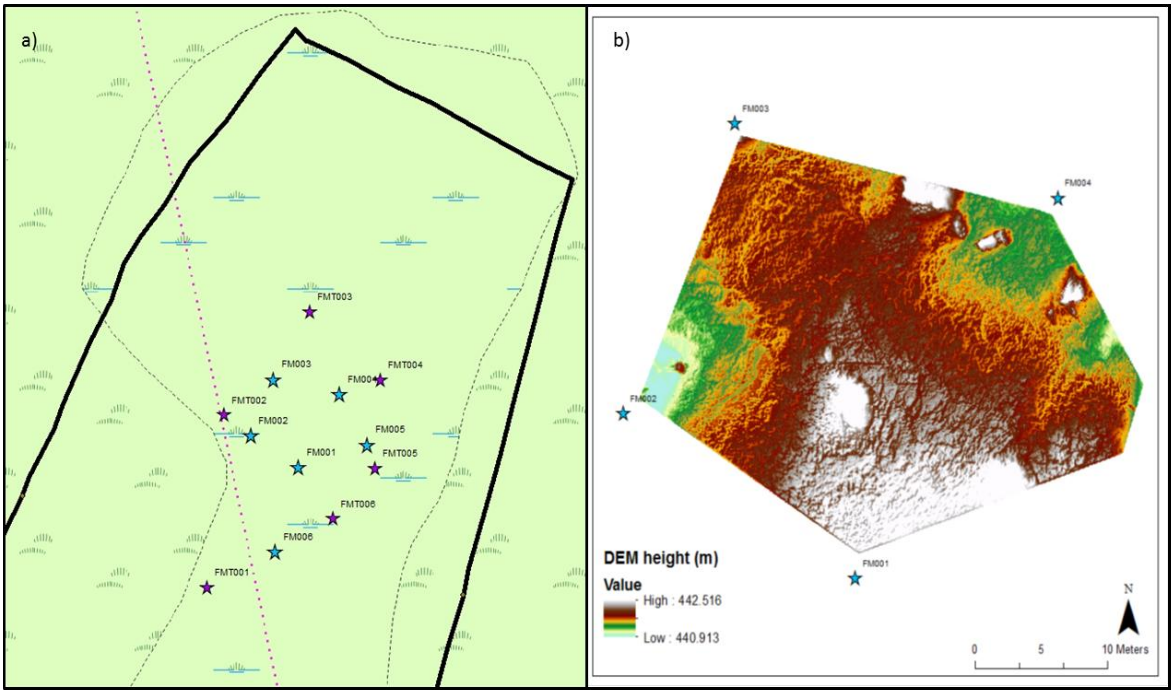


***Figure 4.3 - (a)** Map showing location on the six scan stations. **(b)** An example DEM created from TLS data captured on 6th June 2013.*

4.2.2 Pole transects and erosion pins

Erosion pins have been used extensively in previous studies of peat erosion (Evans and Warburton, 2005). At Flow Moss a series of erosion pins and fixed poles are used to characterise erosion and deposition (Table 4.2). Erosion pins provide a direct method of monitoring changes in the surface height of the peat. Sets of 8 pins measuring 100 mm above the surface were installed by Baynes (2012) in November 2010 in 10 peat haggs at different locations around the site (Figure 4.4 **(a)**). The pins were arranged in two vertical lines of four pins (Figure 4.5 **(a)**). Francis (1990) suggests the arrangement of pins in this way enables a disturbance to a single pin to be more easily identifiable and anomalous readings to be effectively identified. The exposed part of the erosion pins was measured approximately every two weeks. The data collected during this study has been collated with those collected by Baynes (2012) to produce a data set spanning just over a two and a half year period from April 2011 until March 2014.

Although erosion pins are a widely used method of data collection, the method is subject to a number of limitations. Frost heave may cause the pins to rise out of the peat surface during winter; this would suggest that data collected during the summer months may be more representative of erosion processes than data collected during the winter. Fieldwork for this study began in March 2013 and ceased in March 2014, thus providing a full year of data and therefore the effects of frost heave may need to be considered during analysis, however the effect of this should be minimal as the erosion pins were firmly anchored into the peat and substantial frost heave events were infrequent. Couper *et al.,* (2002) provide two further factors which may need to be considered when collecting data using erosion pins. These are deposition of sediment from the upper pins to the lower pins and the movement of the pins due to human or animal interference. A further limitation of the use of erosion pins is the human error which may occur during the measurement. To test this, during one field day all pole transects and erosion pins were measured by two different people and the difference in measurements compared (Figure 4.6).

Further monitoring of changes in the height of the peat surface occurred using pole transects located across the site (Figure 4.4 **(b)**). Again, these were installed by Baynes (2012). The 1.5 m long poles (Figure 4.5 **(b)**) were driven into the ground until they reached the mineral layer at an approximate depth of 1-1.2m (Baynes, 2012). The poles were driven into the peat at this depth so that they were firmly anchored and any changes in exposure of the poles could be directly related to changes in surface height of the peat. Once the poles had been installed, they were cut so the initial exposure was approximately 30cm, limiting the effect of strong winds on the poles (Baynes, 2012). Poles were measured approximately every two weeks. These poles are subject to some of the same limitations of using erosion pins. The poles could be affected by frost heave. However, this study also uses TLS to monitor changes in the surface height of the peat and the erosion pin data can be compared to the data captured using TLS to assess if a change in surface height has occurred or if the poles have been impacted by frost heave.

Table 4.2 *- Number of poles or erosion pins at each location*

Site	Location on map (Figure 4.4)	Number of pins/poles	Located in TLS scan area?
Northern Transect	N1-N9	9	Yes
Southern Transect	S1-S10	10	No
Long Transect	L1-L20	20	Partially
Pool transect	P1-P11	11	No
Erosion pins 1	1	10	No
Erosion pins 2	2	10	No
Erosion pins 3	3	10	Yes
Erosion pins 4	4	10	Yes
Erosion pins 5	5	10	Yes
Erosion pins 6	6	10	Yes
Erosion pins 7	7	10	Yes
Erosion pins 8	8	10	Yes
Erosion pins 9	9	10	Yes
Erosion pins 10	10	10	Yes

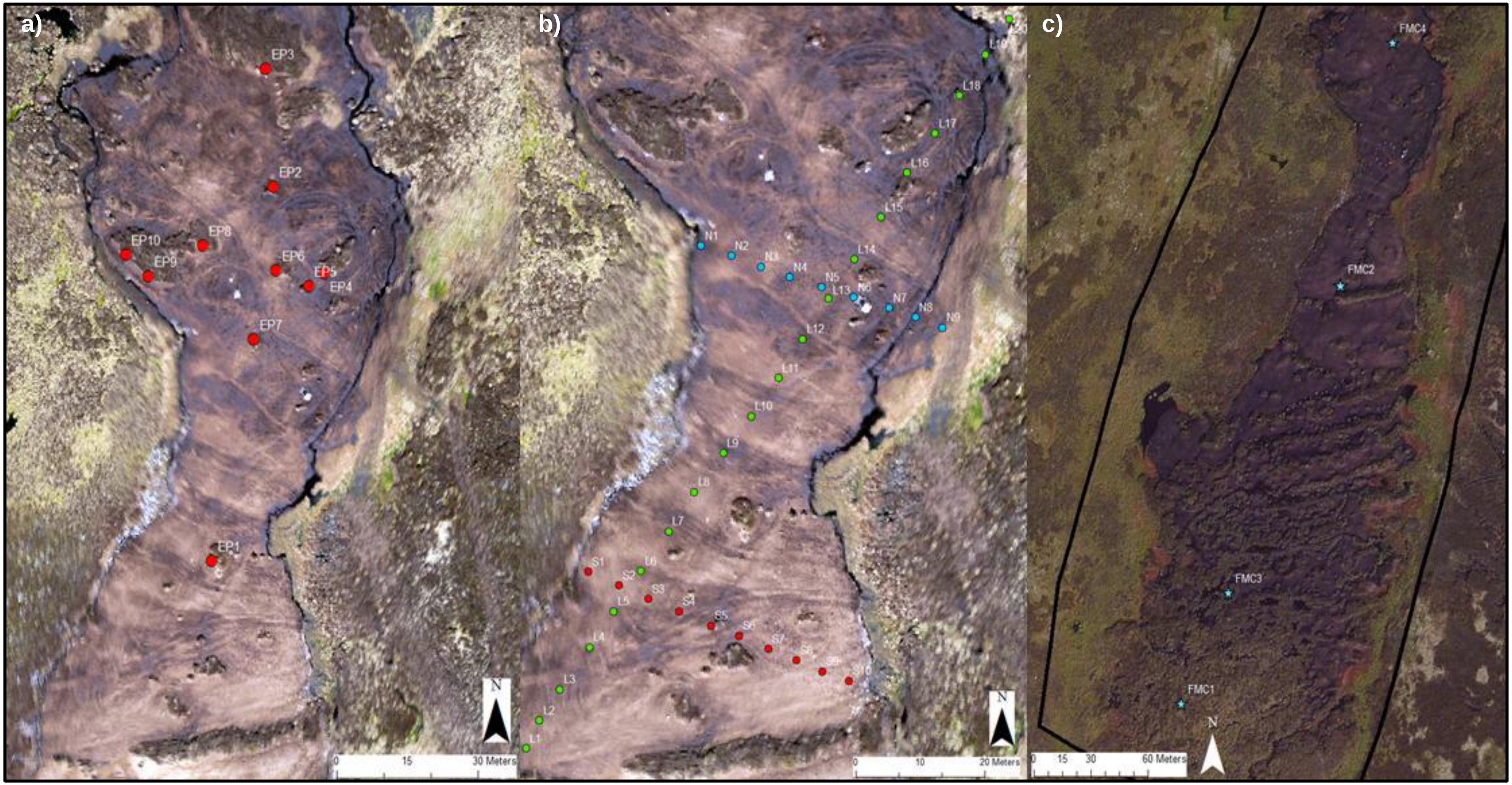


Figure 4.4 - (a) *The location of the erosion pins at Flow Moss* ***(b)*** *The location of pole transects at Flow Moss* ***(c)*** *Flow Moss core locations*

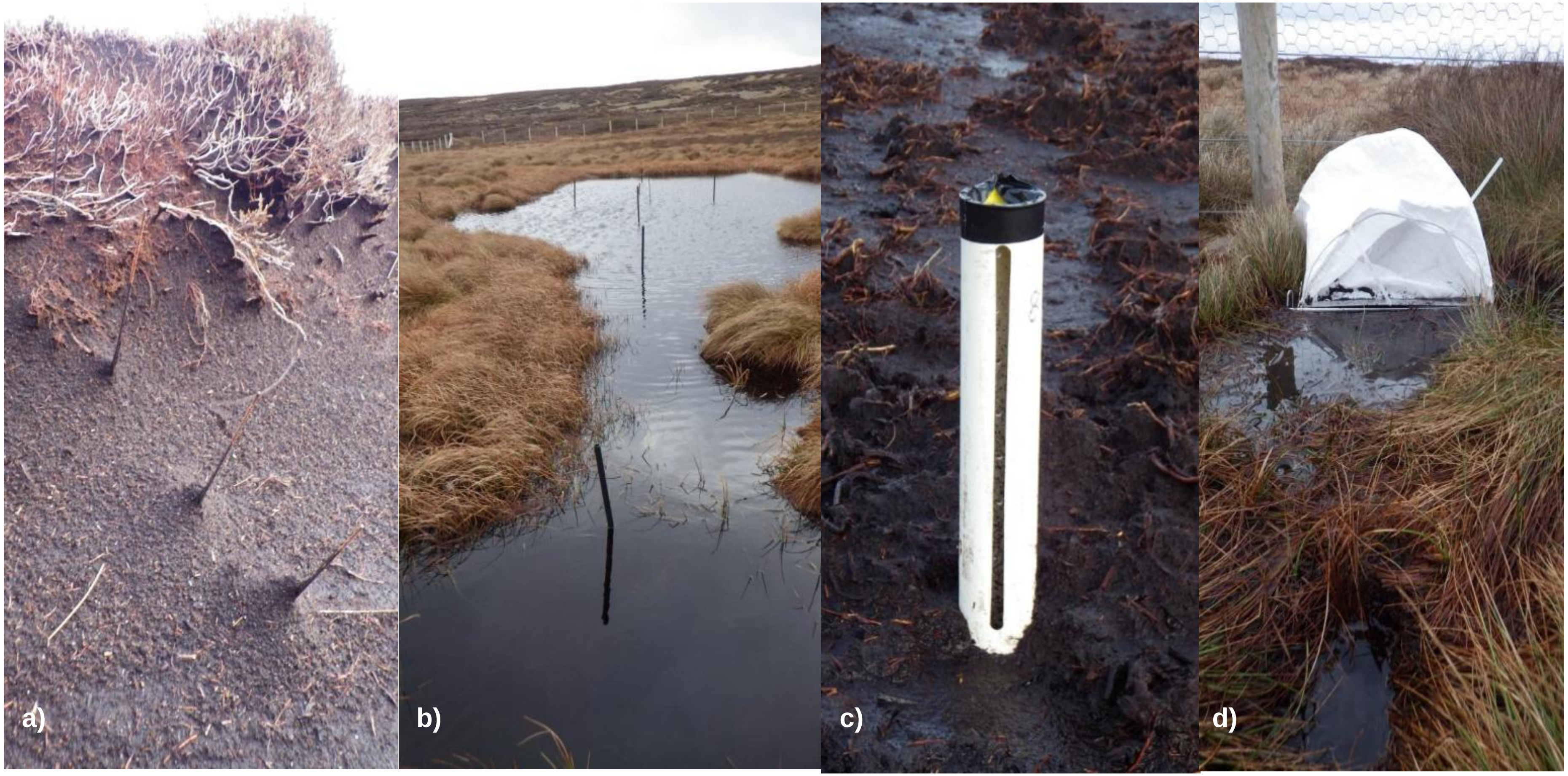


Figure 4.5 *– Field monitoring, examples of* ***erosion*** *pins.* ***(b)*** *Pole transect located in a peat pool* ***(c)*** *mass flux wind sampler* ***(d)*** *fluvial sack trap*

4.2.3 Measurement error of erosion pins and pole transects

In order to quantify the extent of human measurement error when recording pin and pole measurements, an experiment was conducted where all the erosion pins and pole transects at Flow Moss were measured by two different people on the same day with as little time between measurements as possible. The results of this experiment are shown in Figure 4.6. This clearly shows that there is very little difference in measurements, with the exception of two erosion pin values, which can be considered anomalous. The mean difference calculated between measurements was 2.5 mm for erosion pins and 1.8 mm for the deposition pole transects. This is one source of error that will be taken into consideration when using the data for constructing a sediment budget. The data indicate that the erosion pins are subject to a slightly higher level of measurement error than the pole transects. To minimise the impact of this error, all pin and pole measurements during monitoring were recorded by the same person.

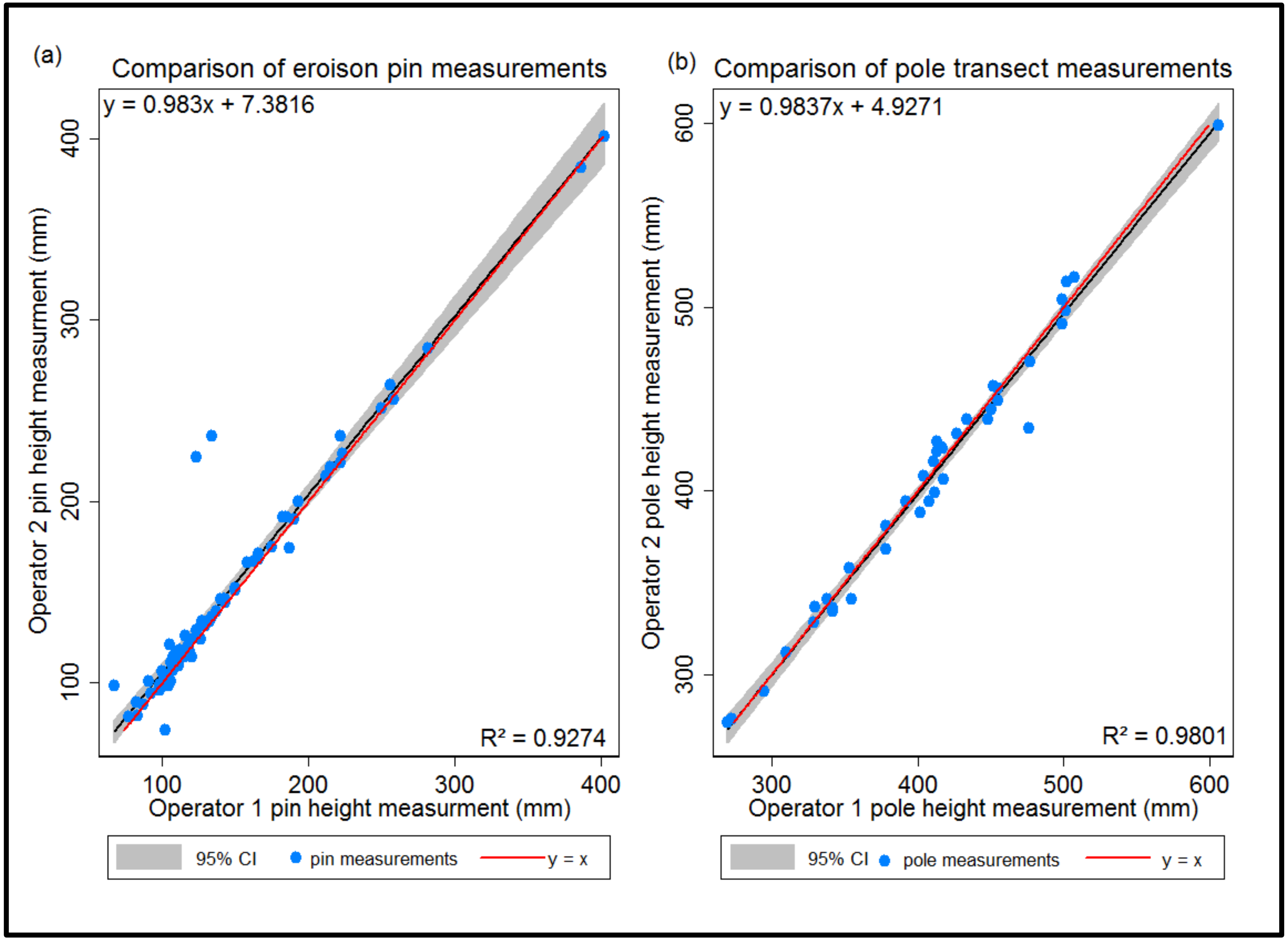


Figure 4.6 - (a) *Scatter plot showing the erosion pin measurements and* ***(b)*** *pole transect measurements recorded by two different people.*

4.2.3 Monitoring peat eroded through aeolian processes

Wind has been recognised as a significant driver of peat erosion and a fundamental characteristic of UK upland environments (Warburton, 2003). Due to the low bulk density of peat, it is highly susceptible to wind erosion and aeolian transport (Evans and Warburton, 2007). In this study, the measurement of peat transported by wind erosion was achieved using passive mass flux samplers. This methodology has previously been implemented by Foulds and Warburton (2007) who installed similar samplers at Moss Flats, a site also located in the North Pennines. The sediment traps are 600mm long plastic tubes with a slot of 250 x 10 mm installed 10 mm above the peat surface (Figure 4.5 (**c)**). Twelve vertical tubes were placed in a circle with a 5m radius at 30° intervals. Eroded peat was collected from all compass directions in a full 360° orientation, thus allowing the direction of surface peat transport to be determined.

Samplers were changed approximately every two weeks. The tubes were taken back to the laboratory, the sediment emptied and dried overnight in an oven at 105° and then weighed to establish the mass of sediment collected. Again, the data collected during this study have been collated with those collected by Baynes (2012). Eroded peat yields can be compared with wind direction data to produce rose diagrams displaying the predominant wind direction, and in which direction most sediment was collected.

4.2.4 Monitoring peat eroded through fluvial processes

Two fluvial sediment traps have been installed at Flow Moss. These have been positioned on the fence line at the most north-western edge of the site to capture sediment lost from off-site active drainage routes. These traps are made of weaved polypropylene bags which catch sediment/peat eroded by fluvial processes (Figure 4.5 **(d)**). The sack traps were designed to allow the water to seep through the sack, but any peat transported in suspension should be trapped. To assess the trapping efficiency of these sacks, an experiment was set up in the lab which aimed to replicate the sack traps used at Flow Moss. A combination of 100 g of peat per litre of water was poured into a polypropylene bag which was suspended over a plastic box. The experiment was left over night to allow the water to drain from the trap. The collected water was poured into weighed beakers and oven dried to establish how much peat had passed through the trap and into the box. The sack trap was emptied in the same way traps collected from Flow Moss would be and the contents of the trap transferred to beakers to be oven dried and then re weighed. By comparing the amount of peat added to the sack with the amount left in the trap at the end of the experiment, a 'trapping efficiency' can be calculated. The results from this experiment identified that the trapping efficiency of these traps was 91.4%; this value can be used to calculate the error of the data collected from the sack traps during this study and is shown in the results section.

Similar to the wind flux samplers, the sack traps were changed approximately every two weeks. The sediment contained within these was oven dried over night at 105° and then weighed. Data collected from the sack traps during this will be collated with data collected by Baynes (2012) to produce a data set from April 2011 to March 2014.

4.3 Characterising the Subsurface Properties of the Peat Store

4.3.1 Ground Penetrating Radar (GPR) data collection

Estimating soil carbon stocks contained within peatlands is reliant on the accurate measurement of peat volume (Parsekian *et al.,* 2012). Holden *et al.* (2002) suggest that traditionally, techniques such as soil coring or pit excavation have been commonly used but these methods are destructive and provide an incomplete characterisation of the peat subsurface often resulting in considerable uncertainty (Proulx-McInnis *et al.,* 2010). Ground Penetrating Radar (GPR) is a geophysical technique for noninvasively identifying changes in dielectric permittivity between soil layers. GPR is non-destructive, and produces more detailed data than those collected using point measurements of depth (Kettridge *et al.,* 2008). GPR involves a transmitting antenna generating a high frequency electromagnetic wave that penetrates the subsurface of the peat. This then returns to a receiving antenna as a sequence of reflections from boundaries between materials with contrasting electromagnetic properties for different properties, such as the boundary between different types of soils or peats with different levels of saturation (Kettridge *et al.,* 2008). This information can be used to calculate a distance to the subsurface layer and thus the depth of the peat.

A preliminary GPR survey has previously been completed at Flow Moss by Baynes (2012). GPR was used to produce nine profiles of subsurface topography that were spaced approximately 50 m apart. Interpolation between these nine profiles provided a means of estimating the total volume of peat. In the present study, the GPR survey of Flow Moss was repeated, but at a much higher resolution. The GPR equipment used during the survey was a Mala RAMAC GPR and a 200MHz unshielded antenna. This was linked to a linked to a Leica 1200 RTK differential GPS which provides a sub-meter position fix, (typically < +/- 0.02m) which surveyed the topography of the site during the collection of depth measurements..

The GPR and GPS antennas were fixed onto a wooden sledge, allowing the system to be dragged across the uneven Flow Moss terrain while minimising loss of contact between the GPR antennae and the peat surface. The DGPS antenna was attached to the centre of the GPR sled so that GPS coordinates were collected concurrently with GPR depth measurements. Depth profiles were collected at approximately 2 m spacing across the entire site from the northwest fence to the southwest fence. This allowed a much more detailed estimation of peat depth to be made from the 104 depth transects which were collected.

The data collected using the dGPS were post processed using Leica Geo Office to produce a file containing the *x, y, z* co-ordinates of where the GPR depth measurements were recorded. DGPS and GPR data can be integrated during post processing to create a realistic sub-surface profile that takes into account the detailed topographic variation of the field site. The data collected using GPR were calibrated using depth probe measurements. To achieve this, four transects were selected and markers placed on the GPR transects. Flags were placed in the ground at these locations and a depth probe used to record the depth of peat at each location. This produced 40 calibration points that could be compared with the GPR results.

4.3.2 GPR data processing

The collected data were post processed using REFLEXW software (Version 7.1.6, Sandmeir, Karlsruhe, Germany).

To process the data a work flow consisting of four steps was followed:

1. Import RAMAC data and convert into REFLEXW-formatted data file.
2. 1D Filtering of individual traces: Subtract mean - the running mean was subtracted from the central point which eliminates low frequencies.
3. 2D filtering: background removal – subtracts an average trace from the profile, can eliminate noise from the whole profile.
4. Topographic correction is applied to the data using the GPS coordinates which are imported into REFLEXW.

Once the data have been processed, peat depths can be extracted from the data by manually picking layers. A mean layer velocity (0.04 m ns^{-1}) is specified which allows picked layers to be converted into peat depths. An ASCII file can then be created and imported into ArcGIS. The surface and subsurface layers were then re-projected to OSGB co-ordinates and fields added to calculate AOD height of the peat layer and the peat thickness.

4.3.3 Peat core analysis

In this project, four peat cores were collected and analysed (Objective 2, Chapter 1) for carbon content, moisture content, bulk density and metals content (Table 4.3). The cores were collected along a transect with the aim of characterising the peat at the site and possibly identifying a hiatus in core stratigraphy which could be indicative of erosion. The cores were collected from both vegetated and bare peat areas in an attempt to establish whether there was a difference in peat properties. Cores were also collected from an area where the peat was thought to be deepest (FMC1) and areas where the peat depth was expected to be shallower (FMC2) to identify changes in peat properties between eroded and intact areas of peat (Figure 4.4 **(c)**). The results from these cores can be compared to assess variability across the site.

Table 4.3 - *Information about the cores collected during this study (Figure 4.4* ***(c)****).*

Core	Date collected	Depth (m)	Grid reference
FMC1	08/03/2013	3.50	NY 80325 53577
FMC2	06/05/2013	1.17	NY 80505 53763
FMC3	24/06/2013	2.50	NY 80446 53619
FMC4	24/06/2013	1.75	NY 80533 53877

4.3.4 Peat properties

The peat cores were subdivided into sections identified from changes in the stratigraphy of the core analysed using the Troels-Smith and von Post classification system. The von Post classification scale is a widely used measure of the extent of decomposition of peat (Grover and Baldock, 2013), while the Troels-Smith classification can be used to describe the darkness, dryness and stratification of the peat (Troels-Smith, 1955).

Stanek and Silc (1977) assessed three different methods (von Post's classification of humification; Munsell colour charts; unrubbed fibre content in percent of total peat and rubbed fibre content in percent of total peat sample) to establish which was most appropriate for classifying the degree of humification of peat. They concluded that all three methods indicate the state of decomposition of the peat and that the von Post method is extremely well suited to field use as it required no equipment and was the least time consuming. However, even though the Troels-Smith and von Post classification methods provide a useful means of classifying peat, both methods are subjective and this may lead to minor differences in consistency. Following the von Post and Troels-Smith classification, the peat cores were divided into subsamples at 3 cm sampling intervals. Each of these were further divided into two 1.5 cm^2 sections, one of which was freeze dried for total carbon and metals analysis and the other used for bulk density, moisture content and loss on ignition analysis.

4.3.5 Carbon content estimation

One of the motivating factors for peatland restoration is the loss of carbon, thus it is essential to gain as much information about the amount of C stored within the peat. To gain information about the carbon content of the peat at Flow Moss, both LOI and Total Organic Carbon (TOC) methods were used.

For LOI analysis, samples that had previously been oven dried and weighed for bulk density analysis were placed in a furnace and heated at 550°C for four hours (Heiri *et al.,* 2001). The samples were then placed in desiccators to cool and reweighed.

LOI is widely used as a method to calculate the approximate amount of organic matter and carbonate mineral content and, indirectly, the amount of organic and inorganic carbon within sediments (Santisteban et al. 2004). LOI is often regarded as a useful tool and one of the most convenient assessment methods for determining the organic matter and carbon content of sediment (Dean, 1974). However, De Vos et al. (2005) question its accuracy for predicting total carbon. Therefore, the peat samples were analysed for total organic carbon (TOC). To do this, the freeze dried peat samples were ball milled and 1 mg of each sample weighed into a tin capsule. These capsules were placed into the Costech CHNS-O combustion reactor, for which De Vos *et al.* (2005) state the detectable limit is 0.1 to 30 mg of organic carbon. Here the samples were heated to 1700-1800° causing the sample to breakdown into its elemental components, N_2, CO_2, H_2O and SO_2. The gas was passed through a gas chromatograph separation column which detects sequentially the amount of each element contained within the sample. During analysis of the samples, a reference material for which the carbon content was known was also analysed to test the accuracy of the machine. This was found to be reporting carbon content values at 99.5% accuracy.

LOI and TOC results were compared for cores from different locations at Flow Moss. This allowed for a spatially distributed data set displaying peat carbon content to be collated. Grove and Bilotta (2013) used both LOI and TOC methods to calculate the fluvial particulate organic carbon of sediment and their results showed that LOI may be a poor indicator of total organic carbon content of a sample.

4.3.6 Metals content analysis

Once the samples had been freeze-dried and ball milled, 250 mg of each sample was weighed out for metal content analysis. The concentrations of metals contained within the peat profile were identified using Inductively

Coupled Plasma Mass Spectrometry (ICP-MS). ICP-MS uses mass spectrometry to detect the presence of metals.

In this study, there are several reasons for analysing the metals content of the peat profile. Firstly, as stated in the site description, Flow Moss is located in close proximity to two large chimneys which were previously used for dry condensation of fumes emitted from lead smelting. This could have led to the contamination of the near surface of the peat. Secondly, information gained from the metals analysis may assist in the understanding of surface erosion occurring at Flow Moss.

When collecting the four peat cores, it was thought that the deepest area of peat was approximately 3.50 m (the length of FMC1), and this was considered the baseline depth. However, the GPR survey has since shown that Flow Moss peat depths reach 4.50 m. Nevertheless, as FMC1 was the most intact peat profile of the four collected cores and measured 3.50 m (which would have been the maximum local depth); this was considered the baseline depth for the comparison of the peat stratigraphy. It is possible that in locations where peat cores of less than 3.50m were collected; the surface of the peat has been eroded resulting in peat of a shallower depth. If this is correct, when the metals results are plotted against each other (using 3.50m as the maximum depth) it should be possible to use spikes in metal concentrations as stratigraphic markers of erosion (accepting that rates of sedimentation will vary locally). The rich lead mining heritage of the area surrounding Flow Moss suggests that the most important metal to consider when examining heavy metal concentrations in peat would be lead. Furthermore, there has recently been growing concern over the mobilisation of lead from upland blanket peat soils to surface waters (Rothwell *et al.,* 2007). A study by Rothwell *et al.* (2007), collected peat cores for down-core profile lead analysis from Alston Moor in the Peak District. Profiles were constructed for the top 30 cm of the cores. The results found that within individual peatland sites there is significant spatial variability in lead pollution records and it is therefore necessary to sample multiple cores to get a true representation of the lead record. Furthermore, the results indicated that far higher lead concentrations were recorded in the upper peat layer. This collaborates results found in other studies (e.g. Livett *et al*., 1979; Cloy *et al.,* 2005) and suggests that lead concentrations could indicate past periods of high emissions (likely due to lead emissions from industry, fossil fuels and vehicle emissions). Lead is a relatively stable metal, and spikes in the peat profile could have been caused by periods of deposition, while troughs could be indicative of erosion.

4.4 Monitoring changes in surface vegetation cover

In this study the extent of vegetation at Flow Moss was mapped using archival UAV data (2011) and contemporary dGPS survey. Unmanned Aerial Vehicles (UAVs) can provide high resolution aerial imagery. Increasingly, UAVs are being used as autonomous and low-cost remote sensing platforms to provide data over a large spatial extent, often which is used in agriculture and ecological mapping (Kleinebecker *et al.,* 2013; Anderson and Gaston, 2013). Examples of recent applications of UAV technology to environmental monitoring include the mapping of vegetation dynamics (Laliberté *et al.,* 2010 and Gademer *et al.,* 2010), precision agriculture (Lelong et al. 2008), and the analysis of post-flood vegetation patterns (Hervouet *et al.,* 2011).

Kleinebecker *et al.* (2013) state that UAV platforms may bridge a gap that exists between the need for an accurate means of monitoring species composition changes and vegetation structure in restored peatlands and the considerable effort that is required for vegetation field surveys.

Objective three of this study (Chapter 1.1) was to ‘Undertake a survey (dGPS) of the vegetation cover at the site and compare results with data collected in April 2011 (UAV survey) and historical imagery (Google Earth 2007) to assess changes in surface vegetation cover’. Initially a repeat UAV survey was planned, however, UAV equipment was not available during this study; therefore, UAV data collected in 2011 (Baynes, 2012) were compiled in Agisoft Photoscan to create a mosaicked image. The data were digitised in ArcMap to establish the extent of the bare peat at Flow Moss in April 2011. A Goggle Earth satellite image was obtained from January 2007, geo-rectified and the area of bare peat digitised from this image to allow comparison of the bare peat area in 2007 and 2011. Finally, a dGPS survey was undertaken at Flow Moss in April 2014 and areas of bare peat and vegetation mapped. Using these three different methods a comparison of vegetation change across three different epochs spanning 7 years was undertaken.

4.5 Summary of methods

This chapter has outlined the methods which were implemented in this study. These form an integrated programme of measurements (Figure 4.1) designed to answer the research questions stated in Chapter 1. This has been done to gain information about how processes operating both at the peat surface and the properties of the subsurface may affect the amount of erosion, and thus carbon, lost from Flow Moss. The temporal continuity of these various measurements is summarised in Figure 4.7 which shows the data collected in this study compared to that collected by Baynes (2012). The monitoring period by Baynes (2012) covered October 2010 – July 2011. This was followed by a one and a half year gap before monitoring for the present study began in March 2013 and ended in April 2014.

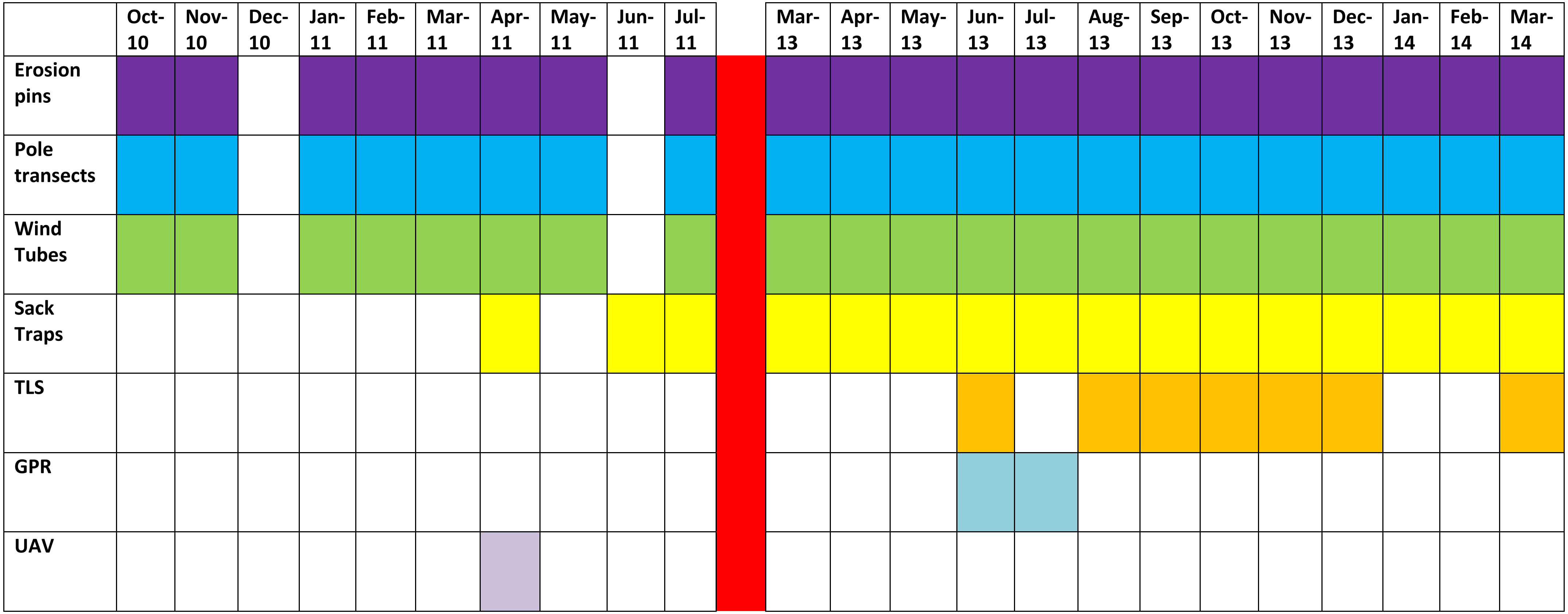


Figure 4.7 *– Timescale for data collection. The red segment indicates the end of data collected by Baynes (2012) and the start of data collection for this study*

5. Flow Moss subsurface results: properties of the peat and carbon store

This chapter discusses the results of the subsurface properties of the peat at Flow Moss. It includes a description of the peat stratigraphy, metal concentration profiles, geometry of the peatland and estimates the amount of peat and carbon stored at the site.

5.1 Peat Cores

In order to fulfil Objective 1: ‘Sample the local peat to establish the variation in bulk density of the peat samples and assess the carbon content and organic matter content using Loss on Ignition (LOI) and Total Organic Carbon (TOC)’ four peat cores were collected and analysed. The peat cores were collected in a transect across the field site from the vegetated peat in the south (Core FMC1), through the bare peat flats (Cores FMC2, FMC3) to the vegetated peat in the north (Core FMC4) (Figure 4.4, Table 4.3). By sampling in this manner, it was expected that FMC1 would provide the most complete record of peat accumulation at the site whilst the other cores would be less complete due to erosion. Cores were analysed for carbon content using both Loss on Ignition and Total Organic Carbon (TOC) methods and metals content was established using ICP-MS (Chapter 4, Section 3.3). The general aim of this approach was to assess whether there are differences in results spatially across the site, and whether changes in peat properties can be used to identify stratigraphic markers of past erosion.

5.1.1 Peat properties

The four peat cores were subsampled into 298 strata, identified from changes in the stratigraphy of the core which had been classified according to the Troels-Smith and von Post classification schemes. The decomposition stratigraphy of the four peat cores, identified using the von Post scale of humification, is shown in Figure 5.1. This method of classification ranges from H1 to H10, with 'H1' defining the least decomposed, fibrous peat while 'H10' defines peat which is the most decomposed, darkest material at the opposing end of the scale. The degree of decomposition of the peat will influence important characteristics such as permeability, water holding capacity, bulk density and fibrosity. These are further impacted by anthropogenic activity, for example drainage, which will accelerate the rate and degree of decomposition, increase bulk density and decrease fibrosity (Hammond, 1981).

Figure 5.1 shows, the humification stratigraphy of the four cores. As expected, the general level of humification increases with depth. This is because the peat deeper in the profile is older and more compacted and has been subject to decay for the longest time. Figure 5.1 also shows that FMC2, which is the shortest core, contains the least decomposed peat with little evidence of more humified basal peat. This core is from the bare peat flats where episodic erosion may have removed the surface peat (Figure 4.4 **(c)**). If the horizon of the peat in FMC2 was truncated due to erosion at an earlier date, when erosion ceased, new peat would begin accumulating. This could have resulted in the observed differences in the decomposition of the peat within the core. In contrast, the cores showing the highest level of decomposition (FMC1 and FMC3) are the cores of the greatest length and were located in vegetated areas, indicative of more intact peat profiles. If it is presumed that the local base of the peat is around 3.50 m (the depth of peat in FMC1 – which is assumed the intact peat profile), then the varying depths of peat in cores FMC2-4 will reflect differences in erosion rates (surface peat removal) and rates of peat accumulation. Dating of the deposits would help resolve these questions but that is beyond the scope of the present project.

The Troels-Smith (1955) classification defines the broad peat stratigraphy identifying physical peat characteristics including degree of darkness, stratification and elasticity. The Troels-Smith method, used alongside the Von Post classification system, provides analysis of the peat properties (Martini *et al*., 2007). This was done using methods outlined in Chapter 4, section 3.4. Table 5.1 provides a summary of the results of the Troels-Smith classification, while Figure 5.1 schematically displays the humification of the four peat cores.

Although all four cores were different lengths, they identify similar key features in the peat stratigraphy. Just below the surface, the peat was classified as *'Tl'*; this describes peat samples which contain roots, intertwined rootlets and rhizomes of herbaceous plants. Towards the base of the cores, the properties were variable, but much of the peat was classified as *'Dg'* (containing fragments of ligneous and herbaceous plants <2mm and >0.1 mm) and *'As'* (Containing clay). Table 5.1 also shows that there is a tendency for elasticity to increase with depth and the degree of stratification in some of the cores is more marked lower in the peat profile.

The decomposition values recorded for the four Flow Moss profiles differ to those found by Baynes (2012) (Chapter 3, Section 5). The results found by Baynes classified all the peat samples as ‘H7’ with no variation in humification with depth. Results collected during the present study show a far greater level of variability with values ranging from ‘H2’ to ‘H9’. There are several possible reasons for the differences in peat properties recorded by Baynes (2012) and recorded in the present study. Firstly, the cores sampled by Baynes (2012) were collected in different locations to the cores collected in this study. The cores classified by Baynes (2012) were collected from the bare peat area, where the peat is likely to have been subjected to erosion, and thus older more humified peat is less likely to be present in the core. This explains why higher humification values were recorded during the present study. Furthermore, even on a local scale, differences in core properties can occur as erosion and deposition dynamics will vary. This is corroborated by the results from the cores sampled during the present study. Cores FMC2 and FMC4 were both collected from the bare peat area at the northern end of the site. However, as can be seen in Figure 5.1, the properties of the cores are very different, with much lower humification values being recorded for FMC2. Finally, the Von Post and Troels-Smith methods are somewhat subjective and this can lead to discrepancies when classifying the peat samples.

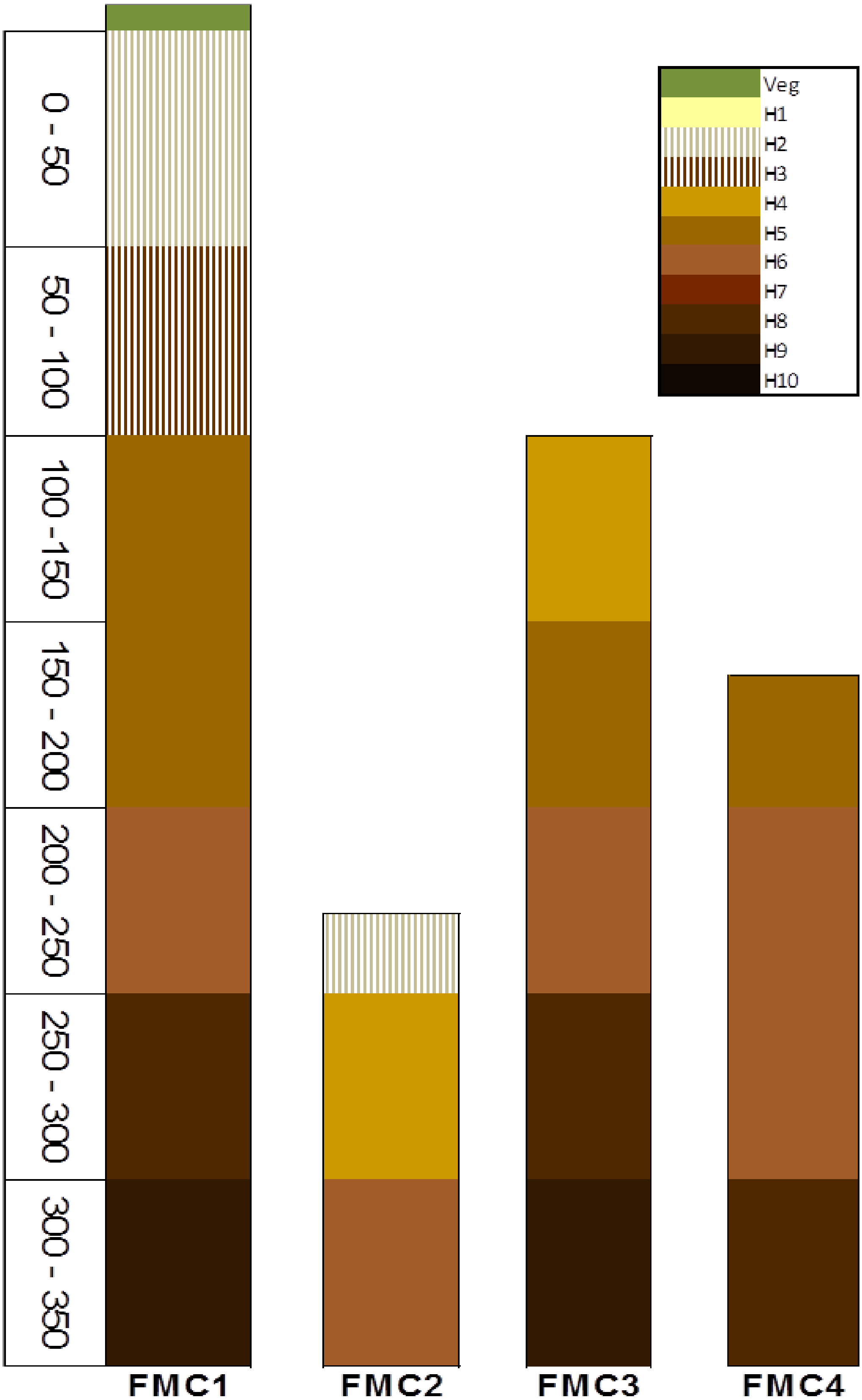


Figure 5.1 – *Stratigraphy of the four peat cores showing the humification profiles. The peat cores were distributed along a transect across the site. The base altitudes for the cores were 445.5, 447.83, 446.5 and 447.25 m.a.s.l respectively. In this figure, cores are aligned using the base of the peat as a consistent horizon.*

***Table 5.1** – Results from the Troels-Smith classification*

Depth (cm) from surface	Code	Darkness	Stratification	Elasticity
		FMC1 – 350 cm		
0-50	Tl	Nig 2	Strf 1 -	1-2
50-100	Dl	Nig 2	Strf 0 - 1	1-2
100-150	Dg	Nig 3	Strf 0 - 1	1-2
150-200	Dg	Nig 3	Strf 0	2-3
200-250	Dl	Nig 3 -4	Strf 0	2-3
250-300	Dg	Nig 3-4	Strf 0	2-3
300-350	As	Nig 4	Strf 0	3
		FMC2 – 117 cm		
0-50	Tl	Nig 2	Strf 2/3	0-1
50-100	As	Nig 2	Strf 1/2	1-2
100-117	Dg	Nig 3	Strf 2	2-3
		FMC3 – 248 cm		
0-50	Tl	Nig 2	Strf 3	0-1
50-100	Ag	Nig 3	Strf 2/3	1-2
100-150	Dg	Nig 3	Strf 1/2	1-2
150-200	Dg	Nig 3-4	Strf 1	2-3
200-250	Dg	Nig 4	Strf 1/2	2-3
		FMC4 – 175 cm		
0-50	Sh	Nig 3	Strf 1	1-2
50-100	Sh	Nig 3	Strf 2	2-3
100-150	Dh	Nig 4	Strf 2	2
150-175	Sh	Nig 4	Strf 1	1

Following the initial von Post and Troels-Smith classification, samples from the four peat cores were analysed for bulk density (Figure 5.2). Bulk density measurements are essential for quantifying the amount of peat stored at in a peatland (Chambers *et al*., 2011). Figure 5.2 shows the range of bulk density values recorded from the four cores, while Table 5.2 shows the descriptive statistics of the bulk density data. The oven dry bulk density of the peat samples ranges from 0.0163 g cm^{-3} to 0.380 g cm^{-3}. Chambers *et al*., (2011) state that peat bulk density is variable and typically, the recorded bulk density of peat in high latitude regions is in the range of 0.05 to 0.2 g cm^{-3}. This is similar to the values from the four Flow Moss Cores.

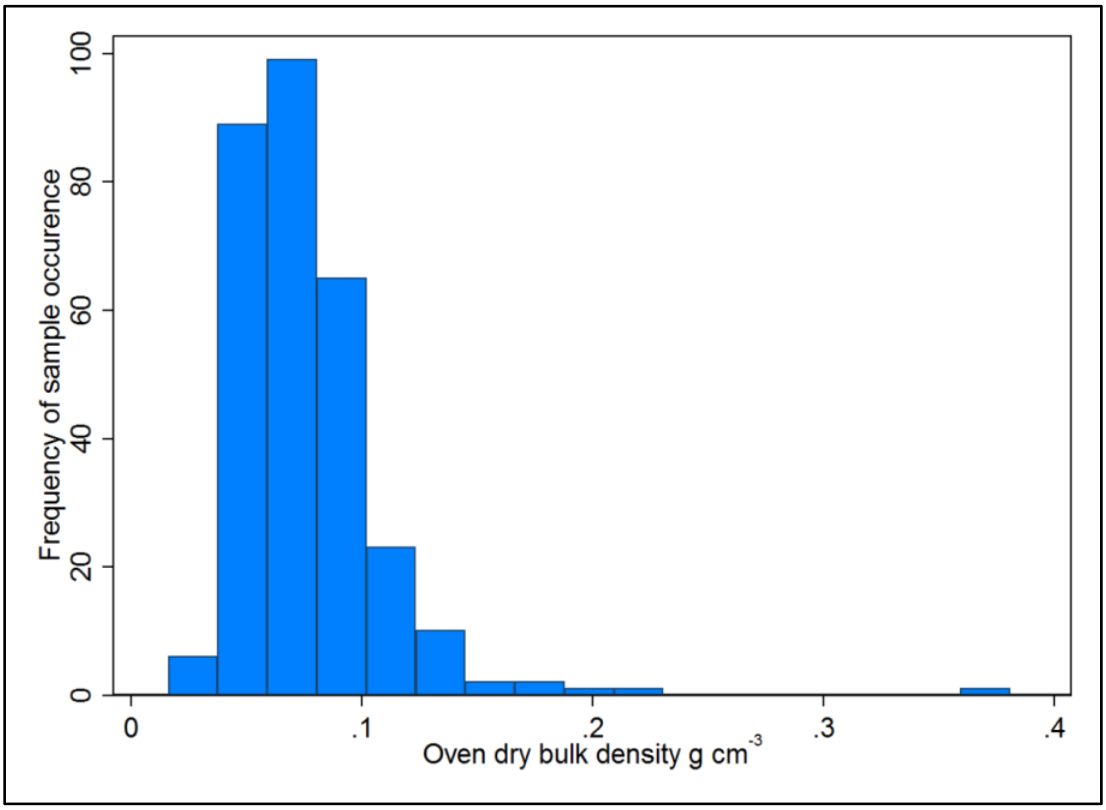


***Figure 5.2** – Histogram displaying the bulk density values recorded from the four peat cores (all samples).*

***Table 5.2 –** Descriptive statistics of the bulk density data*

	Bulk density (g cm^{-3})
Mean	0.078
Minimum	0.016
Maximum	0.381
Median	0.069
Standard Deviation	0.031

The bulk density results were plotted for each core to establish if changes in bulk density with depth could be determined (Figure 5.3). In these plots, the convention is to align the cores at their bases because it is assumed differences in core length are attributed to surface erosion. This figure identifies substantial variation in bulk density within and between cores. Several factors contribute to local variations in peat properties. These include:

- Variation in water content
- Variations in the accumulation and decomposition of peat,
- Differences in nutrient uptake by vegetation,
- Varying intensity of leaching/precipitation (Laiho *et al*., 2003).

Variations in field bulk density primarily are related to water content (Jepsen *et al*., 1997) and a decrease in the water content of peat may lead to an increased bulk density due to compaction during accumulation (Laiho *et al*., 2003). Jepsen *et al*. (1997) state that ordinarily bulk density will increase with depth as the water contained within pore spaces is forced out due to the mass of the overlying sediment. Figure 5.3 supports this hypothesis and shows that although variation of bulk density values exists within the four cores, generally, the bulk density does increase with depth. However, there are local variations in the bulk density which may occur due to the presence of mineral sediment or the trapping of water/gas, leading to a lower bulk density (Jepsen *et al*., 1997). FMC2 shows the greatest variability in bulk density with many spikes in density which are related to enhanced mineral content of the peat. This is consistent with the assumption that FMC2 is a disturbed and eroded profile, as local erosion would wash mineral mater into the disturbed peat.

To establish if a relationship exists between bulk density and moisture content values, the results from the moisture content analysis of the peat cores were also plotted (Figure 5.4). The moisture content data would suggest that there is a slight decrease in the amount of water contained within the peat at greater depths, however, overall there is very little vertical variability within the cores. The moisture content and bulk density data sets were plotted on a scatter plot and a regression line added (Figure 5.5) this figure confirms that an inverse relationship exists between the two variables and as bulk density increases, moisture content decreases. Outliers in the plots correspond to the peaks shown in Figure 5.3 which are typically peat layers with added mineral content possibly reflecting local in wash of sediment from erosion.

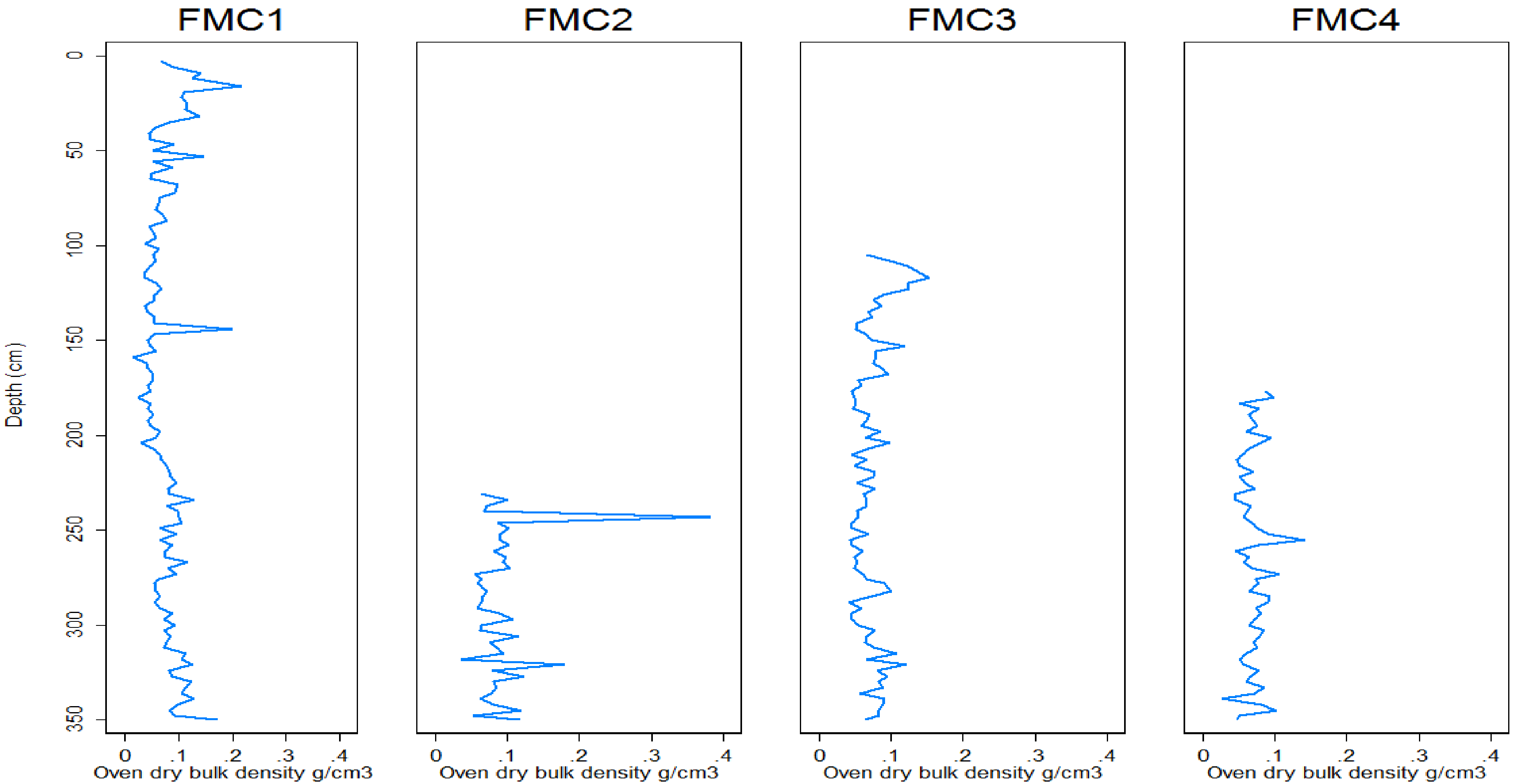


***Figure 5.3** – Bulk density values recorded for the four peat cores. Generally, the bulk density increases with depth.*

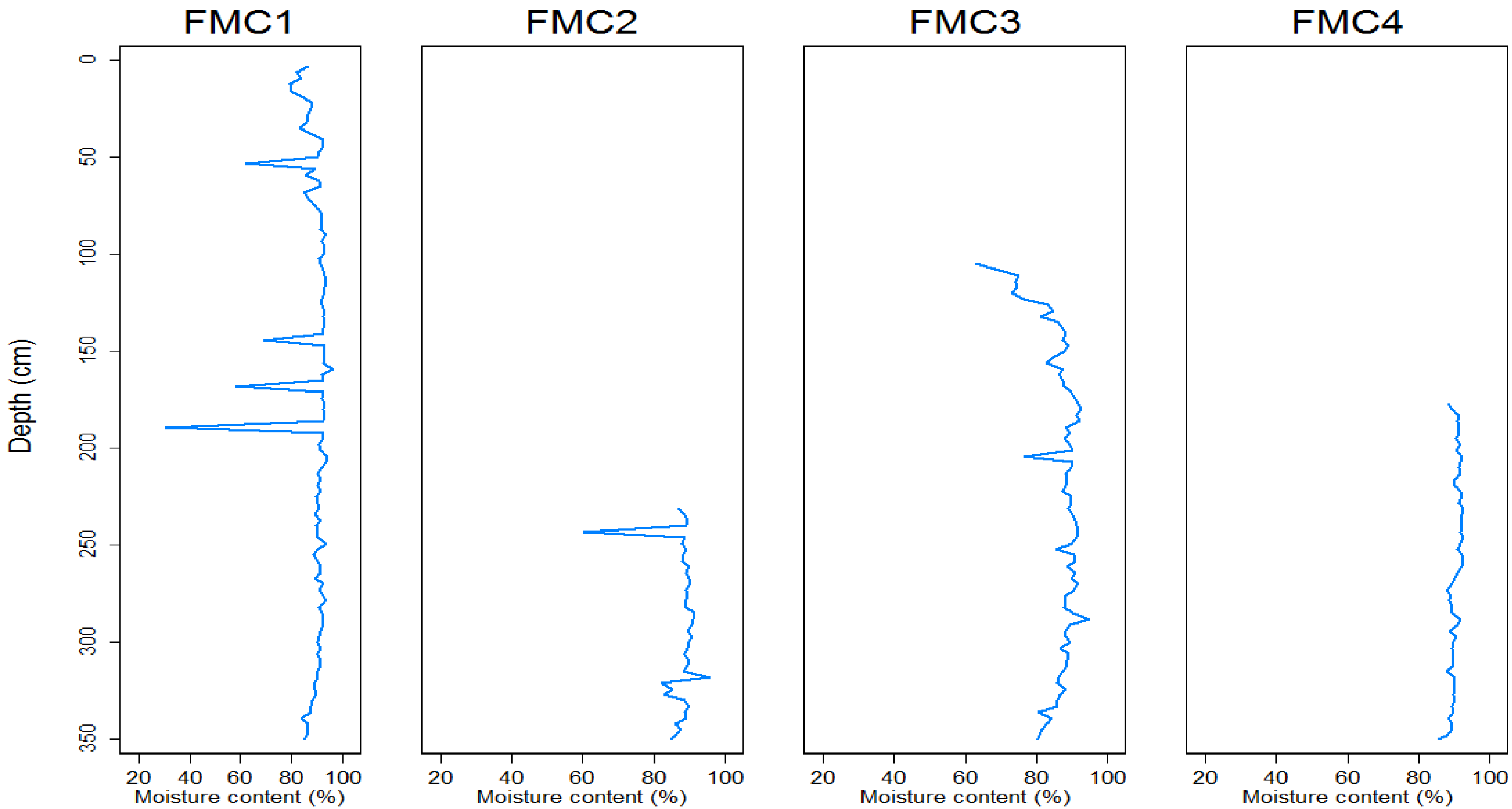


Figure 5.4 *– Moisture content of the peat cores. The data show that there is a very slight decrease in moisture content at the end of the core, however overall, there is little vertical variability.*

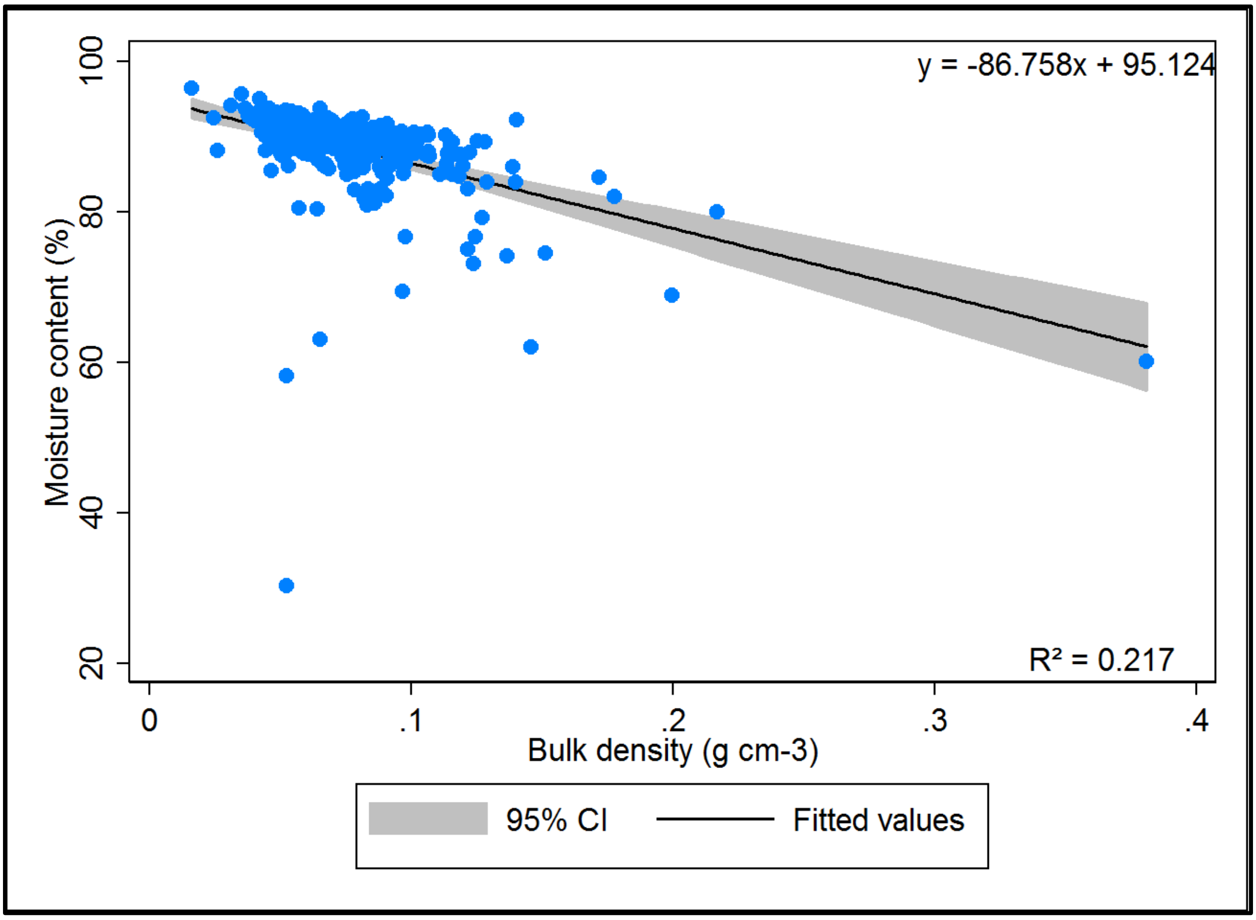


Figure 5.5 *– Moisture content of the peat cores plotted against bulk density values. A negative correlation can be identified.*

The mean dry bulk density value for all peat cores was 0.078 g cm^{-3} with a standard deviation of 0.032 g cm^{-3}. One sample in FMC2 displays a bulk density measurement of well above the mean value at almost 0.38 g cm^{-3}. This outlier has a strong influence over the regression line plotted in Figure 5.5. Nevertheless, the majority of the 'true peat' data cluster around the regression trend in the upper left of the graph where values of moisture content are greater than 80% and bulk density is c. 0.1 or lower.

In Section 5.3.1 these figures will be converted into values of kg m^{-3} and used alongside data collected in the GPR survey to establish the amount of peat stored at Flow Moss.

5.1.2 Carbon content of the peat samples

Following bulk density analysis, samples were subjected to Loss on Ignition (LOI) testing, to establish the organic matter content which can be used as a proxy for carbon content of the samples. The LOI values for the four peat cores are shown in Figure 5.6. The samples were all found to have a LOI value of 50% or greater, with 297 (99.6%) samples containing greater than 70% organic matter.

Similar to bulk density, variations in LOI values occur at similar depths in each core. All four cores show lower values of organic matter content near the surface of the core, which represents the unconsolidated and fresh nature of the organic matter. Similarly, all cores show a decrease in LOI values at the base of the cores. This represents the transition from the organic peat to the clay layer below the peat which contains far less organic matter.

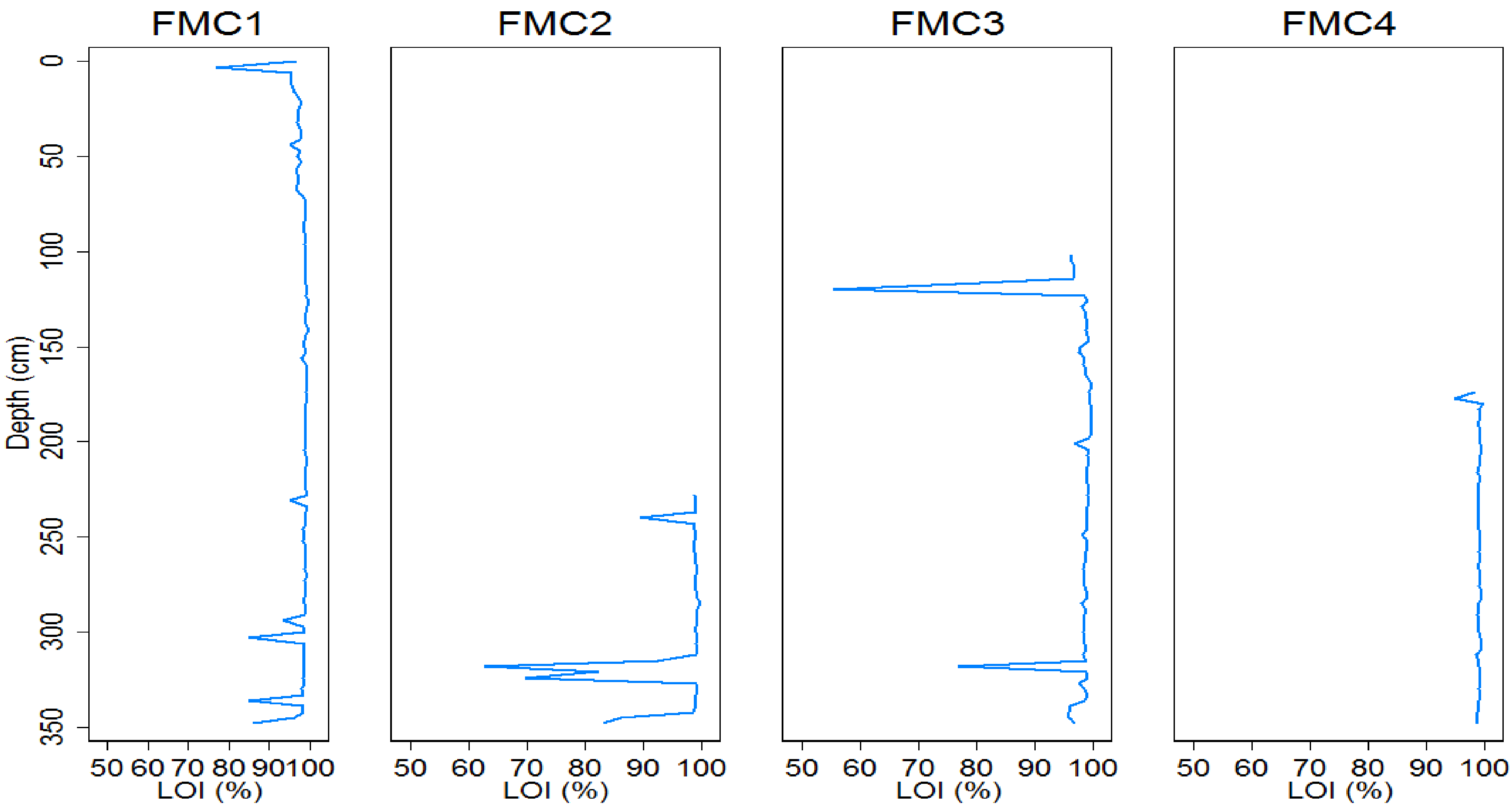


***Figure 5.6** – LOI values of the peat samples*

All of the subsampled peat had high organic matter contents; FMC4 had the smallest range of values, while FMC3 had the largest. Furthermore, Figure 5.6 indicates that FMC2 had the greatest variation throughout the peat profile.

Although LOI provides a convenient method for estimating organic matter content and can be used as a proxy for carbon content, its accuracy has been questioned (De Vos *et al*., 2005). Therefore, the carbon content of the peat was analysed directly using an oxidative combustion method to obtain TOC values (Figure 5.7). LOI values provide an estimate of the organic matter within a sample. To convert this to approximate values of carbon the values would need to be multiplied by a conversion factor, which De Vos *et al.*, (2005) state is 0.58. However, the LOI values are calculated based on the percentage weight loss which occurs when samples are ignited at 550°C. This reduces the organic matter in a sample to ash and carbon dioxide (Veres, 2002). Thus, the percentage of material lost on ignition identifies the percentage loss of combustible material contained within the sample which will ignite above 550°C rather than explicitly the percentage loss of carbon or organic matter. The TOC method (Chapter 4.3.5) heats each sample to 1700-1800°, causing it to breakdown into component parts which can then be directly measured. Although the two methods are dissimilar, they have both been used to provide an estimate of carbon content and it would therefore be expected that similar trends can be identified in both the LOI organic matter content and TOC data.

Similar to the LOI data, the TOC data indicate variations in carbon content in all four peat cores, indicative of periods of erosion and peat accumulation. An increase in carbon content can result from the accumulation of peat, where the addition of organic mass, and therefore carbon occurs (Chambers *et al*., 2011). This would again suggest that similar patterns should exist between the LOI organic matter values and the TOC values. Lower carbon and organic matter contents may suggest that periods of erosion have occurred, which may have prevented the accumulation of peat, and truncated the surface of the peat. A comparison of Figures 5.6 and 5.7 shows similarities between the LOI and TOC data. Both data sets show lower carbon and organic matter values at the surface of the cores. In all four cores, generally, there is a slight increase in carbon and organic matter content with depth, until the lowest 50 cm of the core when a decrease in carbon content occurs. This decrease is where the peat layer ends and the mineral layer below begins.

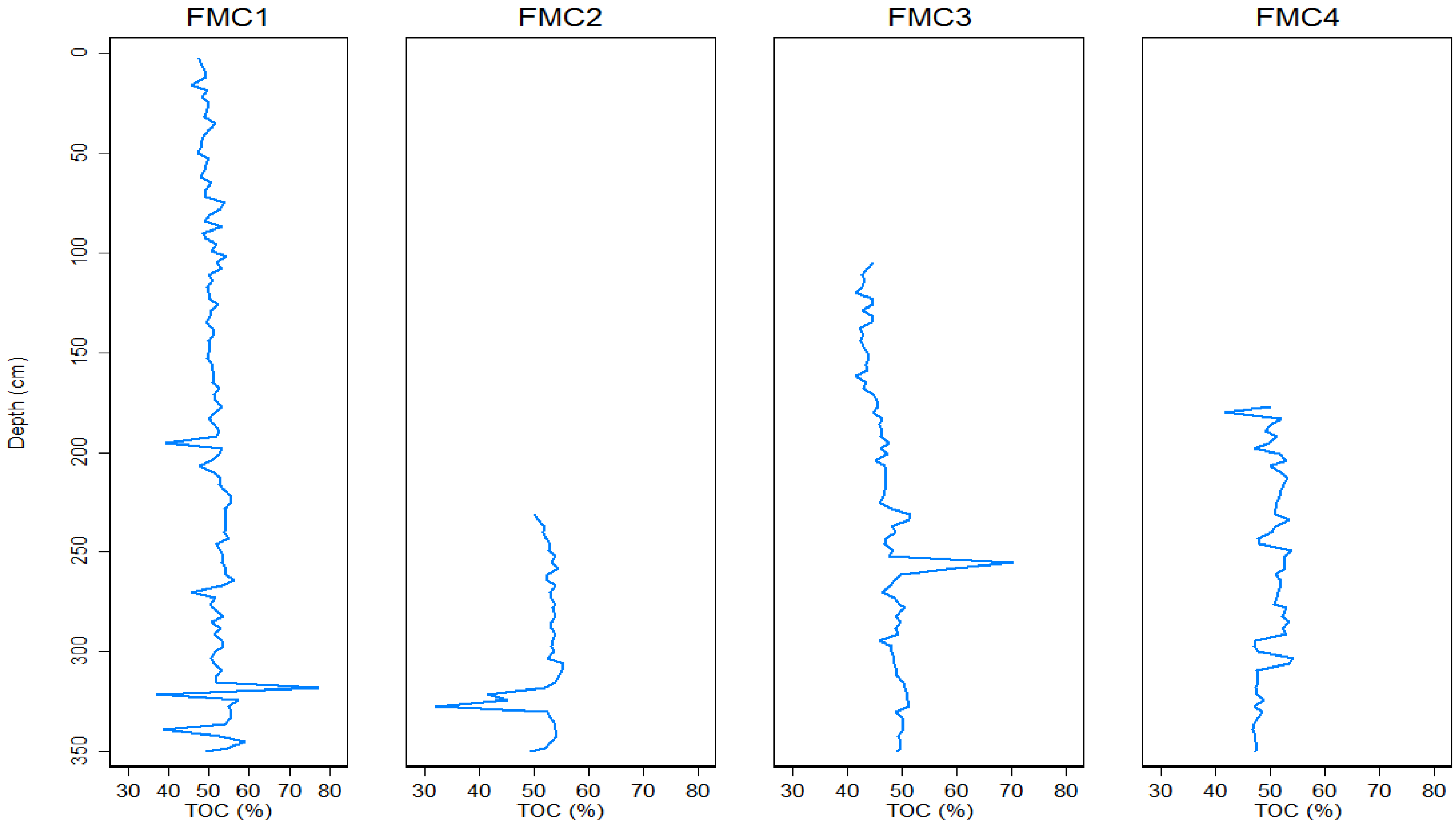


***Figure 5.7** – TOC values of peat samples*

The TOC results were plotted in box plots to identify the median, interquartile range, sample range and outliers contained within the data set. This is displayed in figure 5.8. This figure identifies that that the two cores collected form the bare peat areas (FMC2 and FMC4) are the cores with the least number of outlying data points, the median value of carbon content for the profiles collected from vegetated areas are lower than those of the peat profiles collected from bare peat areas. However, the fact that the two cores from bare peat areas are shorter, could influence the results, as it may be that the peat retained in the profiles is just of a certain type.

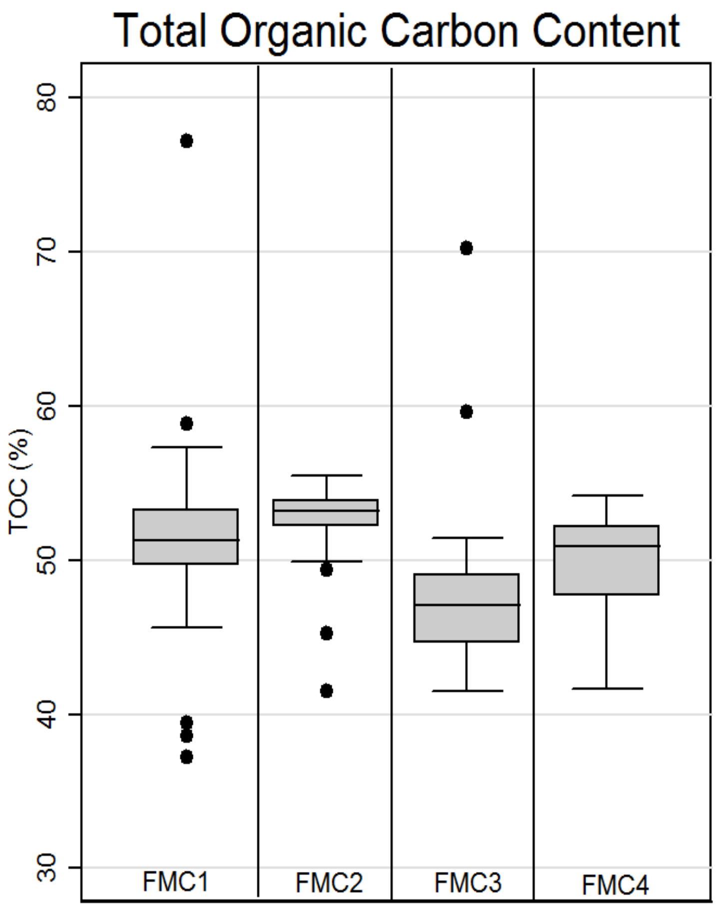


Figure 5.8 – *Box plot showing the variation of the TOC data for the four peat cores including the range, median and interquartile range. Cores are arranged in a transect (Figure 4.4(c)). FMC2 had the highest median TOC values, whilst FMC1, the longest core, had the largest range of values.*

The high level of variablilty recorded in FMC2 could be indicitive of the erosion of the bare peat flats, as it was collected from an unvegetated area. FMC4 was also collected from a bare peat area, however the local area setting for this core (see Figure 4.9) is very different to where FMC2 was collected from, with the peat haggs in close proximity to FMC4 providing some protection from episodes of erosion by wind, wind-driven rain and fluvial action.

5.1.3 Comparison of LOI and TOC values

As both LOI and TOC can be used to estimate carbon content, it would be expected that there would be a correlation between TOC and % organic matter content established using LOI (Figure 5.9).

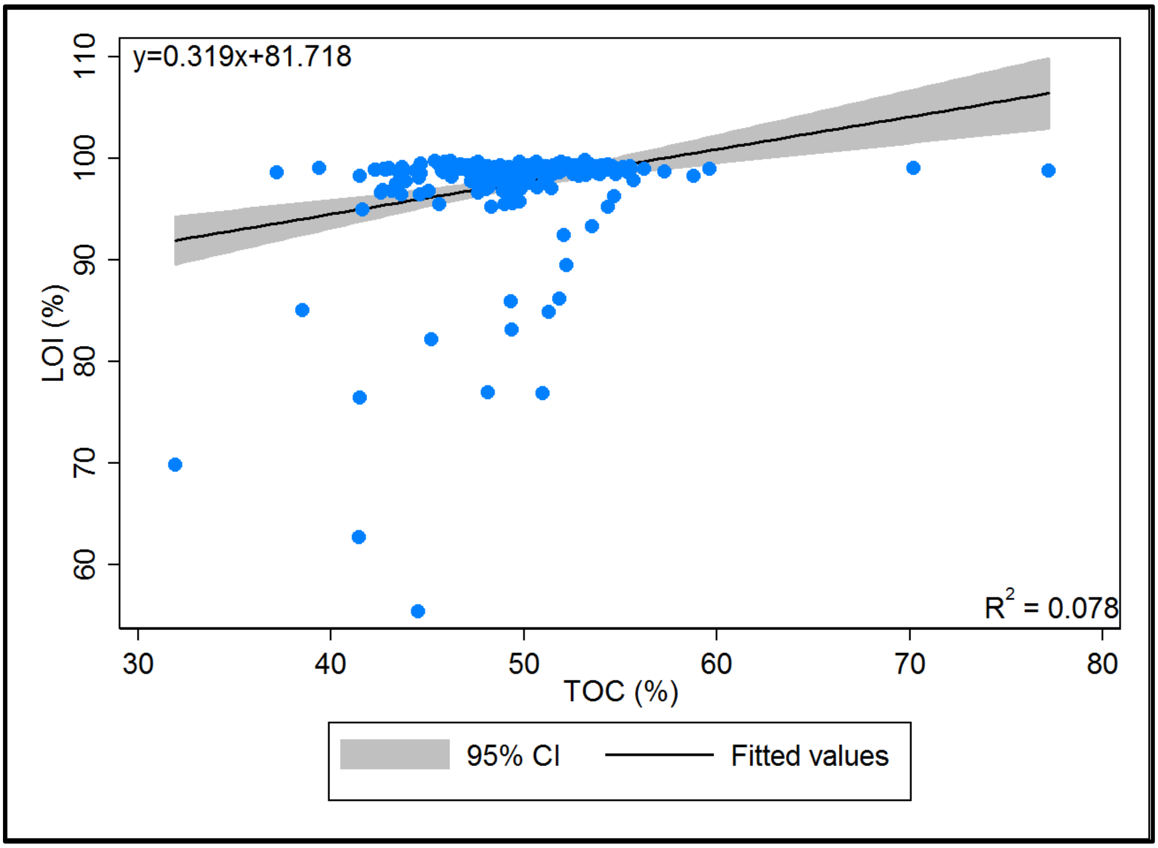


Figure 5.9 – *Scatter plot showing data obtained using TOC plotted against LOI data. The line was fitted using the equation y=0.3193x+81.718.*

An r^2 value of 0.0778 was calculated indicating that there is a weak positive correlation between the two data sets. However, the pattern and scatter in the data suggest the relationship is much more complicated and comparisons between peat soils with high organic content (>90%) and those outliers with lower organic matter contents cannot be captured in a simple linear relationship. Furthermore, the form of the regression equation of the fitted line in Figure 5.9 suggests a simple conversion factor of 50% may not be appropriate for estimating carbon content from LOI in all settings.

To confirm that there is no statistically significant correlation between the data sets, a test of significance was conducted. To establish whether this required a parametric or non-parametric statistical test, the descriptive statistics for each dataset were calculated (Table 5.3).

Table 5.3 – *Descriptive statistics calculated for the two data sets*

Descriptive statistic	Loss on ignition	Total Organic Carbon
Mean	97.71	50.08
Median	98.91	50.38
Skewness	-5.72	0.60
Kurtosis	37.63	8.18
Standard deviation	4.76	4.16

The results demonstrate that the LOI data are non-normally distributed (Table 5.3) and thus a non-parametric statistical test is appropriate. A Spearman's rank correlation (r_s = 0.205) shows there is there is no significant relationship between peat carbon content estimated using LOI and peat carbon content estimated using TOC.

Previous studies have highlighted concerns about the use of LOI for measuring the organic carbon content of sediment samples (Grove and Bilotta, 2013). Sutherland (1998) compared the methods of LOI and TOC for estimating carbon content of fluvial bed sediments. The results suggested that the data collected using a dry combustion analyser (similar to that used in this study, Chapter 4.3.5) had a better level of precision than the data produced using the LOI method. Grove and Bilotta (2013) corroborate this and found that measurements of POC derived using LOI were up to 16 times higher than those derived from oxidative combustion. These authors suggest that oxidative combustion is therefore the preferred method for the measurement of carbon content in sediment samples and, where possible, should be used rather than LOI.

5.2 Metal concentrations profiles in the peat

The metal mining landscape of the Northern Pennines is unique, containing a relatively large concentration of mines possibly dating back as far as the Roman period (Mighall *et al*., 2004). Blanket peats located close to industrial activities have typically been stores of atmospheric pollutants such as heavy metals (Rothwell *et al.,* 2005) and the geochemistry of peat deposits is often indicative of human activities (De Vleeschouwer *et al*., 2010). Smelting, fossil fuel combustion and vehicle emissions have led to peatlands across Europe receiving considerable inputs of trace metals (Rothwell *et al*., 2011). Peat profiles offer a useful archive of atmospheric metal deposition (Shotyk *et al*., 1998) and provide a possible method of recording anthropogenic metal emissions at a relatively high temporal resolution (centennial to decadal) (De Vleeschouwer *et al*., 2010). Furthermore, there has been an increased interest in the metal concentrations found in peat profiles due to concerns about the release of deposited contaminants from upland peat catchments to surface waters (Rothwell *et al*., 2008). Table 5.4 outlines some of the previous studies on metal concentrations found within peat profiles.

Table 5.4 *– A selection of previous studies of metal concentrations found within peat profiles*

Study	Metals	Key points
Livett *et al*., 1979)	Pb	• Major increases in lead concentrations at 8 cm and 6 cm depth
Shotyk *et al*., 2002	Pb	• Highest lead concentrations found near surface of profile
Rothwell *et al*., 2007	Pb	• Lead concentrations highest in near surface peat
Rothwell *et al*., 2010	As Sb Pb Cu Fe Mn	• As and Sb are potentially toxic trace elements resulting from fossil fuel combustion, mining and smelting activities. • Highest metal concentrations found in near surface
Vleeschouwer *et al*., 2010	Pb Ti Cu	• Lead from mining and leaded petrol
Rothwell *et al*., 2011	As Pb	• Arsenic has been deposited due to the presence of this metalloid in coals and ore minerals

The subsamples from the four peat cores were subjected to metals content analysis. This was done for two reasons. Firstly, to assess whether metals concentrations of the peat could be used as stratigraphic markers of past erosion, and secondly, to establish if there were differences in metals concentrations between areas of bare and vegetated peat. Examples of metals used as stratigraphic markers or erosion are provided by Nikitina *et al*., (2014), who used lead concentrations of sediment cores collected from a salt marsh to gain information about erosion caused by tropical cyclones.

In the present study, the concentrations of lead (Pb), antimony (Sb), arsenic (As) and iron (Fe) have been analysed. These metals were selected as previous studies (Table 5.4) have identified them as the metals most likely to be produced during lead mining activities.

5.2.1 Metals concentrations as stratigraphic indicators of erosion

To assess how metal concentration varied with depth, results were plotted against sample depth (Figures 5.10 – 5.13). It was assumed that core FMC1 contained a largely intact peat profile and therefore represented some of the deepest peat at Flow Moss (c. 3.50m); and is considered the baseline for comparison with the other cores. To confirm this assumption a full pollen analysis of the core, accompanied by reference dating would be required but this is beyond the scope of the present study.

The first metal concentrations plotted (Figure 5.10) are those recorded for Pb. The plots show higher levels of Pb in the first 50 cm of both FMC1 and FMC3. Mining is one possible cause for the observed increase in lead concentrations, and Livett *et al*., (1979) state the major increases in Pb often displayed at around 6-8 cm depth are almost certainly a reflection of increased mining and smelting at the time of the industrial revolution. FMC1 shows a double peak in the first 50 cm of the core, and this could be resultant from mining activity in the area. However, mining is not the only possible source of Pb contamination, with fossil fuel and vehicle emissions both offering possible reasons for observed spikes. A rise in Pb concentrations observed in the upper 2 cm of the peat profile may show the effect of the combustion of Pb additives in petrol. Pb is relatively immobile (Cloy *et al*., 2005) in comparison to other metals and therefore can provide a reliable indicator of the levels of Pb the truncated peat was exposed to. Sb and As (Figures 5.11 and 5.12) both display similar trends to those recorded for Pb. This will primarily be due to the production of these metals during the mining and smelting processes. Sb is found in an array of sulphide minerals and coals, and so it would be expected that anthropogenic emissions would be similar to that of Pb and thus similar signals recorded in the peat cores.

Fe shows a very different trend to the other metals (Figure 5.13), all the cores appear to show that Fe concentrations increase with depth. Fe is not a particularly stable metal, and in soils it is known to be readily leached. This would result in increasing concentrations of Fe as the depth of the cores (Figure 5.13).

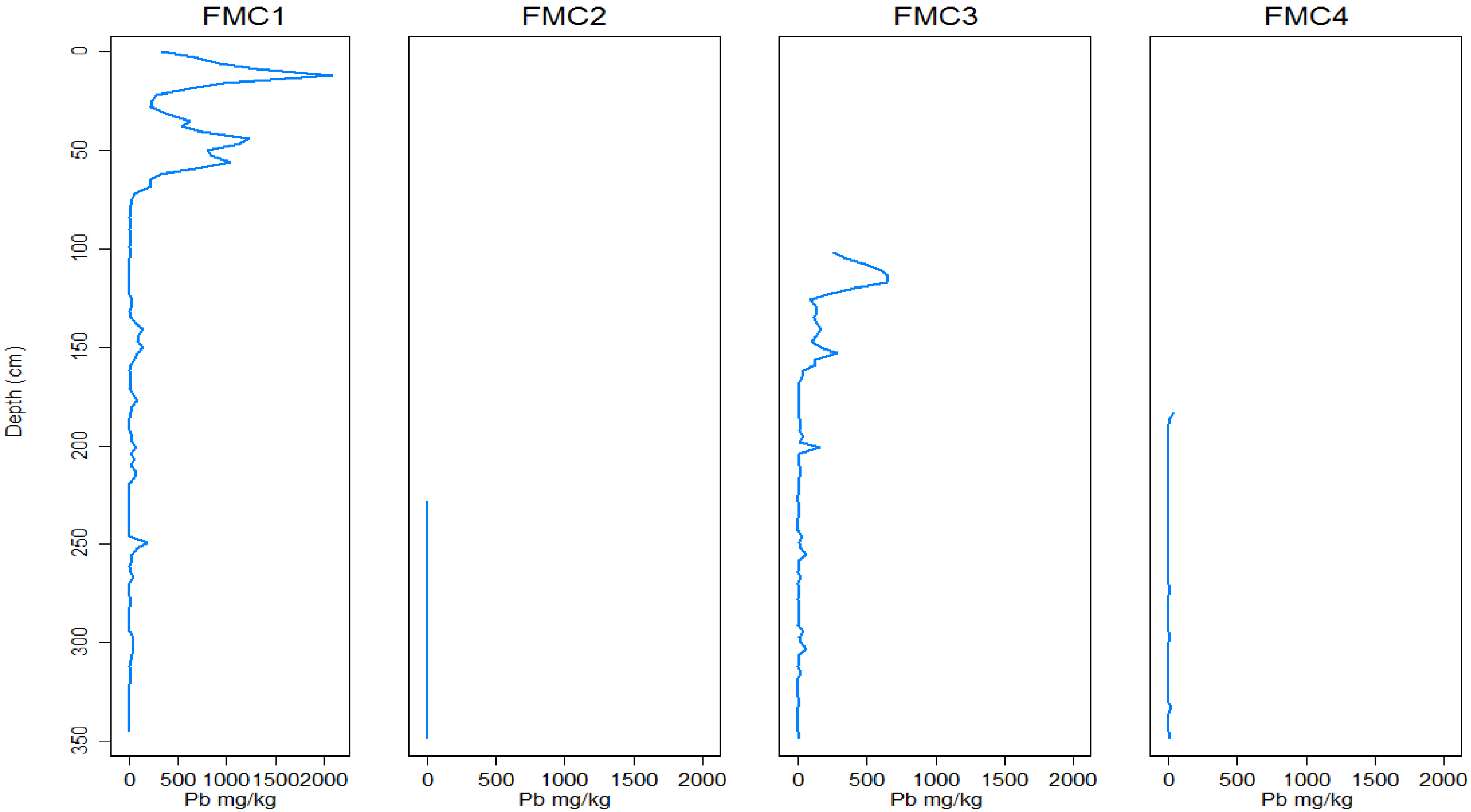


Figure 5.10- *Concentrations of Lead recorded for the four peat cores*

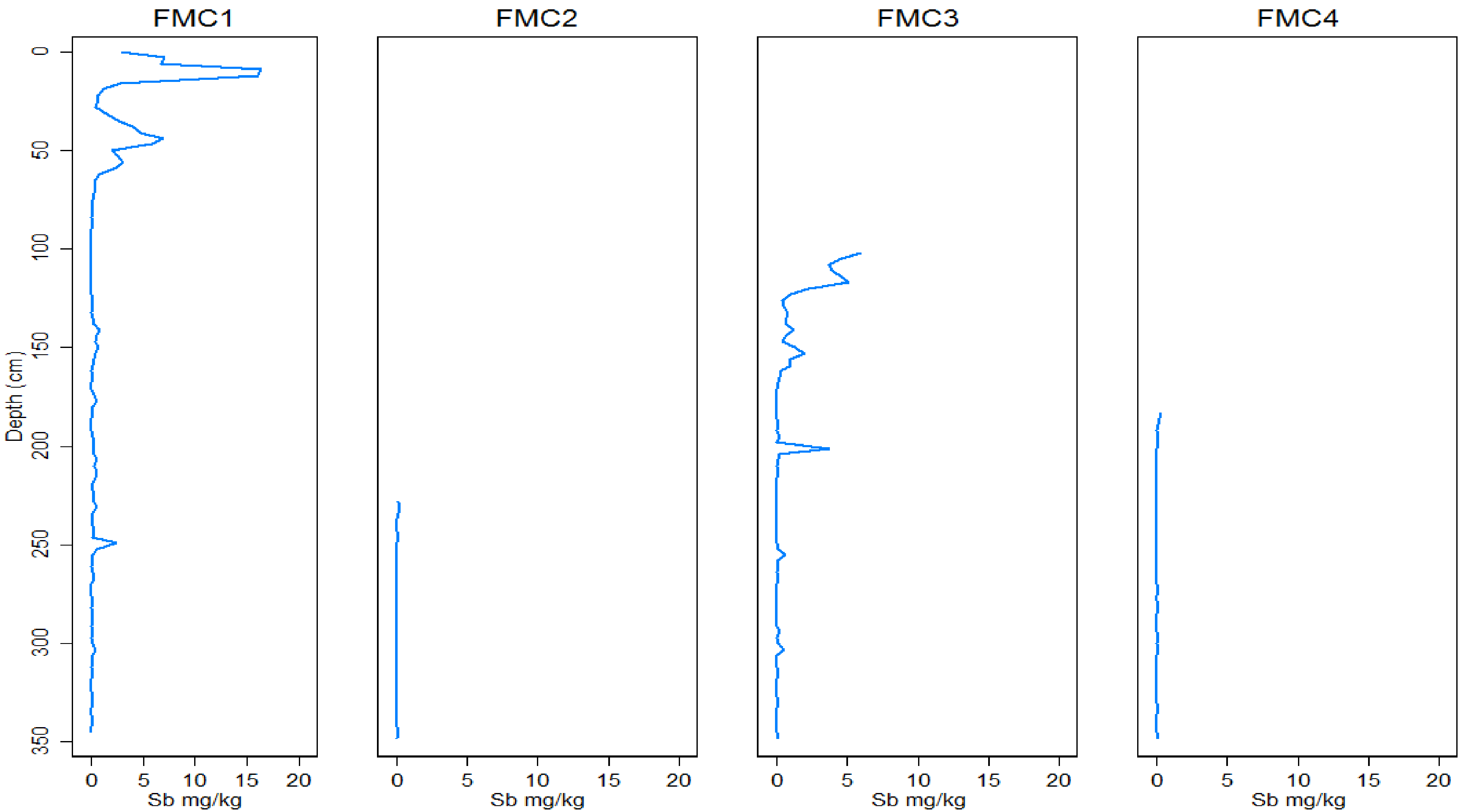


Figure 5.11- *Concentrations of Antimony*

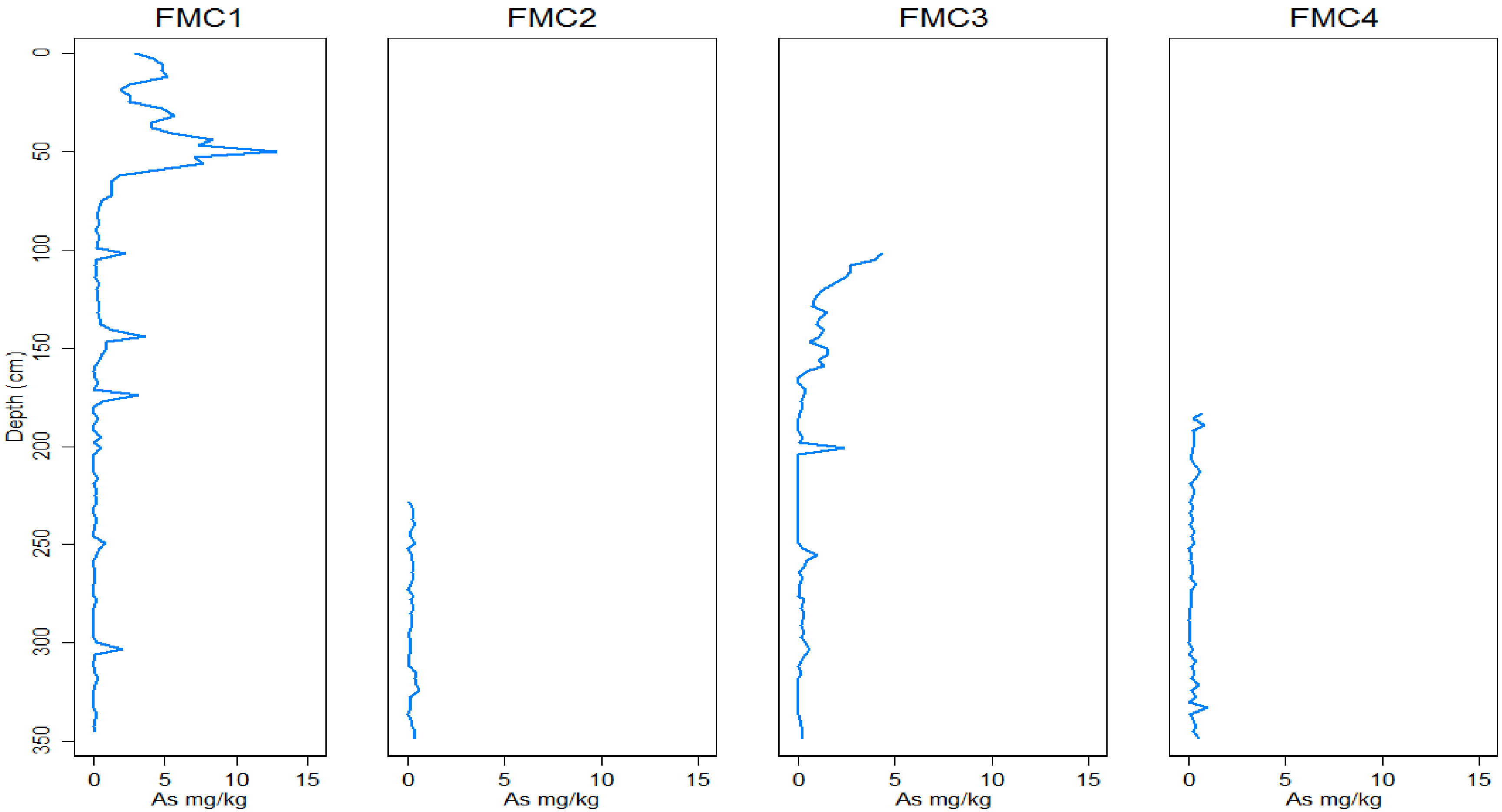


Figure 5.12 - *Concentrations of Arsenic*

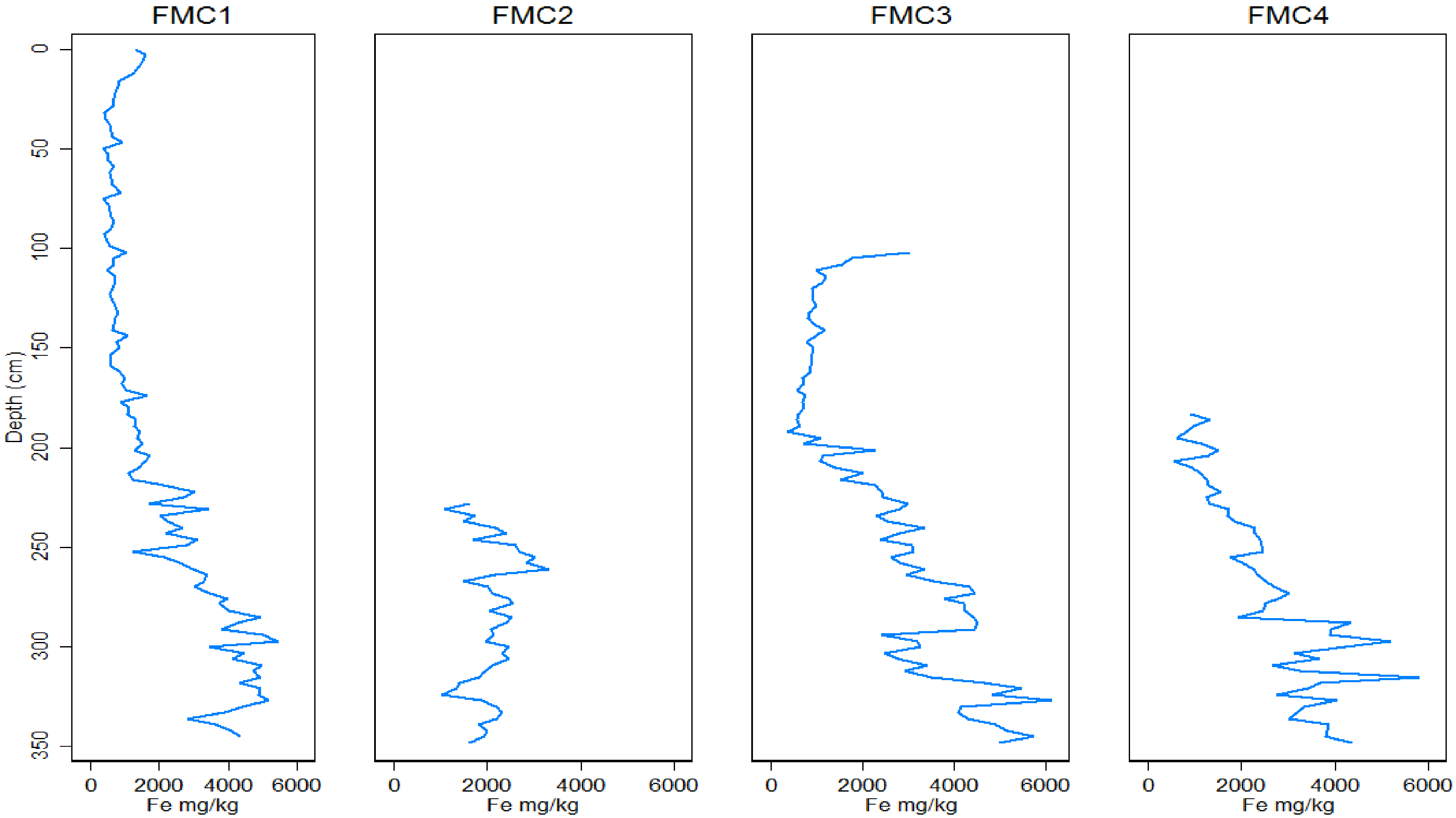


Figure 5.13- *Concentrations of Iron*

When 3.50 m is used as the baseline peat depth, the sampled peat cores appeared to show peaks and troughs of metal concentrations in similar locations. This is especially clear in the results for Pb, Sb and As. A clear example is As, which shows a peak in concentrations between 150 and 200 cm depth. Furthermore, Pb concentrations show a small peak between 250 and 300 cm depth. Although peaks can be identified at similar depths, many of the cores do not match exactly. There are several possible explanations as to why an exact match is not identifiable. The rates of past erosion may not have been uniform, as amounts of erosion can vary even on a local scale. Deposition rates may differ resulting in different sections of the cores being truncated at different times and thus exposed to differing levels of contaminants and peat accumulation.

From the metals plots, it can be identified that some differences exist between the vegetated and non-vegetated cores. FMC2 and FMC4 have lower concentrations of heavy metal contaminants throughout which strongly suggest that the upper part of the peat profile has been removed by erosion. This is supported by the elevated Fe concentrations which are typical of the base of a leached profile and the general peat stratigraphy which shows that the peat is more humified than upper layer peat (Figure 5.1). Pollen analysis and radiometric dating would be useful techniques for confirming this hypothesis.

5.3 Peat depth, peat volume and carbon storage

Estimates of peat depth are essential to gain information relating to peatland development, functioning and carbon storage (Parry *et al*., 2014). Fyfe *et al*. (2014) state that a range of estimates exist for carbon stored in UK peatlands, but uncertainties exist, particularly associated with peat depth and bulk density and very few studies consider complete profiles for peat. In this study, Ground Penetrating Radar (GPR) was used to collect 104 profiles of subsurface topography at a horizontal spacing of approximately 2 m across the full 7 ha area of the Flow Moss study site. This fulfilled Objective 1 of this study ‘to compile a complete peat inventory for the Flow Moss site’.

5.3.1 GPR depth survey

GPR data were post processed using REFLEXW software (Version 7.1.6, Sandmeir, Karlsruhe, Germany) using the methods discussed in section 4.3.3. Examples of GPR radargrams showing the raw data (a), the processed data (b) and the ‘picked’ data (c) are showing in Figure 5.14.

Once the data had been post processed, they were imported into ArcMap software (version 10.2, ESRI) and interpolated to create rasters showing the peat depth and the peat surface height using kriging. This is an interpolation method where the surrounding measured values are weighted to derive a predicted value for an unmeasured location (ESRI, 2014). The cut fill function was used to calculate the volume of peat stored at Flow Moss. The ArcMap raster calculator function was used to subtract the depth values from the surface height values to produce a depth map of the peat at Flow Moss. This was converted to a TIN (Figure 5.15). From this it was possible to determine the spatial variability of peat depth across the 7 ha site. This map identified that the deepest areas of peat are located at the southwest of the site, with shallower depths being recorded towards the middle of the site and beneath the bare peat flats. The map shows that the maximum recorded peat depth was 4.51 m and this was recoded at the southwest end of the site which is covered by vegetated peat.

Once the volume of peat at Flow Moss had been determined, it was possible to combine this information with the mean bulk density value produced during the peat core analysis. The mean value of the peat bulk density was 0.0776 t peat m^{-3} and the standard deviation for the bulk density dataset was 0.0319 t peat m^{-3}. These values were used to calculate the amount of peat stored at Flow Moss. The estimated peat mass stored at Flow Moss is 7973 tonnes. To take into consideration the possible error associated with the bulk density calculations; this value was recalculated with the standard deviation of 0.0319 t peat m^{-3} both added and subtracted. This provided a lower estimate of 4697 tonnes and an upper estimate of 11251 tonnes for the amount of peat stored at Flow Moss.

This figure can be combined with the carbon content data to establish the current size of the carbon store at Flow Moss. The mean value of C for the 298 peat subsamples was 50.21% (+/- 3.64%) This value was used to provide an estimate of 4004 (+/- 16.01) tonnes of carbon stored in the Flow Moss carbon reservoir.

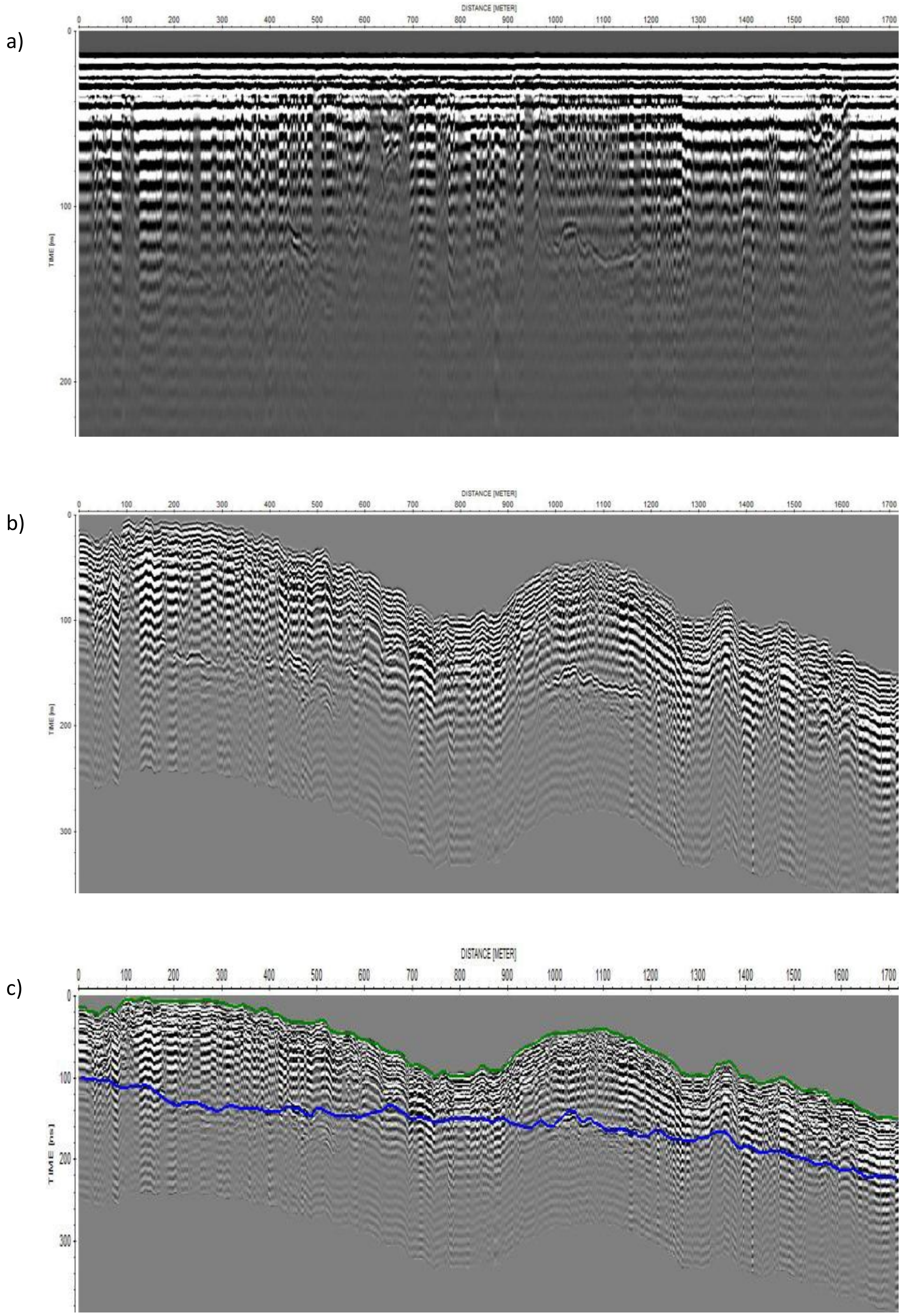


Figure 5.14 – Radargrams from REFLEXW showing the (a) raw data, (b) the processed data and (c) picked data

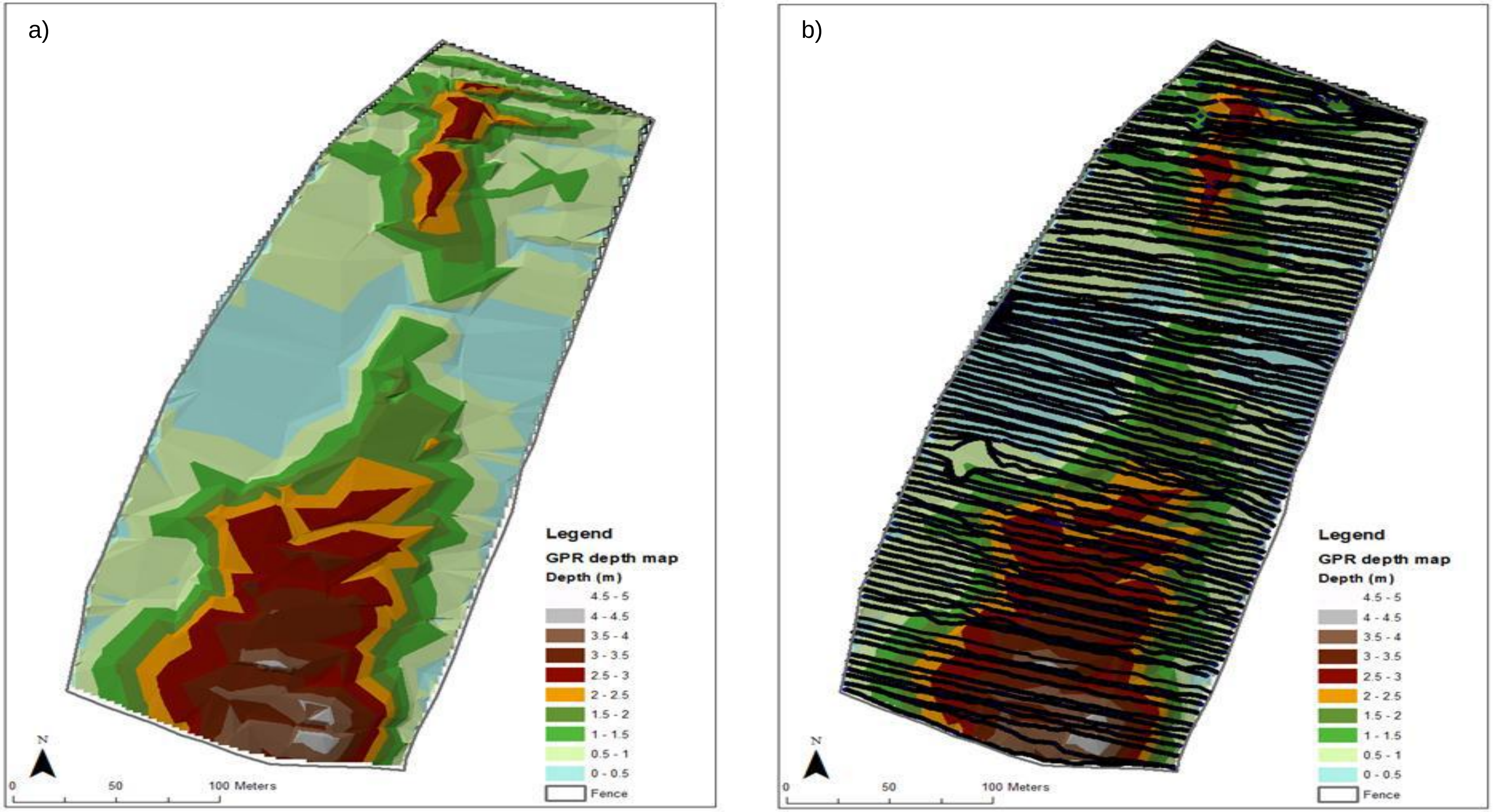


Figure 5.15 – *(a) Map of peat depth created from subsurface topography data collected at approximately a two meter resolution (b) the location of transects for GPR data collection*

5.3.2 Synthetic transect spacing experiment

Previous studies using GPR to map peat depths have collected depth transects at a range of resolutions (e.g. Warner *et al*., 1990), however no examples were found within the literature of previous GPR surveys being implemented at a two meter resolution. To assess the effect that transect spacing had on the calculated peat depths and volumes, a synthetic transect spacing experiment was implemented. To do this, transects of different spacing and layout (Table 5.4) were generated in ArcMap. A raster was created by interpolating data points situated along the newly created transects. Again, the cut fill function was used to calculate the volume of peat and a depth map created to establish the maximum depth and the spatial variability in peat depth. The results of this experiment are shown in Table 5.5. The results demonstrate that the spacing of transects in GPR surveys will influence not only the calculated volume of peat but also the maximum recoded peat depth. The data from Table 5.5 are shown graphically in Figures 5.16 and 5.17. The Figures show that transect spacing impacts volume estimates and maximum depth. Figure 5.16 shows that using the GPR data to replicate a probe survey on a 100 x 100 m grid has a substantial impact on the maximum depth estimate, with this value being 2.56 m. The maximum depth recorded for the GPR survey at a 2 m resolution is 4.52 m, meaning that that data replicating a probing survey underestimate the maximum depth by 1.96 m or 44%. This substantial difference will be caused by only collecting depth measurements in point locations, and suggests that results of peat probe surveys can be providing large under estimates of depth estimations. Figure 5.17 displays the amounts of peat estimated to be stored at Flow Moss calculated using data at different resolutions. The data show that the lower the resolution of the data, the lower the estimate of the amount of peat. Again, it is the data that replicate a 100 x 100 m probe survey that provide the lowest estimated peat amounts, again suggesting that this is not the most appropriate method for estimating the size of peat stores.

Examples of the simulated transect spacing and the depth maps interpolated from these data are illustrated in Figures 5.18 to Figure 5.20. In each of these plots the first panel (Figure **(a)**) shows the depth map, while the second panel (Figure **(b)**) illustrates the transect layout from which the depth map was interpolated. Figure 5.20 **(b)** displays a depth map interpolated from single points spaced on a 100m grid. This point spacing was selected to replicate a peat depth probe survey with data collected on a 100m grid, previous examples of depth probe surveys have used point spacing of various distances ranging from 20-2000 meters (Jaenicke *et al*., 2008, Buffam *et al*., 2010). Figure 5.20 clearly shows a peat depth map of a far coarser resolution than the finely surveyed data (Figure 5.15). Due to the nature of a depth probe survey, it would be too time consuming and labour intensive to employ a depth probe survey at the same resolution as the GPR study conducted here. Table 5.5 and Figures 6.16 and 5.17 indicate that the spacing of transects impacts both the maximum depth estimated from the GPR data and the overall volume estimate. The Figures show that data interpolated from gridded transects produce volume and depth estimates closer to those recorded during the initial GPR survey. This is demonstrated in Figures 5.18 and 5.19 which indicate that GPR data interpolated from transects spaced at a 10 m resolution result in a high resolution depth map, but the data interpolated from the 10 x 10 m grid produces higher maximum depth and volume estimates than transects running parallel across the site. As demonstrated in Table 5.5, conducting a peat depth survey using a 100 x 100 m point grid (as is often used in peat depth surveys) underestimates both the volume of peat and the maximum depth of the peat at a field site, in this case by 26% and 44% respectively. A GPR survey allows the collection of a large amount of data, providing a method superior to depth probe surveys when attempting to quantify the depth and volume of peat stored at a location.

Table 5.5 – *Peat depths, volumes and carbon storage calculated from the synthetic transect sampling experiment*

Sampling spacing and format	**Max depth (m)**	**Estimated Volume of peat (m^3)**	**Amount of peat (tonnes) (0.0776 t m^{-3} density)**	**C stored (tonnes)**	**Carbon estimate difference from GPR survey (%)**	**Amount of peat (tonnes) (0.0776 t m^{-3} density +31.90)**	**Amount of peat (tonnes) (0.0776 t m^{-3} density - 31.90**	**C stored (tonnes) 50.21% +3.64%**	**C stored (tonnes) 50.21% - 3.64%**	**% discrepancy in volume**
Approx. 2m	4.52	102729	7973.89	4003	0	11250	4697	4294	3714	0
5m	4.50	101809	7902.46	3968	0.87	11150	4655	4255	3680	0.9
10m	4.35	100599	7808.53	3921	2.05	11018	4599	4205	3637	2.1
20 m	4.20	98764	7666.10	3849	3.85	10817	4516	4128	3570	3.86
50 m	4.52	93926	7290.61	3661	8.55	10287	4294	3926	3395	8.57
100m	4.2	77054	5980.98	3003	27.48	8439	3523	3221	2785	25
10 X 10m grid	4.47	102060	7921.90	3978	1.32	11178	4666	4266	3689	0.66
20 x 20m grid	4.47	101216	7856.45	3945	1.45	11085	4628	4231	3659	1.47
50 x 50m grid	4.21	98297	7629.82	3831	4.30	10765	4494	4109	3553	4.32
100 x 100m grid	4.21	78895	6123.89	3075	23.19	8641	3607	3298	2852	23.20
100 x 100 m point grid	2.56	76284	5921.17	2973	25.73	8355	3488	3188	2758	26.65

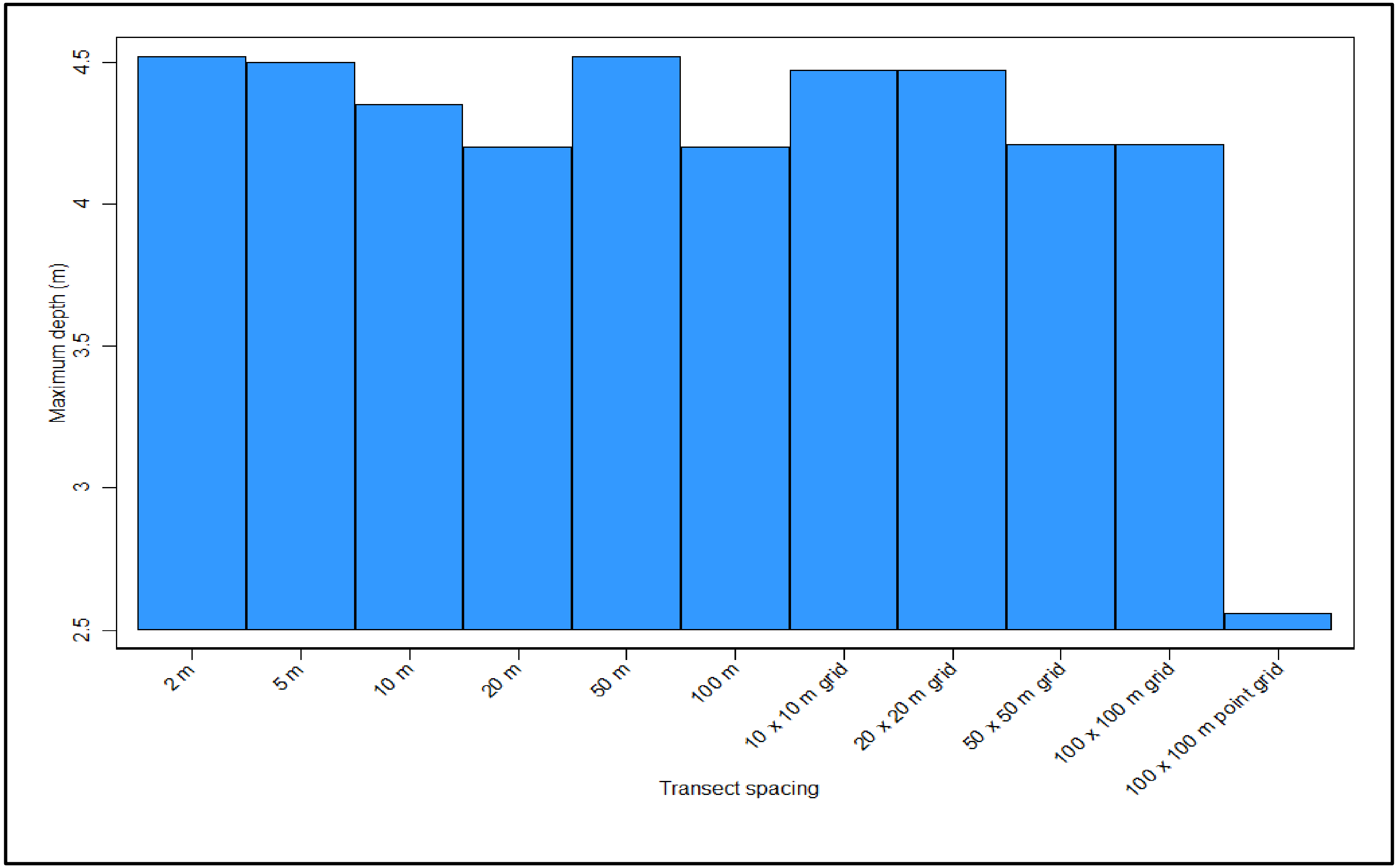


***Figure 5.16 –** a bar chart showing the impact of survey strategy on maximum depth recorded*

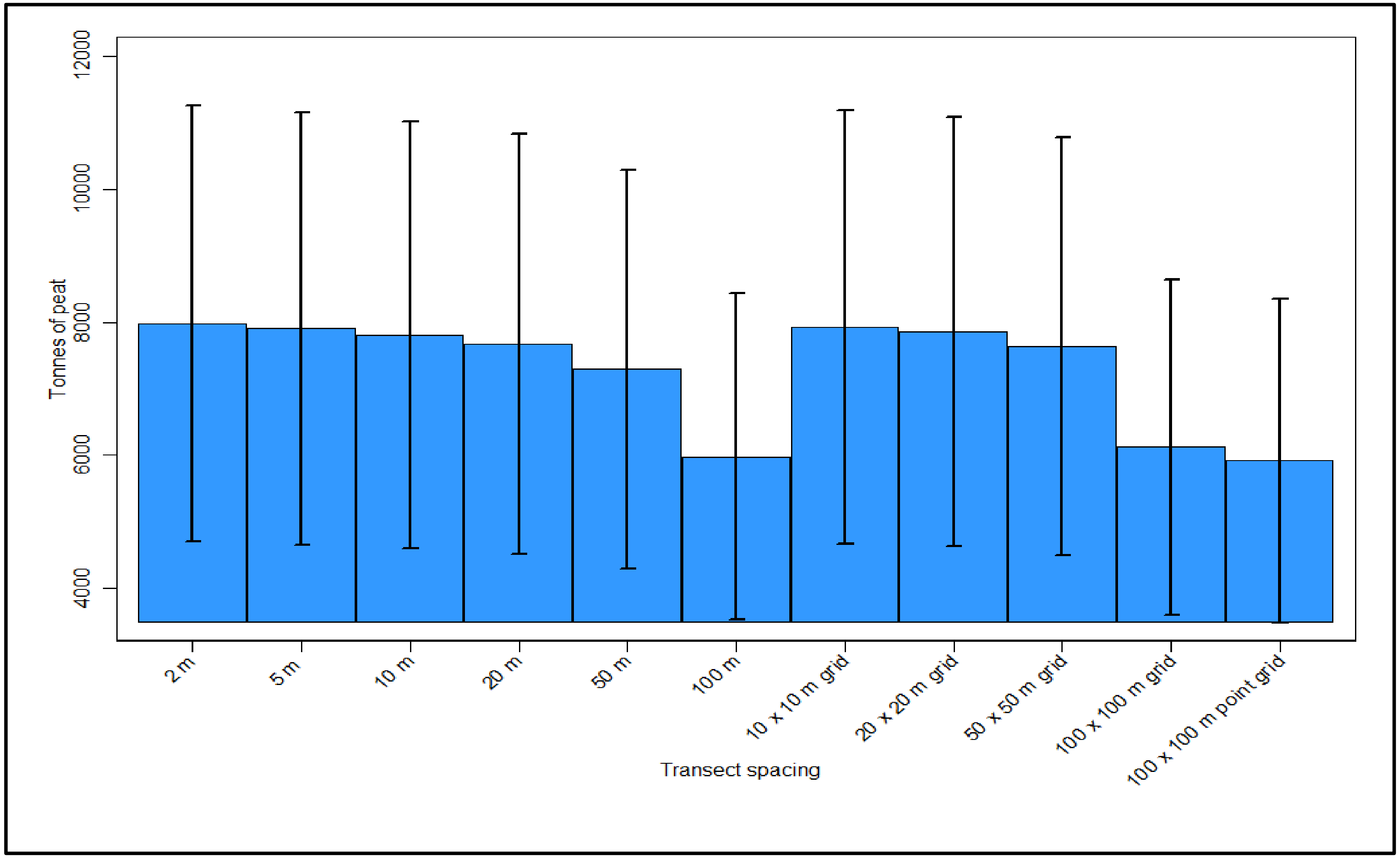


***Figure 5.17** – Bar chart showing the effect of transect spacing and grid spacing on estimates of amounts of peat. Chart shows miniumum and maxiumum amount of peat calculated.*

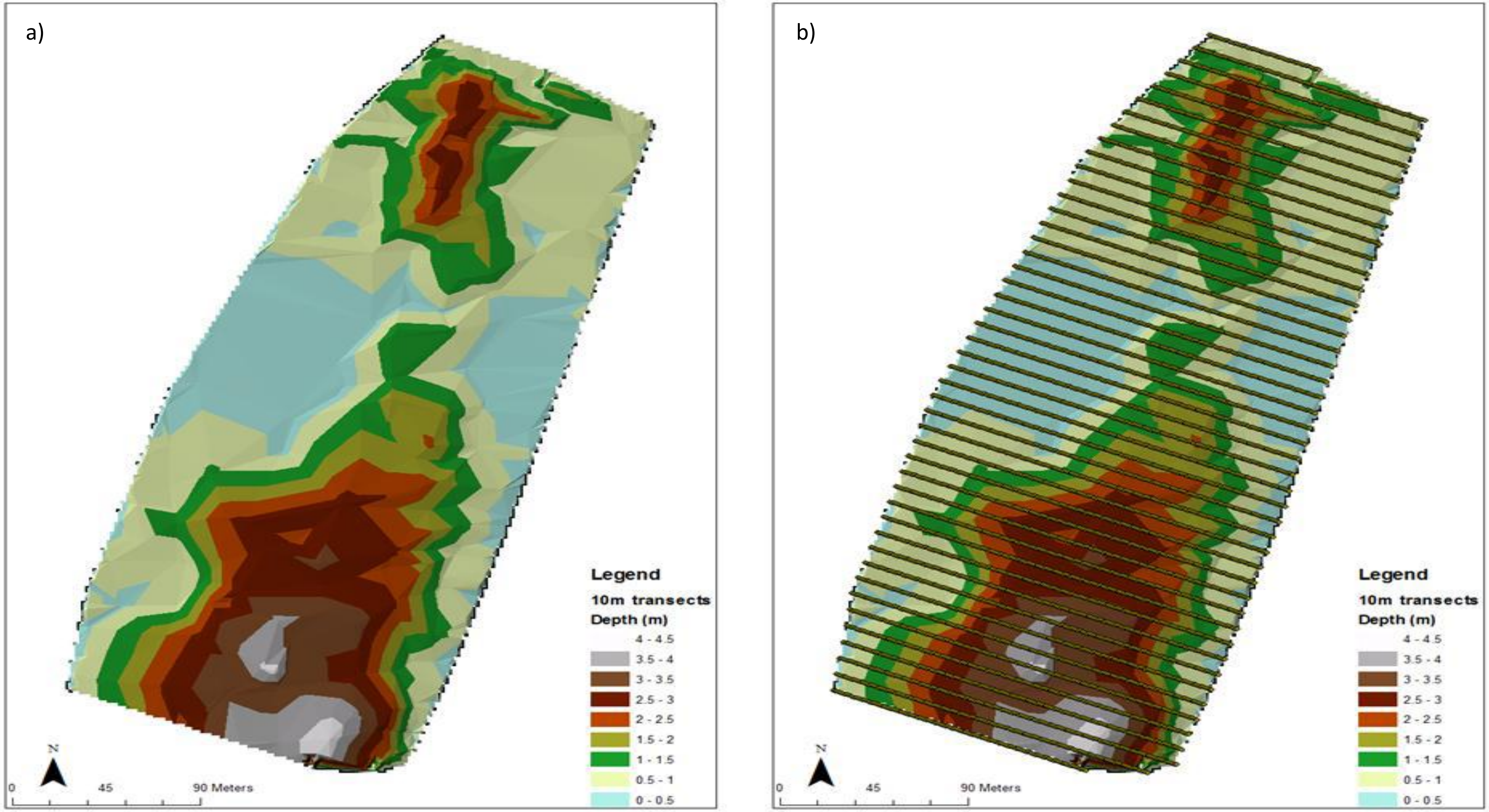


***Figure 5.18** – Depth map interpolated from GPR data using 10m transects*

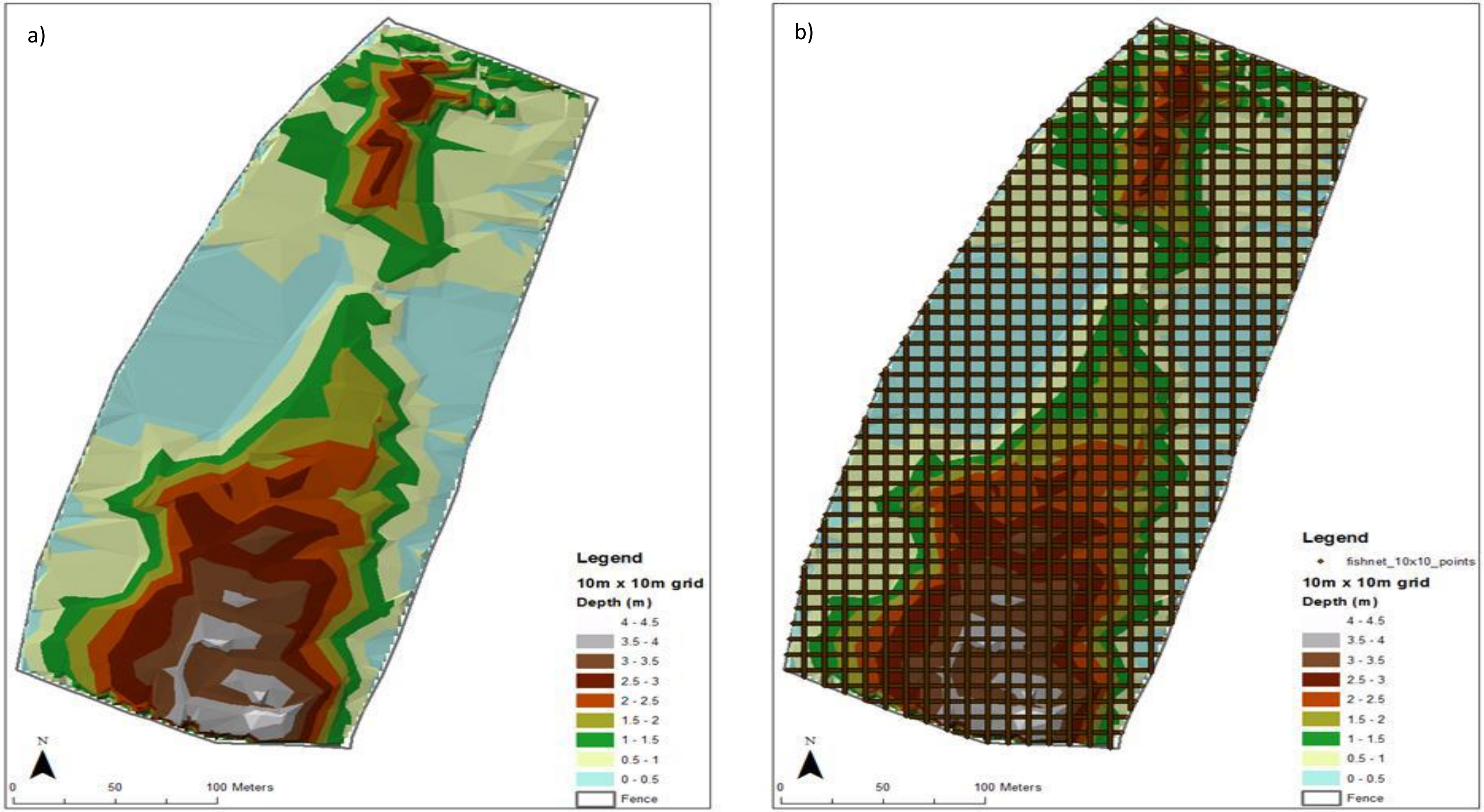


Figure 5.19 – *Depth map interpolated from GPR using 10x10 m gridded layout*

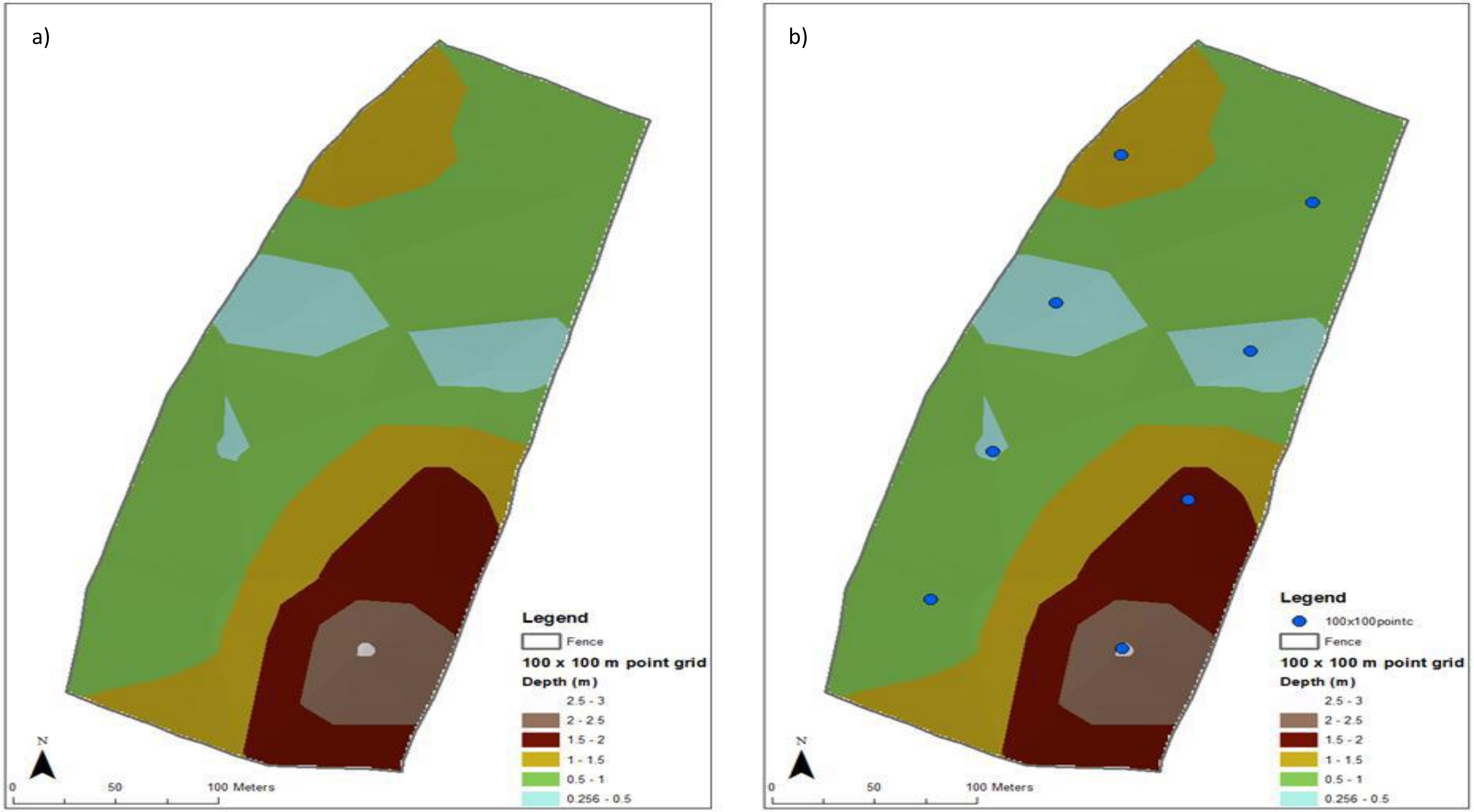


Figure 5.20 – *Depth map interpolated from GPR data on 100m point transects, aiming to replicate a depth survey using a depth probe.*

5.3.3 GPR depth validation

To validate the results obtained using GPR, a depth probe survey was undertaken. Peat depth was measured using a bespoke peat depth probe (a rigid metal depth probe) for ten points along four transects (Figure 5.21). Points were measured in an area of peat where a large range of peat depths were expected.

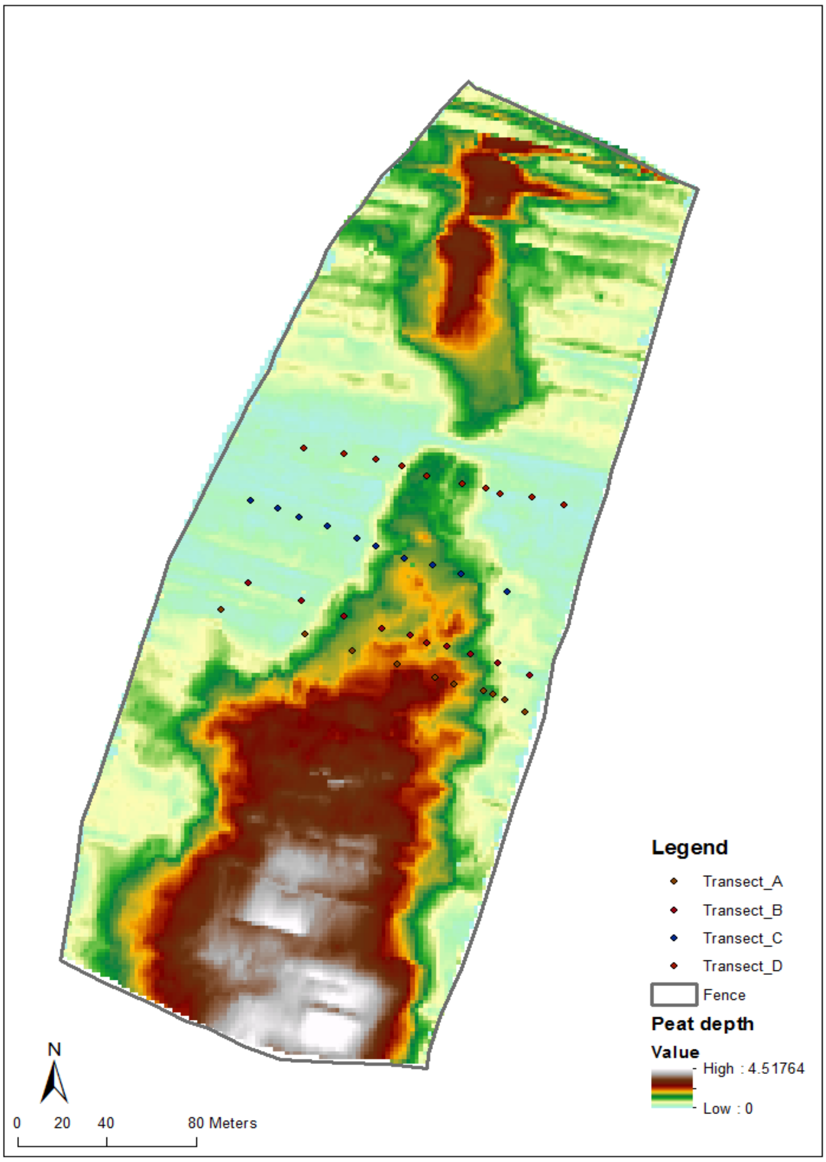


***Figure 5.21 –** Location of the depth probe survey for comparison of GPR depth data.*

The results from the two methods of data collection were compared (Figure 5.22). A linear trend line has been added and an r^2 value of 0.9176 calculated. This indicates that a strong positive correlation exists between the two data sets.

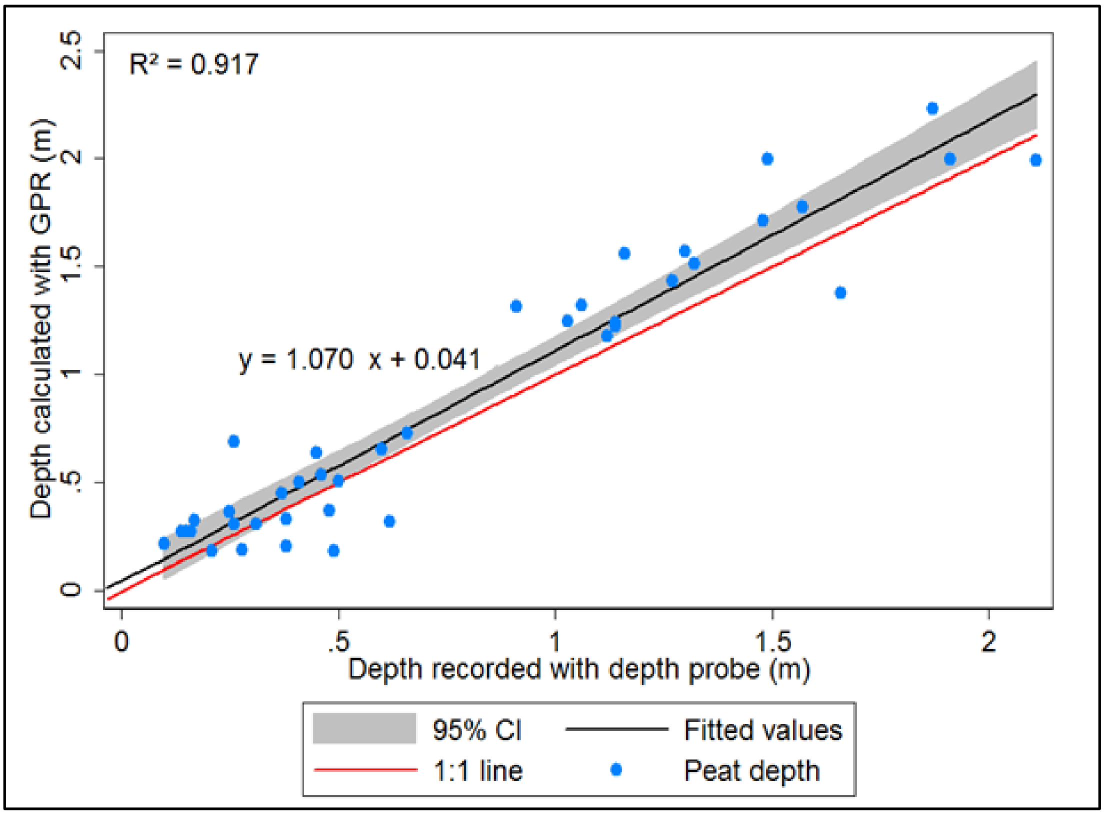


Figure 5.22 – *Depth recorded using GPR plotted against depths recorded using a peat depth probe*

A Wilcoxon test for matched pairs was used to test the null hypothesis 'there is no significant difference between depth measurements recorded using a depth probe and those obtained using GPR'. This test compares the medians of two matched samples. This produced a p-value of 0.00262, meaning the null hypothesis can be accepted at a 95% confidence level (depth range 0 to 2.5 m).

Parry *et al*., (2014) discuss errors which need to be taken into account when using depth probes for GPR calibration. They concluded that depth estimations calibrated using common midpoint surveys were on average 35% greater than calibration using depth probes. Discrepancies between the two were predominantly the result of depth probes becoming obstructed by objects within the peat, and thus not reaching the base of the peat surface. This issue can largely be overcome by multiple probing at a single location, allowing obstructions to be distinguished from laterally extensive stratigraphic horizons such as the base of the peat.

5.3.4 GPR error and uncertainty

During this survey, 104 depth transects were collected. The aim was to follow straight lines across the site, spaced approximately two meters apart. As can be seen from Figure 5.15, this did not always happen. There are several reasons for this. Firstly, during data collection, obstacles were encountered in the field such as deep peat pools, which obstructed some transects. Furthermore, at times it was possible to lose sight of the transect end (marked with a ranging pole) and therefore easy to divert from the planned transect. On occasion, this resulted in transects crossing or being closer together. Nonetheless, due to the 2 m transect spacing, the fact that transects were not exact should have little impact on the collected data because given the density of the lines; almost the entire site was covered at high resolution. This is not a major issue as the errors are easily quantified, as data collected along the GPR traces were individually tagged and spatially referenced using dGPS measurements recorded during GPR data collection. The dGPS error is approximately < +/- 0.02 m.

5.3.5 Carbon storage at Flow Moss

One of the motivating factors for peatland restoration is the loss of carbon and improving carbon sequestration in an attempt to offset emissions. Thus it is essential to gain as much information about the amount of C stored within the peat. To estimate the size of the carbon reservoir at Flow Moss (Research Objective 1), the peat volume data were combined with the average TOC value (50.21%, +/-3.64%). This provided an estimate of 4004 tonnes of C stored at Flow Moss (Table 5.5). This value was recalculated using the lower estimate of the amount of peat stored and the highest estimate to provide a minimum estimate of 2358 tonnes of C and a maximum of 5649 tonnes of C stored. The differing estimates of carbon storage calculated using data produced during the synthetic transect spacing experiment are shown in Table 5.5. The results displayed in Table 5.5 demonstrate that when estimating the amount of carbon stored within a peat bog, sampling strategy will impact the results and even if this is optimised using the most sophisticated and detailed survey methods significant uncertainty still exists in the 'best' estimate. Reporting this uncertainty is essential in any carbon storage estimate.

5.3.6 Chapter summary

This chapter has discussed results collected using methods implemented to gain information about the subsurface properties of the peat at Flow Moss. The data obtained from the peat core analysis suggest there is a large amount of variation in carbon content of the peat both spatially and with depth. Results indicate that carbon content results obtained using an oxidative combustion method to produce TOC values are more accurate than carbon content estimates produced using the LOI method. In the peat profiles (FMC1-4), it is possible to see correlations in the carbon content, organic matter content and concentrations of heavy metals at similar depths. These results suggest that metals concentrations may be useful as stratigraphic markers of erosion. Concentrations of metals are also indicative of historic land use and may provide an approximate date when the peat was truncated (e.g. spikes in lead could be indicative of time periods when lead mining occurred in the areas surrounding Flow Moss). However, as no peat dating could be included in this study it is impossible to know for sure the dates of peat accumulation. To correlate the cores with more certainty, radiocarbon dates together with pollen analysis would provide a more robust chronology of erosion and deposition at Flow Moss.

The high resolution GPR survey has provided a carefully constrained estimate of the volume of the peat stored at Flow Moss. The estimated volume of approximately 7974 (+/- 31.9) tonnes, equates to 4004 (+/- 16.01) tonnes, or 572 (+/- 2.29) tonnes per ha of stored carbon. The synthetic sampling experiment implemented using GPR data demonstrates that sampling design is an important factor when estimating the volume of the carbon store. The results suggest that the GPR methods of data collection are far superior to depth probing when collecting data relating to peat depth/ volume in complex upland terrain.

6. Surface results

This chapter presents results relating to the Flow Moss surface monitoring programme. Details of erosion and deposition patterns are presented and summarised.

6.1 Quantification of Erosion by Aeolian and Hydraulic Processes

The following section quantifies erosion rates at Flow Moss caused by aeolian and hydraulic processes. These data will be used in Chapter 8 to produce an annual sediment budget for Flow Moss.

6.1.1 Aeolian processes

In the UK, wind is a fundamental characteristic of upland environments and a significant factor in peat erosion (Warburton, 2003). The present study quantifies the amount of peat eroded from the bare flats by aeolian processes. Figure 6.1 shows the masses of peat collected from the wind flux samplers for each month between 8th March 2013 and 4th March 2014. Throughout this study, data collected up until the 8th of each month were considered part of the month before (i.e. March 2014 data were collected on 4th, were considered representative of conditions in February). Sediment yields collected during the months of December and January were significantly higher than those collected during the rest of the year which demonstrates a highly episodic delivery of eroded peat by wind erosion which appears to be greatest in the winter months.

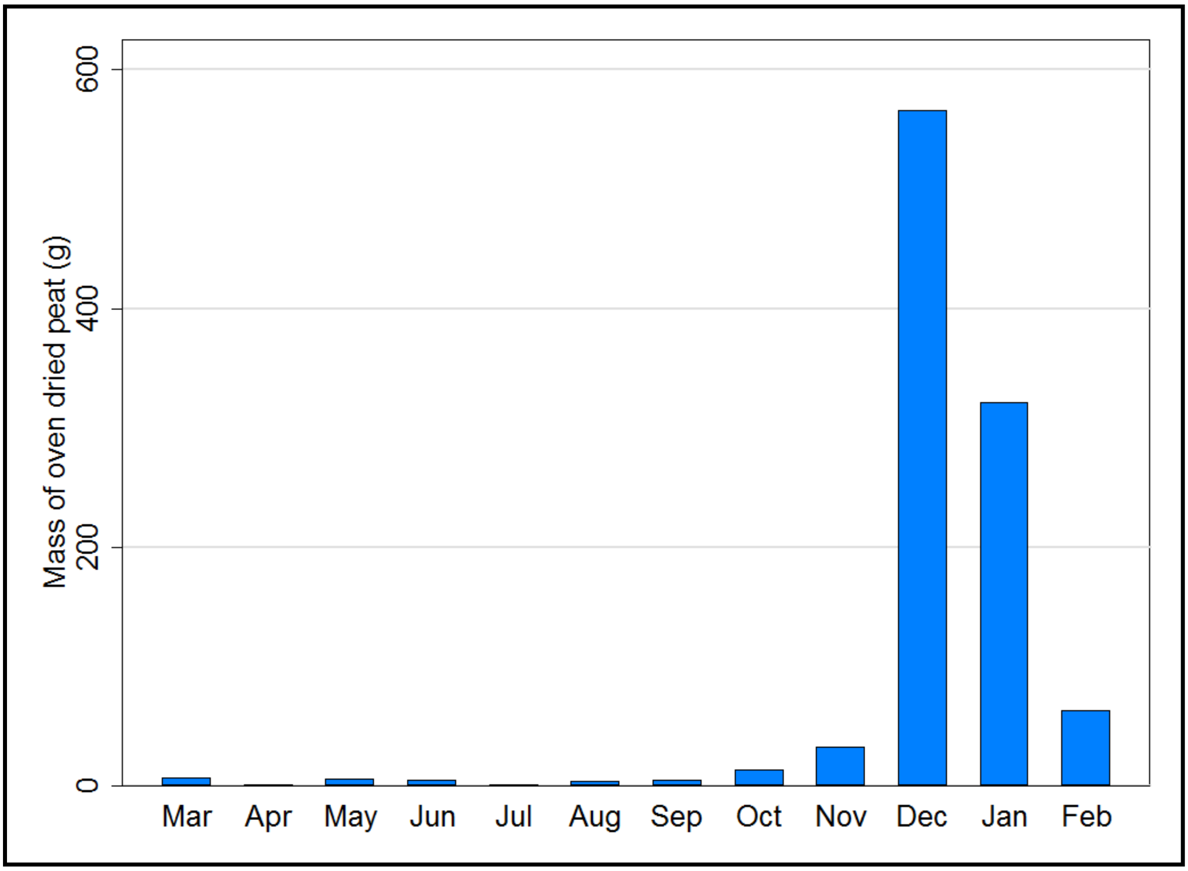


***Figure 6.1**– Amount of peat collected from wind flux samplers each month*

The total yield of sediment collected from the wind flux samplers was 1024g with 86 % of this collected in January and December. The yield of sediment collected in November and December increases dramatically, but then begins to decrease in January and February. This is likely due to antecedent weather conditions.

6.1.2 Fluvial processes

Sack traps were used to monitor the fluvial transport of peat from the main drainage lines exiting the restoration area at Flow Moss. These traps were positioned on the fence line at the most north-western edge of the site to capture sediment lost from the ephemeral drainage routes. Sediment yield data plotted by month (Figure 6.2) show the highest yields (12 g) from the sack traps were recorded in May 2013. The yields recorded during other months are extremely low and rarely exceed 5 g. This fundamentally shows sediment loss via the channel system at Flow Moss is negligible.

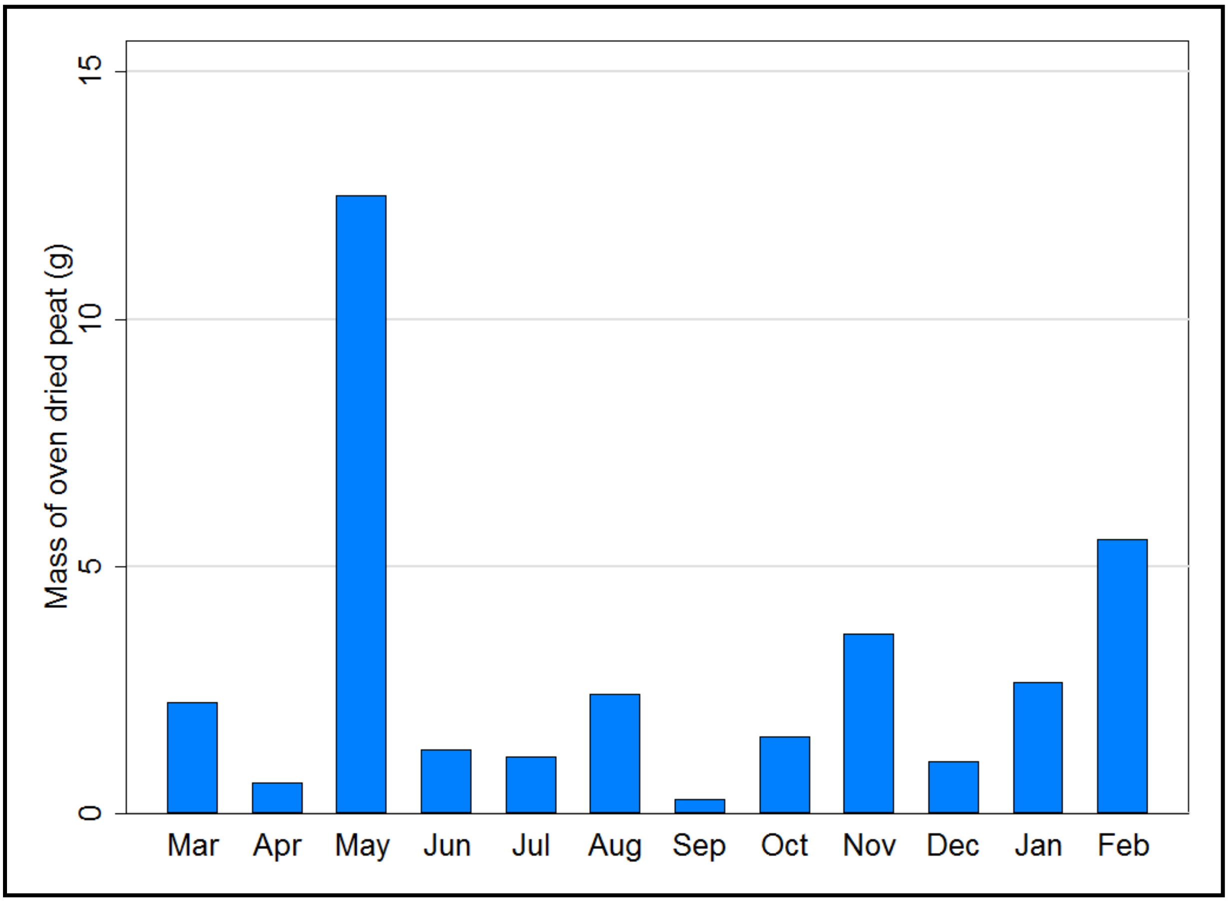


Figure 6.2 *– Bar chart showing the amounts of peat collected from the sediment traps each month during the study. Most sediment was collected from the traps in May 2013.*

Figure 6.3 plots the fluvial sediment yield (g d^{-1}) between March 2013 and February 2014. This shows that during this study, there was very little change in sediment loss from the channel system, averaging 0.096 g d^{-1}. The trend line fitted in Figure 6.3 has a slope of $Y=2x10^{-11}$ e $^{0.00005\ x}$ which is essentially flat.

The total amount of peat collected in the traps over the year was only 36 g. Section 4.2.4 demonstrated that these traps have a trapping efficiency of 91.4% and this number has been used to include error bars on the graph. The value for total yield of sediment collected during the monitoring period was re-calculated including the error factor, with the minimum amount of peat lost during the year being 33 g and the maximum being calculated as 39 g. Such small values effectively show that during the monitoring the export of eroded peat from Flow Moss by the drainage system is negligible.

Comparing the total yields of sediment collected from the wind flux samplers and fluvial sack traps highlights that the yield of peat collected from the sack traps is far lower, 27 times less, than the yield collected from the wind flux samplers. This would indicate that although peat is being mobilised by wind action at the site, much of this is not being transported from the site by fluvial action.

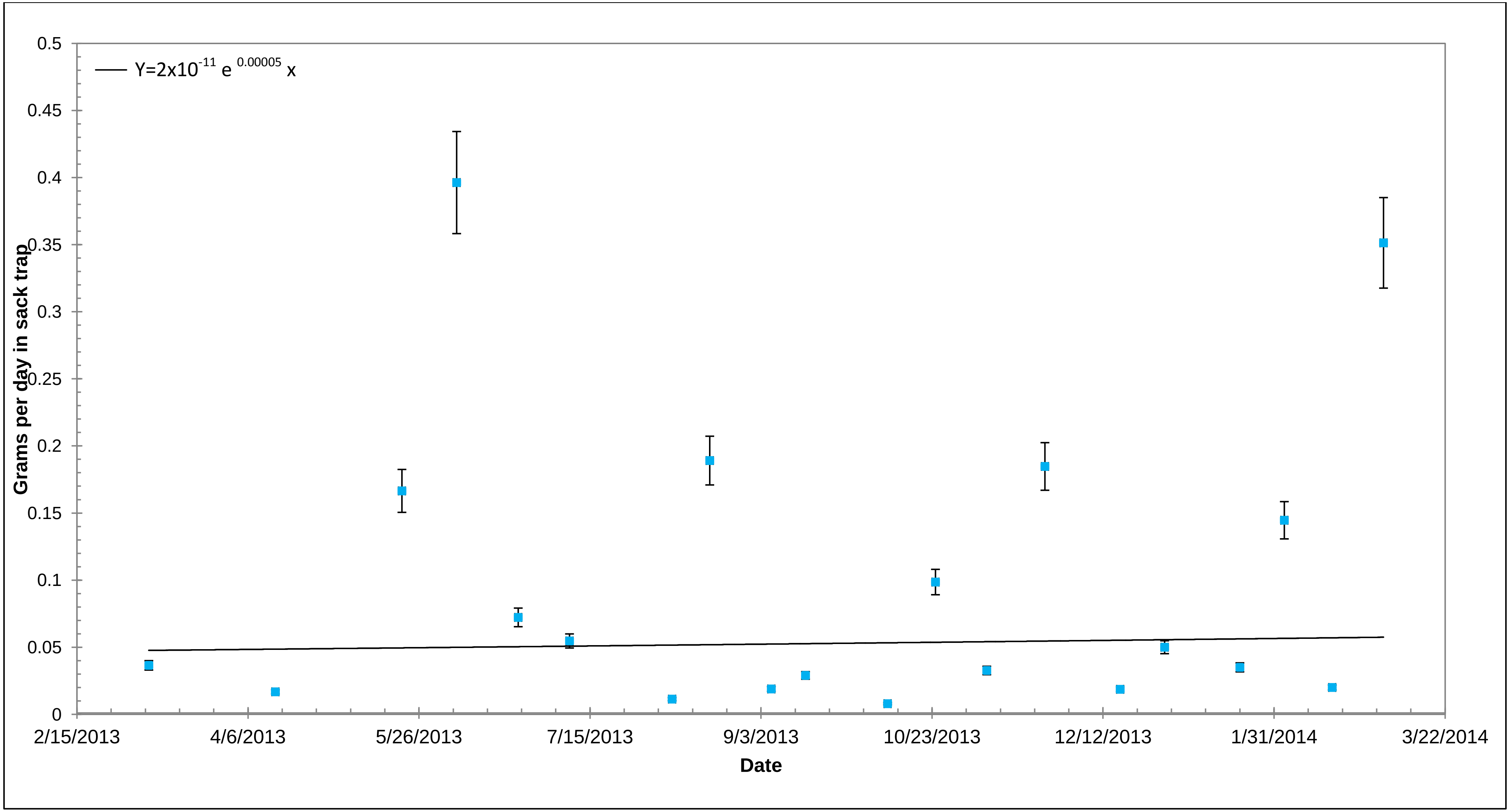


Figure 6.3- *Average amount (grams per day) of peat collected in sack traps between 8th March 2013 and 4th March 2014*

6.2 Quantification of changes in surface height and peat hagg slopes

Variability of the peat surface height in the bare peat area was measured using pole transects and Terrestrial Laser Scanning (TLS). Changes in the elevation of peat hagg face heights were monitored using erosion pins. The following sections present results collected using these methods.

6.2.1 Erosion pins

6.2.1.1 Changes in surface height of peat haggs

Figure 6.4 shows the mean change in pin exposure recorded for each of the 10 sites over the monitoring period. The mean value of change in surface elevation measured using erosion pins was calculated for each site between 8th March 2013 and 4th March 2014. All values were within ±4 mm (Figure 6.4). The mean value of change for the data from all sites combined was found to be -2 mm. These values suggest a small net loss of sediment from the peat haggs, but overall, the peat haggs are fairly stable. However, this is a mean annual value and it would be expected that a seasonal signal is present in the dataset, with more erosion occurring in the winter months (Evans and Warburton, 2007). Figure 6.5 examines the data in more detail and shows the overall change for each site. The basic data are summarised in Table 6.1. Figure 6.5 indicates that the four sites where the greatest variation in pin exposure was recorded are sites Five, Seven, Nine and Ten.

Due to slight differences in the period between measurements, the average change per day was calculated to standardise results and make them directly comparable (Figure 6.6).

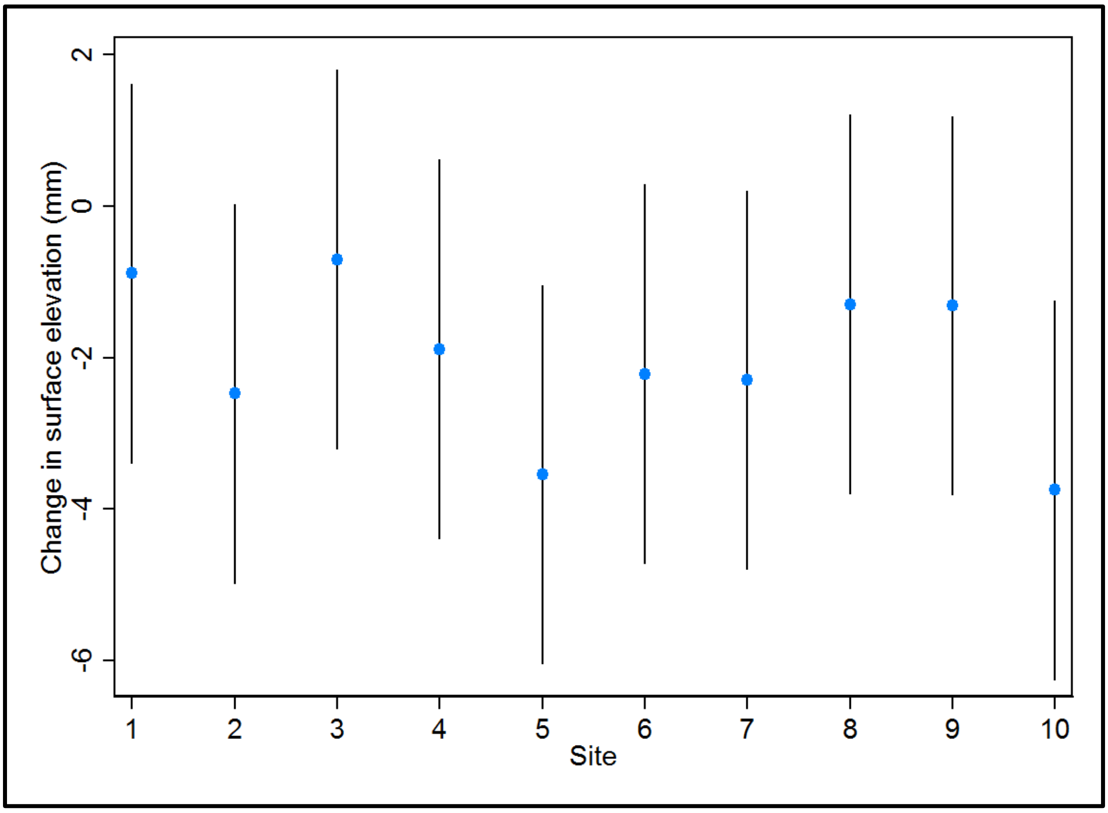


Figure 6.4 - *The mean change in erosion pin exposure for each site recorded between March 2013 and March 2014. The pin measurement error value (2.5 mm) has been used to add error bars.*

Table 6.1 – *Descriptive statistics for the erosion pin data collected during monitoring*

Site number	Mean change (mm)	Median (mm)	Standard deviation (mm)
1	0.8875	1	8.64
2	2.475	1	11.32
3	0.7	0	16.7
4	1.88	1	12.35
5	3.54	0	21.06
6	2.22	0	8.86
7	2.26	2	14.81
8	1.29	1	10.74
9	1.31	0	11.60
10	3.75	2	14.97
All sites	2.0	1	3.383

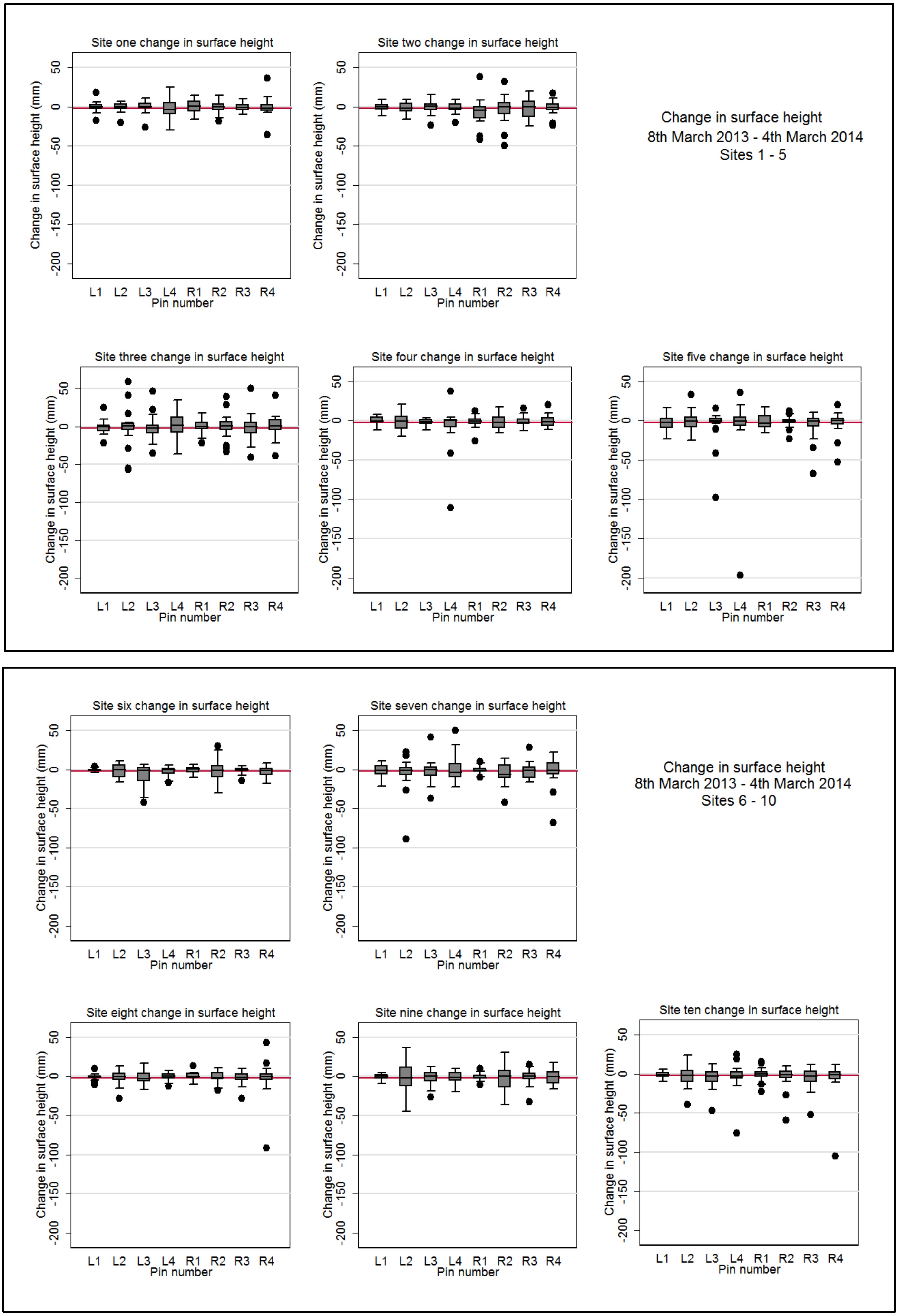


Figure 6.5 *Changes in erosion pin exposure sites 1- 10. The red line indicates the mean value of 2.0 mm.*

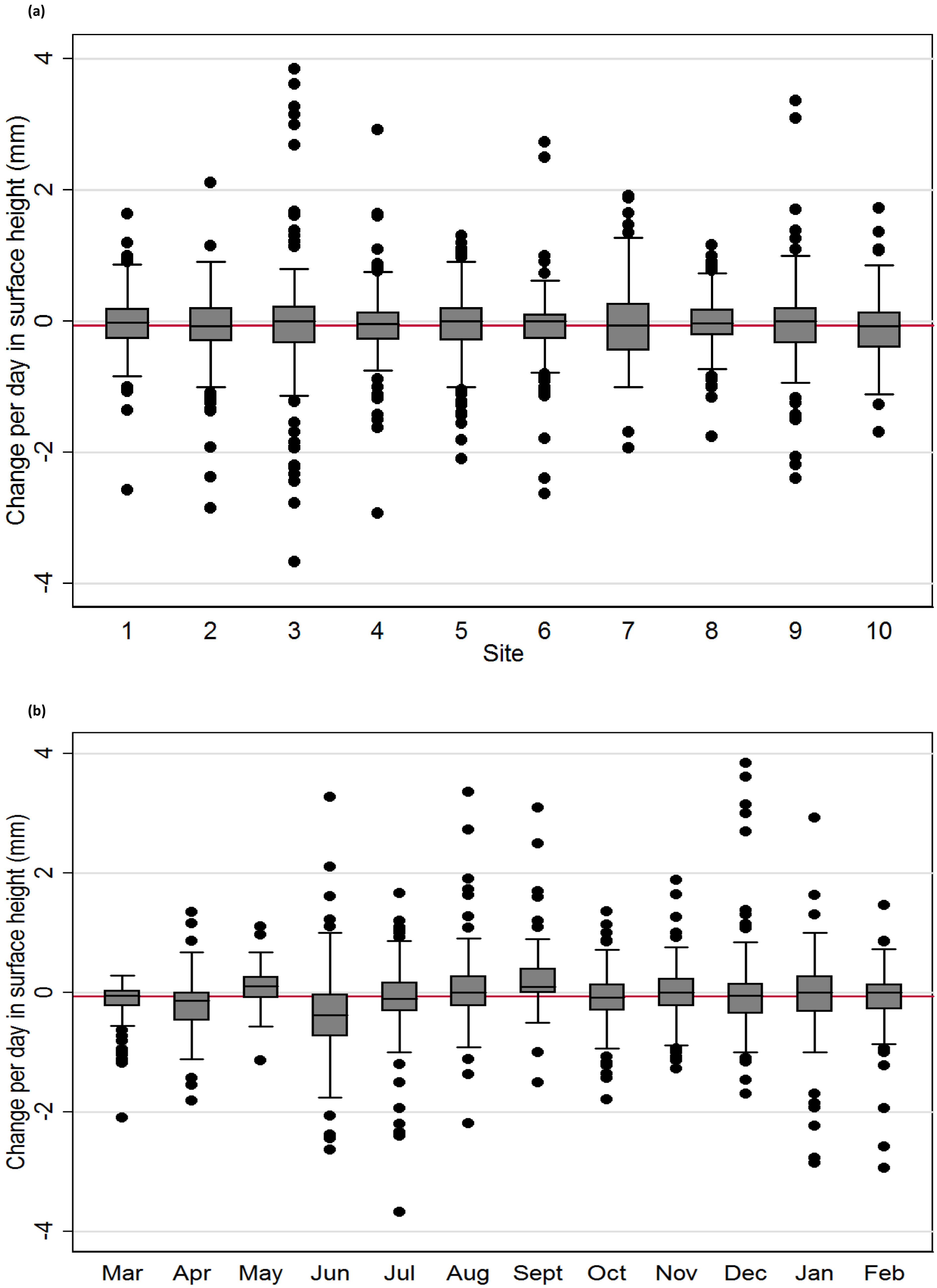


Figure 6.6 – (a) *Change per day of pin exposure plotted by site and* ***(b)*** *change per day for each month. The red line indicates the mean value of change per day of 0.06 mm.*

Figure 6.6 **(a)** confirms, that over the entire monitoring period, there has been little change in surface height of the peat haggs. All change values fall within ± 4mm and 98% of the change values fall within ± 2 mm. Site three would appear to have the greatest variation of recoded changes in surface elevation during the monitoring period. This is probably because site three is one of the most exposed erosion pin sites with little shelter from other peat haggs, leaving it exposed to prevailing south westerly winds (Figure 4.4 **(a)**). Figure 6.6 **(b)** shows change per day of pin exposure plotted by month. Again, this shows that little change (±4 mm) has occurred, but it is possible to see some seasonal patterns within the data set. Figure 6.6 **(b)** highlights that the month where the highest median amount of erosion occurred was June. September appears to be the month which showed the highest median value of deposition.

Analysing pin exposure for each pin in the pin array (Figure 6.7) can potentially be used to infer the mode of erosion on the peat haggs. Greater erosion recorded for the pins at the bottom of the slope (pins L3, L4, R3, R4) would suggest the importance of surface wash; more sporadic patterns would tend to suggest wind erosion or rain drop impact were more significant. Visual interpretation of Figure 6.7 identifies erosion has occurred sporadically across all pins, suggesting erosion from the peat haggs is predominantly resultant from wind and rain. A third potential method of surface change is wind blowing sediment up slope, which would result in a decrease in surface elevation recorded at the base of the slope and an increase at the top of the slope.

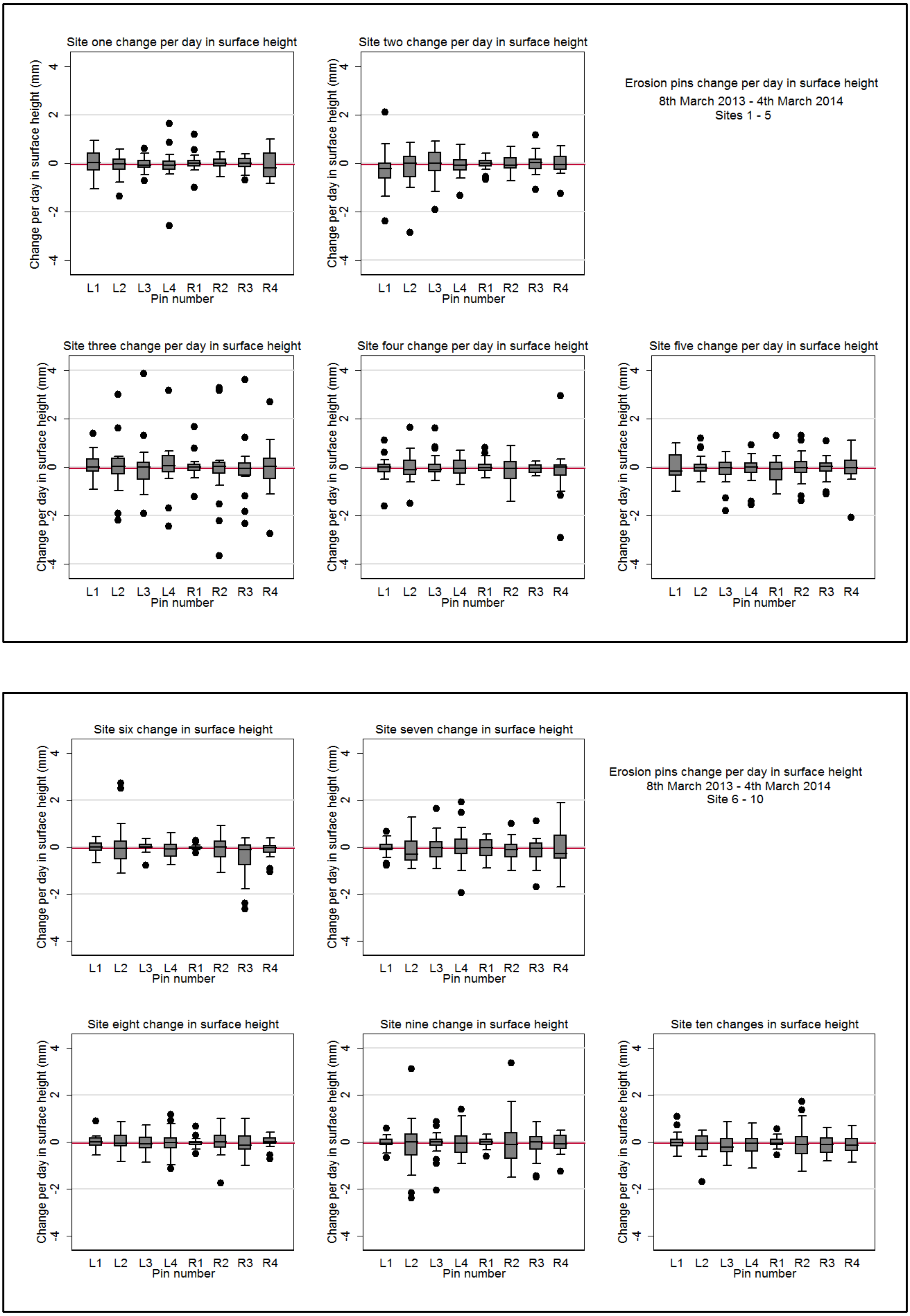


Figure 6.7 - *Erosion pin data for sites 1-10 plotted showing changes for individual pins. The red line indicates the mean change per day value of 0.059 mm*

6.2.1.2 Slope angle of peat haggs and rates of erosion

To assess whether a relationship exists between the slope angle of the peat hagg faces and erosion, the mean values of change over the entire monitoring period (6.8 **(a)**) and mean change per day (6.8 **(b)**) for each site were plotted against measured slope angle.

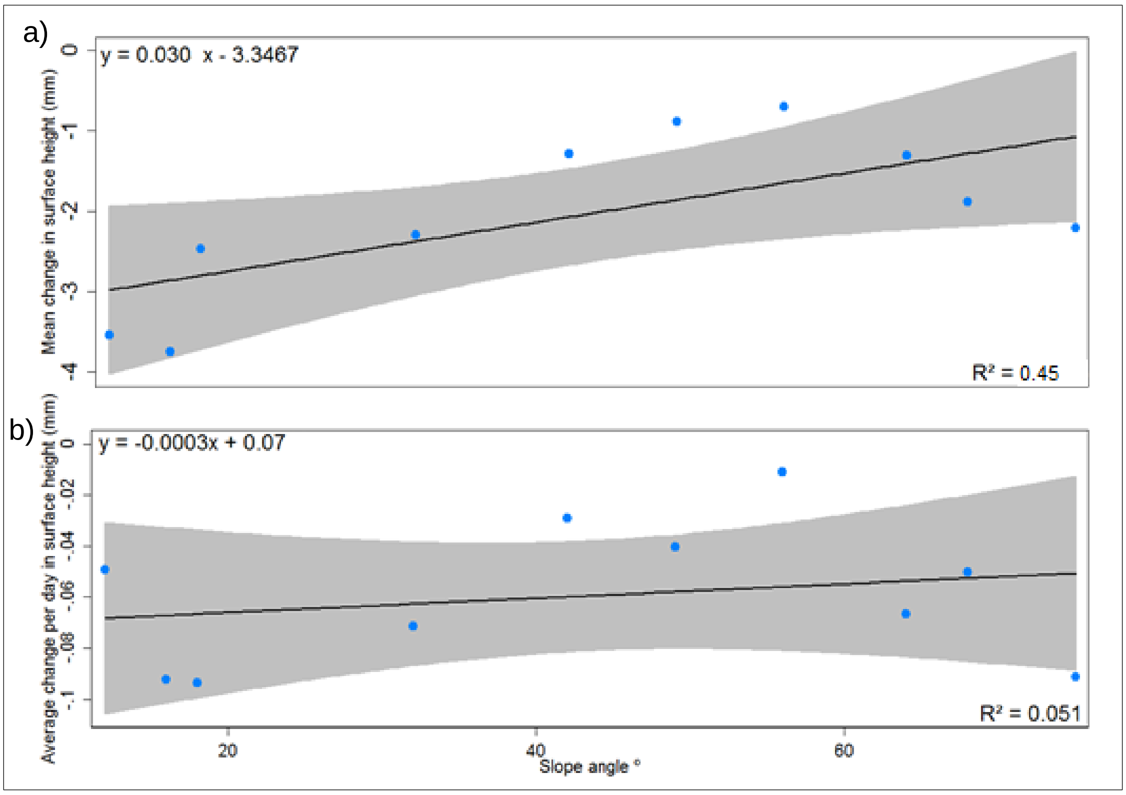


Figure 6.8 – (a) *Mean change in erosion pins over monitoring period plotted against slope angle* ***(b)*** *mean change per day of pins plotted against slope angle.*

Figure 6.8 **(a)** shows a negative relationship between slope angle and erosion, indicating that the erosion pins located on slopes of a lower angle experienced more erosion than those on steeper slopes. Figure 6.8 **(b)** shows only a very slight trend. This pattern may at first sight appear counter-intuitive; however, the haggs with lower slopes are often more susceptible to erosion due to extended periods of water ingress and ponding, increased frequency of frost action, greater trampling by animals and less sheltering from overhanging turf/vegetation. Figure 6.8 **(a)** could potentially identify a 'breakpoint' at 60°, indicating that at angles >60°, the steeper surfaces are undercutting the surface vegetation, which offers some shelter to the hag face.

6.2.2 Changes in surface elevation of the bare peat flats

Three monitoring pole transects were measured approximately every two weeks to record changes in surface height of the bare peat area (Figure 4.4 **(b)** in Chapter 4). The results in the following subsection show changes in surface elevation recorded by the pole transects. Results are compared with surface height data obtained using TLS to assess whether TLS is an appropriate method for monitoring changes in peat surface height. Table 6.2 shows the descriptive statistics of the data obtained for the three transects.

Table 6.2 – *descriptive statistics for the pole transect data*

Transect	Mean (mm)	Minimum (mm)	Maximum (mm)	Median (mm)	Standard deviation (mm)
Northern	1.67	-150	191	2	34.5
Long	1.79	-150	158	1	29.7
Southern	3.94	-96	103	0.5	30.5

The Northern transect contains 9 poles located from the west to east of the bare peat and is located closest to the AWS. The Southern transect is located at the southern end of the bare peat and contains 10 poles from west to east. Finally, the long transect begins at the southwest edge of the bare peat area and contains 20 poles finishing at the northeast end of the bare peat flats (Figure 4.4 **(b)**). Figure 6.9 shows the spatial pattern of net erosion and deposition per day across the peat flats. The Northern transect (N1-N9, located closest to the weather station) shows the most erosion, with all poles except N7 recording net surface lowering. The Southern transect (S1-S10) is of similar orientation and length to the upper transect but shows far less erosion. This could be due to the Northern transect being far more exposed to the prevailing south-westerly winds, however, at the southern end of the bare peat area there is protection from erosion due to sheltering from peat haggs located in close proximity to the southerly transect. The long transect shows areas of both deposition and erosion. Towards the northern end of the pole transect, more deposition is recorded, most likely because the surrounding peat haggs offer some protection from the prevailing wind, but also the local topography falls away to the north, creating a leeward slope which would naturally encourage deposition of material eroded from the more exposed flats (e.g. N1 – N9). Some individual poles within the transects show what would appear to show anomalous values (e.g. N7). Erosion recorded by the pole transects will be highly influenced by the location of individual poles, with local scale erosion and deposition being related to microtopography of channels and individual haggs. The prevailing wind direction during this study was found to be from the southwest (Figure 7.9). The location and topography surrounding Flow Moss results in the site being exposed to wind prevailing from the North, South and West, however, to the East there is a ridge that is approximately 10 m high and offers some protection to the site from wind from that direction. This may explain some of the erosion and deposition patterns seen in Figure 6.9.

Figure 6.10 summarises the changes in surface height for each pole transect during the entire monitoring period and demonstrates that overall the peat flats are relatively stable with the mean value of lowering being 2.4 mm. The pole transect with the greatest amount of surface lowering is the Northern transect located in the largest area of bare peat, and most exposed location (Figure 4.4 **(b)**).

Figure 6.11 plots change in surface elevation per day by month and shows the greatest mean change per day occurred in December 2013. Like the erosion pins, an increase in erosion was observed in June and an increase in deposition in September. From Figure 6.11, it is possible to see a slight sinusoid trend running through the data, suggesting a cyclical pattern within the data set, with more erosion occurring during April, May and June, deposition occurring in August, September, and October and erosion occurring again in November and December. The observed pattern could be resultant from seasonal water table changes, which cause the subsidence and swelling (or “mire-breathing”) in peat (Price, 2003).

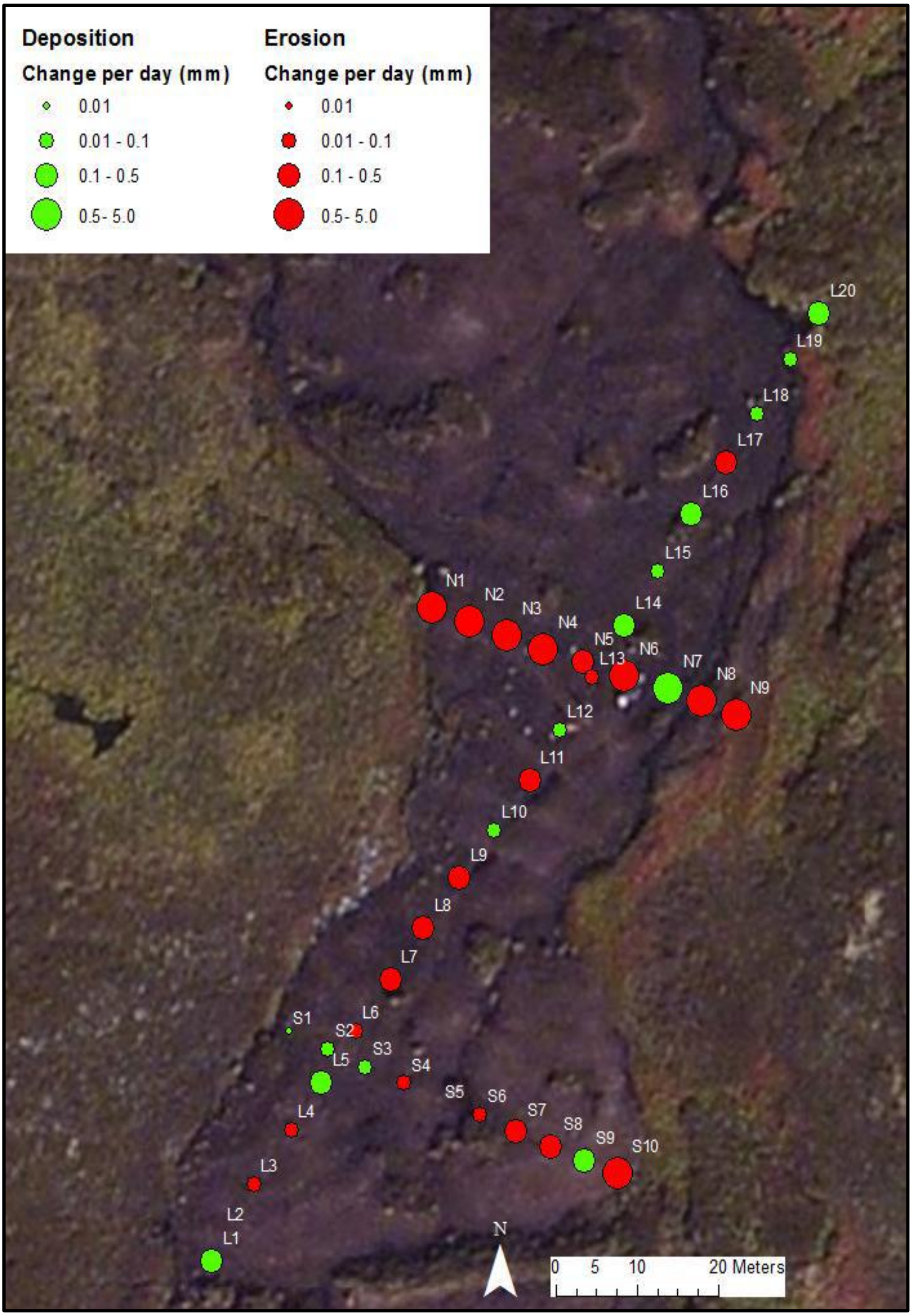


Figure 6.9 – *A graduated symbol plot showing the spatial variation of the change per day in surface height measured using pole transects (March 2013-March 2014). The mean annual change per day for all transects was -0.043 mm.*

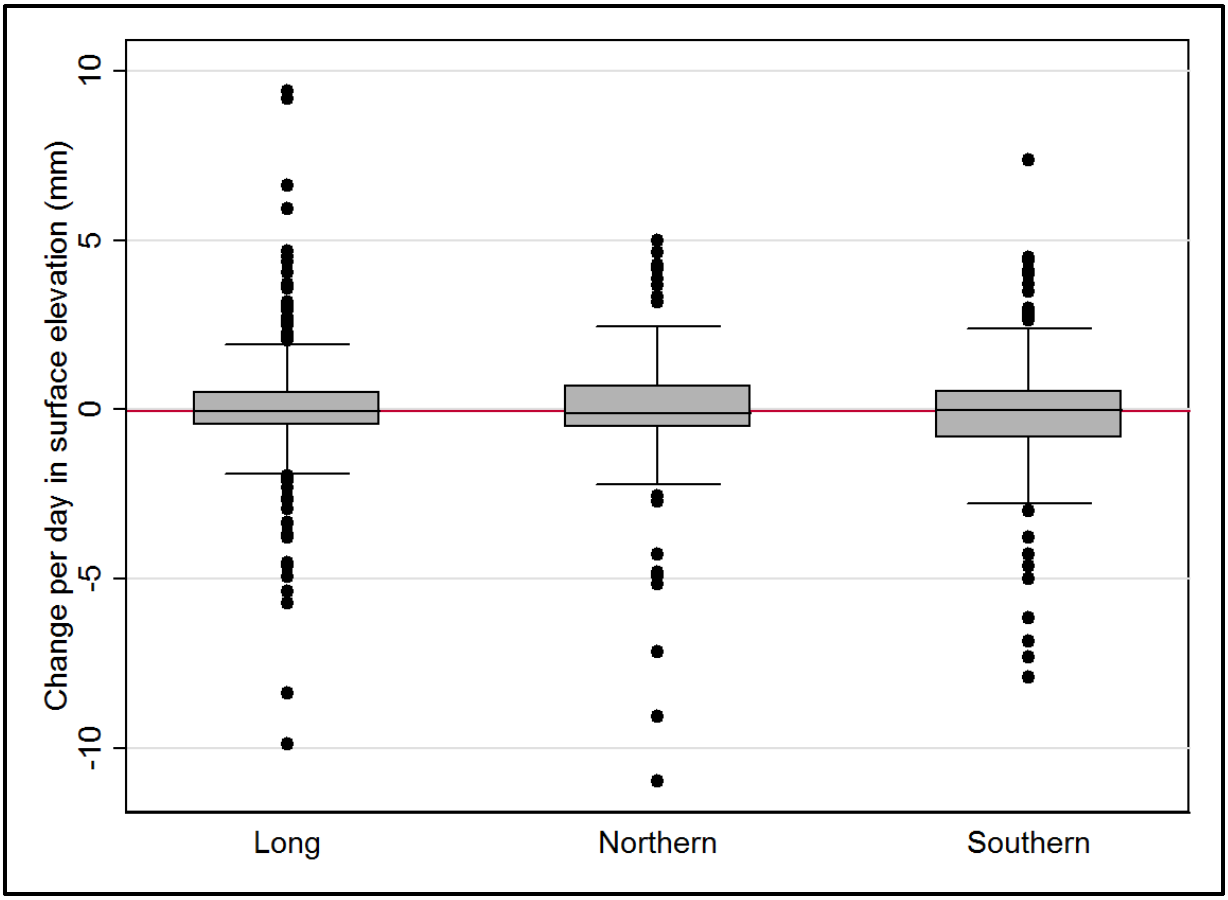


Figure 6.10 - *Box plots showing the change per day in surface elevation recorded using the three pole transects during the monitoring period (Figure 4.4b). The red line indicates the mean value of -0.043mm.*

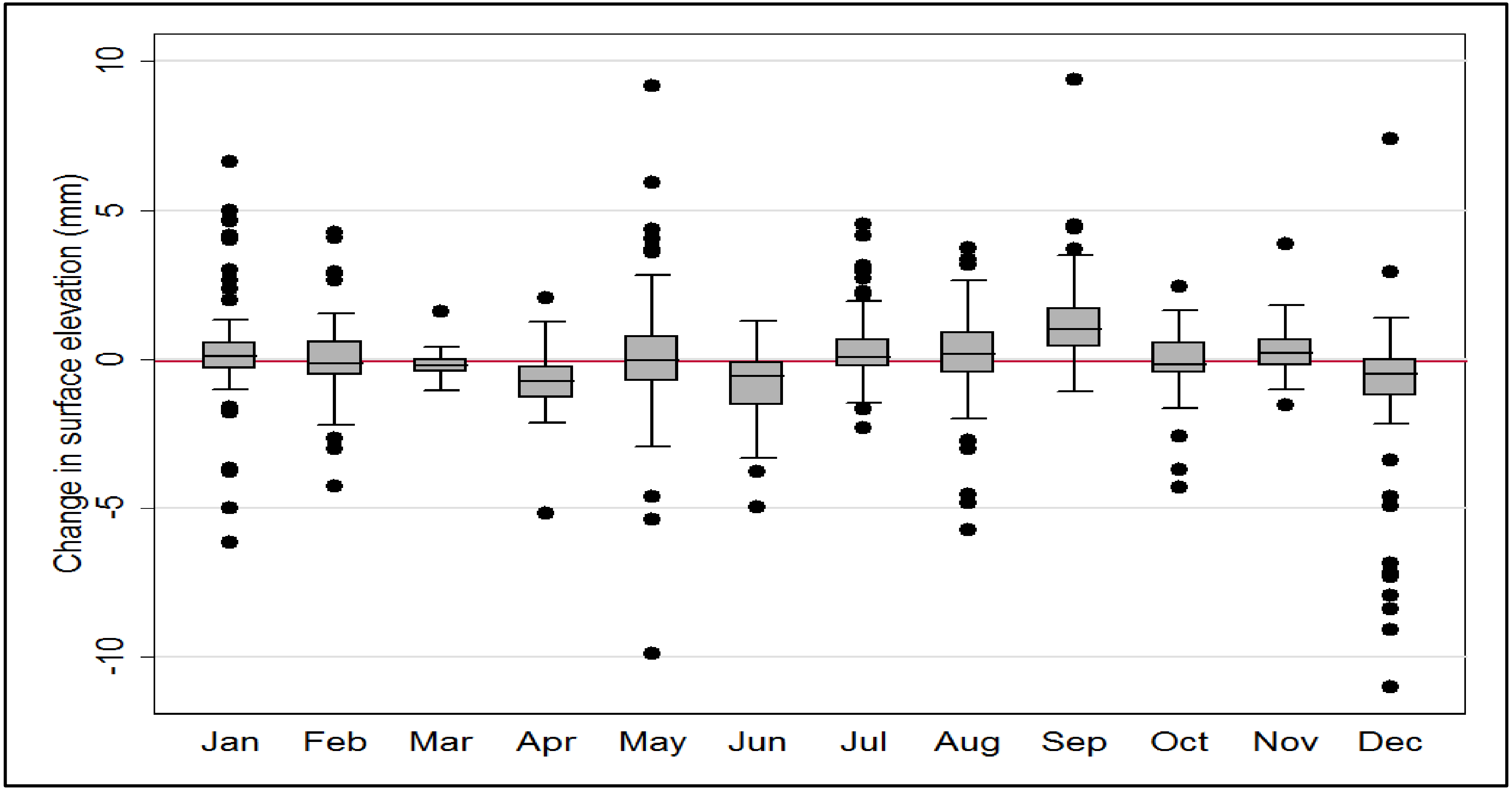


Figure 6.11 *- Box plots showing the average change per day in surface elevation of the bare peat flats plotted by month. The red line is indicative of the mean value of -0.043mm*

6.2.3 Changes in deposition in the main peat pool

11 fixed poles were situated in a peat pool at the northern end of Flow Moss. This pool intercepts all the drainage from the upper catchment and is drained by two small ephemeral streams that are intercepted by the sack traps. Figure 6.12 shows the locations and labels of the deposition poles within the pool.

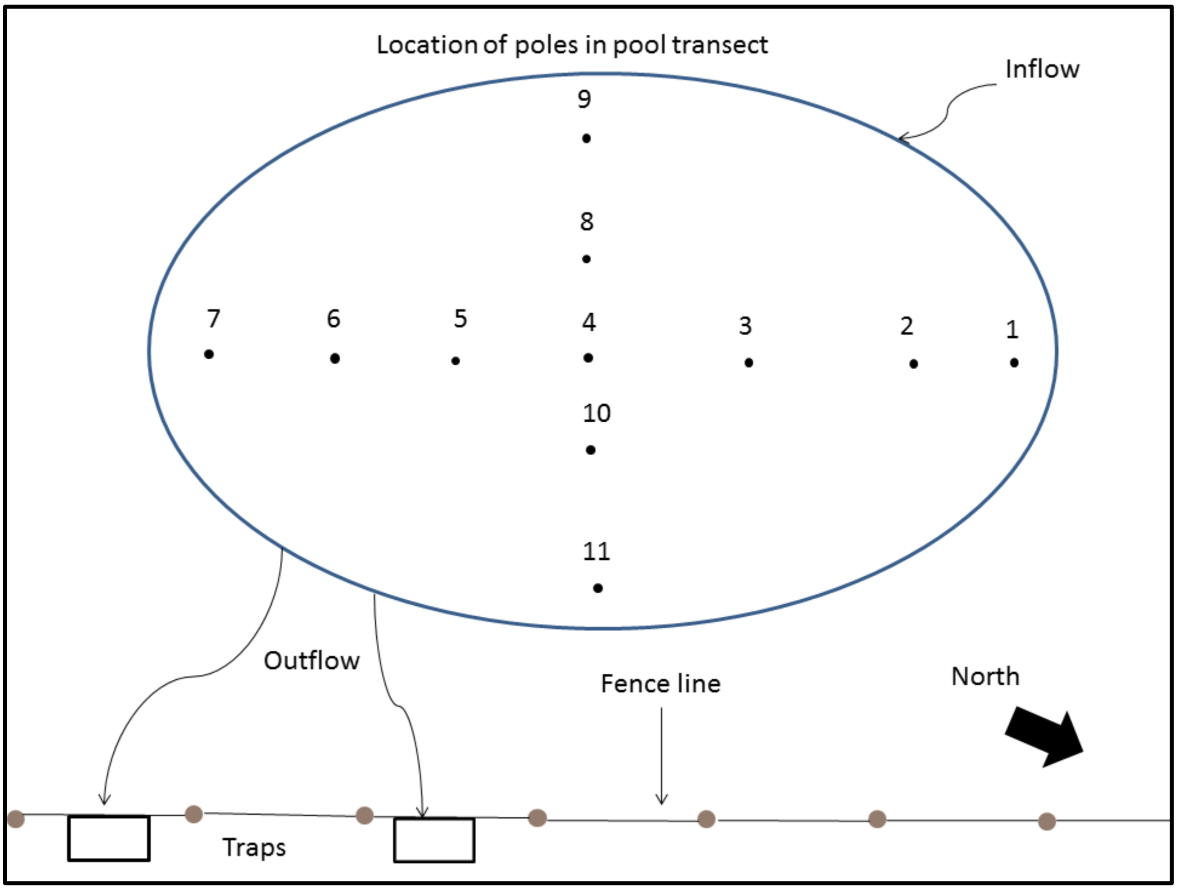


Figure 6.12 – Location and numbering of poles located in the peat pool transect

Figure 6.13 displays the pool transect data plotted by month and indicates that the greatest amount of erosion was recorded for March, while the greatest levels of deposition occurred in June and February. These results are the opposite of the patterns observed from the pole transects on the bare peat area, suggesting that eroded peat from the bare flats is stored in the peat pool.

Pool transect data were compared to both the pole transects located in the upstream bare peat area and the downstream sack traps to establish if a correlation existed between the sites. Correlation statistics (Table 6.3) show only weak relationships. Between the pool and upstream bare peat area there is no statistically significant relationship, while a clearer positive correlation (r^2 0.3313) exists between the pool and the sack traps. Nonetheless, there is no statistically significant relationship between the two data sets. This supports the observations that the connectivity between the bare peat flats and the main drainage lines is only weakly coupled and the export of eroded peat from the site via the fluvial system is slight (due to pool and vegetation trapping of sediment) and only occurs when the ephemeral drainage channels are active following major runoff events.

Table 6.3 - *Correlation statistics for the pool transect data plotted against sack trap data and bare peat data*

	Correlation co-efficient (R^2)	**Spearman's rank (r_s)**
Pool and sack trap data	0.002	0.03
Bare peat and pool data	0.331	0.15

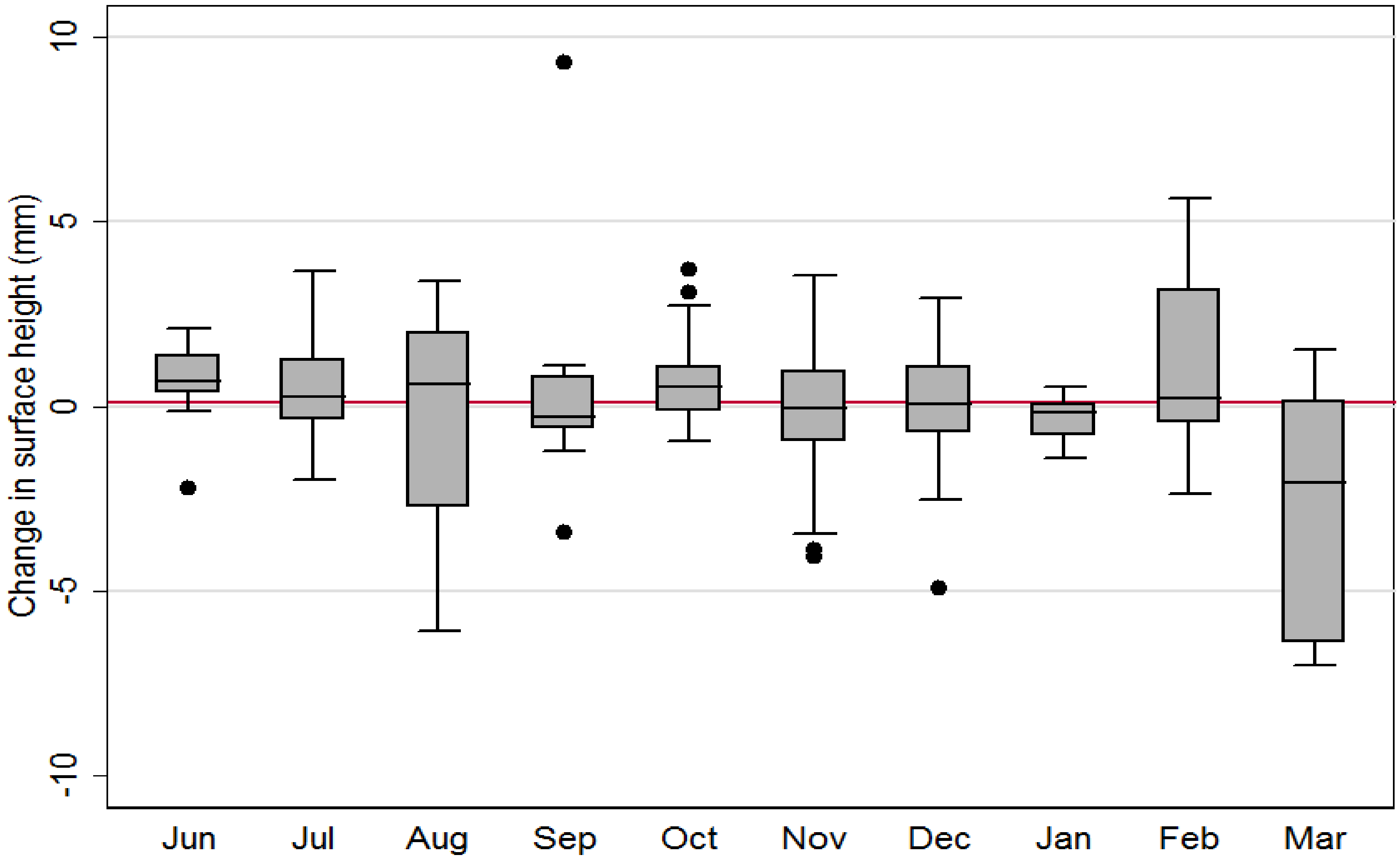


Figure 6.13 – *Change per day of the pool transects plotted by month. Red line indicates the mean value of 0.117 mm*

6.2.4 Terrestrial Laser Scanning (TLS)

Pole transects and erosion pins provide information relating to changes in the surface height of the peat at specific point measurement locations. TLS approaches, however, provide a means of collecting non-destructive, regular observations of the peat surface (Mulder *et al*., 2011) over a spatial scale far greater than that which could be achieved using manually measured local point observations. In this study, an Area of Interest (AOI) located towards the northern end of the bare peat flats was selected for the TLS survey (Figure 4.3 **(b)**). This area was selected as it incorporates a range of surface conditions, including bare peat; vegetated peat and part of a channel which runs through the site. The TLS data were post processed using two different workflows and the methods used to achieve this are outlined in Appendix A. TLS data were collected nine times during a ten month period. The initial aim was to collect TLS data approximately every two weeks, however the times between surveys varied due to equipment availability and windows of suitable scanning weather. The dates of data collection are shown in Table 6.4.

Table 6.4 – *Dates of collection of the nine TLS data sets*

Scan number	Data collection date
1	6th June 2013
2	8th August 2013
3	19th August 2013
4	16th September 2013
5	10th October 2013
6	24th October 2013
7	8th November 2013
8	25th November 2013
9	4th March 2014

6.2.4.1 DEM of Difference

Once the data had been post-processed (using methods outlined in Appendix A1), rasters were imported into ArcGIS, clipped to the same size, and the raster calculator function used to subtract one DEM from another to create a DEM of Difference (DoD) using the work flow process outlined in Figure 6.14. During this process, the earlier DEM (e.g. DEM 8) was subtracted from a later DEM (e.g. DEM 9). In this study, DoDs have been created from DEMs gridded at 20 cm, 2cm and 1 cm resolutions. The resolutions of the DEMs were selected as DEMs filtered at 20 cm and 2 cm have previously been used by Grayson *et al*. (2012) in a study utilising TLS for peatland monitoring. Unfiltered data were exported and gridded at a 1 cm resolution for comparison with results obtained at 2 cm resolution (Figure

6.15). The sizes of the area in Figure 6.18 differ because at higher resolution it is only possible to export smaller areas of the point cloud due to computer memory limitations.

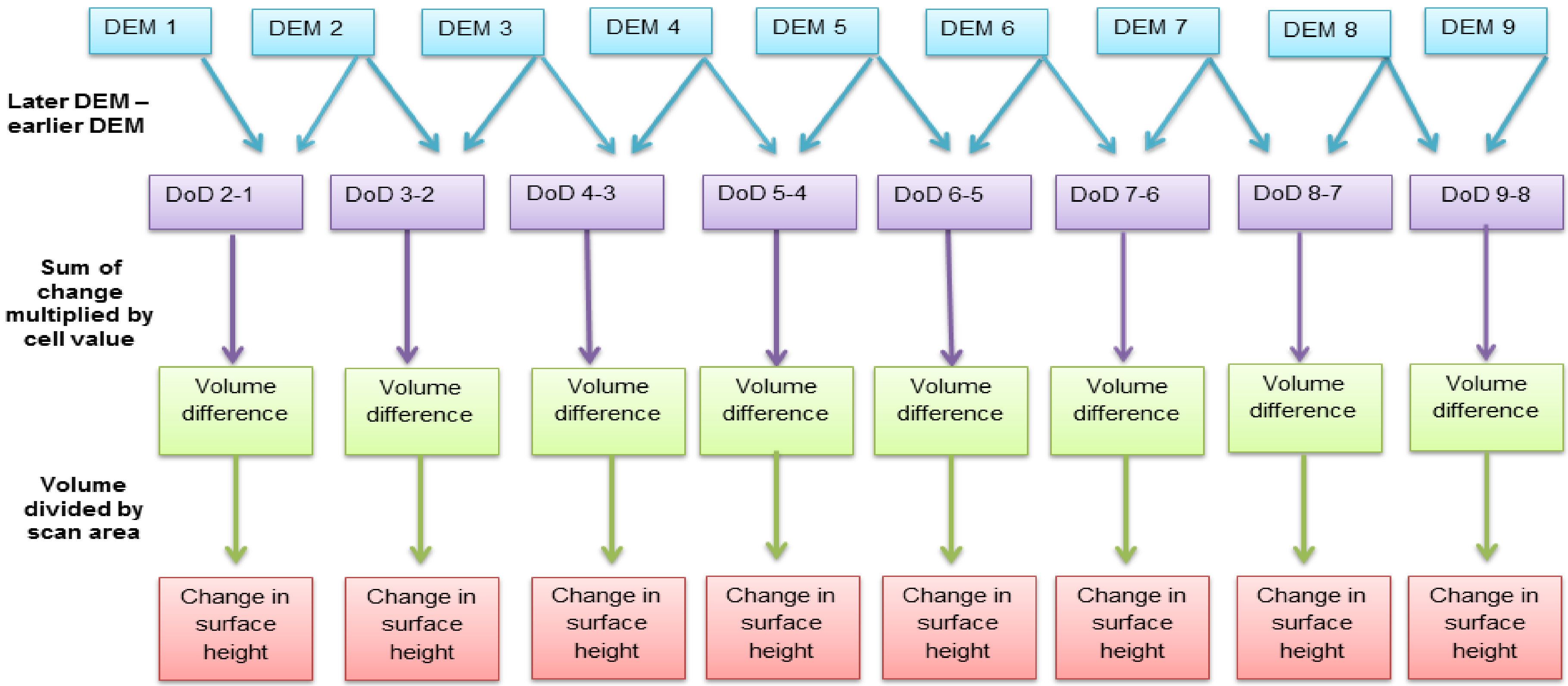
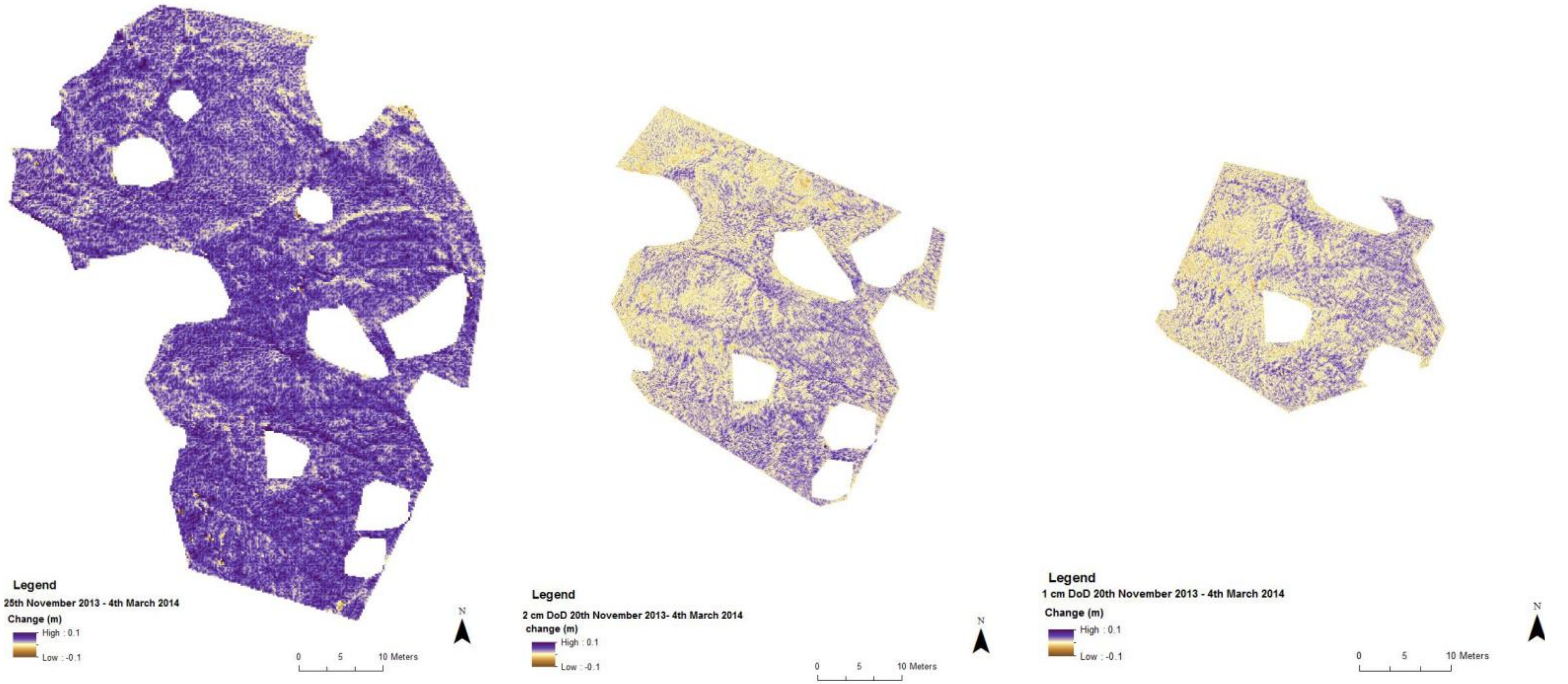


Figure 6.14 - *Processing workflow for calculating the difference between DEMs.*



20 cm DoD **2 cm DoD** **1 cm DoD**

Figure 6.15 – *Examples of DoDs gridded at 20 cm, 2 cm and 1 cm resolutions.*

Several approaches were used when calculating the differences between DEMs. Firstly, DoDs were created for the data at a 20 cm resolution which included areas of bare peat and vegetation (Figure 6.16). This provided an indication of changes over a large area of the site between epochs. The data in Figure 6.16 indicate that between 4th March 2014 and 6th June 2013, deposition has occurred on the bare peat flats, and the height of the surrounding vegetation has increased. Five of the DoDs in Figure 6.16 indicate deposition has occurred on the bare peat flats, while two suggest no change and two indicate erosion. The DoD calculated using data captured at the start and end of the study suggest that overall, deposition has occurred.

Although these DoDs in Figure 6.16 provide an indication of large scale surface change, much of the difference results from changes in vegetation height. Due to the large change that can occur in vegetation height even over a short period of time, the colour stretches of the change maps cannot be easily visually interrogated to identify patterns of erosion and deposition on the peat surface. Figure 6.17 shows examples of two DoDs, from 6th June – 8th August and 25th November – 4th March where the colour scales have been matched. As can be seen, due to the large differences in scale between the DoDs, largely caused by changes in vegetation height, at this scale the important detail of changes on the bare peat surfaces cannot be easily seen.

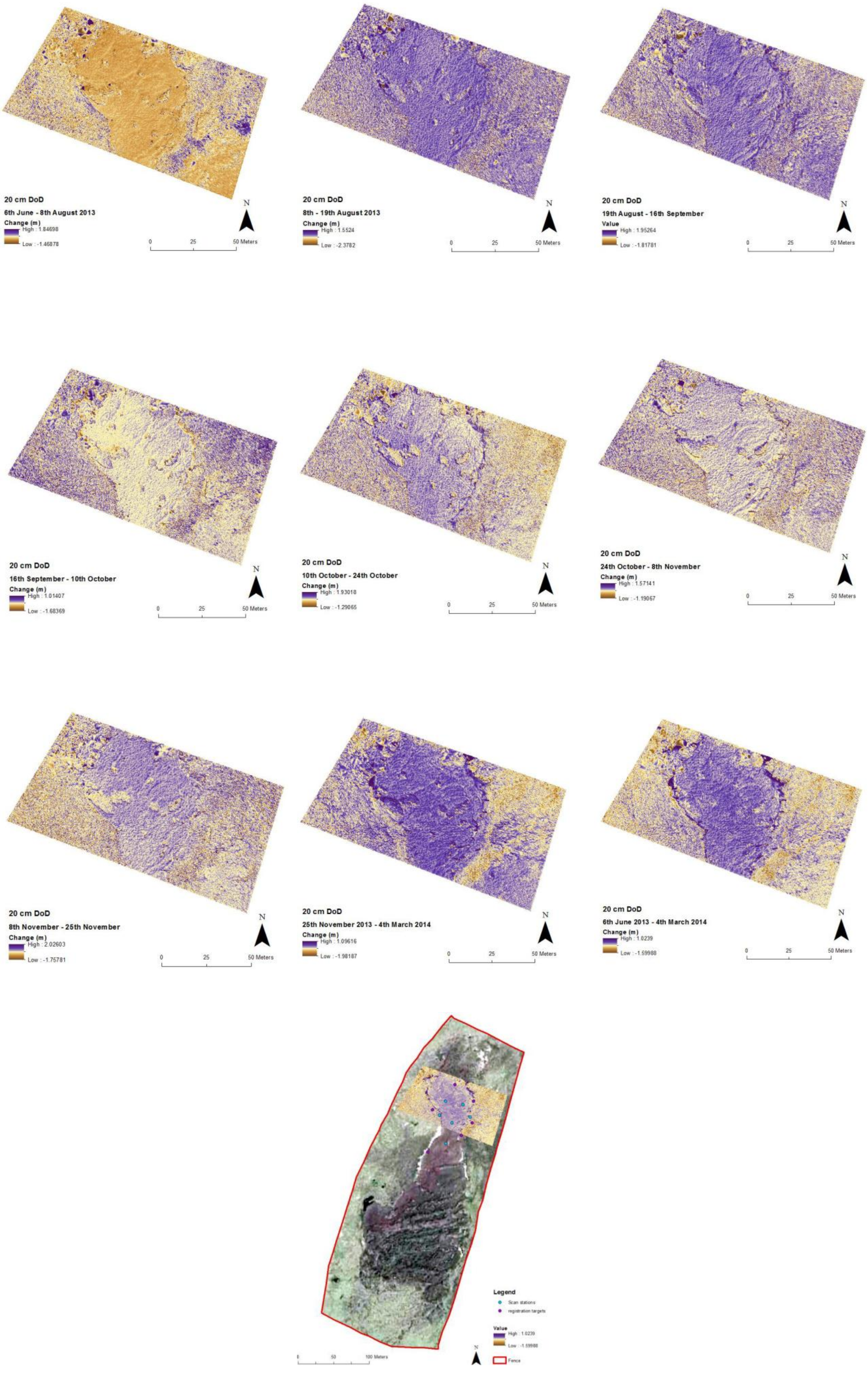


Figure 6.16 - *DoDs created from DEMs at a 20cm resolution, vegetation included*

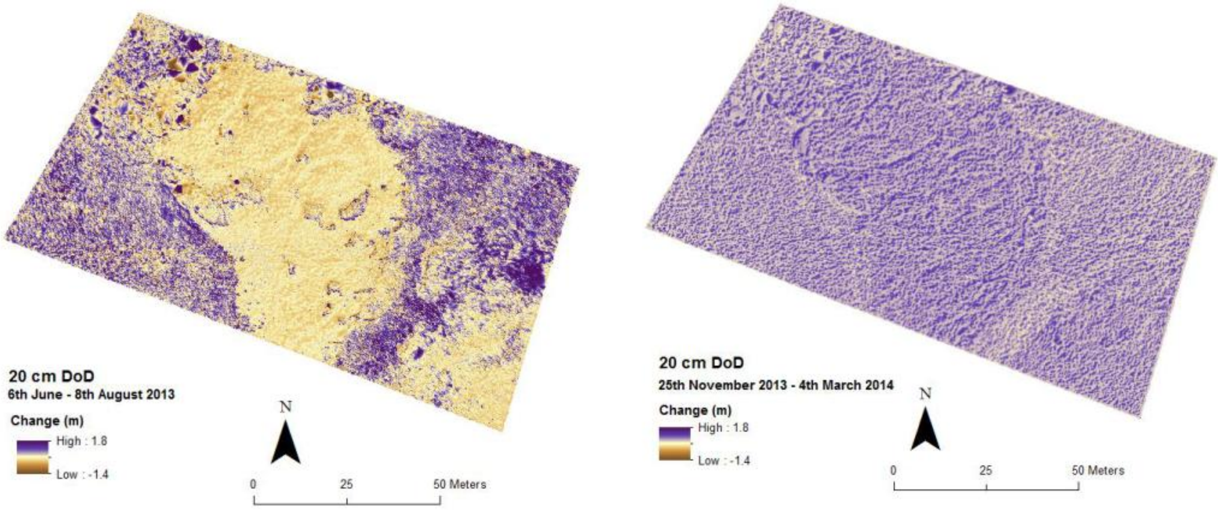


Figure 6.17 – *Examples showing the impact of large changes in vegetation height on the resolution of the colour stretch used in the DoDs.*

To create comparable maps, areas of vegetation were removed. Areas of vegetation, peat haggs and water were identified and mapped during a dGPS survey. The collected data were then used to generate a shape file in ArcGIS which masked out areas of vegetation and peat haggs. Areas of standing water located on the periphery of the bare peat were also masked out, as the water acts as a reflecting surface and can result in areas of 'no data' in the DEM. The DoDs generated for the 20 cm resolution data are shown in Figure 6.18 while the DoDs generated for the 2 cm and 1 cm resolution data are in Appendix A (Figure A2 and A3). The colour stretches are matched allowing comparison between the change maps and qualitative analysis of where erosion and deposition has occurred within the scanned area. Figure 6.18 shows only small sub-centimetre changes have occurred across the bare peat flats during the period of monitoring and four of the DoDs indicate that a small increase in peat surface height has occurred between TLS surveys. Most deposition would appear to have occurred between 25th November 2013 and 4th March 2014. The DoD in Figure 6.18 shows that the most erosion occurred between 16th September 2013 and 10th October 2013 (DEM 5- DEM 4). It would also appear that in the majority of cases very little change occurred. To quantify changes occurring in peat surface height, the data are used to calculate the mean change in surface height across the TLS survey area.

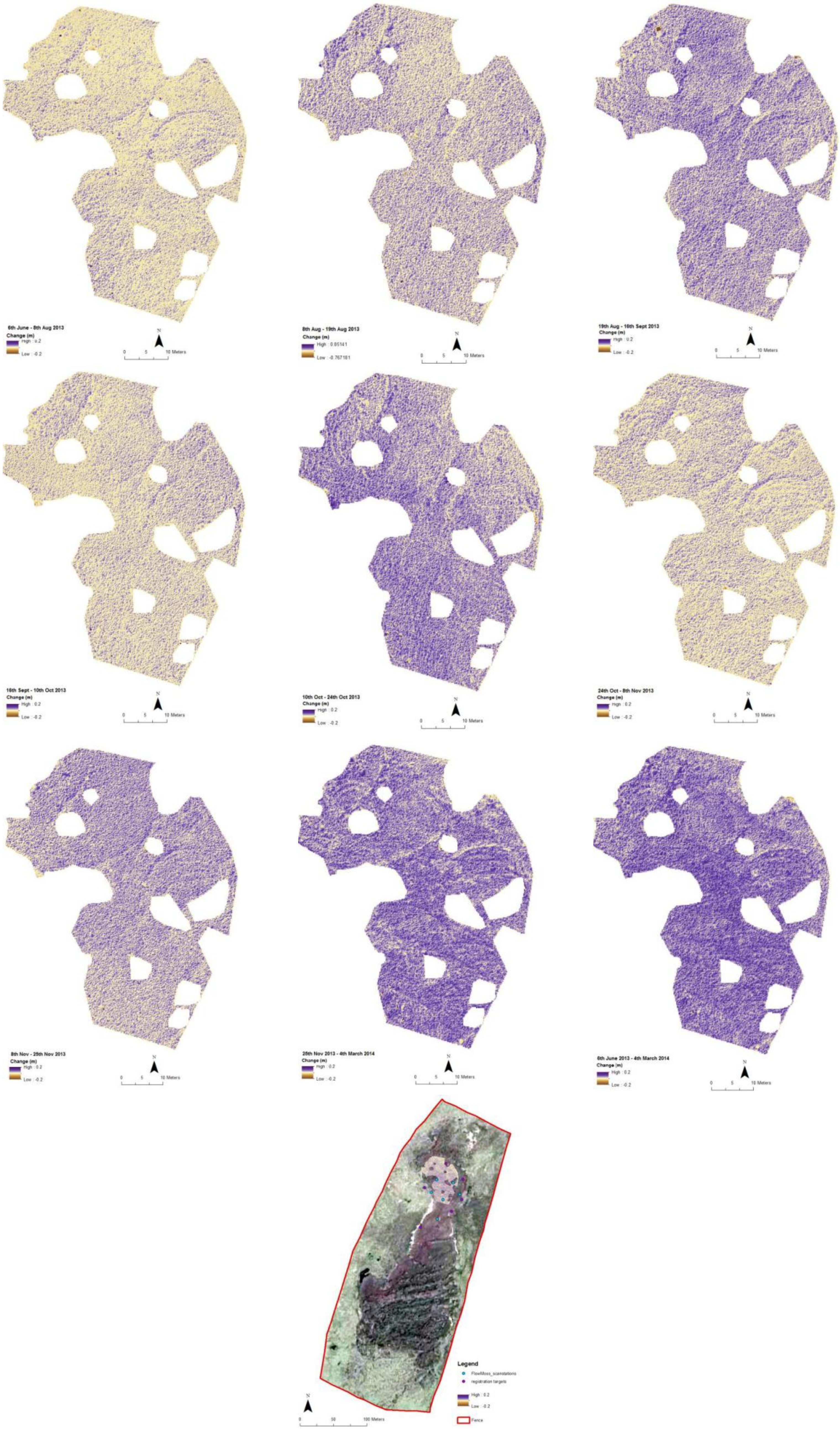


Figure 6.18 – *DoDs created from DEMs at a 20 cm resolution with areas of vegetation and peat haggs masked out.*

6.2.4.2 Change detection calculated using TLS

From the DoDs, it is possible to gain information relating to areas of erosion and deposition, and some of the small topographical features which have formed between scans. However, simple visual comparison of DoDs does not provide quantitative values such as changes in surface height or values of sediment loss or gain. This is calculated by measuring the volume difference between two DEMs. Figure 6.19 and Table 6.5 show the change in surface height calculated from the 20 cm DoDs, the 2 cm DoDs and the 1 cm DoDs. The values along the x axis in Figure 6.19 correspond to the specific DoD (Table 6.4). For example 9 – 8 is the DEM created from data collected on 25th November 2013 (8) subtracted from the DEM created from data collected on 4th March (9). Figure 6.19 shows that the surface height fluctuates between epochs, but in general there is an increase in surface height from the first set of data collected using TLS (on 6th June 2013) and the last set of data (collected 4th March 2014). The calculated surface heights shown in Figure 6.22 have been calculated from DoDs of different sizes (Figure 6.18) and therefore, the estimates of change in surface height would not be expected to be the same. This shows that when using TLS for peatland monitoring, survey area can impact on the results and should be carefully considered.

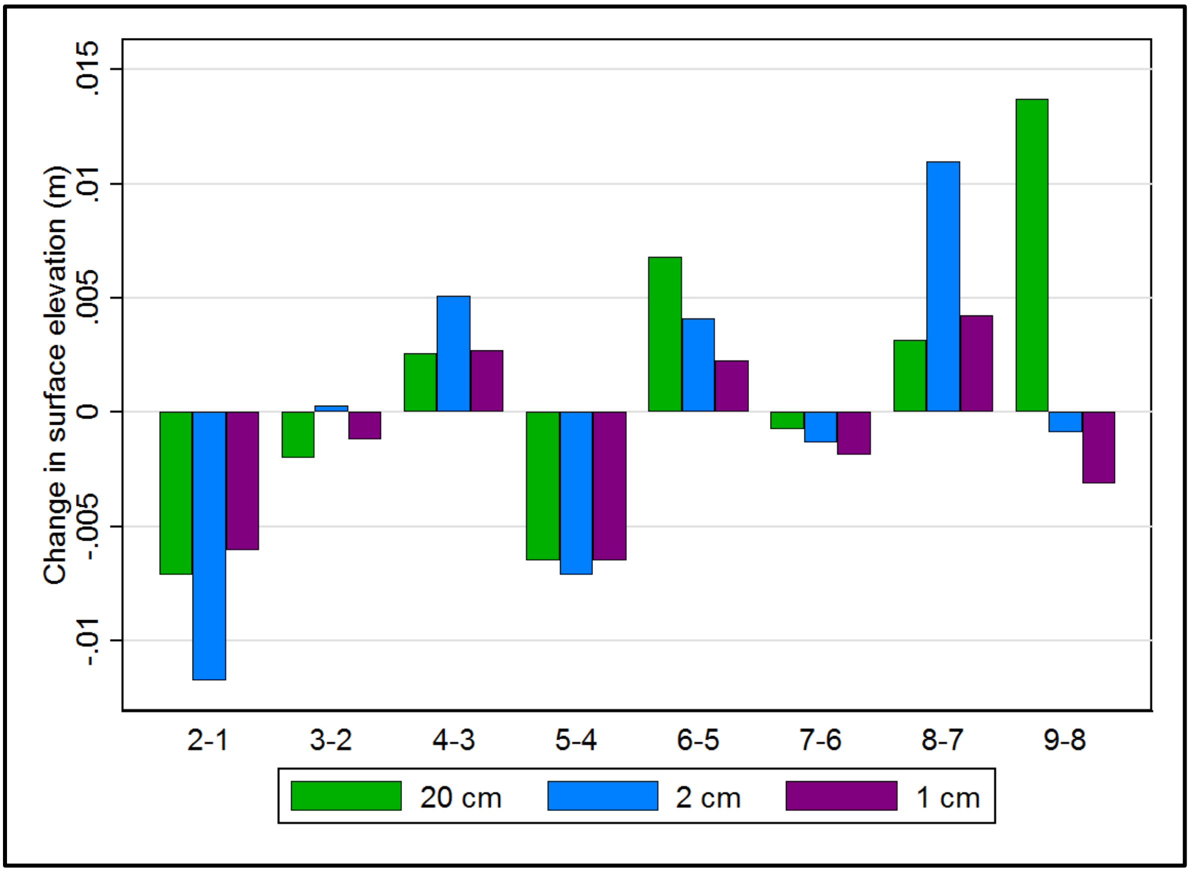


Figure 6.19 *- Graph showing the change in surface elevation measured using TLS filtered and gridded at three different resolutions.*

Volume estimates determined from TLS can be multiplied by the peat bulk density (0.078 g cm^{-3}) to give the change in the mass of peat in the scan area. This can be divided by the DoD area (m^2) to produce an average value of kg of peat loss or gain per m^2. The value of change per m^2 determined from the TLS data can be scaled up to the full area of the peat flats by multiplying by the area of bare peat mapped from the dGPS survey (Table 6.5).

Table 6.5 – *Changes in surface height and volume change estimates calculated from the TLS data*

	20 cm resolution		**2 cm resolution**		**1 cm resolution**	
DoD	**Average change in surface height (m)**	**Change in amount of peat (tonnes)**	**Average change in surface height (m)**	**Change in amount of peat (tonnes)**	**Average change in surface height (m)**	**Change in amount of peat (tonnes)**
2-1	-0.00714	-5.975	-0.01178	-9.86	-0.00606	-5.07
3-2	-0.00203	-1.699	0.00030	0.25	-0.00118	-0.99
4-3	0.00258	2.158	0.00509	4.26	0.00272	2.27
5-4	-0.00649	-5.426	-0.00714	-5.97	-0.00650	-5.44
6-5	0.00681	5.698	0.00410	3.43	0.00226	1.89
7-6	-0.00076	-0.636	-0.00134	-1.12	-0.00188	-1.57
8-7	0.00316	2.640	0.01099	9.19	0.00422	3.53
9-8	0.01369	11.450	-0.00088	-0.74	-0.00313	-2.62
9-1	0.01620	13.552	0.00845	7.07	0.00188	1.57

Although the data above were calculated from DoDs with the majority of vegetation and peat haggs removed, the DEMs still contained minor items of monitoring equipment and smaller dispersed patches of vegetation which could impact on the change results. Furthermore, although these estimates are scaled up for the whole 7 ha site, the DoDs used to estimate the volumes differ in size (Figure 6.15), due to trade-offs between spatial scale and resolution. Therefore, to calculate change values from the DEMs in a more rigorous fashion, a 96 m^2 area of the point cloud known to contain no vegetation, monitoring equipment or peat haggs was selected. The data were gridded at a 20 cm, 2 cm and 1 cm resolution. The DEMs created from TLS data gridded at a 2 cm resolution are displayed in Figure 6.20.

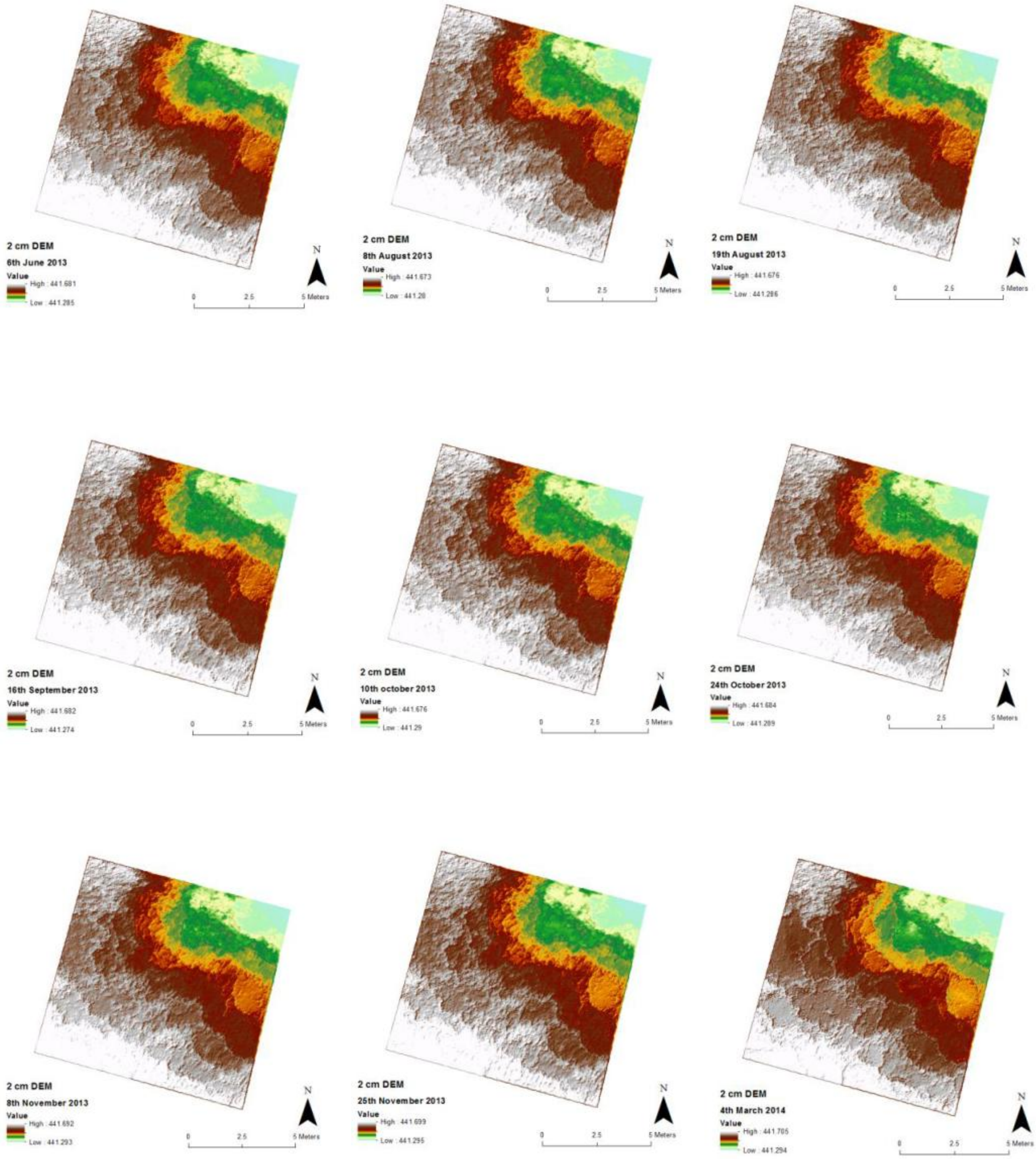


Figure 6.20 – *DEMs created from TLS data collected at nine different epochs and filtered and gridded at a 2 cm resolution.*

The DEMs were used to create a DoD (Appendix A) and average changes in surface elevation were calculated for comparison between different resolutions (Figure 6.21).

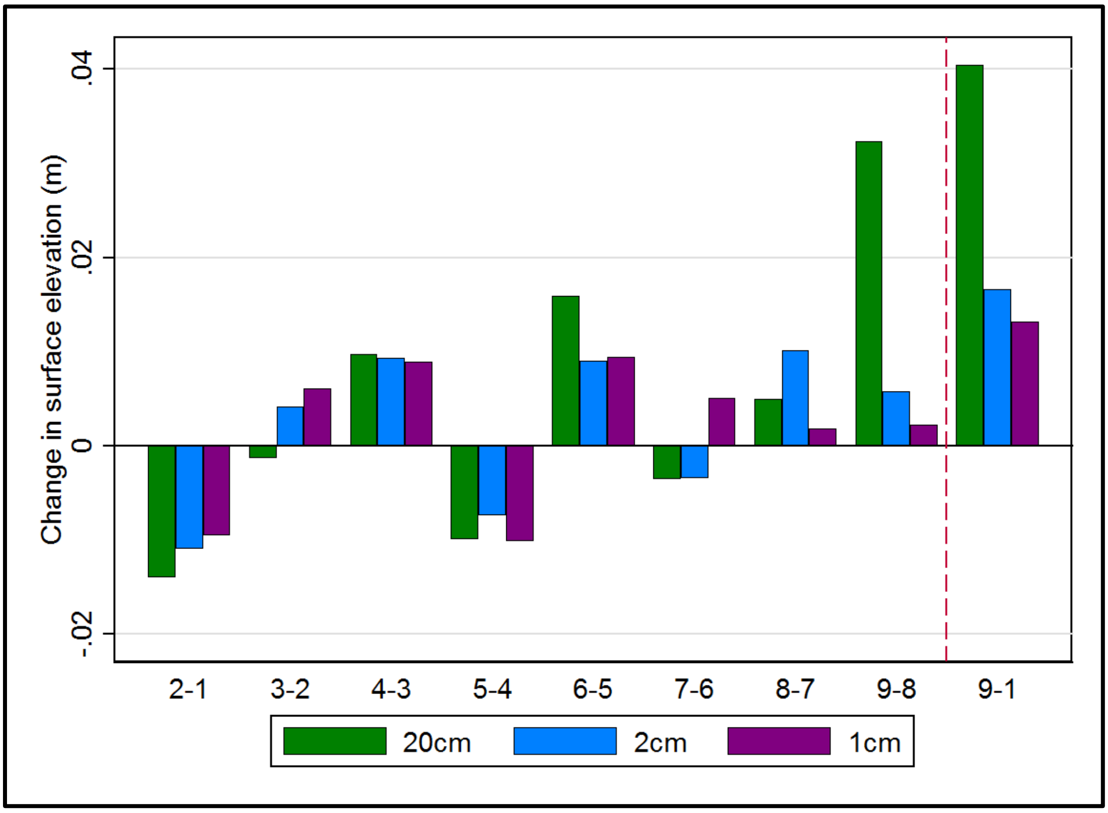


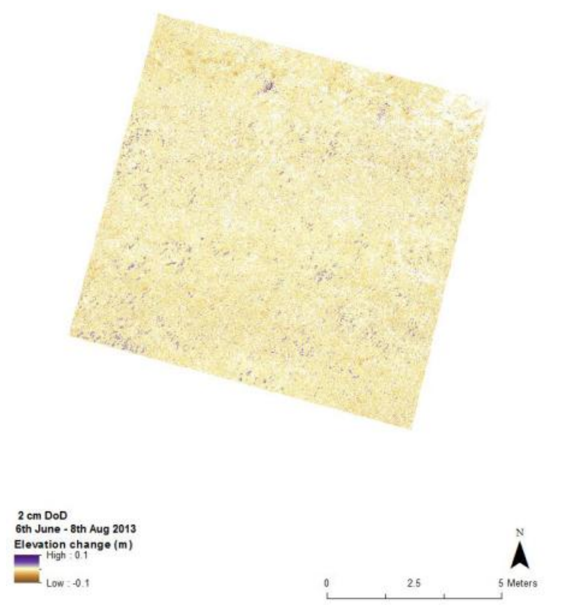
Figure 6.21 *- Graph showing the change in surface elevation measured using TLS filtered and gridded at three different resolutions. The red line divides individual survey epochs from the total monitoring period assessment results.*

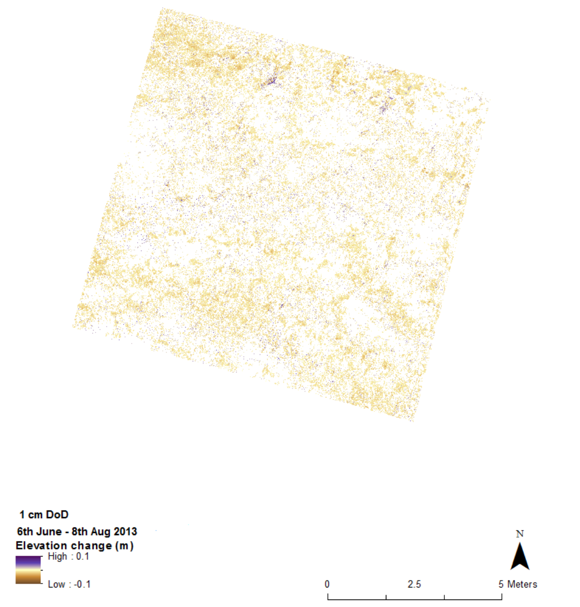
Figure 6.21 shows that even though the changes in surface elevation have been calculated from the same area of DoD, there are still significant differences in the calculated surface height. The data indicate that the highest amounts of change are produced using the data gridded at a 20 cm resolution and the lowest from data gridded at a 1 cm resolution. In seven cases in the data above, the values estimated from the 20 cm DoD are greater than those estimated from 2 cm and 1 cm DoDs. The figure suggests that the resolution of the DEM has an important impact on the magnitude of the change results. This is partly because of the different cell sizes used for the different resolutions. A DoD at a 20 cm resolution will resample the data and produce a number representative of all the values that fall within the 20 cm pixel. The data at a 2 cm resolution will contain pixels one tenth of the size of the 20 cm cell and therefore, an area the same size as the 20 cm cell will contain 10 times as many data points, data gridded at a 1 cm resolution will contain 20 data points in an area the same size as the 20 cm cell.

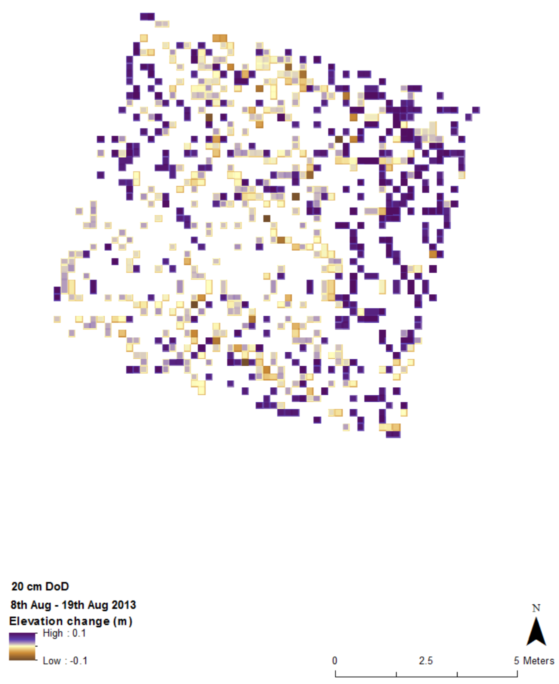
To account for error in the DoDs, a threshold value of +/-0.0071 m (the MSA value produced during post processing) was used to mask out the data points showing change which was considered to lie outside the detectable range of the TLS (Figure 6.22). In Figure 6.22, the column on the left displays data at a 20 cm resolution, the centre column data at a 2 cm resolution and the column on the right, data gridded at a 1 cm resolution. From the coarser 20 cm resolution DoDs, it is possible to identify general characteristics of change that has occurred (i.e. if erosion or deposition has occurred). However, from the higher resolution DoDs (those created from data gridded at a 2 cm and 1 cm resolution) much more detail can be seen and it is possible to pick out smaller geomorphological features such as micro-terraces forming on the peat surface. These are particularly evident in the DoDs created from data captured on 4th March 2014 (9) with the data collected on 25th November 2013 (8) subtracted. These terraces were not visible in all the DoDs, because as is shown in Figure 6.17, due to small scale changes occurring between some of the data collection dates, putting the DoDs on a comparable colour scale can remove some of the detail of the

DoDs. Nonetheless, if there was no change in height between the DEMs, a change of 0 would be shown in the DoD.

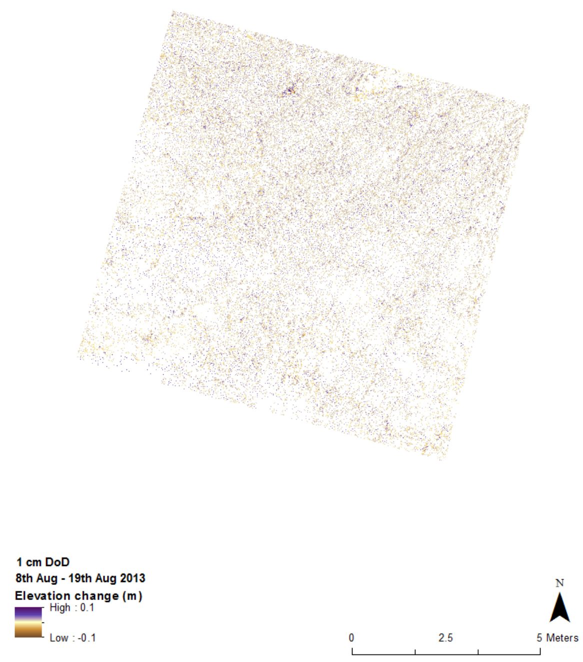

20 cm DoD
6th June - 8th Aug 2013
Elevation change (m)
High: 0.1
Low : -0.1
N
0
2.5
5 Meters


2 cm DoD
6th June - 8th Aug 2013
Elevation change (m)
High: 0.1
Low : -0.1
N
0
2.5
5 Meters

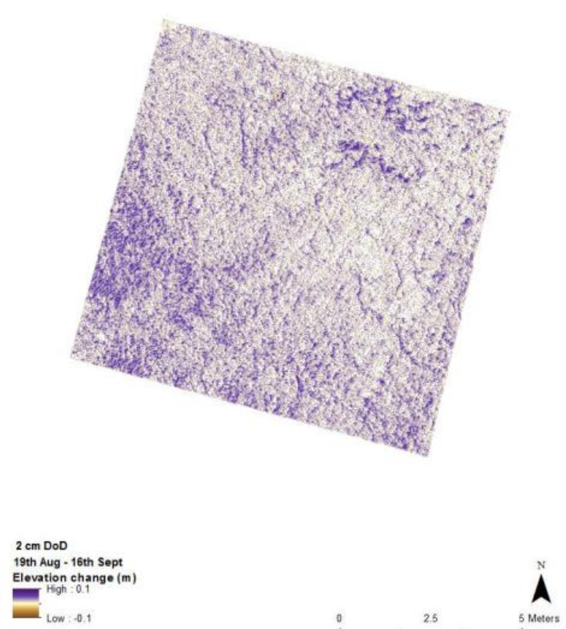

1 cm DoD
6th June - 8th Aug 2013
Elevation change (m)
High: 0.1
Low : -0.1
N
0
2.5
5 Meters

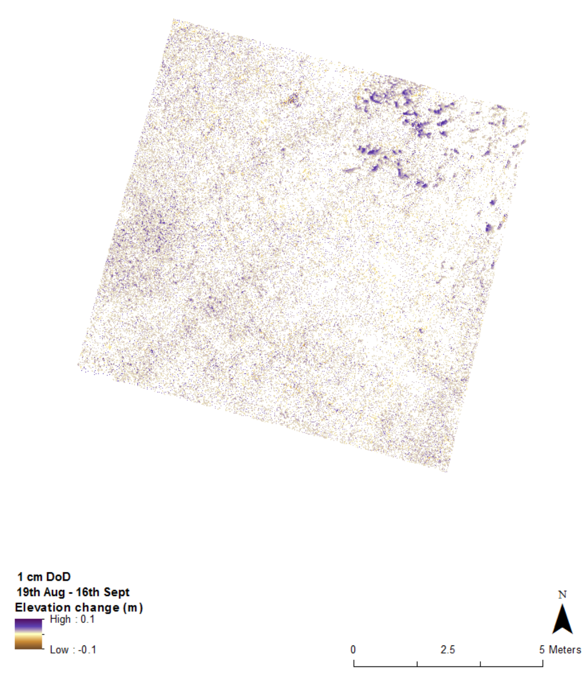

20 cm DoD
8th Aug - 19th Aug 2013
Elevation change (m)
High: 0.1
Low : -0.1
N
0
2.5
5 Meters

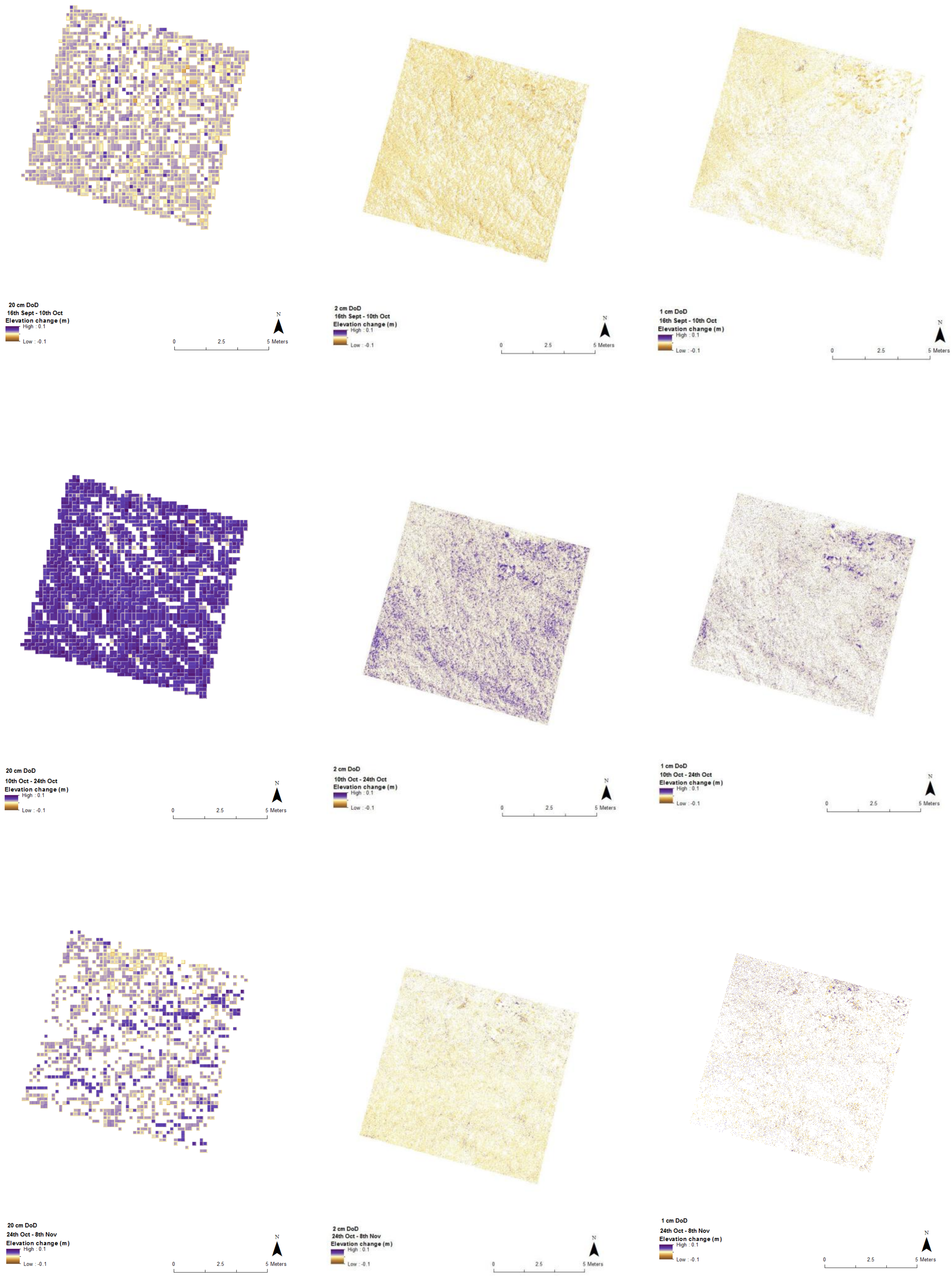

2 cm DoD
8th Aug - 19th Aug 2013
Elevation change (m)
High: 0.1
Low : -0.1
N
0
2.5
5 Meters


1 cm DoD
8th Aug - 19th Aug 2013
Elevation change (m)
High: 0.1
Low : -0.1
N
0
2.5
5 Meters


20 cm DoD
19th Aug - 16th Sept
Elevation change (m)
High: 0.1
Low : -0.1
N
0
2.5
5 Meters


2 cm DoD
19th Aug - 16th Sept
Elevation change (m)
High: 0.1
Low : -0.1
N
0
2.5
5 Meters


1 cm DoD
19th Aug - 16th Sept
Elevation change (m)
High: 0.1
Low : -0.1
N
0
2.5
5 Meters


20 cm DoD
16th Sept - 10th Oct
Elevation change (m)
High : 0.1
Low : -0.1
0
2.5
5 Meters
N
2 cm DoD
16th Sept - 10th Oct
Elevation change (m)
High : 0.1
Low : -0.1
0
2.5
5 Meters
N
1 cm DoD
16th Sept - 10th Oct
Elevation change (m)
High : 0.1
Low : -0.1
0
2.5
5 Meters
N
20 cm DoD
10th Oct - 24th Oct
Elevation change (m)
High : 0.1
Low : -0.1
0
2.5
5 Meters
N
2 cm DoD
10th Oct - 24th Oct
Elevation change (m)
High : 0.1
Low : -0.1
0
2.5
5 Meters
N
1 cm DoD
10th Oct - 24th Oct
Elevation change (m)
High : 0.1
Low : -0.1
0
2.5
5 Meters
N
20 cm DoD
24th Oct - 8th Nov
Elevation change (m)
High : 0.1
Low : -0.1
0
2.5
5 Meters
N
2 cm DoD
24th Oct - 8th Nov
Elevation change (m)
High : 0.1
Low : -0.1
0
2.5
5 Meters
N
1 cm DoD
24th Oct - 8th Nov
Elevation change (m)
High : 0.1
Low : -0.1
0
2.5
5 Meters
N

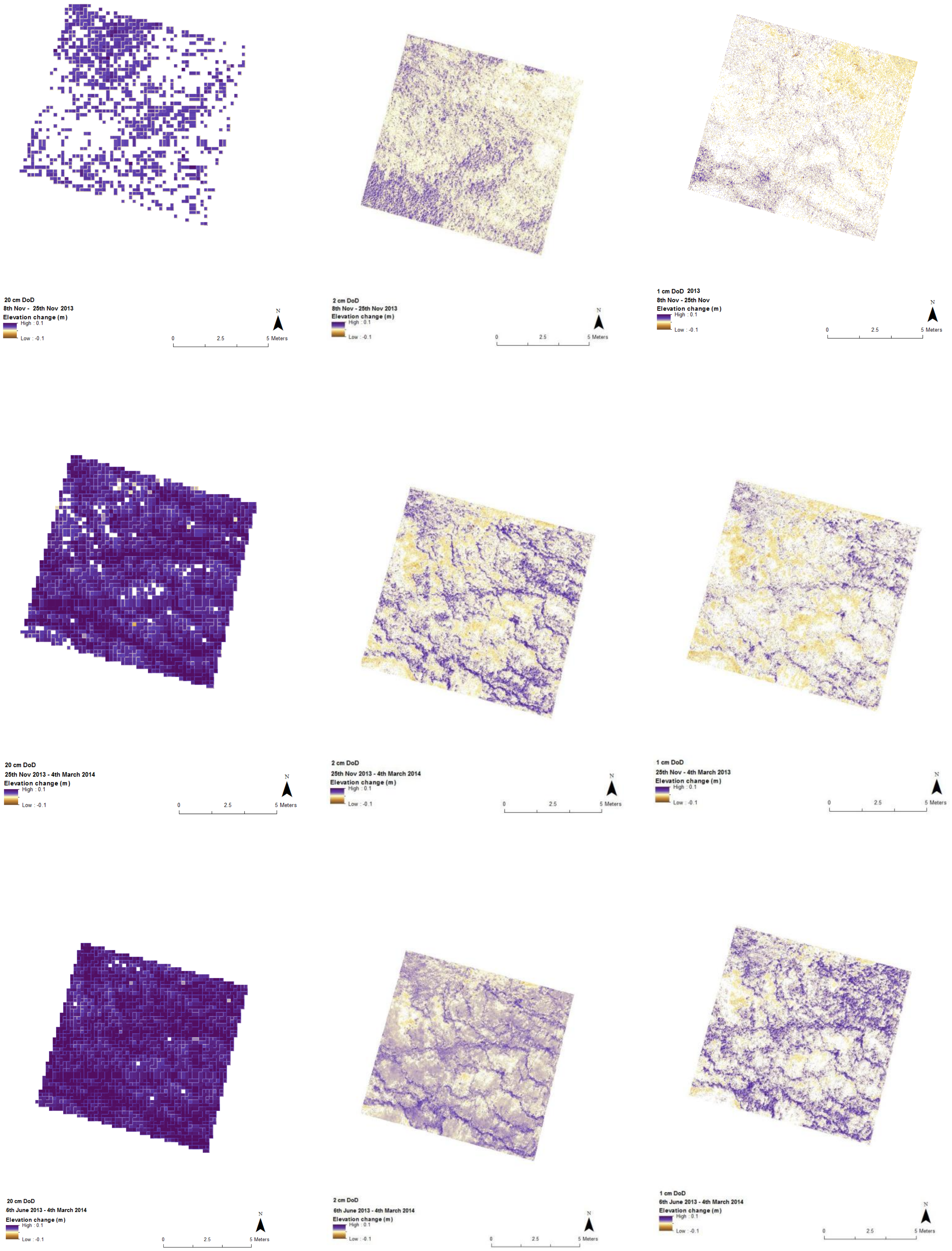


Figure 6.22 – *DoDs of the same area of bare peat created from DEMs gridded at 20 cm, 2 cm and 1 cm. Values within +/- 0.0071 m (highest MSA value) have been considered erroneous and removed from the DoDs.*

In some of the DEMs from which the DoDs were created, the micro-terraces were more pronounced. This could be due to weather conditions between survey dates, or detached vegetation and brash could have collected around the terrace crests. From these DEMs, it is possible to map the movement of the terraces over time by digitising some of the terrace fronts and comparing them with the DEM collected from the previous TLS survey. An example showing the terraces digitised from the DEM created from data captured on 4th March 2014 and compared to terraces recorded in the DEMs created from TLS data collected on 25th November and 8th November 2013 is shown in Figure 6.23. Figure 6.23 shows that over a short period of time (c. 4 months) some clear movement can be observed in the terraces and ‘creep behaviour’ can be seen with the fronts oscillating back and forth. In some areas the terrace fronts have advanced by up to 690 mm, while in others they were found to have retreated by up to 440 mm. Over such a short period of time, a distinct consistent progression is not evident. However, if the movements of the terraces were to be mapped over longer timescales, patterns could be established.

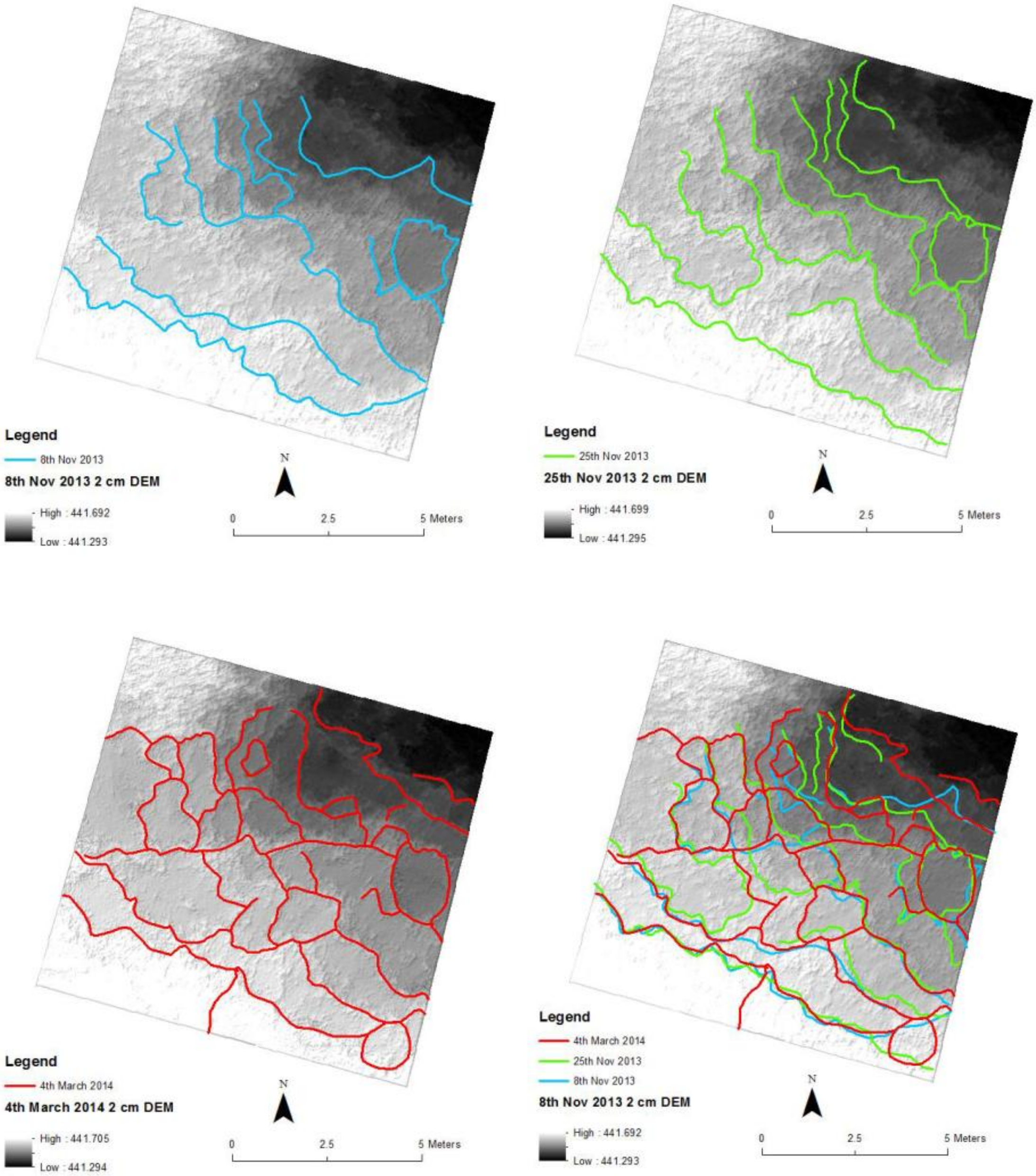


Figure 6.23 – *Peat surface micro-terraces digitised on the DEM from* ***(a)*** *8TH November 2013,* ***(b)*** *25th November 2013 and* ***(c)*** *4th March 2014* ***(d)*** *shows all digitised peat terraces on the DEM created from data captured on 4th March 2014.*

Once the error values had been removed from the DoDs, the changes in surface height were recalculated and compared (Figure 6.24). Figure 6.24 shows that, in all cases, recalculating surface height with the error values removed results in an increased surface height compared to the change values calculated from the DoDs with the error values included. The reason for this is that removing all values that fall within the 0.0071 m threshold increases the mean of the height distribution which could result in a slight over estimate of the change which is occurring. Figure 6.24 also shows that smaller differences exist between the values calculated for the 2 cm and 1 cm DoDs than the 20 cm. This is due to the cell size of the DoDs. As can be seen in Figure 6.22, removing values within the 0.0071 m threshold has the greatest impact on the DoDs at a 20 cm resolution. This is because a cell size of 20 cm is ten times larger than the 2 cm cells, meaning that if the value of the 20 cm cell is within +/- 0.0071 m , the entire 20 cm cell will be removed, which would be covered by 10 cells at a 2 cm resolution, or 20 cells at a 1 cm resolution. Figure 6.24 demonstrates that removing data within the error threshold has the least effect on the DoDs created from DEMs gridded at a 2 cm resolution. To assess whether there was a significant difference between the change values calculated from DoDs containing the error values and those calculated from the DoDs with the error values removed, the two data sets were compared and an R^2 value of 0.89 was calculated. This shows a strong positive correlation between the data sets. To test if this was statistically significant, a spearman's rank correlation co-efficient of value of r_s 0.96 was calculated, suggesting a statistically significant association between the data sets and showing that there is no significant difference between change values calculated from DoDs containing values considered within the error threshold, and change values calculated from the DoDs with these removed.

Subsequent to removing the error values, the DEMs were used to calculate change values where TLS point clouds had been registered in difference projects to establish if this resulted in differences (Appendix A2).

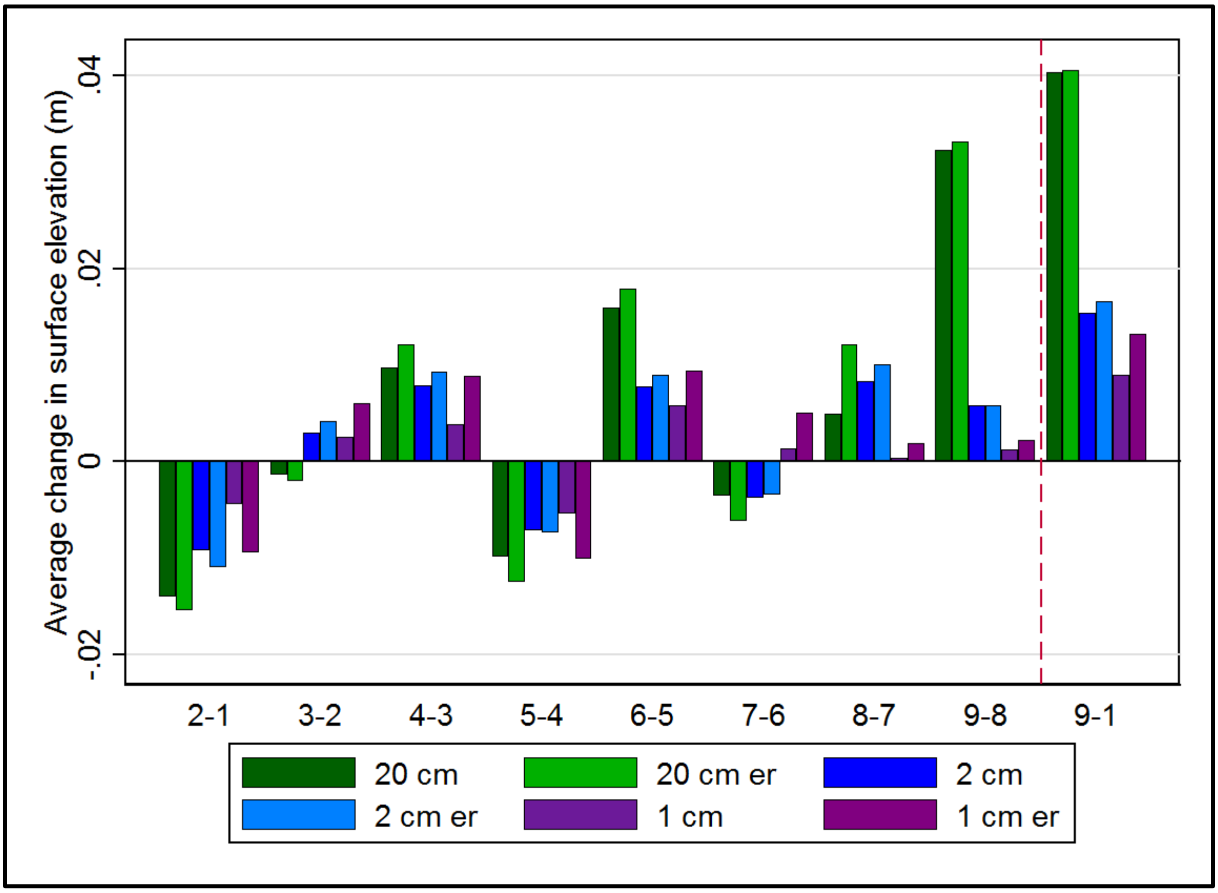


***Figure 6.24 –** A bar chart showing the change in surface elevation calculated using DoDs filtered at three different resolutions and calculated using DoDs with the error values (+/- 0.0071 m) removed (labelled as 'er' on the plot). Data calculated for the entire TLS monitoring period are shown to the right of the red dashed line.*

In this study, data handling became a significant issue due to large file sizes. The unfiltered data gridded at a 1 cm resolution only allowed a small area of the point cloud to be exported for analysis, whilst the data filtered and gridded at the coarsest (20 cm) resolution were not detailed enough to see the formation of micro topographical features on the peat surface. Furthermore, the data indicate, that in the majority of cases, change in surface height calculated from the DoDs created from DEMs at a 20 cm resolution is greater than the values calculated from DoDs created from data at a 2 cm and 1cm resolution. As data filtered at a 20 cm resolution have proven too coarse to identify micro topographical features, it was decided that DoDs at a finer resolution should be used to calculate change detection values. The point density of the point clouds was found to be 0.8 per cm^2, and therefore gridding the data at 1 cm could lead to over-sampling. Due to this, the DoDs created from data at a 2 cm resolution were deemed most appropriate for the change detection and are used in this study to characterise the geomorphic changes in the peat surface. During this study, two different work flows were used for the post processing of the TLS data (Appendix A1, Figure A1). During post processing (Appendix A2), it was found that the different processing work flows resulted in different change results calculated from the DoDs. Therefore, it was decided that data sets which had been registered together prior to filtering would be used for the change detection in this study, as by registering the different resolution point clouds in the same project, it resulted in all data sets having a consistent value for registration error.

6.2.4.3 Comparison of poles and TLS data

To establish whether the changes in surface height calculated using TLS are an accurate representation of changes measured directly on the ground, TLS data were compared with changes in surface height measured from the pole transects. The mean differences in surface height recorded from the three pole transects were calculated for the same epochs as the DoDs from which the TLS height differences were calculated (Table 6.6 and Figure 6.25).

Table 6.6 – *Change in surface height calculated using TLS data and change in surface height recorded from pole transects*

DoD number	Mean change in surface height calculated using TLS gridded at a 0.02m (m)	TLS Error value +/- (m)	Mean change in surface height recorded with pole transects (m)	Pole transects error +/- (m)
9-1	0.018732	0.0071	0.006256	0.0018
9-8	0.002036	0.0071	-0.00882	0.0018
8-7	0.001793	0.0071	0.008692	0.0018
7-6	0.005091	0.0071	0.001692	0.0018
6-5	0.009404	0.0071	0.001538	0.0018
5-4	-0.00993	0.0071	-0.00795	0.0018
4-3	0.008778	0.0071	0.015692	0.0018
3-2	0.006535	0.0071	0.000283	0.0018
2-1	-0.0087	0.0071	-0.00431	0.0018

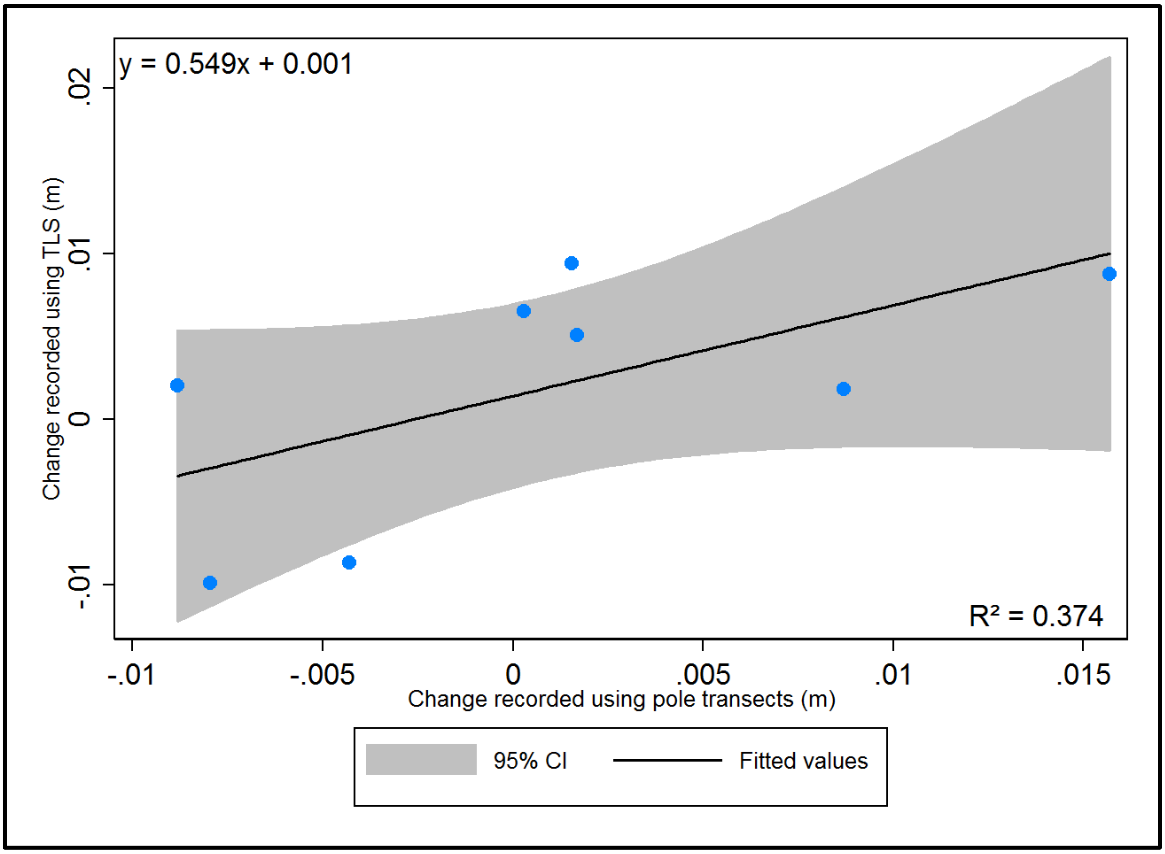


Figure 6.25 – *Average change in surface height calculated using TLS plotted against average change in surface height recorded using pole transects (r^2 0.3737).*

A positive relationship exists between the average changes in surface height calculated using TLS and the average change in surface height measured from the deposition pole transects (r^2 0. 3737), however this would not be considered statistically significant (spearman's rank = r_s 0.5). The regression equation for the graph above indicates that a ratio of 0.5 exists between surface height changes calculated using TLS and surface height changes measured using pole transects. An exact match is not expected as measurements calculated using TLS and values measured in the pole transects cover different specific areas of the peat flats (Figure 6.9) and there is a large amount of spatial variation in erosion and deposition across the site

(Figure 6.9). For a more direct comparison, a strip of the DoD measuring c. 45 m^2 was selected which had a pole transect (the Northern transect, Figure 4.9) located within it. Vegetation on the edge of the bare peat margins was omitted so surface height changes were calculated using eight of the nine poles.

The average change in surface height calculated from TLS for this area was (with the error values of +/-0.0071 m removed) plotted against the average change values recorded from the pole transect (Figure 6.26). This graph shows a strong positive correlation between the two data sets (r^2 0.74) and an r_s value of 0.95 indicates there is a statistically significant relationship between the two data sets, suggesting that TLS is an appropriate method for monitoring changes in surface height of peat. The regression equation for the data shown in Figure 6.26 indicates that there is a ratio of 0.67 between the surface height change data calculated from the TLS and the data directly measured from the pole transects. This is 0.13 larger than the ratio of 0.54 shown between the two datasets in Figure 6.25. There is likely to be a difference in ratios between Figure 6.25 and 6.26 which could have occurred because the values of change in surface height have been calculated from DoDs located in different areas and as previously mentioned, rates of erosion will be highly variable even on local scales. Furthermore, differences could stem from the removal of the values considered to fall within the error values in the TLS DoDs. Differences can also occur, because the pole transects record one point measurement, while the TLS DoD contains many measurements over a larger area. As shown in Section 4.2.3, it is also possible for errors to be made when measuring the poles and this could lead to further discrepancies. The ratio of 0.67 between the measurements from the poles and the measurements from TLS shown in Figure 6.26 could have resulted from the area of the DoD used to calculate the changes in surface height. Although only a small section of the DoD was used to calculate differences in surface height, this inevitably covered a larger area than the single point measurements recorded using the pole transects.

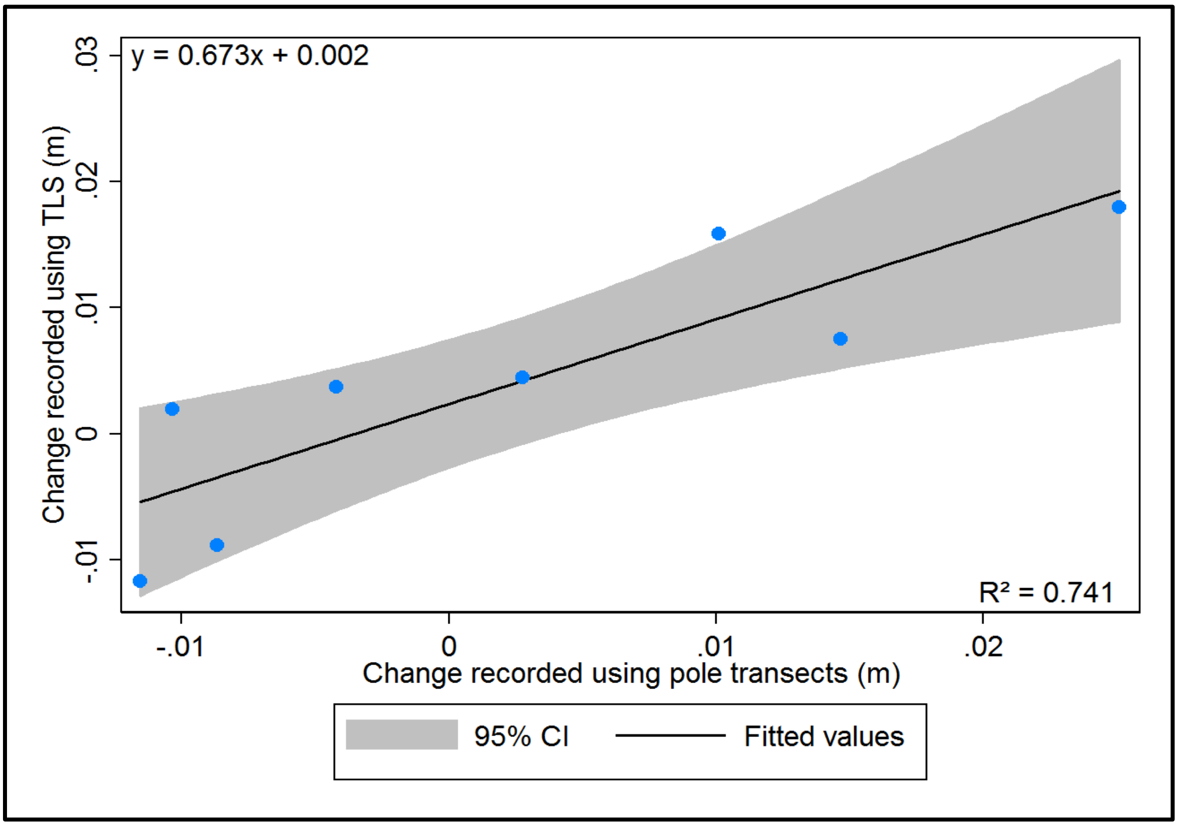


Figure 6.26 – *Average change in surface height calculated using TLS plotted against average change in surface height recorded using pole transects (r^2 0.74).*

Although the poles confirm that the TLS is appropriate for recording changes in the surface height of the peat, both the data from the pole transects and the change in surface height calculated using TLS show an increase in peat surface height has occurred between 6th June 2013 and 4th March 2014.

There are several possible reasons for this. Firstly, a net flux of material across the site could have occurred and this transect could be located within an area of net accumulation (Figure 6.9). Other possible reasons include: mire breathing, peat desiccation resulting in increased surface roughness and frost heave. Off-site wind derived inputs could have potentially contributed to some of the observed increase in surface height, however this is unlikely because beyond the perimeter of Flow Moss, exposures of bare peat are rare and the area of eroded peat within the field site is surrounded by an extensive buffer zone of well-established heather vegetation. The most likely reason for what appears to be a net gain in surface height between June and March is water table levels prior to TLS surveys. As is shown in Chapter 7 (Figure 7.8), the lowest recorded water table values occurred during June (approximately 60 cm below surface), while some of the highest values were recorded preceding the TLS survey in March 2014 (approximately 18 cm below surface). The fluctuating water table heights could result in the peat surface contracting during dryer periods and expanding on wetting up (Price, 2003).

The values calculated from the TLS DEMs would appear to contrast with the data shown in Figure 6.9 which show erosion in the Northern transect. This is because the DoD calculating change for the entire study period has been created from data collected in March 2014 with the data collected from June 2013 subtracted from it. The average change per day value shown in Figure 6.9 is the mean change per day recorded across the entire study period and takes into account variation occurring between the start and end date of monitoring. These values are likely to differ from those shown in Figure 6.9, because as shown in Figure 6.11, during some months (December 2013 and January 2014) there was a large amount of variation in the data collected to monitor the surface height of the bare peat. This would not be reflected in the DoD that simply looks at the surface height at the start and end date. Therefore, values of mean change per day were calculated using the TLS data (Table 6.7).

Table 6.7 – *Average change per day in peat surface elevation calculated using TLS and pole transects*

Epoch	Days between Survey	Change in surface elevation from 96m^2 2 cm TLS DoD (m)	Change per day from TLS (m).	Change recorded using pole transects (m)	Change per day from pole transects (m)
9-1	272	0.018732	0.000069	0.006256	0.000023
9-8	99	0.002036	0.000021	-0.00882	-0.000089
8-7	16	0.001793	0.000112	0.008692	0.000543
7-6	15	0.005091	0.000339	0.001692	0.000113
6-5	14	0.009404	0.000672	0.001538	0.000110
5-4	24	-0.00993	-0.000414	-0.00795	-0.000331
4-3	28	0.008778	0.000314	0.015692	0.000560
3-2	11	0.006535	0.000594	0.000283	0.000026
2-1	61	-0.0087	-0.000143	-0.00431	-0.000071

The average change per day values in Table 6.7 indicate that during the TLS monitoring period, a slight net increse in surface height of 0.000187 m measured using TLS and a net increse in surface height of 0.000108 m calculated using the pole transect has occurred. These values have been established by calculating the average of the change per day value for each epoch (the data shown below the dotted line in Table 6.7). These values of change per day will still differ from those shown in Figure 6.9, as the measurements collected during the TLS surveys (from June 2013 – March 2014) and the measurements collected using the pole transects (from March 2013 – March 2014) covered different periods of time. Subsequently, a value of mean change per day was calculated by subtracting the values of surface height recorded during the first TLS survey (6th June 2013) from the values of surface height recorded during the last TLS survey (4th March 2014) and dividing this by the number of days (272) of covered by the TLS monitoring period. This is labled as 9-1 in Table 6.7. The differences shown between values calculated from the first and last dates of the TLS monitoring period and values calculated from data collected at a higher temporal resolution would indicate that annual monitoring surveys can miss important erosion dynamics occuring within the year, and for a true representation of what is happening to the peat surface, monitoring at a high, typically monthly, temporal resolution is required.

6.2.4.4 Limitations of TLS for peatland change monitoring

The strong positive correlation between the direct measurements and TLS change in surface height results suggest that TLS measurements can provide a method for quantifying change in the surface height of peatlands. However, there are several caveats that need to be considered. One of the key questions arising from this study is whether the small changes in the height of the peatland surfaces lie within the detection limits of the TLS. Although changes in surface elevation (erosion and deposition) are small during the period of measurement the correspondence of direct measurements and TLS data clearly demonstrate that such changes can be captured with these techniques (within the limits of detection). The two methods in combination provide a valuable means of capturing local and spatial variations in patterns of sediment transfer.

In this study, the MSA value (0.0071 m) was used to filter out data so that changes smaller than the error values were removed from the DoD and not summed in the final calculation (Figure 6.22). It is unlikely that all the values that fell below the error threshold were erroneous and it is therefore possible that change values estimated from the DoDs with the threshold values removed were an over estimate. This was, to some extent, mitigated against by calculating the change values both from DoDs containing the points falling within the threshold values and the DoDs with these removed to establish whether there was a significant difference between the two datasets. An r^2 value of 0.89 and a spearman's rank correlation co-efficient of r_s 0.96 was calculated, showing there is no statistically significant difference between the two data sets shown in Figure 6.24, suggesting that removing the error values does not have a significant impact on the datasets.

TLS data were captured from a portable survey tripod and semi-permanent targets. This could lead to an additional error in the data. Although the tripod was carefully positioned over the fixed scan stations and the target locations were fixed, there may have been a slight difference in tripod height or target height during each field survey due to changing surface conditions and small scale earth or vegetation movements. To a degree, the MSA registration in RiScan should overcome issues relating to tiepoint registration, however, to fully mitigate this limitation, fixed scan stations and targets would be required. However, the addition of fixed scan stations and target locations may not be practical for all TLS monitoring applications and could be particularly problematic in peatland environments where stable ground control stations are difficult to locate.

The addition of large permanent objects with a flat surface located within the scan area would help reduce issues of registration error, as when the plane patch filter was run, these would provide surfaces to which the plane patch points could be assigned and then matched during the MSA processing (Appendix A2). In this study, attempts were made to locate as many plane patch points as possible within the fence line, as the fence was known to be a fixed object; however the fence was located on the perimeter of the scanned area limiting the number of fixed points within the area of interest.

DEMs created from data at a 2 cm and 1 cm resolution can be useful in defining some of the micro-topographical features of the peat surface (Figure 6.20). These are especially prevalent in the DoDs for the March dataset (9) – the November dataset (8) and for the DoD showing overall change between June 2013 and March 2014. The formation of features such as peat micro-terraces were recorded after longer periods of time had elapsed and significant weather events had occurred between TLS data capture. Figure 6.23 shows that using the raw DEMs, it is possible to map the movement of the peat terraces overtime.

The change in surface height of the peat correlates with results shown by the pole transects (R^2 =0.74). This suggests that TLS is an appropriate method for quantifying changes in surface height. Both the pole transects and the results from the TLS surveys show that a small net increase (< 2 cm) in peat surface height has occurred at Flow Moss. One possible reason for this is that eroded peat was deposited within the TLS survey area. However, Grayson *et al.* (2012) discussed other factors which could explain observed increases in peat surface height. Firstly, they suggested that the formation of needle ice during cold conditions could result in the peat surface height being elevated. This is only a temporary condition, so TLS data captured when needle ice was present would result in higher surface elevation measurements than when there was no needle ice. Furthermore, increases in peatland elevation could result from fluctuations in water table height, and if the peat wets up after a period of dryness, the surface height could increase. Price (2003) states that this is the results of the collapse of large pore spaces, leading to surface subsistence during dry conditions, and the refilling of these pores (and thus expansion of the peat) once the water table level begins to rise.

6.3 Changes in the spatial distribution of bare peat and surface vegetation cover at Flow Moss

To establish the extent of the vegetation change (re-vegetation) over time, the area of bare peat at Flow Moss was mapped. The fence at the perimeter of the Flow Moss site, the AWS, and the location of the six scan stations were used to ground truth the data and for registration of the images. Three different methods were used to map the extent of the bare peat between three different epochs:

1. Google Earth satellite imagery (50 cm resolution) captured in January 2007 was obtained and the extent of the bare peat digitised (Figure 6.27 **(a)**).

2. Bare peat areas were digitised from UAV data at a 3.7 cm resolution collected in April 2011 (Baynes, 2012) and geo-referenced using differential GPS data (Figure 6.27 **(b)**)

3. In April 2014, a differential GPS survey was undertaken at Flow Moss and areas of bare peat and vegetated peat haggs were mapped (6.27 **(c)**).

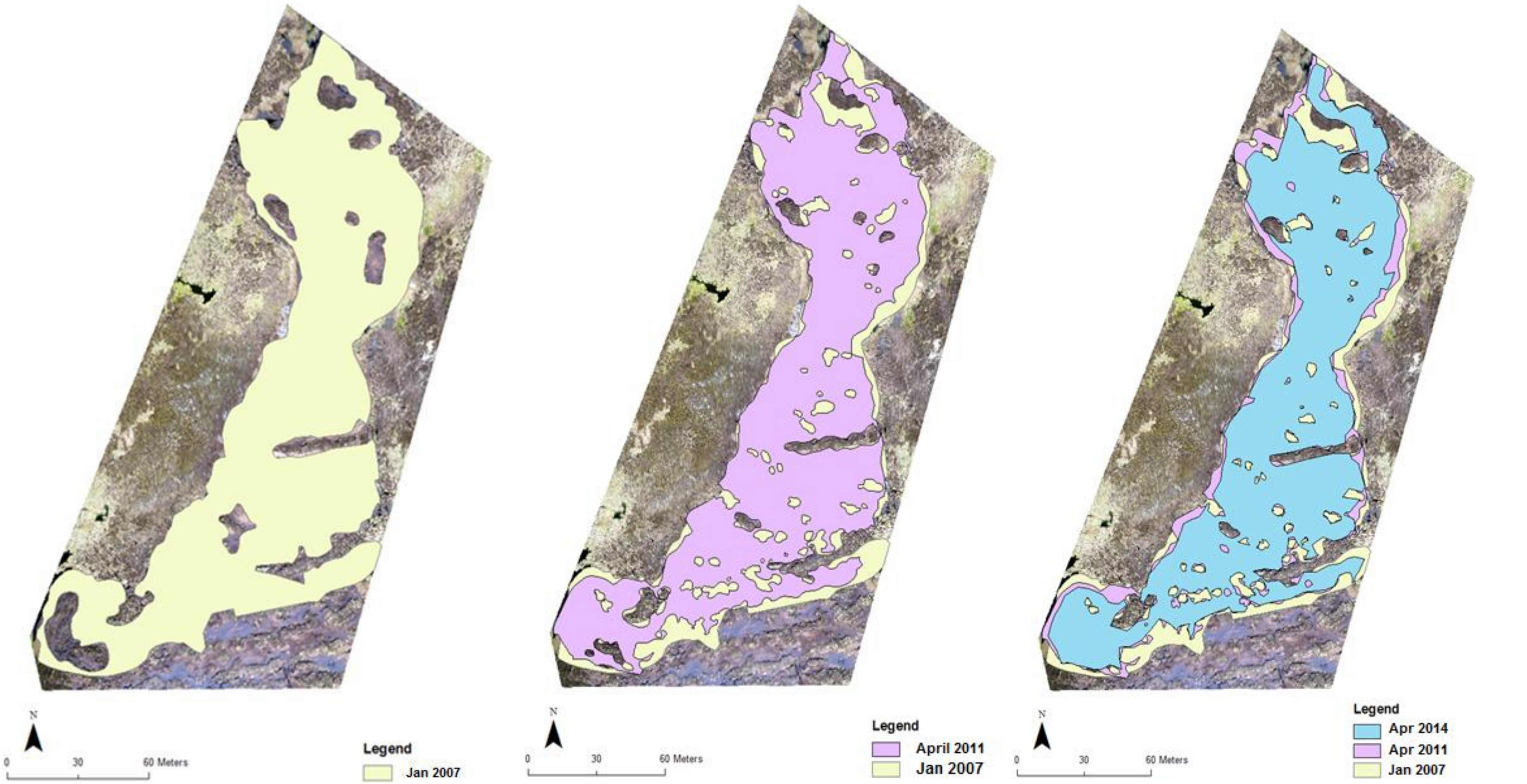


Figure 6.27 – *The bare peat area digitised using three different methods over three different epochs.* ***(a)*** *2007 Google Earth satellite data, (****b)*** *2011 UAV data and* ***(c)*** *2014 GPS data. The peat maps show that the bare peat area reduced in size by 2966 m*^{2} *or 21.64% between January 2007 and April 2014.*

The area of bare peat, summarised in Table 6.8, from the image captured in January 2007 was 13754 m^2, while the area of bare peat from the April 2011 UAV image was 11754 m^2. This shows a reduction of 1999 $m^{2,}$ a 14% reduction of the bare peat area that was present in January 2007. The area of the bare peat measured during the GPS survey in April 2014 was 10777 m^2. This is a further reduction of 997 m^2 or 12% between April 2011 and April 2014. Overall between January 2007 and April 2014 the area of bare peat reduced by 2966 m^2, which is a reduction of 22% of the bare peat area in 2007. The data show that vegetation is laterally encroaching from around the margins of the bare peat. This would suggest that the restoration methods attempting to re-establish vegetation at Flow Moss have been effective. The widest extent of bare peat (located at the northern end of the site) was measured and the change in maximum width calculated (Table 6.8). It was found that the widest extent had reduced by 3.28 m from 83.87 m to 80.59 m between January 2007 and April 2011 and further reduced by 9.92 m to 70.67 m by April 2014.

Table 6.8 – *Summary table of the changes in bare peat area at Flow Moss*

Date	**Area of bare Peat (m^2)**	**% change since 2007**	**Max width (m)**	**Max width change (m)**
January 2007	13754	0	83.87	
April 2011	11754	15	80.59	3.28
April 2014	10777	21.6	70.67	9.92

Although the data show a clear reduction in the amount of bare peat at Flow Moss, there are limitations to the data used. Firstly, Google Earth imagery was only available for January 2007, while the other two sources used data collected during April. This could have resulted in some seasonal change in vegetation, however, both the dGPS and UAV data were collected during early spring (April) which is before vegetation would begin to establish and therefore the seasonal change should be minimal. Furthermore, since the majority of the vegetation change has occurred at the margins of the bare peat, it is unlikely that all the change recorded is due to seasonality and more likely that vegetation has begun to re-establish. A further limitation of the data used stems from the spatial resolution of the images. The UAV image was of a higher resolution (3.7 cm) than the Google Earth satellite imagery (50 cm resolution), This could account for some of the differences in the vegetation cover maps, as small peat haggs were less distinguishable in the coarser resolution Google Earth imagery. The most accurate method was the dGPS survey as data were collected from the field and the error value was < +/- 0.02 m. However, ground survey is also the least practical for collecting data over a large spatial scale and there is a trade-off between data resolution and the spatial extent over which the data can be rapidly collected.

6.4 Surface results chapter summary

The results presented in this chapter provide an insight into the surface changes which have occurred during monitoring at Flow Moss.

Hagg and peat flat erosion:

- Erosion pins suggest that the peat hagg faces at Flow Moss are relatively stable with all measurements of change in pin exposure falling between +/- 4 mm (average +/- 2 mm). Although locally variable

(+/- 10 mm), the pole transects indicate that the bare peat flats are relatively stable, with the mean change for the study period being 0.059 mm and all changes in surface height falling within +/- 10mm.

Surface height monitoring and Terrestrial Laser Scanning

- There is a strong positive relationship between surface height data collected using TLS and data collected using pole transects. This suggests TLS offers a potential method for monitoring erosion of peatlands over a larger spatial scale than point measurements provided a strict ground control protocol is observed.
- There are limitations to the methods which need to be taken into consideration when implementing a TLS survey. Both the TLS data and pole transect data suggest that between June 2013 and March 2014, an increase in peat surface height was recorded at Flow Moss. Antecedent environmental conditions will be analysed alongside surface change data in the next chapter in an attempt to explain this further.

Wind Flux

- The data collected from the sediment traps during this study indicate that a seasonal pattern exists in wind transported peat. Higher yields of sediment were collected during winter. In Chapter 7, a comparison will be made between windward and leeward facing traps to identify if wind direction impacts on sediment movement.

Channel loss

- Sack trap data suggest that very little sediment is lost from the site through the fluvial system. The yields of sediment collected from the sack traps were found to be 27 times less than yields collected from the wind flux samplers. During monitoring for this study, there was found to be very little change in the yields of sediment collected from the channel system.

Bare peat and re-vegetation

- The mapping of the changes in the extent of the bare peat which has occurred during the last 7 years shows that vegetation is beginning to laterally encroach from the margins of the bare peat, thus reducing the bare peat area and suggesting that re-vegetation attempts at Flow Moss have had some limited success A reduction of 12% of the bare peat area has occurred since restoration began in 2010.

In Chapter 8, the main findings from this chapter will be combined and used to develop a sediment budget for Flow Moss during the monitoring period.

7. Environmental variability and drivers of erosion

This chapter discusses the relationship between local environmental conditions and erosion processes at Flow Moss. Environmental conditions will impact on erosion processes, as increases in wind and rain will enhance particle detachment (Chapter 2, Section 3). Data collected using the AWS at Flow Moss (Chapter 4, Section 1) from 1st March 2013 until 10th March 2014 are discussed. The chapter describes the environmental conditions recorded during monitoring and combines these with geomorphological surface change data (Chapter 6, Section 1) to provide a basis for establishing which environmental factors are the key drivers of erosion at Flow Moss. Figure 7.1 summarises the main periods of sediment movement and identifies that the greatest yields of peat eroded by wind were recorded in December 2013 and the highest yields from the sack traps were collected in May 2013. The average change per day in surface elevation recorded using erosion pins and pole transects has been plotted and shows the highest levels of erosion of the peat hagg faces occurred during April and June, this correlates with the results from the pole transects which show the highest change per day value for the erosion of the bare peat flats occurred in April and June. Both the erosion pins and pole transects recorded the highest levels of deposition during September.

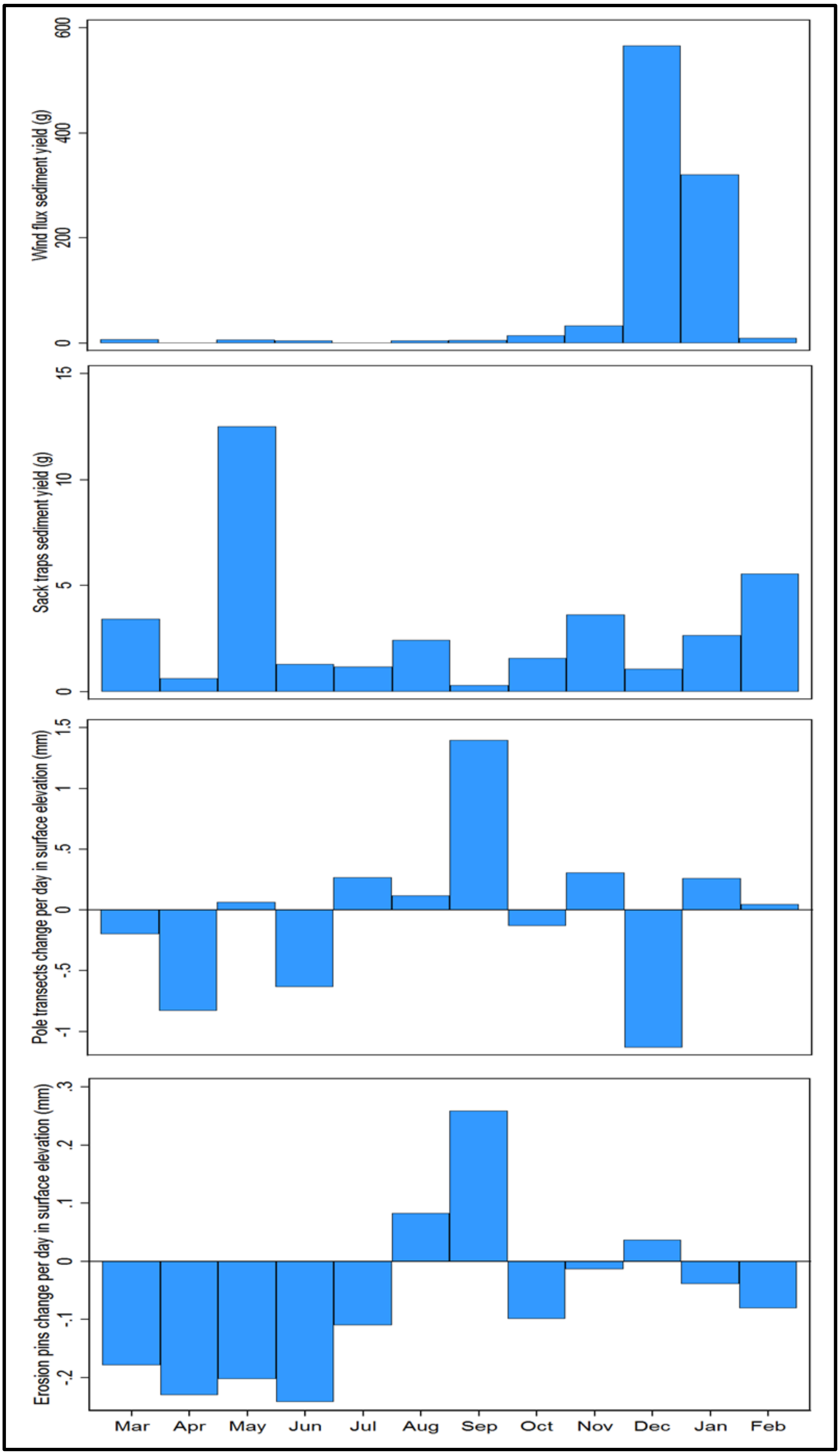


Figure 7.1 – *A diagram summarising sediment movement occurring during each month of this study*

7.1 Climate conditions

There are several environmental variables which can impact on erosion processes. Understanding the factors driving erosion is important so information can be used to design restoration methods tailored to suit the specific peatland (Parry *et al.,* 2014).

Data were collected quasi-continuously (at 30 minute intervals). In an attempt to create a full record, short periods of missing data have been interpolated with data obtained from the UK Environmental Change Network weather station located at Moor House (550 m.a.s.l.), which is located approximately 20 km to the south of Flow Moss. Data have been concatenated to produce hourly values of rainfall, temperature, wind speed, wind direction and water table height.

7.1.1 Rainfall

The hourly rainfall record for Flow Moss between 1st March 2013 and 10th March 2014 is shown in Figure 7.2. The total rainfall between 1st March 2013 and 10th March 2014 was recorded as 1000.2 mm with the average daily rainfall being 2.66 mm. Data from Moor House indicate that between 1931 and 2000, the average annual precipitation was 1982 mm (Holden and Adamson, 2001) which is 982 mm higher than Flow Moss. As Flow Moss is approximately 100 m lower in altitude than Moor House and there is a strong decreasing rainfall gradient away from the main Pennine ridge, it is expected that precipitation will be considerably less. For comparison, the estimated annual precipitation for Alston (11 km southeast) is 898 mm (1981-2010 average, UK Met Office). Data collected by Baynes (2012) recorded a rainfall amount of 414 mm over a 200 day period. This was extrapolated to provide an estimate of 756 mm annual precipitation. This is less than the amount recorded during this study, although Baynes (2012) stated that 2010 was a particularly dry year.

The rainfall data plotted in Figure 7.2, show 11 significant periods where rainfall intensity exceeded 5 mm hr^{-1}, which was considered to be the threshold level above which rainfall events were classified as high intensity. A previous study by Foulds and Warburton (2007) found that maximum peat flux rates were recorded in association with rainfall intensities typically of 4-5 mm hr^{-1}. Baynes (2012) selected 5 mm hr^{-1} as the threshold value for rainfall events, so for consistency with the above two studies, a threshold rainfall value of 5 mm hr^{-1} was also selected for this study. The rainfall data are re-plotted by month (Figure 7.3) to establish during which month the highest number of high intensity rainfall events occurred. Figure 7.3 identifies that the greatest number of periods where rainfall exceed 5 mm hr^{-1} occurred in July, September and December, with the greatest number of these events (5) occurring in December 2013.

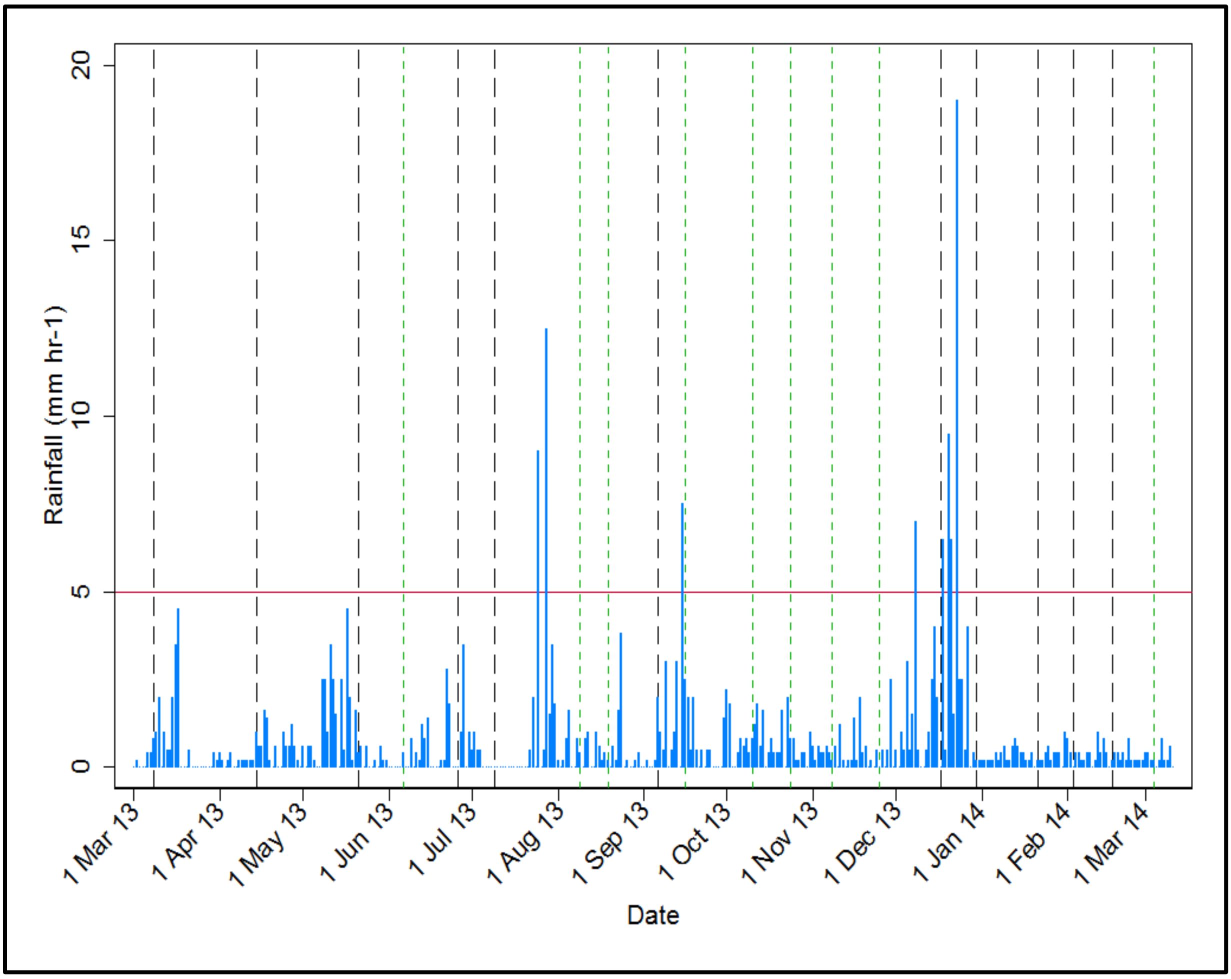


Figure 7.2 – *Hourly rainfall recorded at Flow Moss between 1st March 2013 and 10th March 2014. The black dashed lines represent dates when sediment collection from the traps and measurements of erosion pins were taken. The green dashed lines indicate the dates where TLS survey occurred alongside regular monitoring. The red line shows the rainfall intensity threshold of 5mm hr^{-1}.*

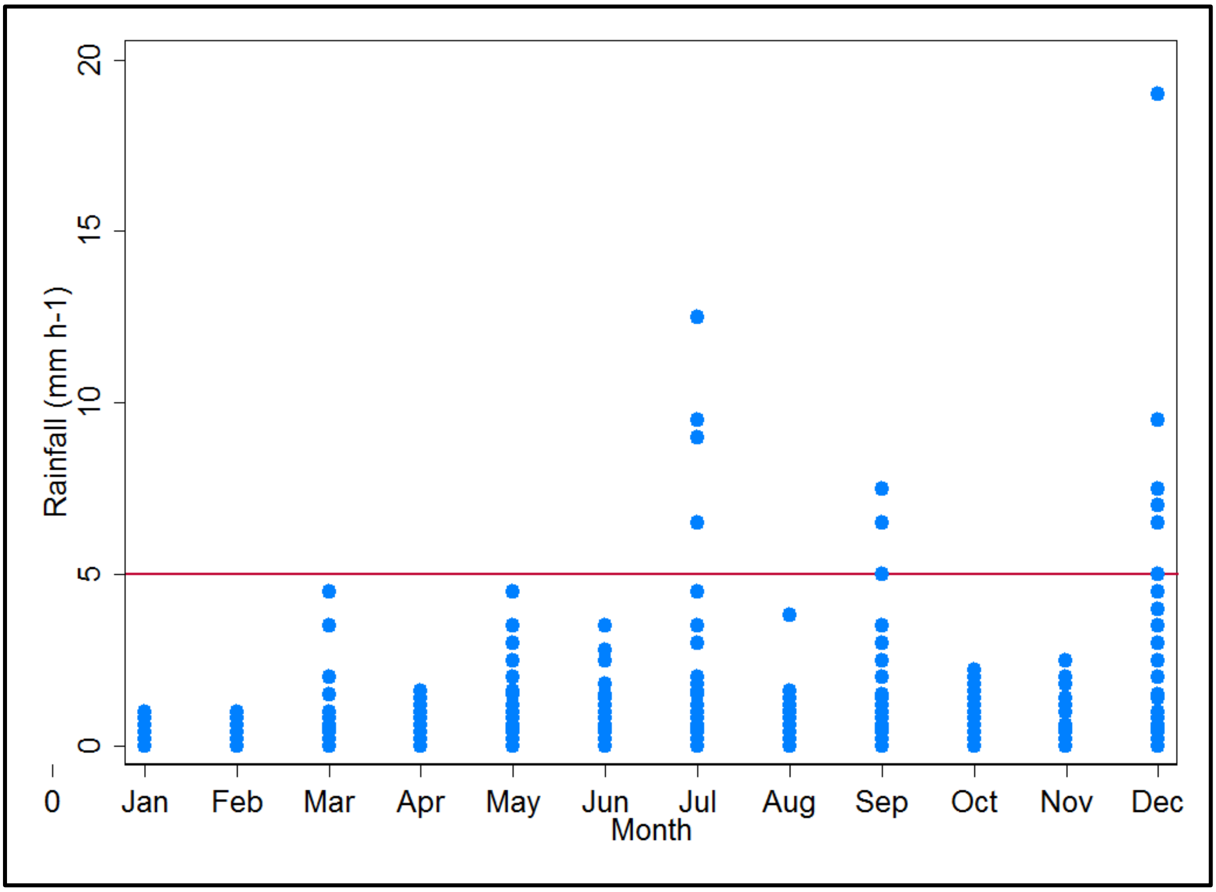


Figure 7.3 – *Hourly rainfall plotted by Month to identify during which months the most events exceeding 5mm hr^{-1} occurred.*

7.1.2 Temperature

The Flow Moss temperature record (Figure 7.4) ranged from a minimum of -6.60°C to a maximum of 23.32°C and shows a strong cyclicity in the annual temperature range. The mean daily temperature during the study period was 6.49°C (Table 7.1) which is 4.69°C warmer than the mean daily temperature of 1.8°C recorded by Baynes (2012). The difference in mean temperature is predominantly due to the different sampling periods over which data collection occurred. Therefore, the mean temperature recorded for this study between 19th November to 25th March and 8th April to 14th April was calculated to provide a mean value comparable to the one recorded by Baynes (2012). This value was 2.54°C, which is only 0.74°C higher than the mean temperature recorded by Baynes (2012). The average annual temperature at Moor House between 1931 and 2000 was 5.3°C, which is 1.19°C lower than the average annual temperature recorded at Flow Moss during the present study. Due to the higher altitude of Moor House, it is likely that warmer temperatures will be observed at Flow Moss (lapse rate 0.65˚C / 100 m).

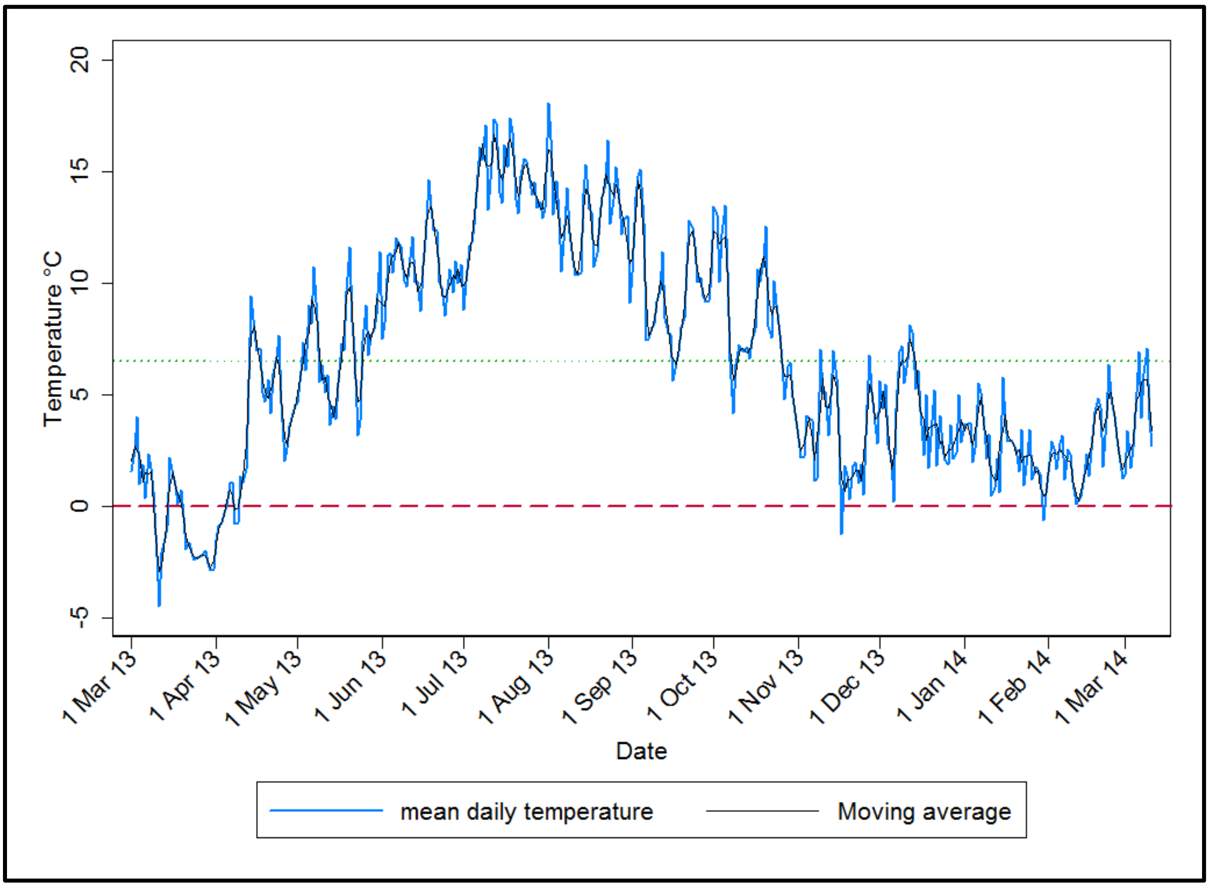


Figure 7.4 – *The record of mean daily temperature recorded at Flow Moss. A daily running average has been fitted to the data set. The green dotted line indicates the mean annual temperature of 6.49°C, while the red line highlights 0°C.*

The number of frost days is an important factor to take into account when examining drivers of erosion. Freeze-thaw activity will result in a higher availability of sediment for transfer. To identify the number of times the peat may have been susceptible to frost disturbance, the temperature record was used to calculate the number of frost days (a frost day is defined as a period of 24 hours in which the mean temperature is ≤ 0°C). The hourly temperature record shows that the number of hourly values where negative air temperatures were recorded was 812. To calculate the number of frost days, the mean daily temperature was calculated and times when this value dropped below 0°C considered a frost day. There were 25 frost days recorded during this study, which fell in March, April, November and December 2013 and January 2014.

The AWS used during this study was not able to directly measure the number of days when lying snow was present on the ground. Therefore, to obtain this information, the time-lapse imagery was examined and the number of days where snow was visible were counted. From the time lapse photos, it was established that lying snow was present for 53 days of this study (Figure 7.5). Figure 7.5 shows that the month where the greatest number of snow days was observed was March 2013, snow was present until April 2013 and then no further snow recorded until November 2013.

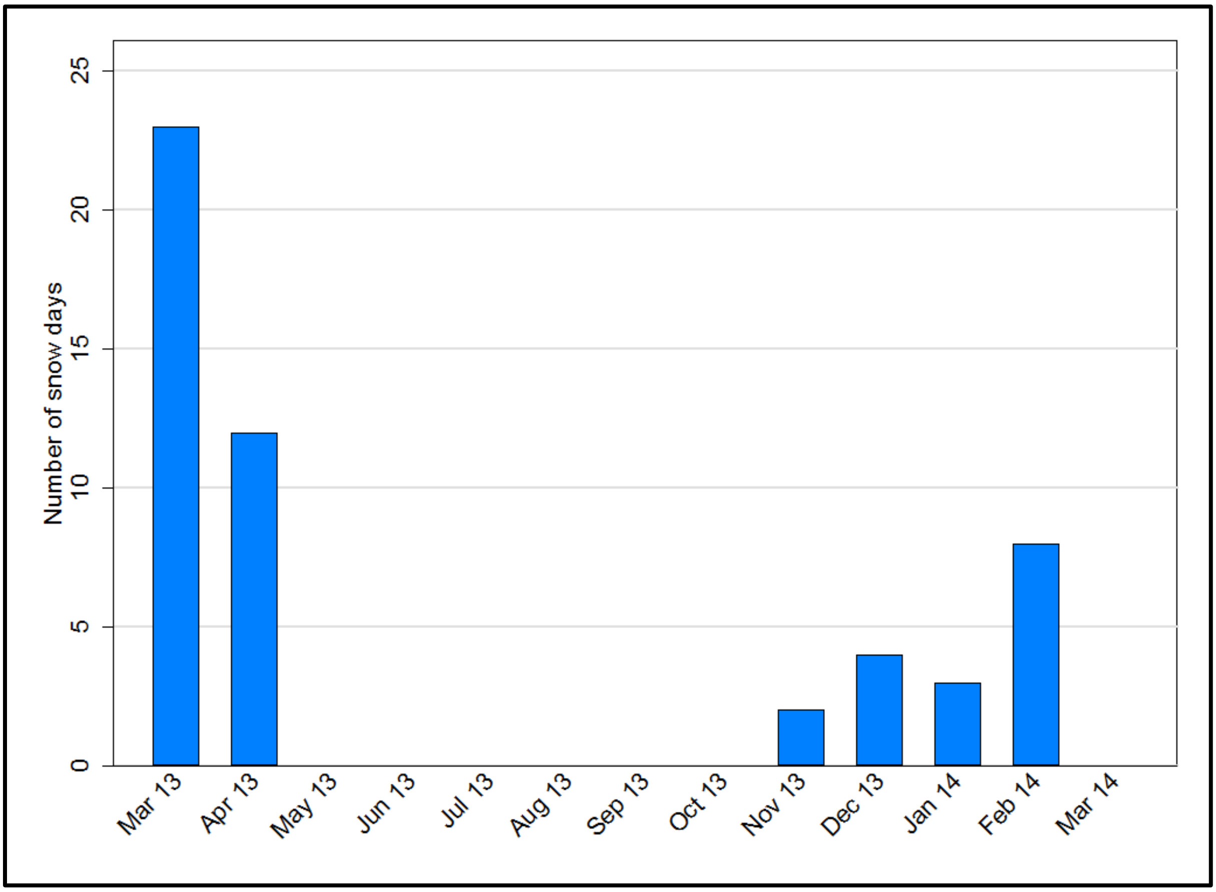


Figure 7.5 – *The number of snow days counted from the time lapse imagery which occurred between 1st March 2013 and 4th March 2014.*

7.1.3 Wind speed and direction

In addition to rainfall, wind-speed and direction have the potential to impact on erosion processes, as higher wind speeds are more likely to entrain and transport peat detached by rainfall. The AWS data were used to identify periods where high wind speeds were recorded (Figure 7.6).

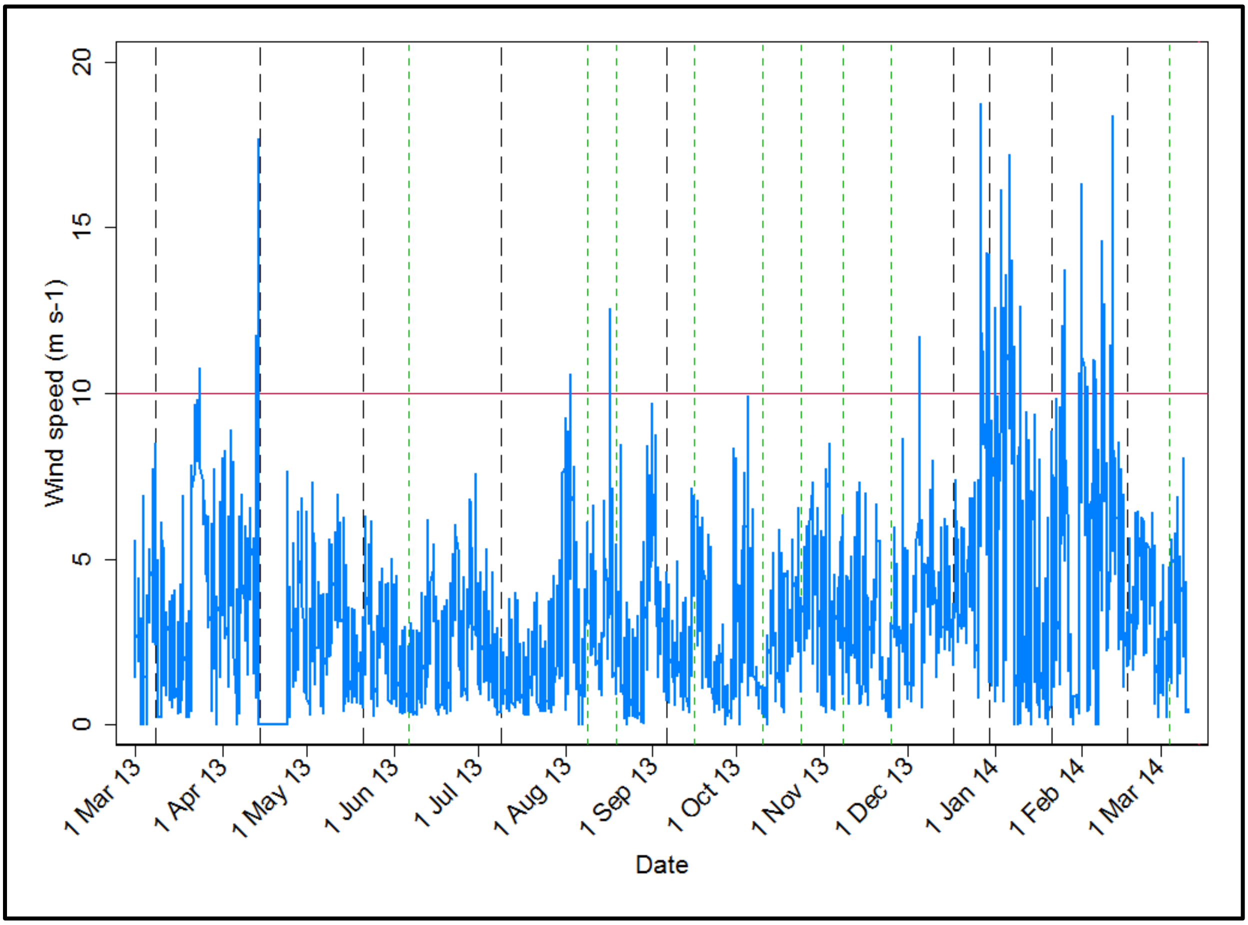


Figure 7.6 – *Hourly wind speed recorded between 1st March 2013 and 4th March 2014. The black dashed lines represent dates of normal monitoring including sediment collection from the traps and measurements of erosion pins, while the green dashed lines indicate the dates where TLS survey occurred alongside normal monitoring. The red line shows wind speed greater than 10 m s^{-1}. During the period of monitoring there is missing data for the wind speed record from 15th April – 25th April 2013.*

High wind speed events were categorised as times where the wind speed exceeded 10 m s^{-1}. Warburton (2003) suggested that the main events which transport peat on an upland peat flat in the North Pennines were events that exceeded a threshold value of friction velocity of 1 m s^{-1} which approximates to a wind speed value measured by the AWS of 10 m s^{-1}. Furthermore, only 3% of wind speeds recorded during this study were found to be above this figure, suggesting that this value is a true representation of a high wind speed event in comparison to the majority of the data. The recorded hourly wind speed values have been plotted by month in Figure 7.7, identifying 255 times in the hourly record where wind speed exceeded 10 m s^{-1}.

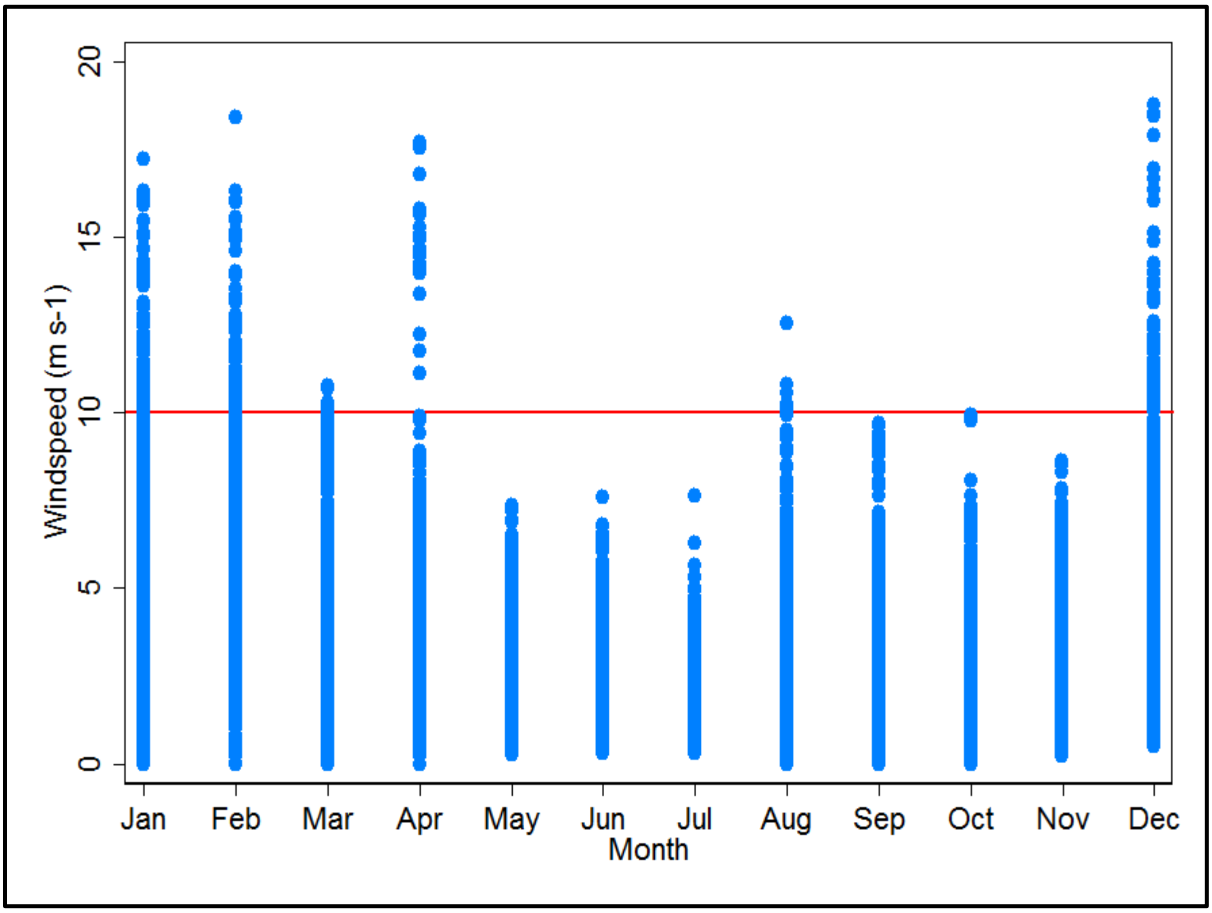


Figure 7.7 – *Hourly wind speed plotted by Month to identify during which months the most events exceeding 10 m s^{-1} occurred.*

Although both wind speed and rainfall can influence erosion events, a combination of heavy rainfall concurrent with high wind speeds will result in greater yields of sediment due to the process wind-splash erosion (Warburton, 2003). Therefore, periods where heavy rainfall coincided with high wind speeds were identified as it would be expected that these conditions would precede times when the greatest amounts of sediment were recorded in the wind flux samplers. Figure 7.8 shows the wind speed series plotted alongside rainfall. From this Figure it is possible to identify three potential periods of exacerbated erosion where high rainfall intensities (≥5mm hr^{-1}) coincide with high wind-speeds (≥10 m s^{-1}). Figure 7.8 shows that at the end of December 2013, there was an extended period of high rainfall and wind speeds which occurred over several days.

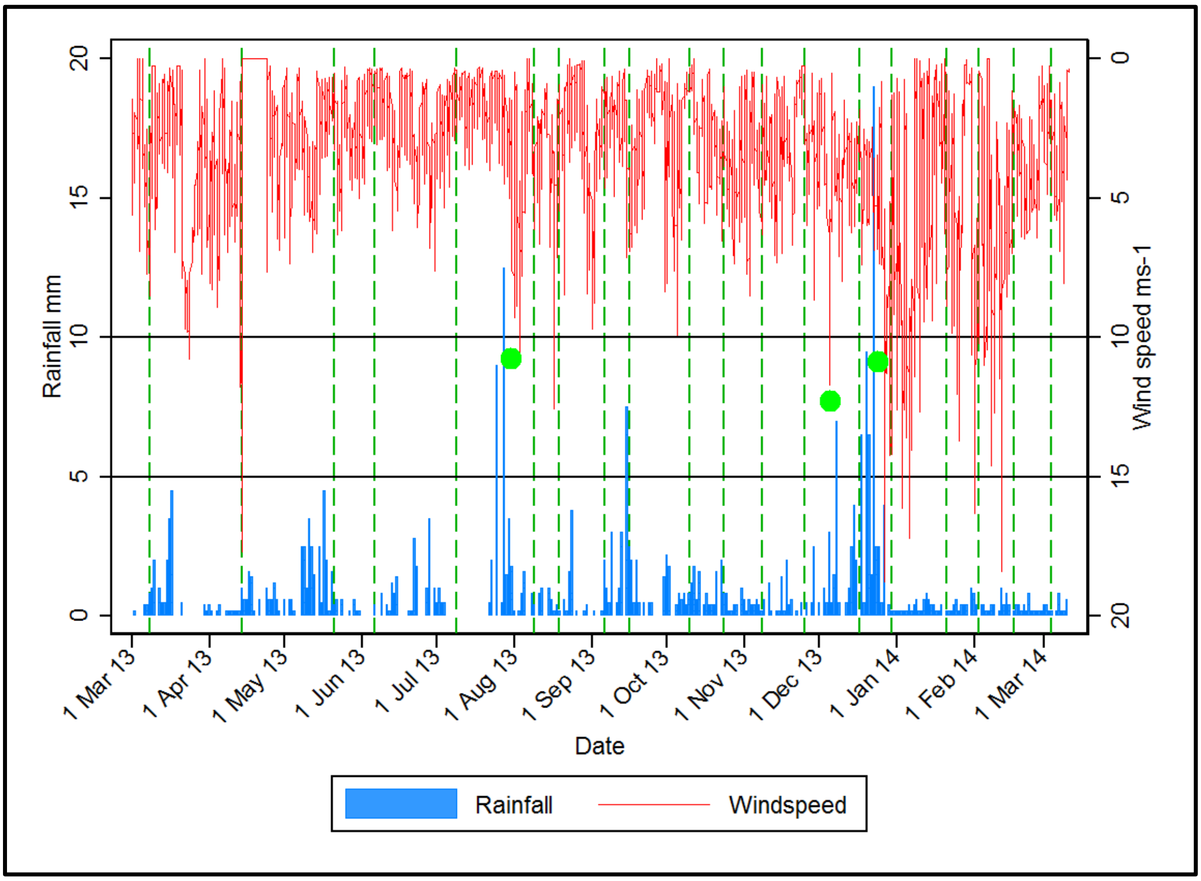


Figure 7.8 – *The hourly record of wind speed and rainfall for Flow Moss. The black lines represent the rainfall and wind speed thresholds. From this Figure three periods of high rainfall and wind speed occurring concurrently and between field days can be identified (represented by the green dots).*

In addition to wind speed and rainfall, Warburton (2003) noted that wind direction is an important control on erosion processes of bare peat. This is due to the erosive energy being dominated by the prevailing wind direction.

Figure 7.9 shows the distribution of wind by direction during the entire monitoring period (8/3/13-4/3/14). The mean wind direction recorded during this study was 219°.

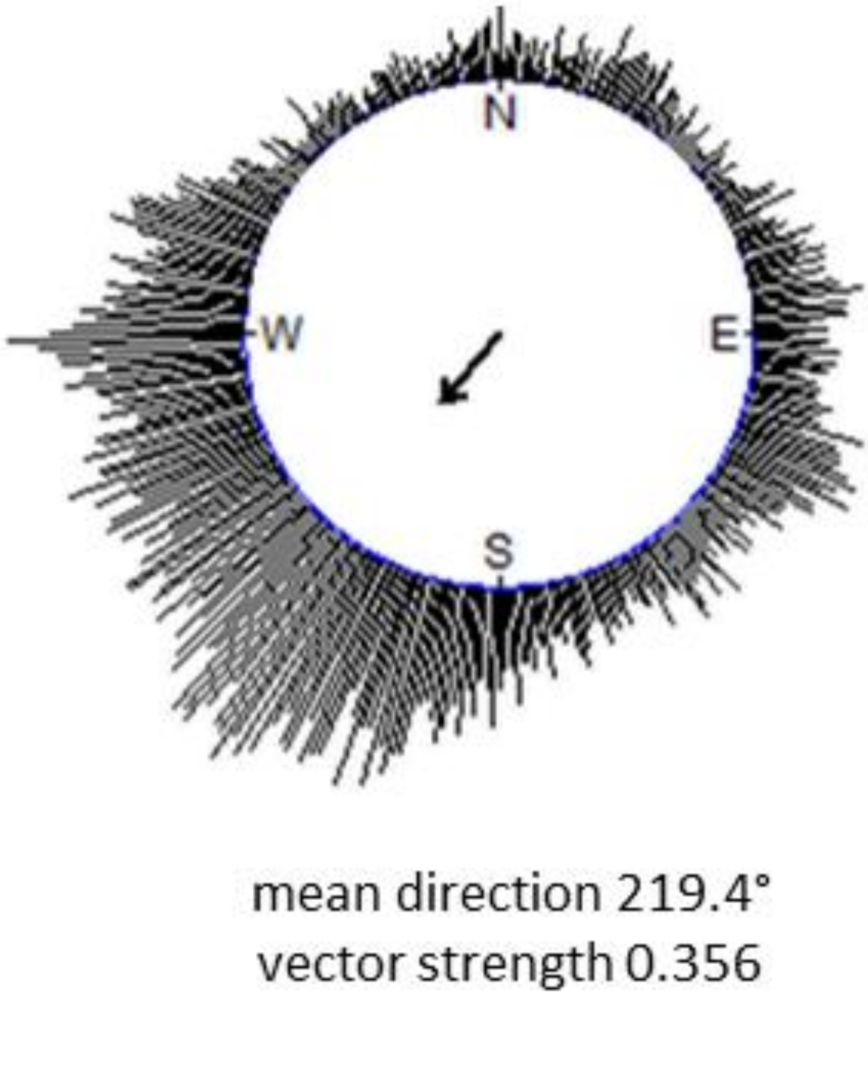


Figure 7.9 – *The distribution of wind direction recorded between 1st March 2013 and 11th March 2014. Prevailing wind is from the southwest.*

The prevailing wind direction is from the southwest with the dominant wind direction varying between 190 and 270º. Winds from other directions are recorded less frequently, with there being some local winds from the southeast. The location and topography surrounding Flow Moss results in the site being exposed to wind prevailing from the North, South and West, however, to the East there is a ridge that is approximately 10 m high and offers some protection to the site from wind from that direction.

7.1.4 Variations in water table height

An automatic pressure transducer recorded measurements of water table height every 15 minutes during the monitoring period. These were concatenated to produce hourly values of mean water table depth. A time series showing hourly recordings of the water table depth is shown in Figure 7.10. Due to temporary faults with the data logger, there are periods of missing data within the water table record. These occur between 1200 on 8th March and 0500 on 29th March 2013, 0900 on 6th May 2013 and 0400 on 18th May 2013, 0900 on 25th June and 1200 on 30th July 2013, 1400 on 6th September and 0600 on 26th September 2013 and 1200 on 23rd November and 0600 on 27th December 2013. In total there were 2904 hourly water table values missing from the 9008 hour record.

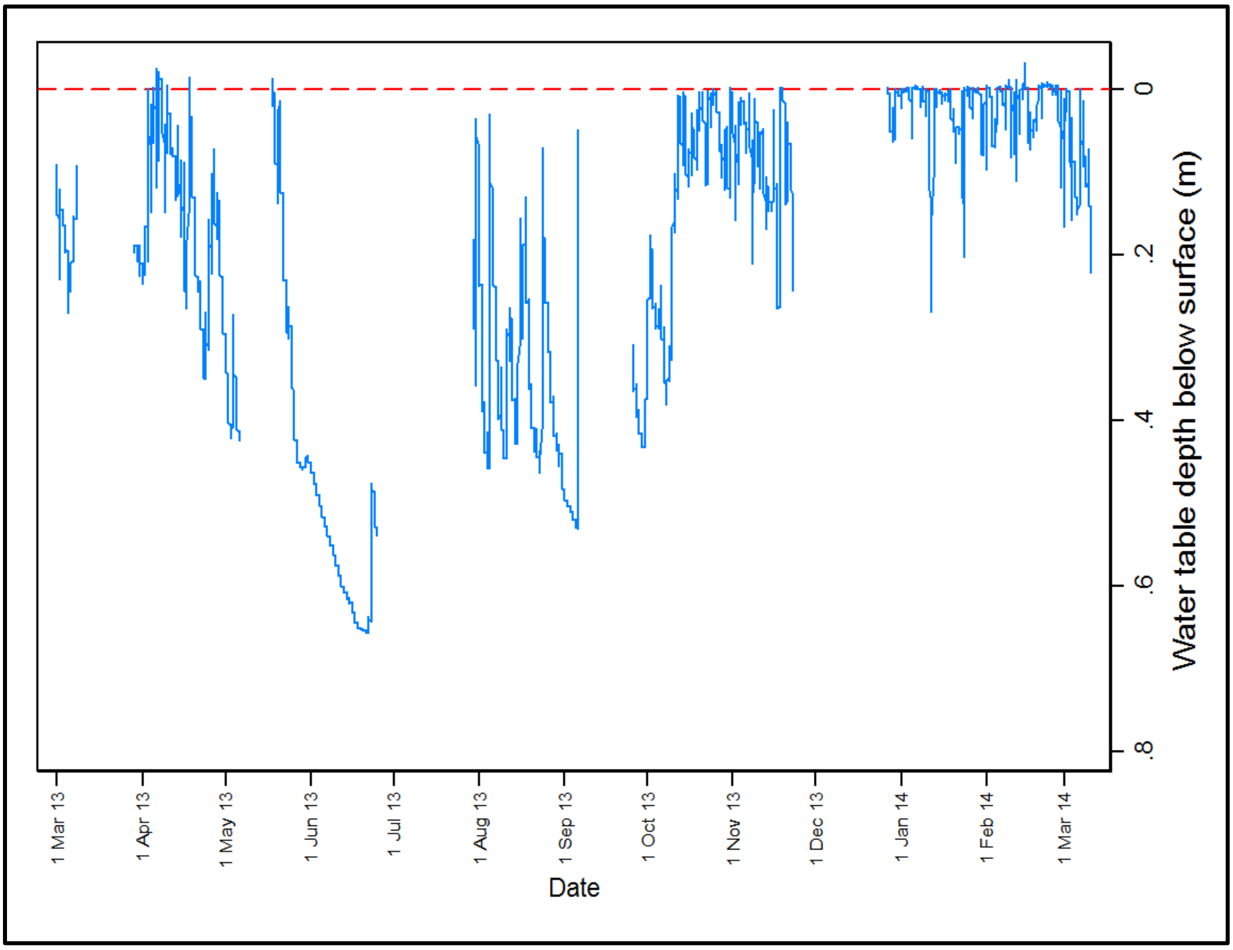


Figure 7.10 – *Water table depths below the peat surface. The red line indicates the height of the peat surface.*

Figure 7.10 demonstrates that for the majority of the year, the water table depth was greater than 0.6m below the peat surface with measurements dropping below this during the summer months. The greatest drop in water table height was experienced in June. During the winter months (November – March) the water table is maintained above 0.8 m. The data show that 5.4 % of the recorded values suggest that the water table height was above the surface of the peat. This is likely to have resulted from recordings when water ponding on the peat surface occurred.

To establish the relationship between recorded rainfall and the depth of the water table below the peat surface, the hourly recordings of water table depth were plotted with hourly recordings of rainfall (Figure 7.11).

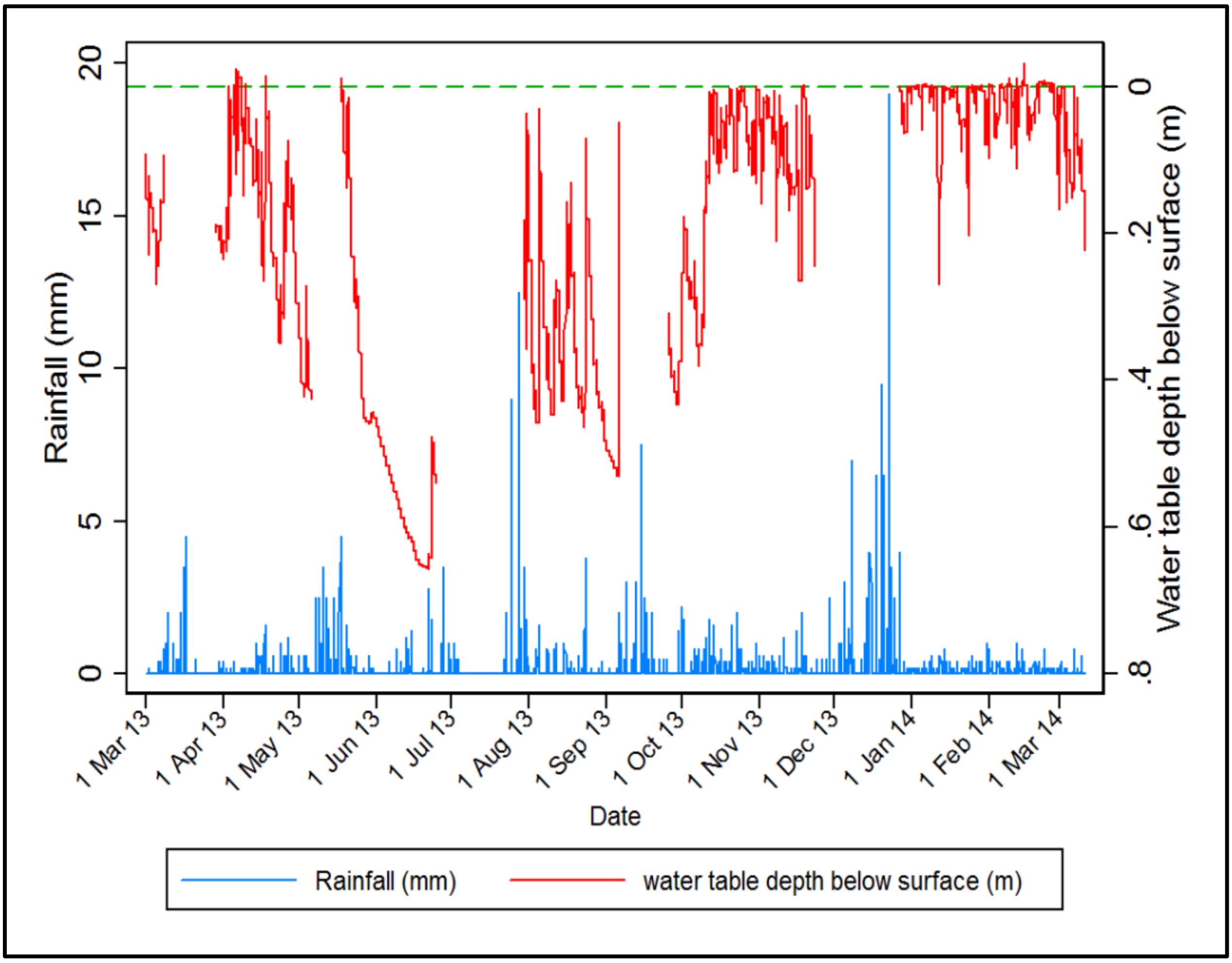


Figure 7.11 – *The Flow Moss hourly water table depth and rainfall record.*

When comparing the water table fluctuations and rainfall amounts, in some instances, a rapid response in water table height can be observed following high rainfall events. This suggests rapid rewetting following rainfall events, but slow drying. Decreases in water table depth are shown during the summer months where recorded temperatures were higher. The data demonstrate that rainfall amount has an impact on the water table level. The rapid response of the water table shown in the data is typical of a 'flashy' regime (Holden and Burt, 2003), which Holden et al., (2004) state would be expected in many upland peat catchments. However, Daniels *et al.,* (2008) state that antecedent water table level is linked to storm flow runoff characteristics. One of the limitation of this study is that water table data were collected only from within the bare peat area. The water table behaviour in vegetated areas could be very different to the bare peat and collecting measurements from both vegetated and bare peat areas for comparison could have overcome this limitation.

7.1.5 Summary of weather conditions

Overall, the results recorded relating to environmental conditions during this study indicate that weather conditions experienced at Flow Moss are fairly typical of a UK upland environment with very cool temperatures (as low as -6.6°C) during the winter alongside periods of heavy rainfall. Table 7.1 compares environmental conditions recorded by Baynes (2012) and environmental conditions recorded during this study. The value shown for the mean temperature for the present study has been calculated using the same time period as Baynes (2012).

Table 7.1 – *Comparison of environmental conditions recorded during the study by Baynes (2012) and this study. n.d. = no data.*

Environmental condition	Baynes (2012)	Present study
Total Rainfall (mm)	756	1000
Average temperature (°C)	1.8	2.5
Number of Snow days	n.d.	53
Mean wind direction (°)	224	219

Figure 7.12 provides a summary of the weather conditions recorded during the study period. This clearly shows that the total rainfall amounts recorded during December 2013 were far higher than those recorded during any other month. Furthermore, the three months where the highest values of mean wind speed were recorded were December 2013, January 2014 and February 2014. As displayed in Figure 7.1, the largest recorded yields of sediment collected from the wind flux samplers were recorded in December 2013 and January 2014, indicating that wind and rainfall may impact sediment mobilised by wind. There were three periods of time when rainfall events of $\geq$5mm hr^{-1} coincided with wind speeds of $\geq$ 10m s^{-1} (Figure 7.8) and it is hypothesised that during these epochs the greatest yields of sediment lost by erosion would occur due to higher levels of energy available for particle detachment and transport. Two other factors may also have resulted in increased erosion of the peat surface. The drying out of the peat surface may result in increased sediment availability due to surface desiccation of the peat and Freeze-thaw action may have led to the detachment of peat from the surface and further sediment availability for transportation around the site. The next section of this chapter links the sediment yield data with the data relating to environmental conditions to identify the key drivers of erosion at Flow Moss.

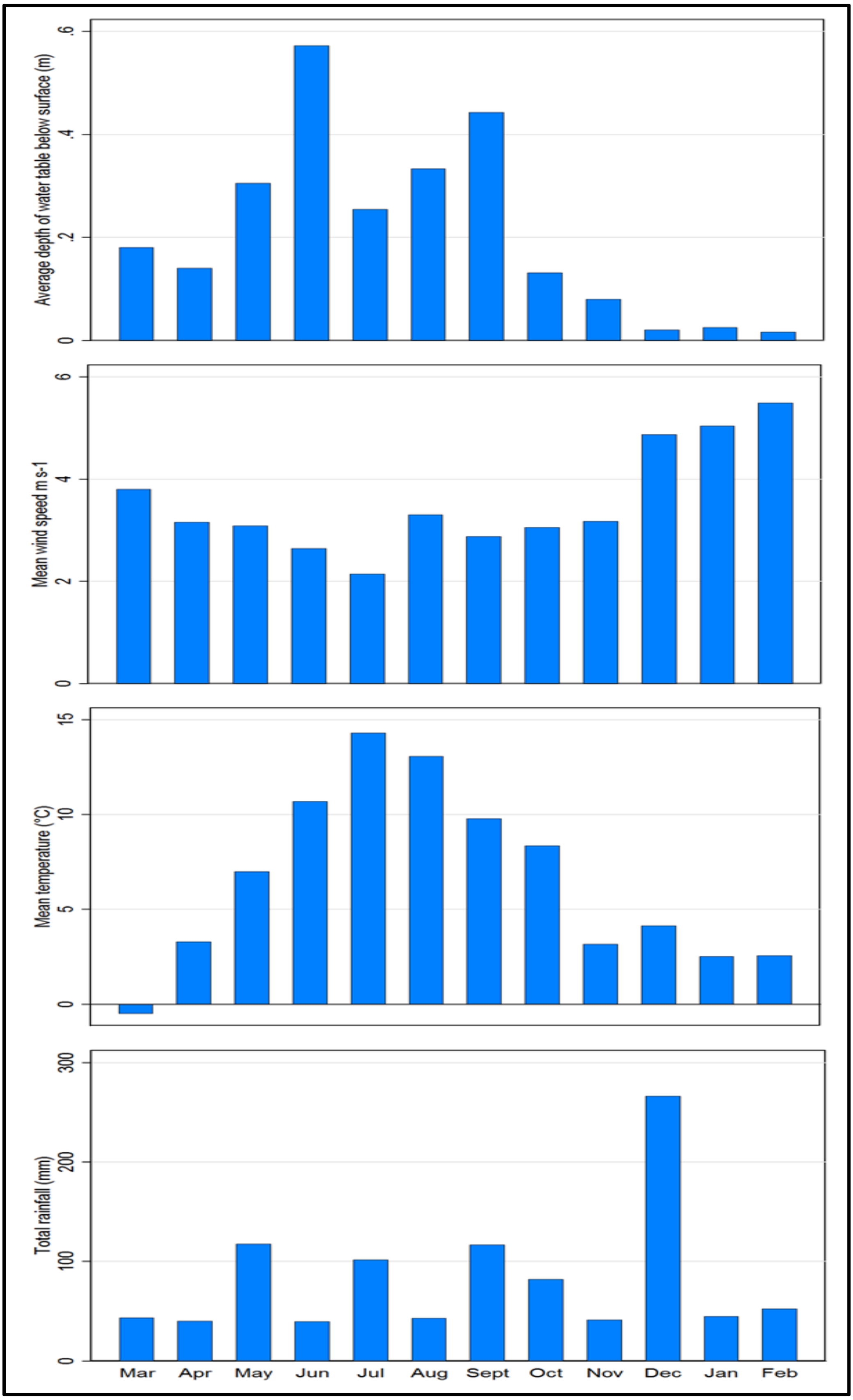


Figure 7.12 – *A summary of the monthly weather conditions recorded during monitoring.*

7.2 The impact of environmental conditions on eroded peat yields

The following section of this chapter compares peat yields with data relating to environmental conditions and identifies the key processes driving erosion at Flow Moss.

7.2.1 Aeolian processes

Previous studies by Warburton (2003) and Foulds and Warburton (2007) have identified that climatic conditions preceding an erosion event are a crucial factor in controlling the nature of erosion by aeolian processes. This was further investigated by Baynes (2012) who concluded that the amount of erosion occurring through aeolian processes is dependent on relationships between wind direction, wind-speed and rainfall intensity. The data in this study were analysed to corroborate if this was the case. Results are summarised in Table 7.2.

To assess the impact of wind direction on the mass of peat collected in the flux samplers, sediment yields were compared alongside wind direction data. Example plots are displayed in Figure 7.13, with the remaining data displayed in Appendix B.

Table 7.2 – *Results from wind direction data and sediment trap collection data*

Period	Sediment yield (g)	Sediment yield per day (g)	Mean wind direction (°)	Mean wind speed (M s^{-1})	Predominant sediment flux (°)
8th March -14th April 2013	0.81	0.02	106.5	4.31	90
14th April – 21st May 2013	5.71	0.15	164.1	2.48	180
21st May – 6th June 2013	0.4	0.03	186.5	2.61	180
6th June – 24th June 2013	4.25	0.24	189.7	2.54	330
24th June – 9th July 2013	0.68	0.05	246.4	2.87	270
9th July – 8th Aug 2013	3.54	0.12	198.7	2.53	240
8th – 19th August 2013	0.7	0.06	238.1	3.58	270
19th Aug – 6th Sept 2013	1.52	0.08	220.1	3.23	60
6th Sept – 16th Sept 2013	3.37	0.34	234.3	2.53	180
16th Sept – 10th Oct 2013	1.14	0.05	186.8	2.67	120/210
10th Oct – 24th Oct 2013	12.93	0.92	130.8	2.67	240
24th Oct – 8th Nov 2013	29.25	1.95	222.9	3.71	30
8th Nov – 25th Nov 2013	3.63	0.21	230.8	3.04	240
25th Nov – 17th Dec 2013	246.19	11.19	240.1	3.72	180
17th Dec – 30th Dec 2013	319.71	24.59	203.7	5.83	270
30th Dec 2013 – 21st Jan 2014	234.52	10.66	181.1	5.53	330
21st Jan – 3rd Feb 2014	86.45	6.65	162.8	5.43	240
3rd Feb – 17th Feb 2014	9.15	0.65	186.4	6.09	330
17th Feb – 4th March 2014	53.84	3.59	201.9	3.56	330

The results shown in Table 7.2 and Figure 7.13 indicate examples where the yields of sediment collected from the flux samplers appear to correlate with the prevailing wind direction recorded for these periods. However, the majority of the other data sets (Appendix B) appear to follow a more random distribution with just one or two sediment traps recording high sediment yields. This indicates local conditions are impacting on the sediment trap array, and to some extent this relates to the environmental conditions preceding the emptying of the wind flux tubes. For periods where snow days are recorded (1^{st} – 8^{th} March 2013, 8^{th} March-14^{th} April 2013, 14^{th} April – 21^{st} May 2013, 8^{th} November – 25^{th} November 2013, 25^{th} November – 17^{th} December 2013, 17^{th} December – 30^{th} December 2013, 30^{th} December 2013 – 21^{st} January 2014 and 3^{rd} – 17^{th} February 2014) there is less of a correlation between yields and wind direction. Snow cover provides protection to the bare peat surface from wind erosion, so on many of the days the peat would have been protected by the snow and drifts of snow may interfere with the trapping efficiency of the samplers. Furthermore during the course of the measurement period the surface of the peat underwent significant micro-topographical changes due to frost heave, local water erosion and surface desiccation. This often results in large local changes in the relief of the peat surface (10-30 mm in places). If this occurred in the vicinity of the tube traps, sediment yields would be affected as the local sediment flux could be greatly enhanced or reduced. Figure 7.14 shows the total sediment yields collected plotted against the mean wind direction for the entire study. This Figure shows that there are some correlations with the predominant wind direction and the yields of sediment collected by the wind flux samplers. As shown in Figure 7.14, the greatest sediment yields were collected from tubes facing towards the southwest, which was also the overall prevailing wind direction.

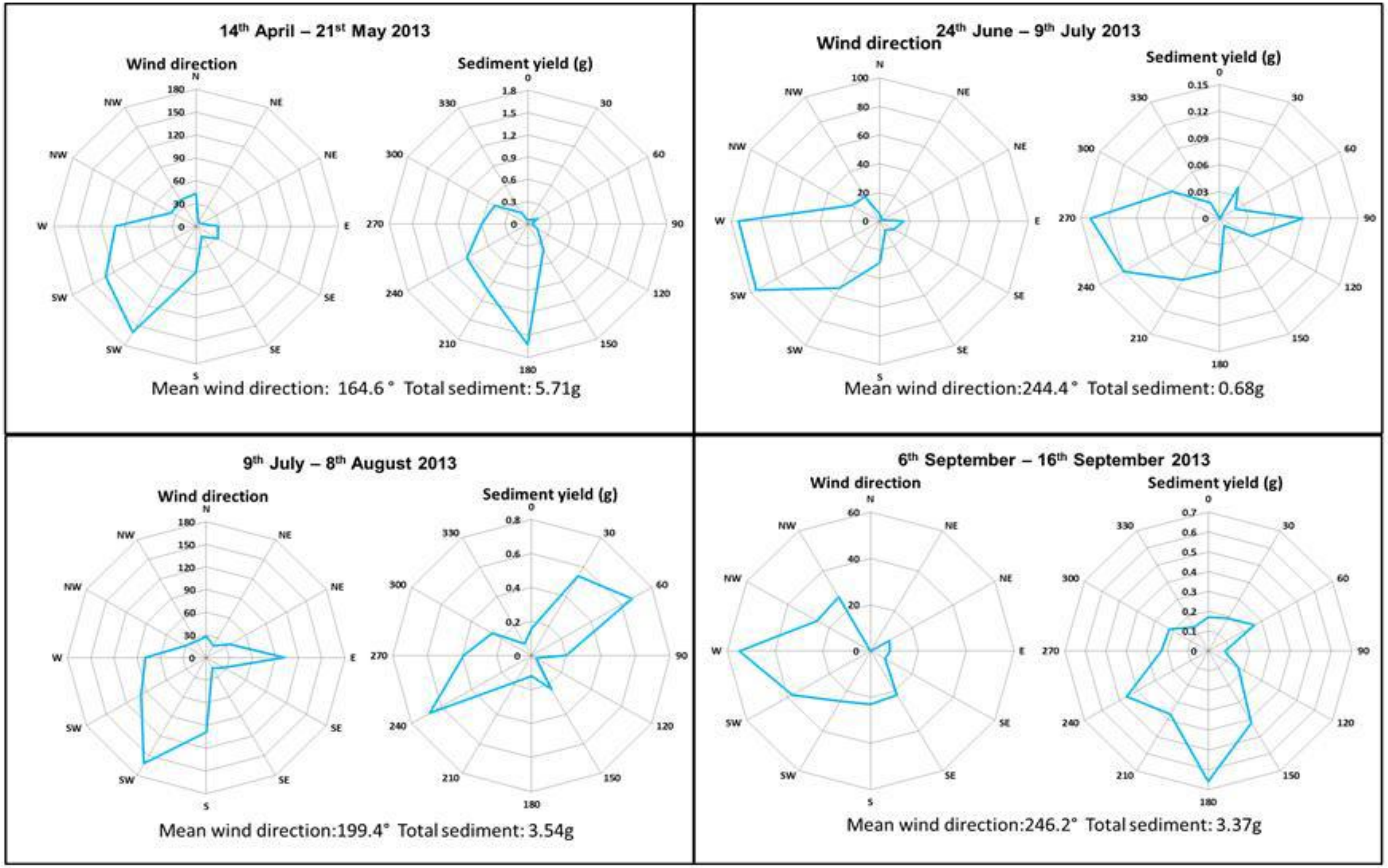


Figure 7.13 – *Example plots showing wind directions and sediment yields recorded during monitoring.*

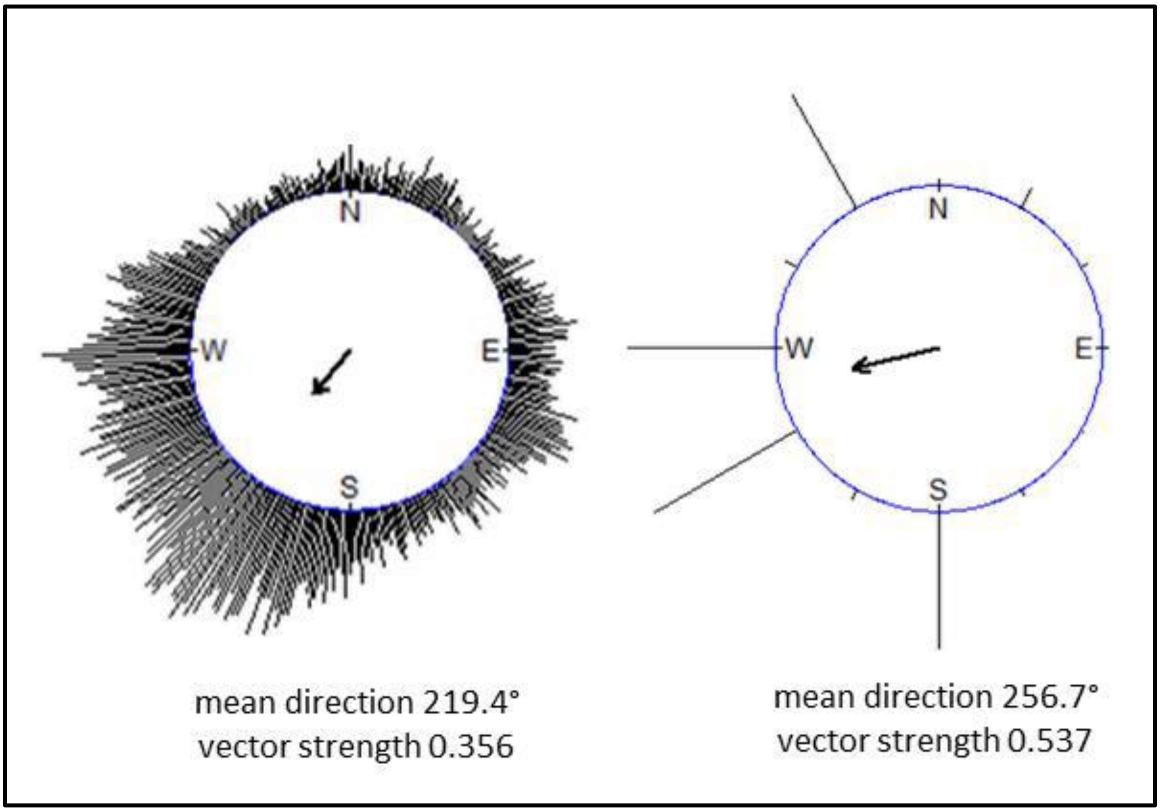


Figure 7.14 – *Wind direction and sediment yields recorded between 8th March 2013 and 4th March 2014.*

Warburton (2003) observed that during a study at Moss Flats, the samplers that were windward facing had a peat flux of 3 to 12 times greater than the leeward facing samplers. This indicates that wind direction is important in controlling peat flux by aeolian processes. The study by Baynes (2012) corroborates this and found that the distribution of peat collected from the wind flux samplers matched the distribution of wind directions measured concurrently. To assess whether a greater yield of sediment was collected from windward or leeward facing samplers, the total yield of sediment collected from windward facing samplers (tubes placed at 180-270°) was plotted against the total yields of sediment collected from leeward facing samplers (tubes covering 0-90°). This is shown in Figure 7.15.

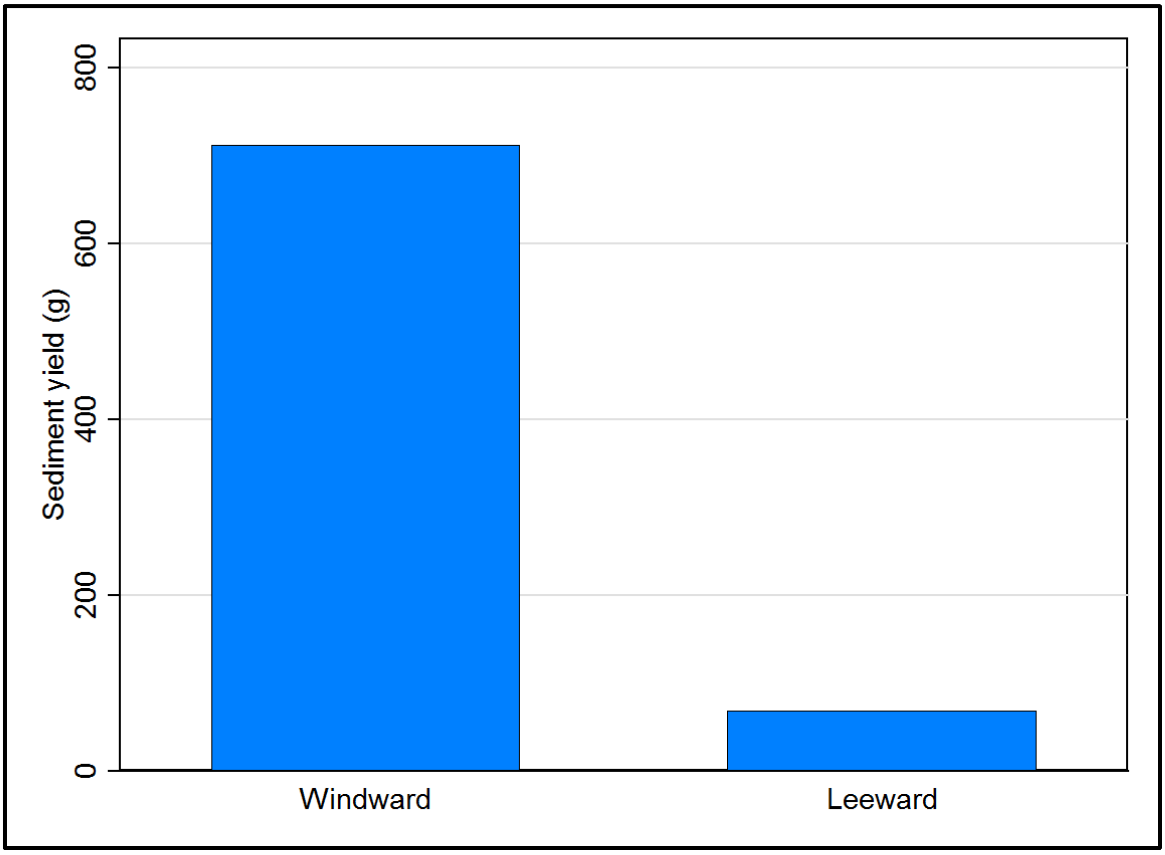


Figure 7.15 – *Total yield of sediment collected from windward facing and leeward facing samplers*

The total yield from the 3 windward facing samplers' was 712.19 g which is over 10 times greater than the 67.6 g of sediment collected from the leeward facing samplers. This suggests that wind direction is an important factor in erosion processes, but during the period of monitoring local erosion may also have significantly altered the spatial distribution of sediment flux.

Further evidence of wind direction impacting erosion dynamics can be seen from the TLS DEM, in which distinctive areas of microtopography, known as peat terraces, are identifiable (Figure 7.16 **(a)**).

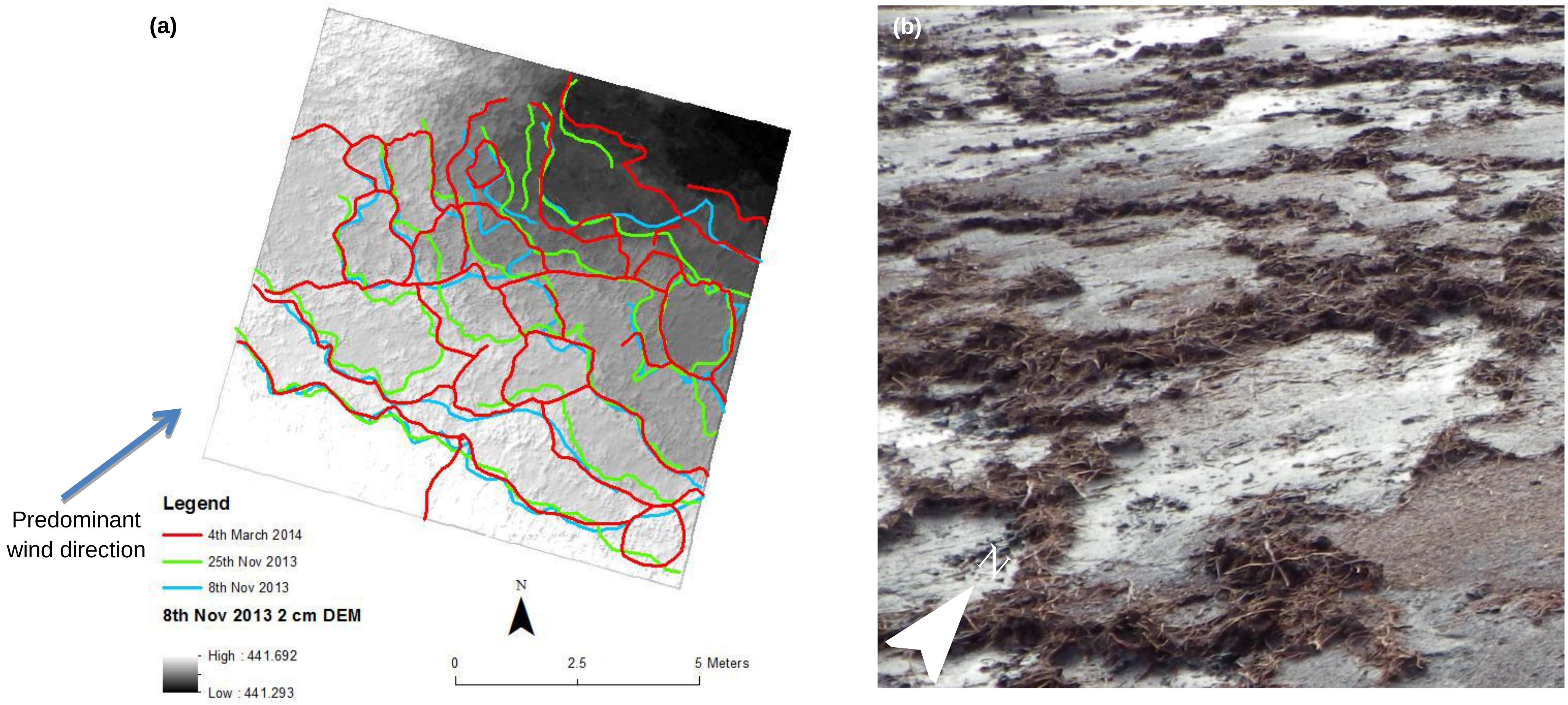


Figure 7.16 – (a) *TLS DEM with shape files of digitised terraces overlaid, showing terrace movement between 8th November, 25th November 2013 and 4th March 2014.* ***(b)*** *A photograph of the peat terraces – terrace fronts are picked out by fibrous peat and brash accumulations.*

A photograph of the terraces is shown in Figure 7.16 **(b)**. Prior to the formation of the terraces, the peat surface was much flatter and had a far smoother appearance. 2 cm resolution TLS DEMs created from data captured on 8th November 2013, 25th November 2013 and 4th March 2014 were examined and the location of terraces mapped. These could then be compared to identify movement that occurred between the three epochs. The terraces appear to be ‘creeping’ in the same direction as the prevailing wind measured between these epochs. This would indicate that detached peat is being transported in the direction of the prevailing wind.

To assess whether wind speed has an impact on the yield of sediment collected from the wind flux samplers, the mean wind speed recorded prior to data collection is plotted against sediment yield collected from the tubes (Figure 7.17).

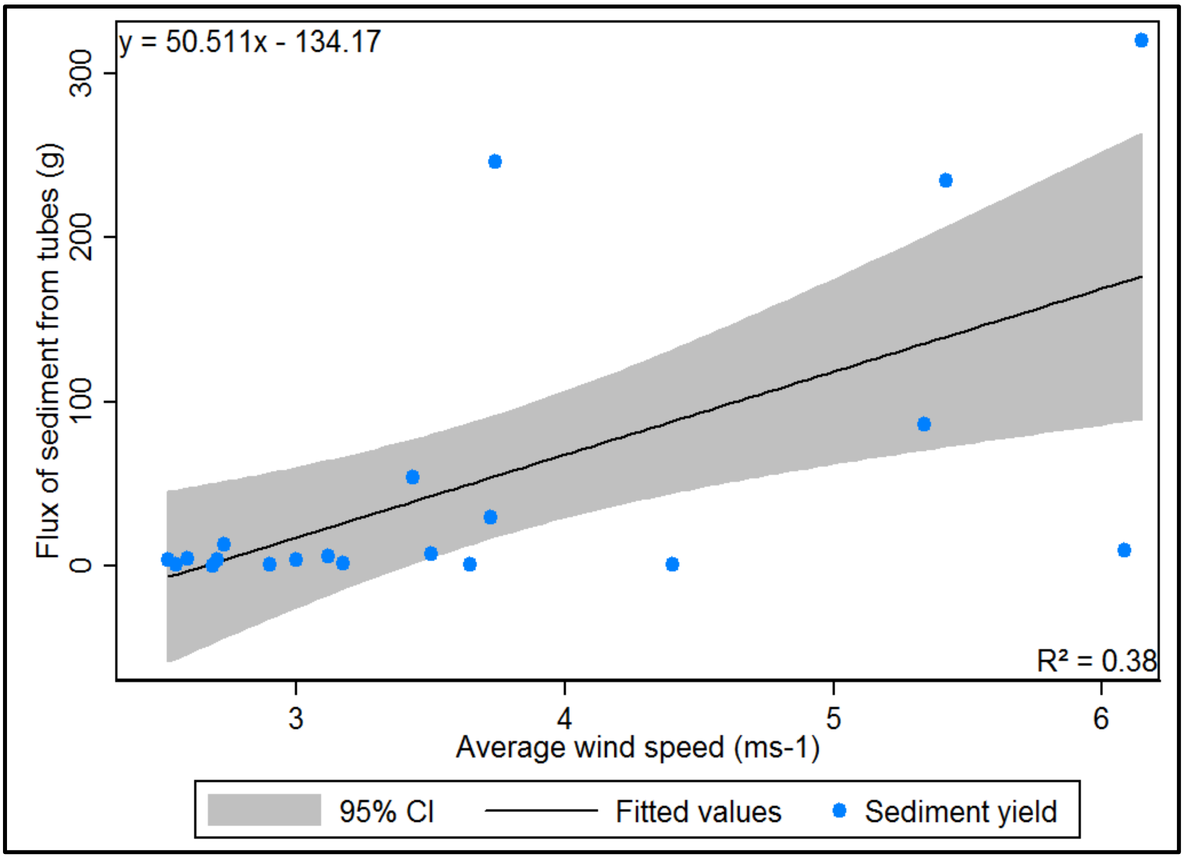


Figure 7.17 – *Yields of sediment plotted against the mean wind speed.*

Figure 7.17 shows a weak positive relationship between mean wind speed and sediment yield. Although this correlation is weak, it nevertheless demonstrates that sediment flux has a directional component imparted by the prevailing wind direction and speed. The scatter in the plot is related to local factors which add to the heterogeneity of sediment transport over the rapidly changing bare peat surface.

Previous studies have found rainfall to be a key driver of erosion (Evans and Warburton, 2007). Therefore, total rainfall amounts recorded between the collections of sediment from the traps, were plotted against sediment yields to establish if a correlation existed (Figure 7.18).

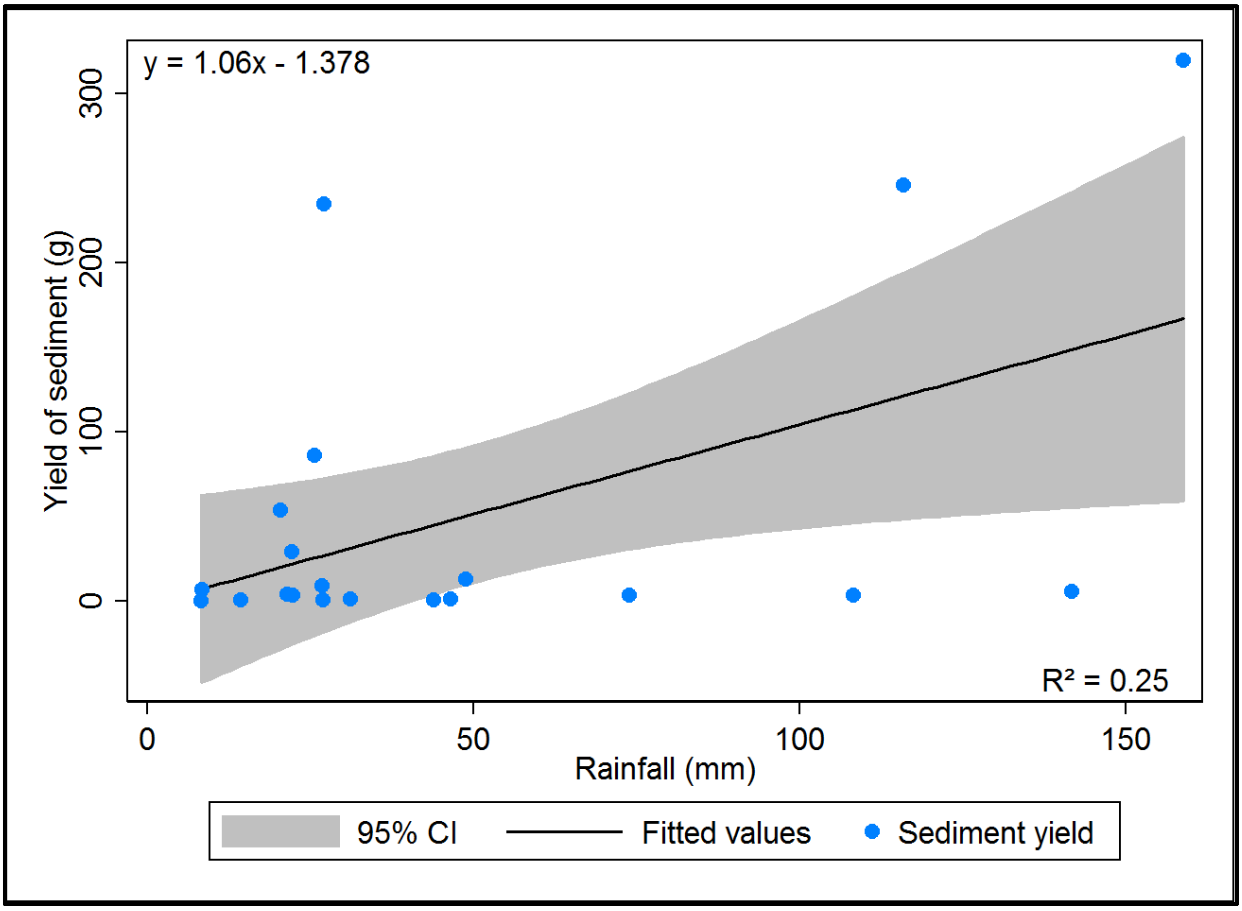


Figure 7.18 – *Preceding total rainfall plotted against sediment yield from flux samplers.*

The results show that a weak positive correlation exists between rainfall amounts and yields of sediment collected from the wind flux samplers but the scatter pattern of the data indicate that this relationship is composed of a few points of high sediment yield but a majority of points which form a baseline of low sediment yields spanning a large range of rainfall totals (5-140 mm). This implies rainfall intensity or short periods of intense rain may be a better explanation of this pattern. Therefore, the number of hours during which the wind speed or rainfall was greater than the specified threshold (5 mm hr $^{-1}$ for rain fall, 10 m s^{-1} for wind speed) were calculated and the amount of times these occurred between sediment trap emptying plotted against the collected yields (Figure 7.19 and 7.20).

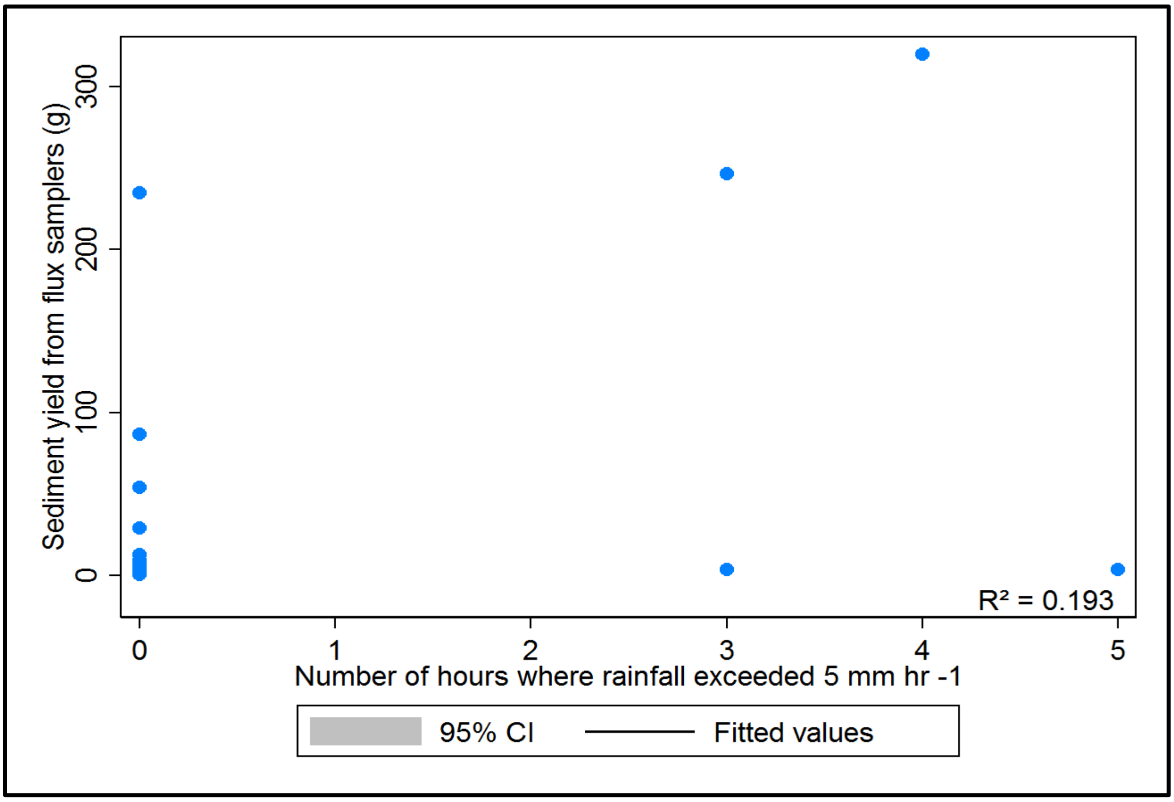


Figure 7.19 – *Number of hours where rainfall exceeded 5 mm hr^{-1} plotted against sediment yields.*

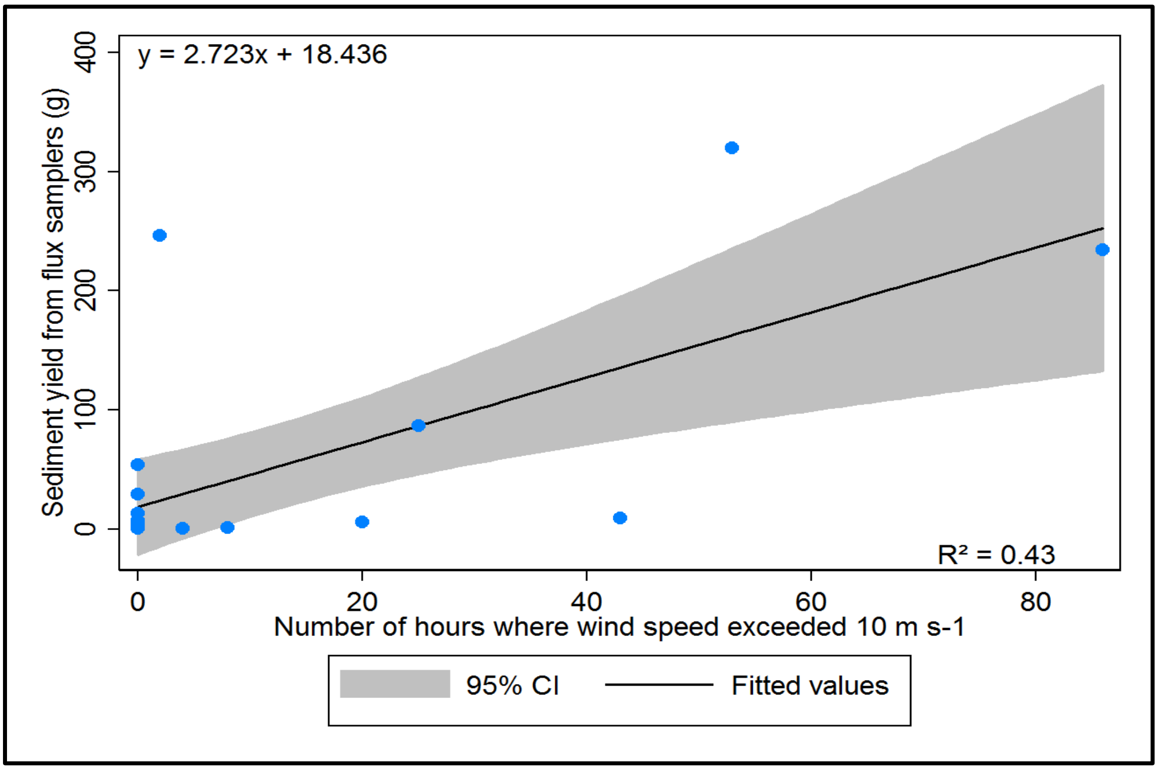


Figure 7.20 - *Number of hours winds speed exceeded 10 m s^{-1} plotted against sediment yields.*

Figures 7.19 and 7.20 provide two key pieces of information. Firstly, as shown in Figure 7.19, there is a weak positive relationship (R^2 = 0.19) between rainfall intensity and collected sediment yields. A large amount of scatter is shown in the plot suggesting there is little or no relationship. However, the relationship between the number of hours high wind speed events occurred and sediment yields is far stronger (R2 = 0.43) suggesting that although both high intensity rainfall events and high wind speed events have an impact on yields of sediment, wind speed results in more erosion than rain.

Baynes (2012) found that the largest yields of sediment collected from the wind flux samplers were collected following periods where high wind speeds coincided with high rainfall amounts. To assess whether the same patterns exist in the data collected for this study, the total rainfall amounts and mean wind speed occurring before the wind flux samplers were emptied were ranked from one to 20. The rank of the wind speed and the rainfall were then added together to provide a number representing the severity of the wind and rain event. These were then plotted against yields of sediment collected from the wind flux tubes (Figure 7.21).

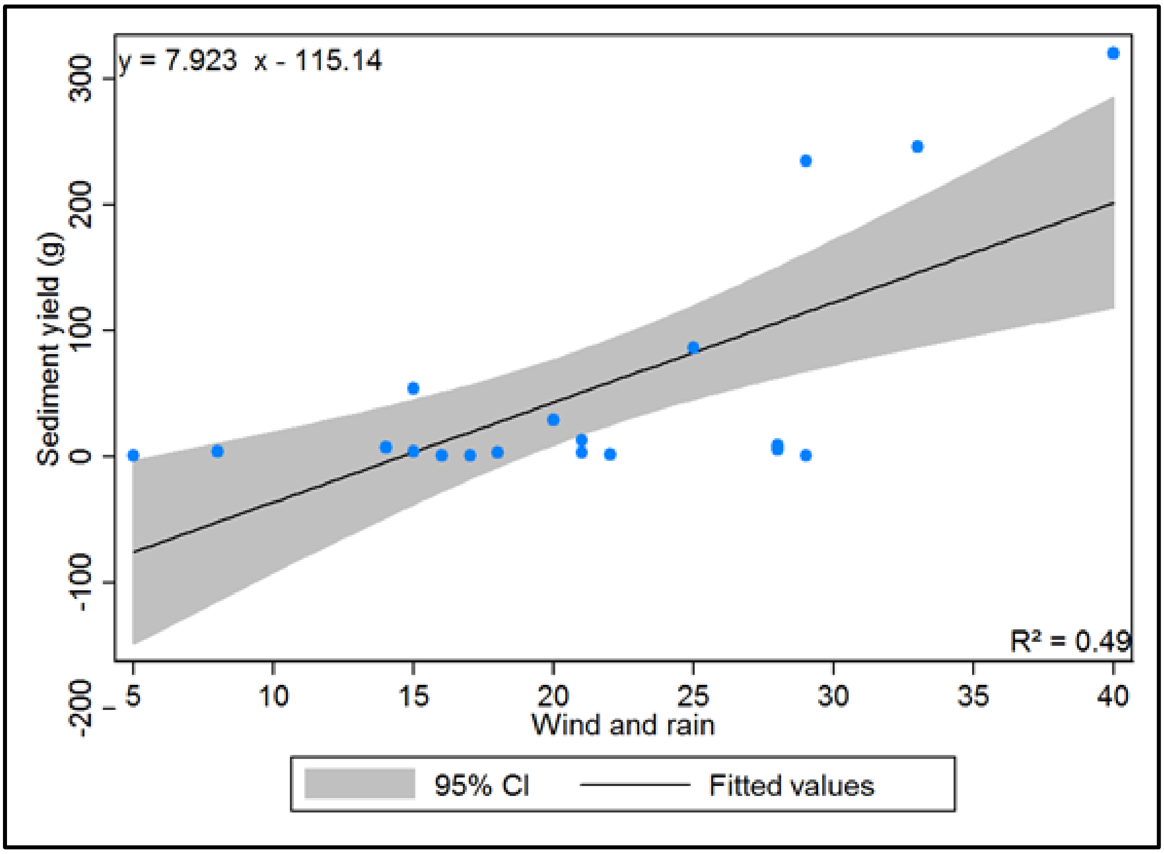


Figure 7.21 – *Yields of sediment collected from the wind flux samplers plotted against storm events.*

Figure 7.21 shows that a positive relationship exists ($r^2 = 0.49$) between the intensity of the wind speed / total rainfall and yields of sediment collected from the wind flux samplers. To confirm this statistically, a Spearman's rank correlation coefficient of 0.497 was calculated with a p value of 0.026. Therefore, the correlations between the two variables can be considered statistically significant. Subsequently, the number of hours where high windspeed events and high rainfall events occured between sediment trap emptying were combined and plotted against the collected sediment yields (Figure 7.22).

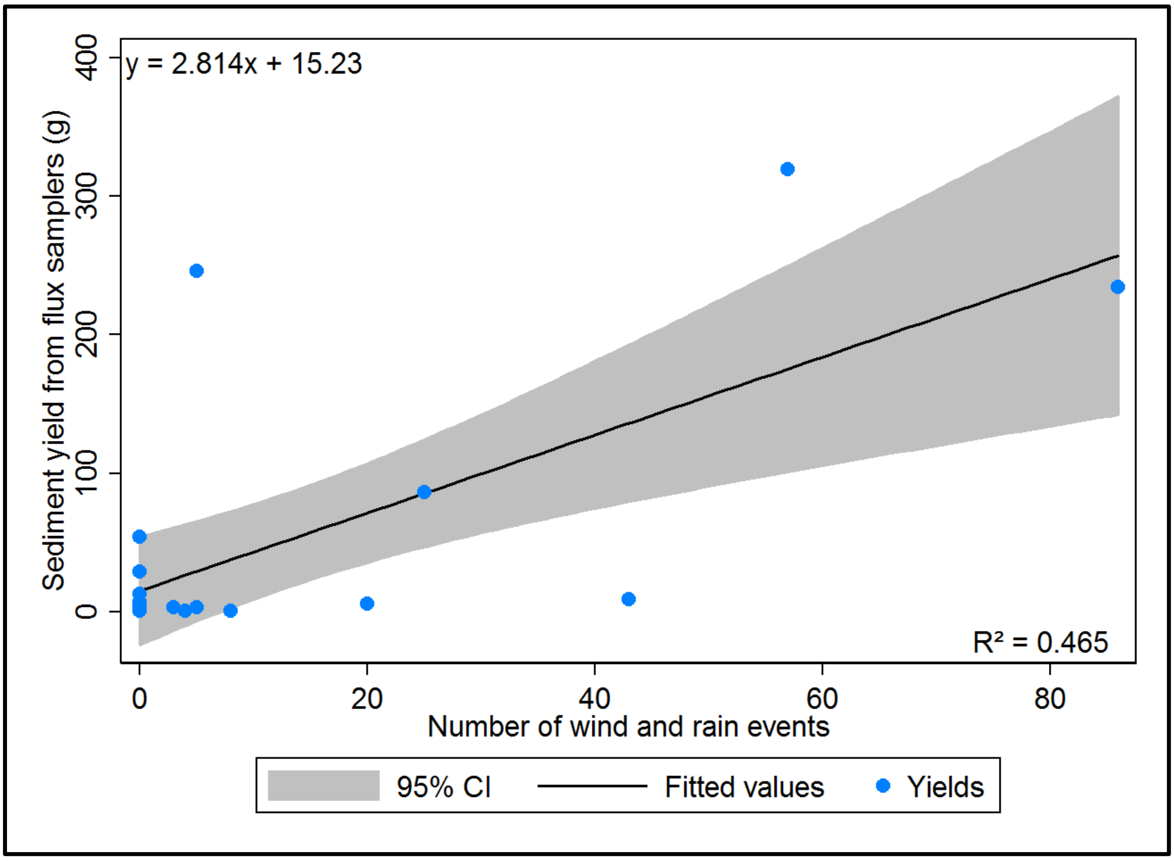


Figure 7.22 – *the number of hours where high wind speed and high rainfall events occurred plotted against sediment yields.*

Figure 7.22 shows a positive correlation between hours of high intensity wind speed and rainfall events, and sediment yields collected from the wind flux tubes (R^2 = 0.46). This suggests that although high wind speed and high rainfall intensity events can impact on erosion dynamics independently, the biggest increases in erosion occur following events where these have acted together. Oblique rainfall will lead to the detachment of peat particles from the surface, which results in a greater level of sediment availability for sediment transportation by aeolian processes and wind-driven rain.

7.3 Chapter summary

The results in this chapter demonstrate that the environmental conditions recorded at Flow Moss during the monitoring period are typical of a UK upland environment with low temperatures and high rainfall recorded during the winter months. The data displayed in Section 7.2 indicate that in the majority of cases, antecedent wind direction impacts the distribution of sediment collected from the wind flux samplers, however local variation would be expected. Figure 7.13 and 7.14 indicate that in many cases the predominant wind direction matched the direction from which the greatest yields of sediment were recorded in the wind flux samplers. This is further corroborated by the results shown in Figure 7.15 which identified the total yields of sediment collected in the windward facing samplers (712.19 g) to be 10.5 times greater than the 67.6 g of sediment collected from the leeward facing samplers. Furthermore, Figure 7.16 shows that the peat terraces digitised from the TLS DEMs would appear, in many cases, to be moving in the same direction as the predominant wind direction. Although these were only mapped over a short period of time (c. 4 months), it indicates that wind direction may impact the formation of geomorphological formations on the peat surface. All these factors outlined above indicate that wind direction is an important factor in erosion processes.

The data displayed in Figures 7.17 and 7.18 indicate a weak positive correlation between mean wind speeds and sediment yields ($R^2 = 0.38$) and total rainfall and sediment yields ($R^2 = 0.25$). This indicates that both wind speed and rain fall can impact on sediment yields lost through erosion. Furthermore, the data in Figure 7.21, where ranks of storm events have been plotted against sediment yields, show the strongest relationship (R^2 = 0.49) indicating that it is events where high wind speeds and rainfall occur concurrently that result in the greatest yields of sediment lost through erosion. This suggests that rainfall may be leading to the detachment of peat from the surface which is mobilised and transported by high wind speeds.

8.0 Discussion

This chapter collates the results presented in Chapters Five, Six and Seven. Data relating to surface changes are used to construct an annual sediment budget (Figure 8.1) for Flow Moss, and information gained from this combined with peat carbon content data and carbon flux estimates from other studies (Roulet *et al.,* 2007, Nilsson *et al.,* 2008*,* Evans and Lindsay 2010, Worrall *et al.,* 2011) measuring dissolved organic carbon and gaseous $CO_{2,}$ to provide an estimated annual carbon budget. This can be used alongside measurements of the size of the peat carbon reservoir at Flow Moss to identify whether Flow Moss is currently a sink or a source of carbon.

Data collected during the study are combined with those collected by Baynes (2012) to provide an extended record from April 2010 to March 2014 identifying longer term patterns of sediment loss which is used to assess whether restoration measures implemented by the North Pennines AONB peatlands programme are reducing sediment and carbon lost through erosion.

8.1 Construction of a Sediment budget for Flow Moss

A sediment budget quantifies erosion, sediment storage and processes linking these (Slaymaker, 2003). As discussed in Section 2.4, sediment budgets provide a valuable tool for understanding sediment loss and this information can be used to assign priorities for restoration strategies aimed at erosion control. Table 8.1 lists the key fluxes of the sediment budget at Flow Moss recorded during the monitoring period and these are summarised in Figure 8.1.

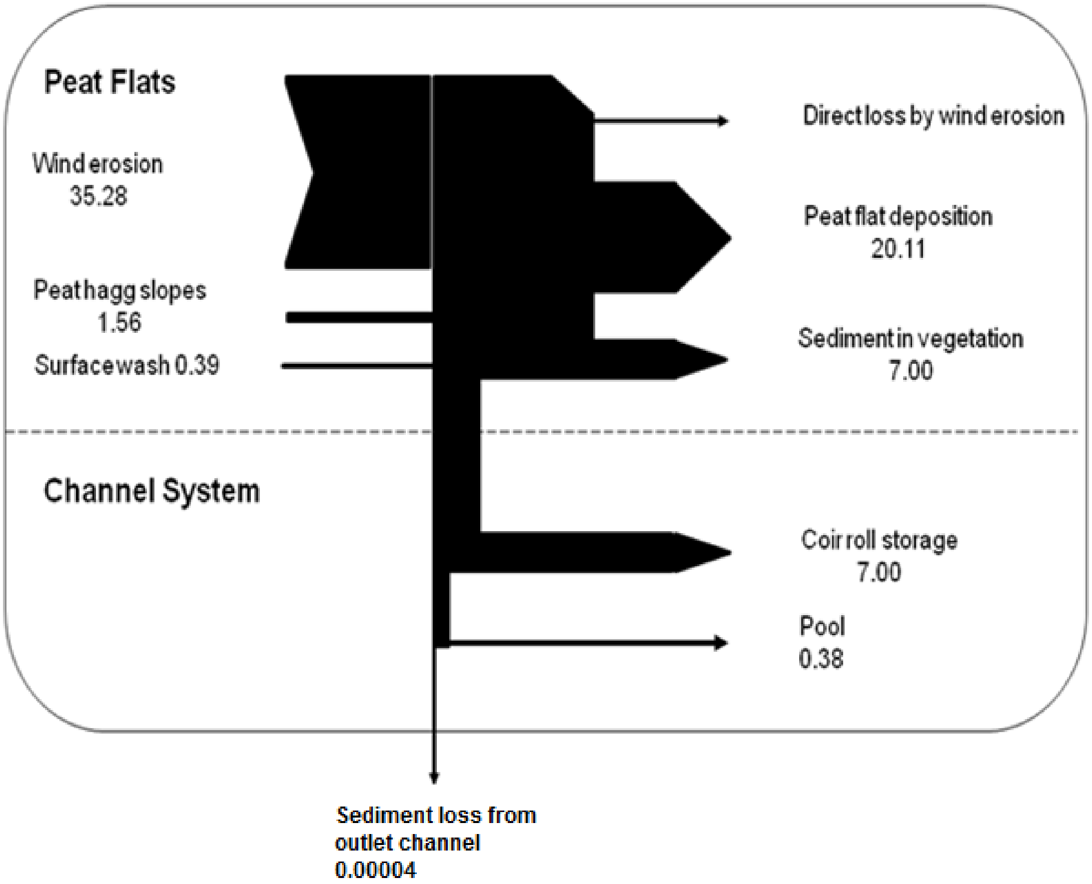


Figure 8.1 *- A schematic representation of the Flow Moss Annual sediment budget. All values are in tonnes*

An important part of this study was the use of erosion pins to monitor changes in surface height of the peat haggs (Chapter 4.2.2). An example of using erosion pins for peatland monitoring is provided by Evans and Warburton (2005). Data measuring erosion of gully walls were scaled up to provide a sediment yield for the Rough Sike catchment. Negative pin values were considered as random error, or as evidence of local deposition. Similarly, Couper *et al.* (2002) discuss the correct use of erosion pins and state that negative values should be assumed to be either an indication of local deposition or error within the dataset. Baynes (2012) calculated change in pin exposure twice, once including and once eliminating negative readings. As the sediment budget from this study will be compared to the one by Baynes (2012) the same method was implemented and the yields of sediment loss from the peat haggs were calculated twice, once including negative values and once with the negative values removed.

When the negative values of erosion pin exposure are included, the results suggest that 3.01 t a^{-1} of peat have been deposited onto the peat hagg slopes. However, this is unlikely to be a true representative as local deposition will be occurring (sediment from the top of the slope collecting around erosion pins at the base). Furthermore, during field observations, on several occasions, pedestals around the base of the erosion pins (Figure 8.2) were observed. These resulted in a false negative recording of change in pin exposure. Therefore, in the sediment budget, the value calculated excluding negative pin readings of 1.56 t a^{-1} of erosion was used. Nevertheless, it is feasible to expect the surface of the peat to expand and contract (Price, 2003); consequently excluding negative pin recordings may lead to an overestimation of sediment yield eroded from the hagg faces.

Table 8.1 – *Table of calculated annual sediment fluxes for Flow Moss*

Process	**Sediment flux (t a^{-1})**	**Quantified error (t a^{-1}) and ((%))**	**Assumptions**
Wind erosion of peat flats	35.28	± 8.7 (23.9)	This value assumes that erosion though aeolian processes is uniform and data collected from the wind flux samplers are representative of the entire bare peat flats
Erosion & Deposition on hagg slopes with (negative values included)	3.01	± 0.40 (13.3)	This assumes the slopes used for monitoring are representative of the entire study area. It is assumed that negative readings of change in pin exposure (deposition) are correct.
Erosion from hagg slopes with negative values removed.	1.56	± 0.40 (25.6)	This assumes the slopes used for monitoring are representative of the entire study area. It further assumes that negative erosion pin readings (deposition) are erroneous.
Hydraulic peat transport (at catchment outlet)	0.000036	±0.0000031 (8.6)	Assumption that peat is not being hydraulically transported from the site in other areas.
Deposition in pools	0.38	±0.017 (4.5)	This figure assumes that the measured pool is representative of other pools.
Erosion & Deposition on bare peat flats	20.11	±0.29 (1.4)	This value assumes that the data collected from the pole transects are representative of the entire bare peat flats.
Deposition in vegetated areas	7.00	1.75 (25)	This assumes that unaccounted for sediment is trapped and stored within the vegetation and not lost from the site. It is further assumed that the deposition recorded for the peat flats has occurred due to deposition and not increases in surface height due to frost heave and fluctuating water table levels.
Entrapment by coir rolls	7.00	1.75 (25)	This assumes that equal amounts of the peat unaccounted for in the sediment budget is entrapped by the coir rolls or trapped within vegetated areas.

Figure 8.2 – *Pedestals observed at the base of some erosion pins during field visits. These could lead to false negative values of change in pin exposure.*

The data collected during this study (Table 8.1) were up-scaled to provide a value representative of the sediment flux for the entire bare peat area. This was done using a series of simple scaling equations relating the sampling area to the total area of bare peat (Appendix C). All values were standardised to tonnes (mass) and multiplied by 0.98 (the proportion of the year covered by the measurement period) to calculate tonnes per year.

The sediment budget in Figure 8.1 identifies several important features. Firstly, a substantial amount of sediment is transferred within the site through aeolian processes, suggesting the bare peat flats are geomorphologically very active. A small amount of this sediment will be lost from the site through wind erosion; however as Warburton (2003) suggests, sediment eroded by wind splash processes will normally travel only short distances of between 1 and 10 m once particles have been mobilised and up to 50m when dry dust is blown. These distances are short in comparison to the large area of bare peat at Flow Moss (10777 m^2) which is surrounded by extensive vegetated areas providing ample opportunity to trap some of the eroded peat before it leaves the site (Figure 8.3 **(a)**). Therefore during wind erosion events, the majority of the eroded peat will be redistributed within the bare peat flats. This is confirmed by the large amount of sediment (20.11 t a^{-1}) which is deposited on the bare peat flats. This indicates that eroded sediment is being redistributed on the bare peat flats and in the surrounding vegetation with only smaller amounts entering the channels at the periphery of the bare peat and being lost though fluvial processes. The very small yield of sediment (0.000036 t a^{-1}) collected from the sack traps clearly demonstrates this. Furthermore, the low yields of sediment collected in the sack traps are indicative of the importance of the pools in trapping eroded peat and, more recently, the success of the coir rolls installed to dam the ephemeral channels. Sediment which enters the channel systems may be caught up in these rolls and prevented from leaving the site (Figure 8.3 **(b)**).

It is estimated that the difference in eroded peat yields between the amount of sediment transported by wind erosion and that deposited on the peat flats (Table 8.1) is approximately 14 tonnes. The most likely fate of this peat, which is not directly accounted for in the sediment budget, is deposition in the vegetation surrounding the bare peat and trapping behind the coir rolls

(Figure 8.3 **(b)**). Therefore, as an initial approximation, the mass of peat trapped by both the coir rolls and vegetation surrounding the bare peat has been estimated as approximately 7 tonnes each (± 25%). This should be investigated further particularly because it is possible that the estimate of the amount of peat re-deposited in the bare peat area is an overestimate. Some of this change in surface height will be the result of changing water table levels and frost heave that have caused an increase in surface elevation but been incorrectly recorded as deposition.

Figure 8.3 – *Examples of eroded peat being trapped by* ***(a)*** *vegetation at the margin of bare peat* ***(b)*** *coir rolls.*

The sediment budget (Figure 8.1) estimates the amount of peat transferred from the slopes of the peat hags to be 1.56 t a^{-1}. Peat from the haggs will be transported to the bare peat flats where most of it will be deposited. Some of the unmonitored peat haggs are located in close proximity to the channel system and it is likely that in these areas, some of the eroded sediment will enter the channel system directly. This was not monitored during this study, but volumes of peat lost from the haggs into the channel system are unlikely to significantly alter the sediment budget, and a proportion of this eroded sediment will be trapped by the coir rolls.

Although the sediment budget quantifies the main components of sediment transfer and loss, it is important to note several limitations with this approach. When scaling up the processes monitored at Flow Moss, it is often assumed that processes act uniformly across the site, and this is unlikely to be the case in all instances. Furthermore, the sediment budget was created from data collected at the central and northern end of the Flow Moss site, where the conditions and erosion dynamics may be different to those recorded at the more vegetated southern end of the site. Erosion dynamics can vary widely even on local scales (Evans and Warburton, 2007).

8.2 The Flow Moss Estimated Carbon budget

One of the three key research questions (Chapter 1, Section 1) of this study was ‘How can peatland carbon stores be accurately assessed, and what is the local carbon store at Flow Moss?’ To answer this question, a detailed subsurface investigation was undertaken (Chapter 5, Section 3) and the data produced used to estimate the amount of peat and carbon stored at Flow Moss. Using the results from the subsurface investigation and the sediment budget a carbon budget is constructed for Flow Moss in order to estimate the amount of carbon sequestered, and assess whether the site is currently acting as a net sink or source of carbon.

8.1.1 Flow Moss Carbon reservoir

The results in Section 5.3.1 estimate the volume of peat stored at Flow Moss to be 102730 m^3. When combined with bulk density results, this equates to a mass of 7974 tonnes of peat, which when combined with the TOC results of 50.21%, provides an estimate of the size of the carbon store as 4004 tonnes of carbon or 572 tonnes per ha. Although this value indicates the size of the carbon store at Flow Moss, it is not representative of a carbon budget.

8.2.1 Flow Moss Annual Carbon Budget

Baynes (2012) used a simple calculation to assess the amount of carbon loss from areas of bare peat and the amount of carbon sequestration by areas of vegetated peat at Flow Moss. The same calculation will be used here to create a carbon budget for this study period and the results compared to those calculated by Baynes to establish changes to the carbon budget which may have occurred. In common with the study of Baynes (2012), this study did not directly measure dissolved organic carbon (DOC) lost from the site, or the gaseous fluxes from the peat. To overcome this, Baynes used values from previous studies (Table 8.2) and the knowledge of the extent of areas of vegetated peat and bare peat (mapped from the UAV in 2011 by Baynes (2012) and with the differential GPS in the present study).

Table 8.2- *Rates of carbon emissions and sequestration recorded during previous studies*

Study	**Emissions from bare peat area g C m^{-2} yr^{-1}**	**Carbon fixation in vegetated areas g C m^{-2} yr^{-1}**	**Area of peat**
Roulet *et al* (2007)		21	28 km^2
Nilsson *et al*., (2008)		20 – 27	6.5 km^2
Evans and Lindsay (2010)	56	20.3 ± 4.0	c. 5km^2
Worrall *et al*., (2011)	272 ± 15 to 522 ± 59		c. 5km^2

The values recorded for carbon fixation in the studies outlined in Table 8.2 are not substantially different; therefore to calculate the carbon budget for Flow Moss, a mean value of the three studies (22.075 g C m^2 per year) was used. Two predictions of carbon emissions from bare peat areas were found within the existing literature. These were 56 gCm^2 per year estimated by Evans and Lindsay (2010) and 272 to 522 g C m^2 per year estimated by Worrall *et al*. (2011).

There is a large difference in the values quoted in the studies above, therefore the emissions from the bare peat at Flow Moss will be calculated three times for a low emission scenario (using the value of Evans and Lindsay (2010) of 56 g C m^2 per year), a medium emission scenario (using the lower value of Worrall *et al*. (2011) of 272 g C m^2 per year) and a high emissions scenario (using the higher value of Worrall *et al*. (2011) of 522 g C m^2 per year). The estimates of carbon fixation and emission rates for Flow Moss are shown in Table 8.3.

Table 8.3- *Calculated rates of carbon sequestration and emissions for Flow Moss.*

Surface cover	**Carbon flux rate (gC m^{-2} yr^{-1})**	**Area (m^2)**	**Total carbon stored/lost per year (tC yr-1)**
Vegetated peat	22.075	59223	1.31
Bare peat (low)	56	10777	-0.24
Bare peat (medium)	272	10777	-2.93
Bare peat (high)	522	10777	-5.63

The data shown in Table 8.3 can be combined with data from the sediment budget to establish the amount of carbon lost or sequestered at Flow Moss. The mid-range value of the three scenarios outlined in Table 8.3 is used in the carbon budget calculation (Figure 8.4). From Figure 8.4 it is possible to identify that the annual carbon fixation rate of the vegetated areas at Flow Moss is 1.31 t C yr^{-1}, while the carbon emissions for Flow Moss are currently approximately 2.93 t C yr^{-1}. This would indicate that Flow Moss currently emits 1.63 t C yr^{-1}indicating that the site is a net source. However, if the carbon budget is recalculated using the lower emissions value of 56 m^{-2} yr^{-1}, it would suggest that only 0.24 tonnes of carbon are emitted through gaseous exchange each year leading to a total carbon loss of approximately 0.25 t C yr^{-1}, 1.07 t C yr^{-1} less than the net rate of carbon fixation. The POC loss from the site through erosion is significantly smaller than the amount of carbon lost from the peat through gaseous processes. Even though the bare peat area at Flow Moss is still geomorphologically very active, overall the total amount of carbon lost from the site through erosion is negligible in terms of the overall carbon balance. Nonetheless, even small losses of carbon from peatlands can lead to a significant impact on global CO_2 levels if they were to occur globally and therefore peat and carbon loss through erosion should always be mitigated against as much as possible.

More carbon may be being sequestered in the area surrounding Flow Moss than is accounted for in the carbon budget. When looking at the carbon budget diagram displayed in Figure 8.4, it is important to note that the magnitude of the gas exchange for the vegetated area of the peat is artificially constrained by the area defined by the fence line at Flow Moss. The fence defines a clear land parcel in terms of the restoration area, however the extent of the vegetated peat is likely to extend beyond this boundary.

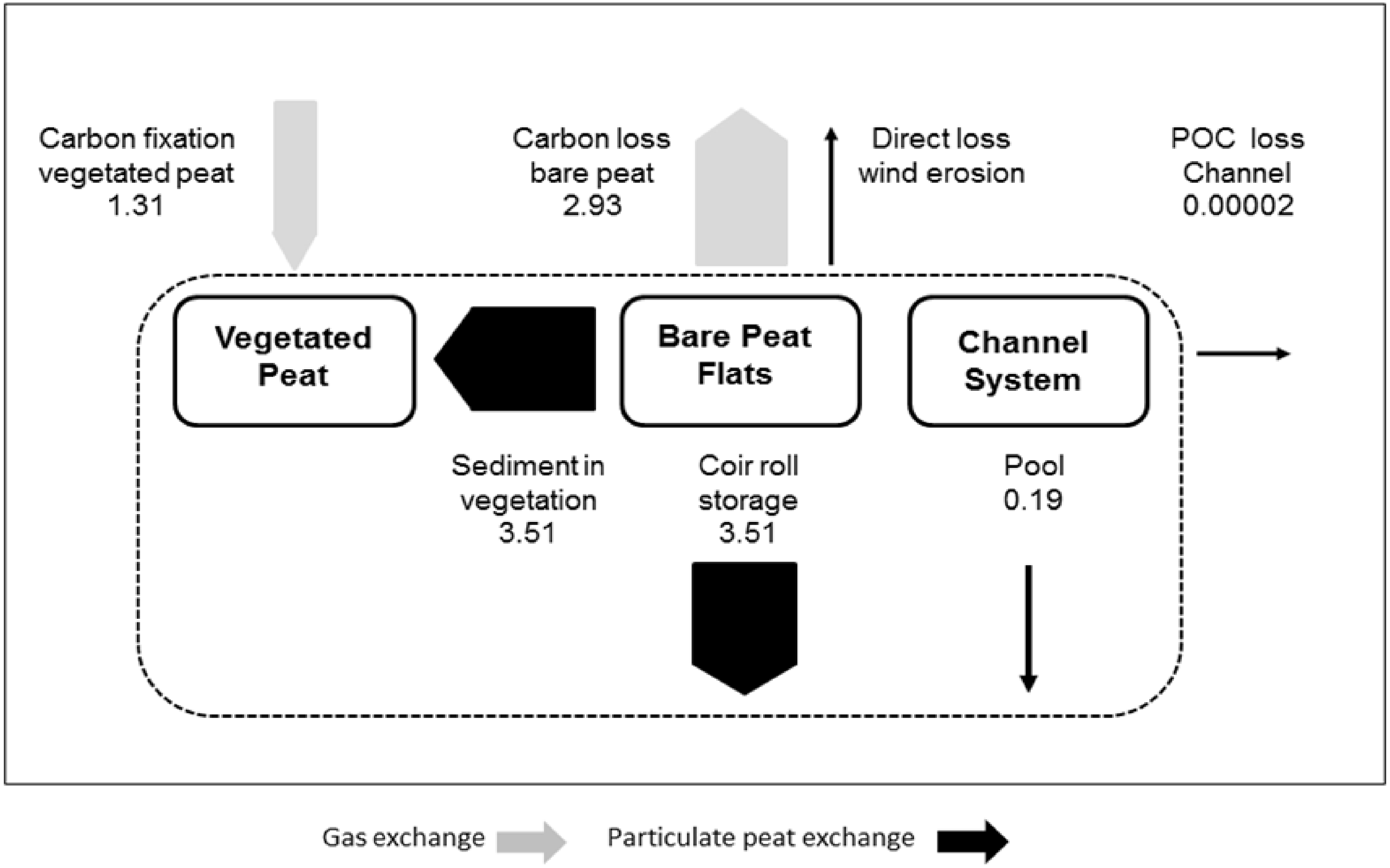


Figure 8.4 – *The annual carbon budget for Flow Moss identifying rates of carbon fixation and emissions from the site. All values are in tonnes C per annum.*

8.2.2 The future of the Flow Moss carbon store

The role of peatlands in moderating atmospheric CO_2 concentrations and mitigating carbon emissions through sequestration is becoming widely recognised (Fyfe *et al*., 2014). Many peatland restoration and monitoring projects are motivated by the large proportion of global CO_2 emissions which are released from soils and a small change in emissions could substantially impact upon future climate change scenarios (Biasi *et al*., 2014). This is one of the motivating factors behind many peatland restoration and monitoring projects. Lal (2003) discusses how despite their global significance, erosion-induced carbon emissions into the atmosphere still remain a misunderstood and poorly quantified component of the global carbon cycle. Kuhn *et al*. (2009) corroborate this and state that the exchange of greenhouse gases between terrestrial ecosystems and the atmosphere represents one of the greatest uncertainties in our understanding of the global carbon cycle.

The current carbon store at Flow Moss is approximately 4004 tonnes. Using the 'worst case scenario' value of 522 g C m^{-2} yr^{-1} for emissions from bare peat, the Flow Moss carbon store is estimated to lose approximately 0.1 % of stored carbon annually, whilst using the mid-range value from the carbon budget, the carbon store at Flow Moss is decreasing by c. 0.05% each year. In contrast, using the 'best case scenario' of 56 g C m^{-2} yr^{-1}, shows the Flow Moss carbon store is increasing by c.0.03% per year. This is likely to be an underestimate as these values are small and do not account for formation of any new peat.

Table 8.4- *Carbon budget estimates calculated using knowledge of areas of bare and vegetated peat (from the dGPS survey) and carbon emission values from Evans and Lindsay (2010) and Worrall et al. (2011).*

Emissions scenario	Emissions (g C m^{-3} per year)	Change in carbon store (% per year)
Low	56	+0.03
Medium	272	-0.05
High	522	-0.1

Using the values in Table 8.4, it is possible to project the impact that re-vegetation could have at Flow Moss (Figure 8.5). One of the aims of the current restoration project at Flow Moss is to re-vegetate the bare peat areas. If full vegetation cover was to be achieved, the rate of carbon fixation at Flow Moss would be 1.55 t C yr^{-1}. The re-vegetation would further reduce the yields of sediment lost through erosion of the bare peat by offering protection from erosive agents such as wind and rain.

The re-vegetation and future carbon balance predicted, for Flow Moss, illustrated in Figure 8.5, indicates that as the bare peat extent decreases, the carbon emissions will decrease until the two reach a point of balance in approximately 20 years' time. Subsequently, Flow Moss would become a net store of carbon, sequestering almost 1 t C yr^{-1} by 2040.

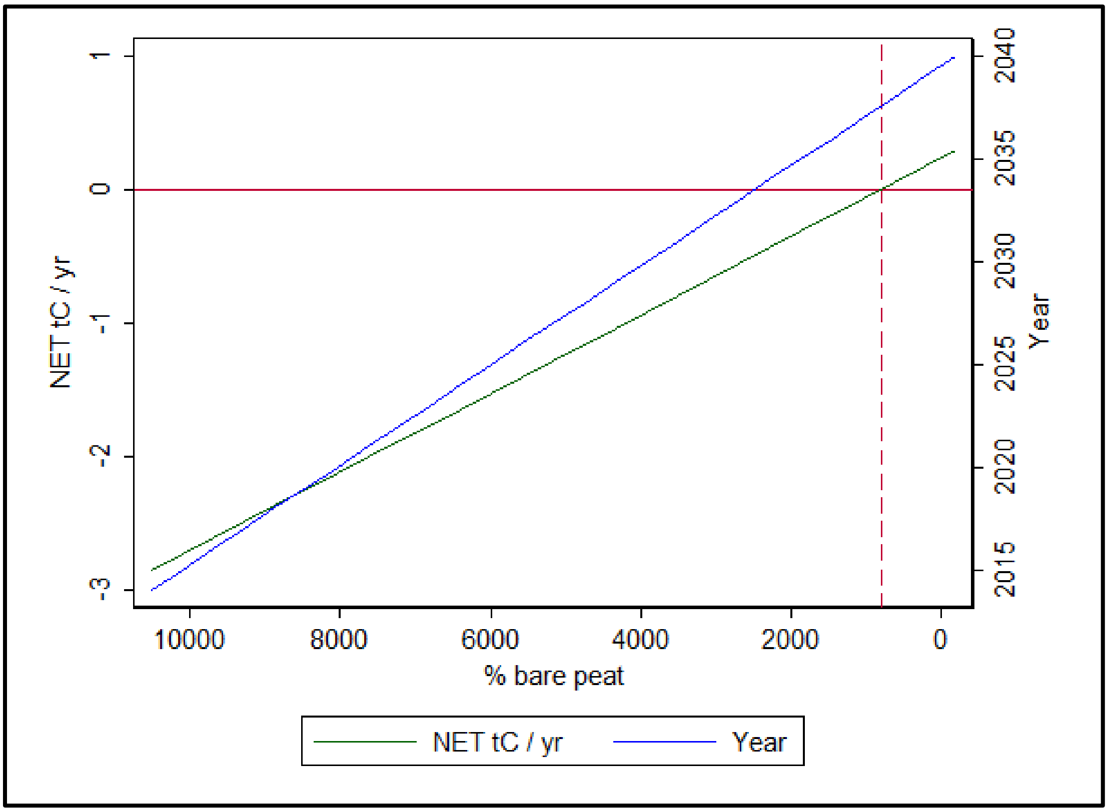


Figure 8.5 – *The predicted Flow Moss carbon balance and vegetation cover. Continued re-vegetation is predicted to result in Flow Moss switching from a net carbon source to a net sink in approximately 20 years.*

The values discussed above indicate that the carbon store at Flow Moss is relatively stable, however even small emissions could have a negative impact on future climate scenarios. An annual 0.1% loss of stored carbon from global peatlands would result in emissions of between 0.32 and 0.4 Gt C yr-1 being added to the atmosphere. LeQuere et al. (2009) state that 1 Gt C is equivalent to 0.47 ppm atmospheric CO2 therefore a 0.1 % loss equates to between 0.15 and 0.188 ppm of CO2 being added to global atmospheric CO2 concentrations annually. This estimate is based on the Flow Moss carbon budget. However, Flow Moss is currently undergoing restoration, and it is likely that the emissions figure for global peatlands is significantly higher as degraded and non-restored peatlands will be releasing more carbon.

Nonetheless, there are caveats that need to be considered when attempting to restore degraded peatlands. The restoration of peatland water tables will, over longer time scales, increase CO_2 sequestration. However, initially it will lead to increased emissions of CH_4 (Couwenberg, 2011) which has a global warming potential of 86 times that of CO_2, and therefore could have serious implications for climate change mitigation.

Future restoration projects will face increasing pressure from possible changes in climate. The UK Climate Projections (UKCP09) provides projections of future climates for high, medium and low greenhouse gas emissions (UKCP09, 2014). The current UKCP09 climate projections indicate that under future climate scenarios, a decrease in summer precipitation and an increase in winter precipitation could occur in the UK (UKCP09, 2014). Increased winter precipitation may result in more extreme rainfall events which would increase more sediment availability for transport. In contrast, increasing temperatures and decreasing precipitation levels in the summer could lead to a reduction in water table height. This can accelerate soil respiration rates (Couwerberg, 2011), leading to a net loss of soil C, higher concentrations of atmospheric CO_2 and thus further global warming due to positive feedback mechanisms (Luke and Cox, 2011). Furthermore, Ritson *et al.* (2014) state that reduced rainfall will result in increasing DOC concentrations from peatlands. Additionally, a decrease in the water table can result in surface desiccation and a greater yield of sediment availability for transport by physical processes (Goulsbra et al., 2014).

8.3 Recent changes in erosion dynamics at Flow Moss

The data collected during this study can be combined with that collected by Baynes (2012) to provide an extended data set which spans from October 2010 to March 2014, therefore providing a longer perspective on temporal changes in the erosion dynamics at Flow Moss. In addition, the annual sediment budgets constructed in both studies can be compared to identify changes in the nature of peat erosion dynamics and assess the effectiveness of the current restoration measures implemented at Flow Moss.

8.3.1 Recent changes in erosion rates

Four sets of data (wind flux samplers, pole transect, erosion pin and sack trap data) are directly compared to determine key changes which have occurred in the erosion rates at Flow Moss. Pole transect data, from Baynes (2012) and this study, were standardised to make them comparable (Figure 8.6) by calculating the rate of change per day and plotted by month to identify if a seasonal signal was present.

The data displayed in Figure 8.6 identify that variability in pole transect measurements exists both within, and between the two data sets. The data collected during the present study would appear to show more variability than the data collected by Baynes (2012). This is potentially due to seasonal differences, with higher levels of erosion occurring during December 2013 and an increase of deposition recorded during September; data were unavailable for these two months in the study by Baynes (2012).

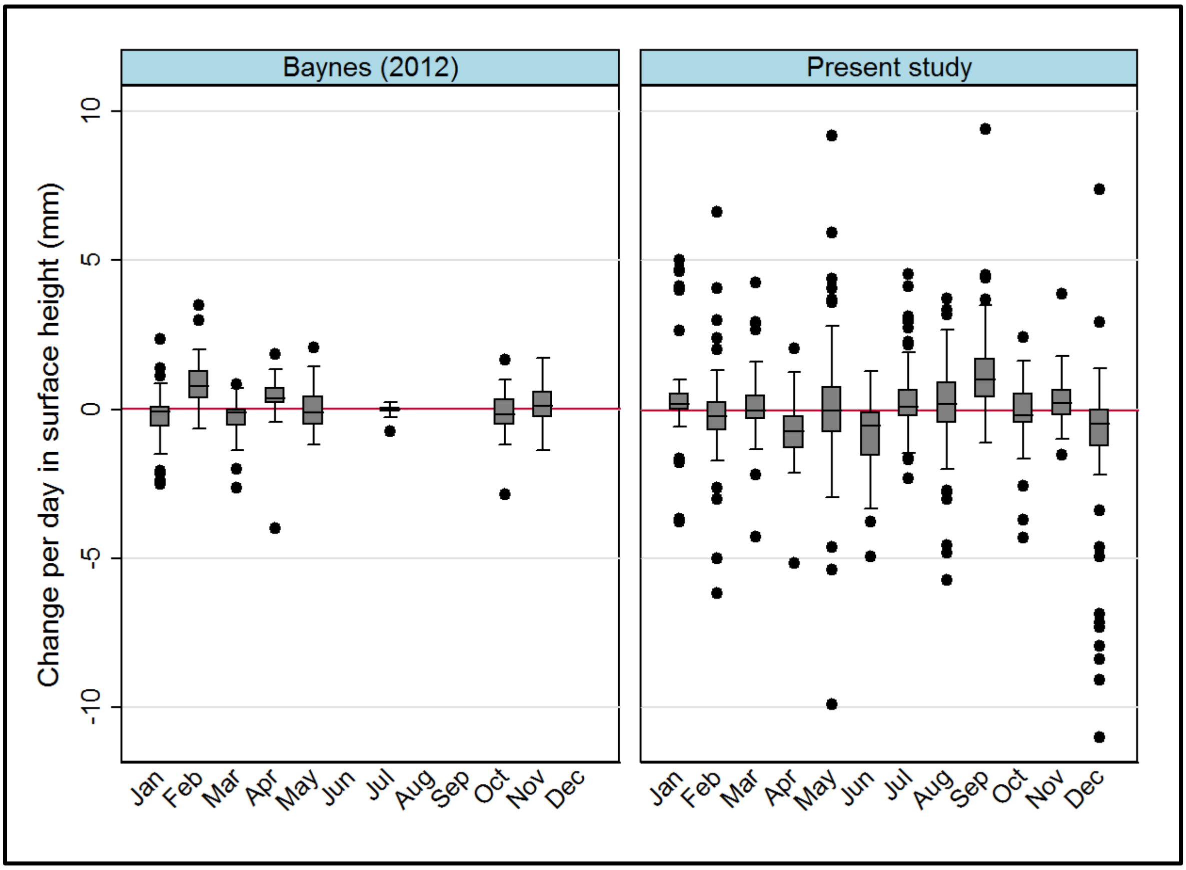


Figure 8.6 – *Box plots showing changes in surface height recorded using fixed pole transects by Baynes (2012) (measurements from November 2010 – July 2011) and during the present study (measurements from March 2013 – March 2014) The red line indicates the mean value of change per day (0.027 mm for the study by Baynes and -0.43 mm for the present study).*

Subsequently, the change per day in surface heights recorded using erosion pins were plotted (Figure 8.7). The mean values of -0.017 mm for the study by Baynes (2012) and -0.059 mm for the present study are indicated on the plots by the red lines. Figure 8.7 shows that in both data sets, there appears to be a seasonal signal, corroborating results outlined in Chapter 7 and suggesting that environmental conditions influence erosion patterns. Both studies have shown an overall slight net surface lowering recorded using erosion pins, with the mean value for the present study suggesting more erosion has occurred than in the study by Baynes (2012). There are several possible explanations for this difference. Firstly, as shown in Figure 8.6, the month where the greatest surface lowering of the peat haggs was recorded during this study was June. No data were available for June in the study by Baynes (2012). Furthermore, Baynes (2012) recorded an extensive snow cover at Flow Moss from November 2010 until January 2011, this resulted in no data being available during this period in the study by Baynes, and the thick layer of snow may have offered some protection to the peat surface from erosive agents such as wind, resulting in less erosion occurring during these months. Figure 8.7 shows that there was far more variation in surface height data collected during this study, again, this was likely due to differences in monitoring periods and weather conditions. To confirm whether this was the case, AWS data for both studies could have been compared alongside the measurements from the pole transects.

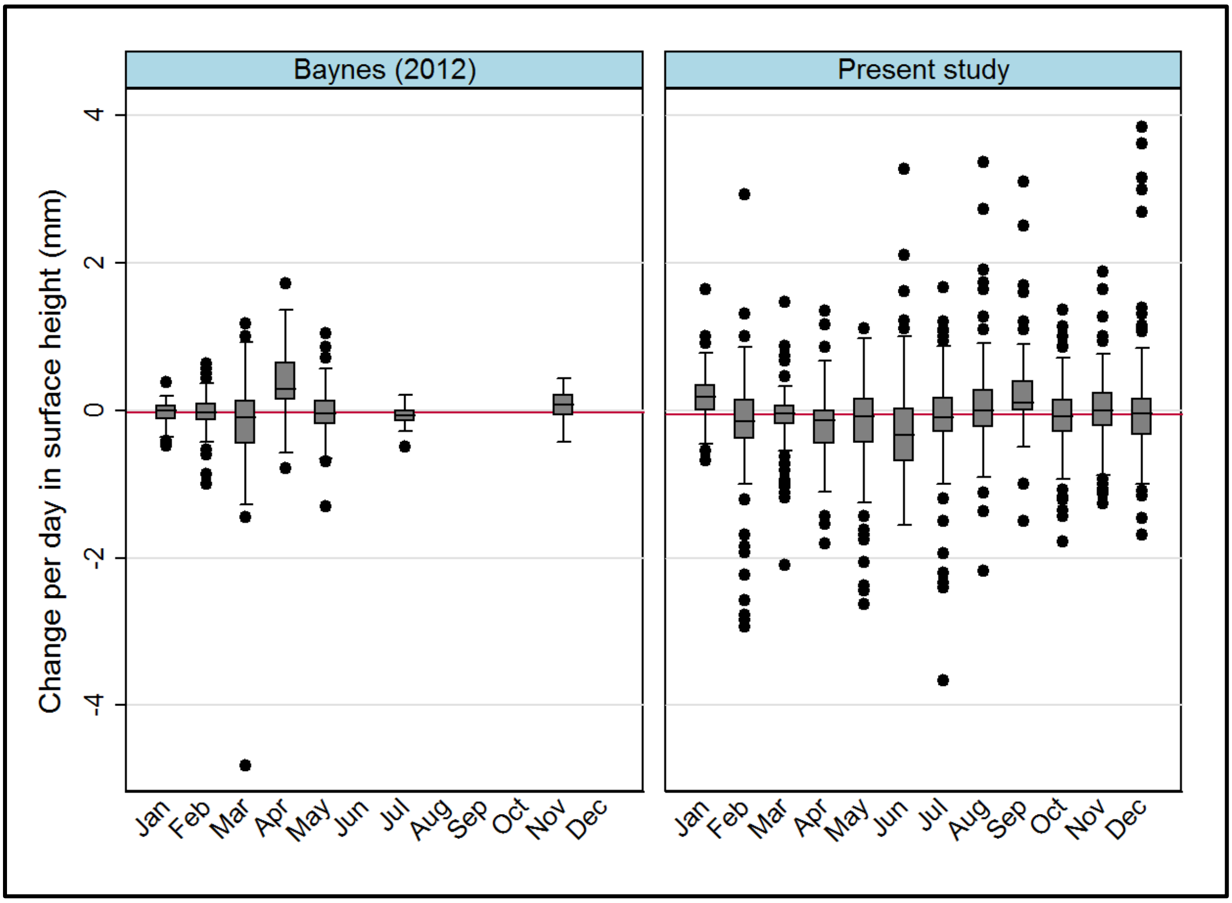


Figure 8.7 – *Erosion pin data recorded by Baynes (2012) (measurements from November 2010 – July 2011) and during the present study (measurements from March 2013 – March 2014).*

Both figure 8.6 and 8.7 indicate that there has been very little change in rates of erosion of the surface height of the bare peat since 2010. The mean rate of change per day in pole exposure of 0.027 mm for the study by Baynes and -0.043 mm during this study confirms this.

The total yields per month collected from the wind flux samplers for both this study and the study by Baynes (2012) are shown in Figure 8.8. This figure identifies that for the months where data were collected in both studies, the yields from the traps are of a similar magnitude, with the exception of the results for January. One possible explanation for this difference is the large amount of snow that Baynes (2012) reports occurred at Flow Moss between November 2010 and January 2011. During the present study, the greatest yield of sediment was recorded during December, which was a period of the highest recorded wind speeds and rainfall; however, no data were available for this month in the study by Baynes (2012) due to extensive snow cover. Furthermore, smaller yields would be expected as the snow cover would have offered a layer of protection to the peat surface, reducing the effect of erosive agents such as wind. The high yields of sediment recorded in December 2013 and January 2014 for the present study and the absence of data for the study by Baynes (2012) offer an explanation as to why, in the sediment budget, the sediment yields from aeolian processes recorded during this study are significantly higher than those recorded by Baynes (2012).

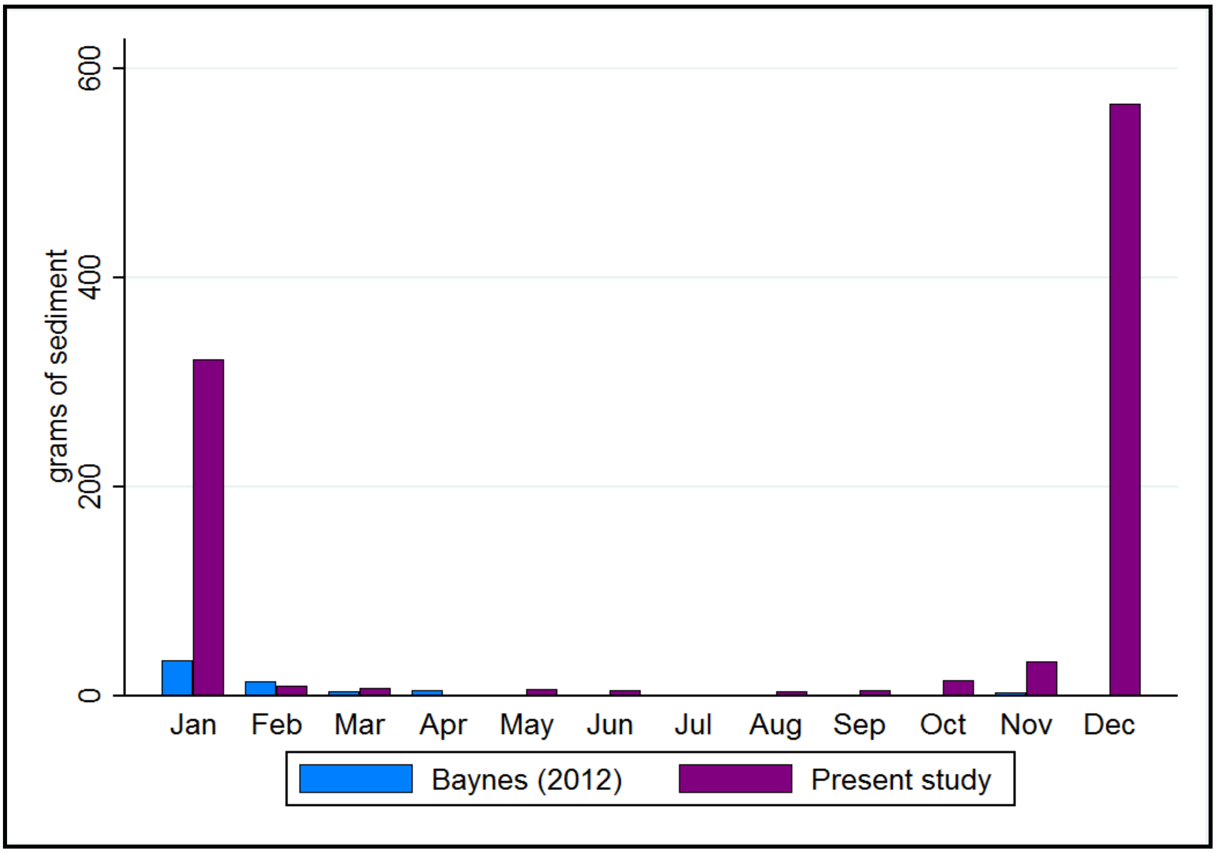


Figure 8.8 – *Sediment yields collected from the wind flux samplers each month.*

Figure 8.9 shows data collected from the sack traps at Flow Moss. Although the deposition pole transects demonstrate little change has occurred in the bare peat flats, when looking at yields collected from the sack traps, over the extended monitoring period a clear decrease can be observed. This shows that since the sack traps were installed in April 2011, there has been a reduction (approximately 98%) in sediment yield collected from the sacks, and exiting the site though fluvial processes.

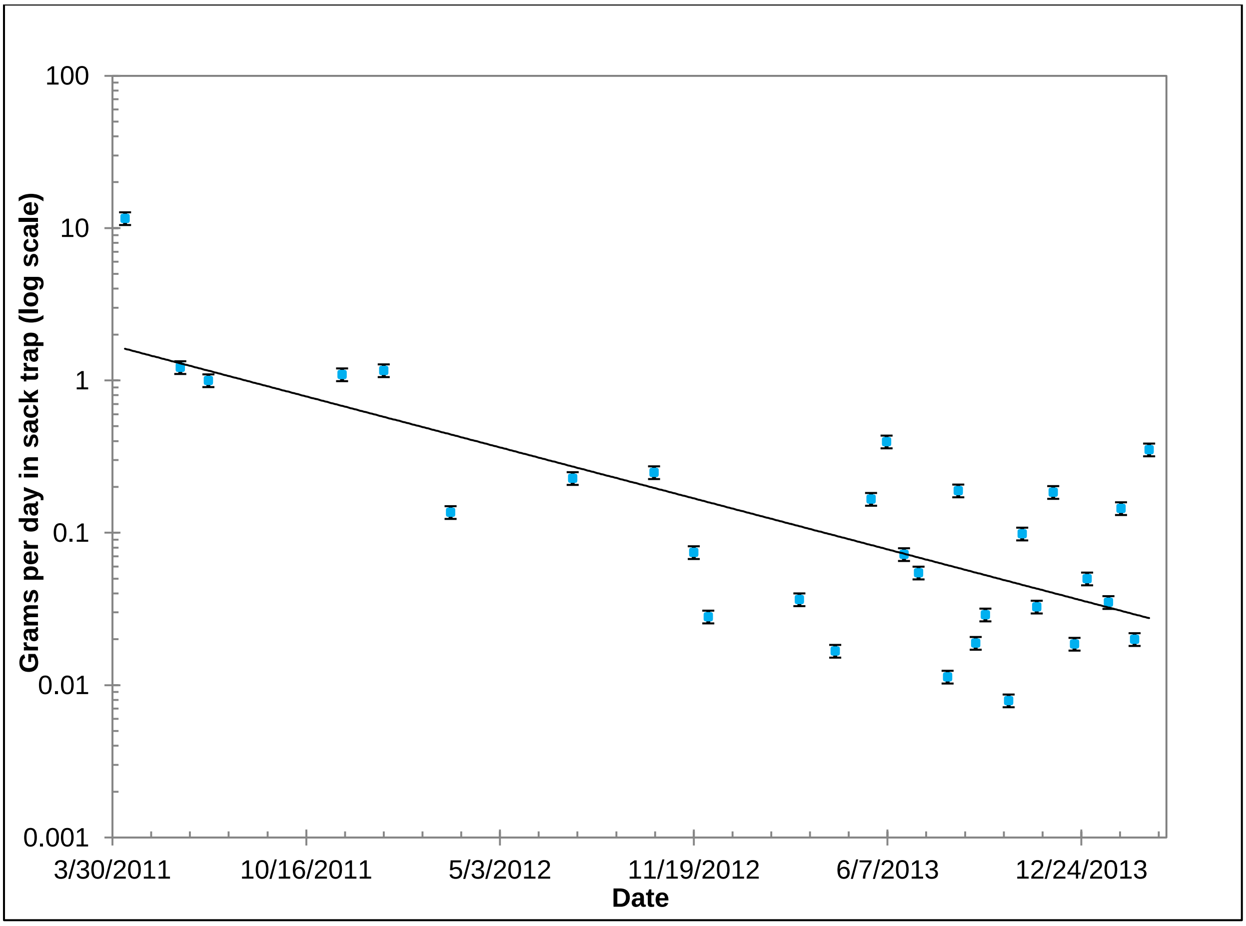


Figure 8.9 – *Data collected from the Flow Moss sack traps. A 98% decrease in sediment yield has occurred since April 2011*

There are two possible explanations for the observed decrease in erosion through fluvial processes. Firstly, encroachment of vegetation (Section 6.4) from the margins of the bare peat flats will increasingly intercept and store eroded sediment. Increasing vegetation can trap material before it reaches stream channels and will result in less erosion by reducing rain splash and offering protection to the peat surface from wind erosion. Changes in vegetation cover could partly explain why there is a seasonal signal displayed in the data with slightly higher yield being collected from the traps during the winter months. More recently, the coir rolls which were installed at the site in February 2013 to manage surface water flow on the margins of the bare peat may also be having an impact by damming the peat channels, reducing surface water velocities and encouraging local sedimentation.

8.3.2 Comparison of the Flow Moss Sediment budgets

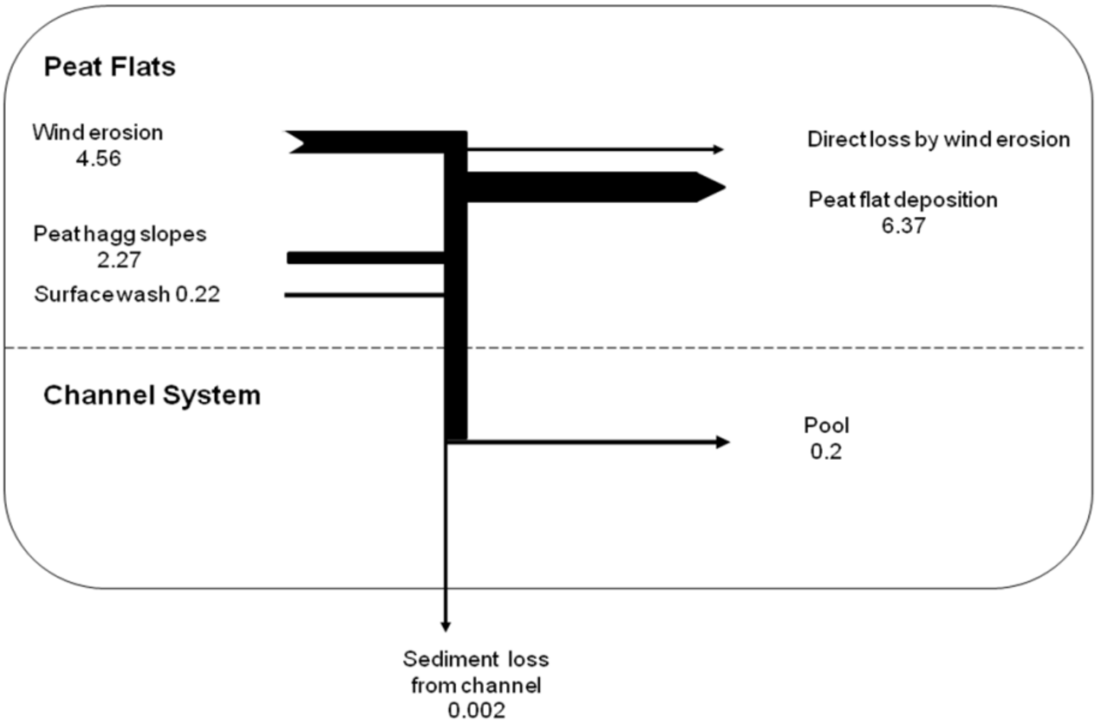
To assess how the erosion dynamics at Flow Moss have changed during the four years of restoration, a sediment budget was constructed from data collected by Baynes (2012) and compared with the sediment budget constructed using data collected in the present study. The sediment fluxes recorded during both studies were scaled up using the same methods (see Section 8.1) and annual values calculated (Table 8.5).

Table 8.5 — *Comparison of peat erosion process rates calculated from data collected in this study and data collected by Baynes (2012).*

Process	**Sediment flux ($t\ a^{-1}$) - 2010 Baynes (2012)**	**Sediment flux ($t\ a^{-1}$) -2013/14 Present study**	**% change**
Wind erosion of peat flats	4.56	35.28	+ 774
Loss from peat hagg slopes	2.27	1.56	- 69
Fluvial transport (at catchment outlet)	0.0018471	0.000036	- 98
Deposition in pools	0.21	0.38	+ 181
Deposition on bare peat flats	6.37	20.11	+ 316

The data shown in Table 8.5 are used as a basis of Figure 8.8 and both clearly demonstrate a substantial difference in the sediment flux from wind erosion of the peat flats in contrast to other processes. There are several possible reasons for this. Firstly, as previously discussed, during the 2010 study period of Baynes (2012) there were several months where the bare peat was covered by a thick layer of snow; this could have provided a layer protecting the bare peat flats from erosion by wind and rain. Furthermore, the data (Table 8.5) indicate that deposition of peat on the bare peat flats contrasted markedly between the two sediment budgets with a difference of + 316% in the latter budgeting period. Both studies identify that the main mechanism of erosion at Flow Moss is caused by aeolian processes; but equally both budgets show that large amounts of peat are redistributed within the bare peat flats, and only a very small amount of sediment is actually lost from the site. The data in Table 8.5 demonstrate a 98% decrease in the yield of sediment collected in the sack traps at the catchment outlet. This large decrease is likely the result of increasing vegetation cover and greater peat trapping by the coir rolls. Increased deposition in the peat pool (an increase of 181%) also explains the falling yield. Finally, although every attempt has been made to standardise data collection during the two sediment budget periods, inevitably some differences occur. For example, the lying snow cover in December 2010 meant data collection was impossible and so there is no data available for the wind flux tubes from this month. The sediment budgets from the two different years indicate that there may be a large amount of inter-annual variability. This would suggest that for a true indication of what is occurring, sediment budgets need to be implemented spanning several years rather than shorter amounts of time.

(a)



(b)

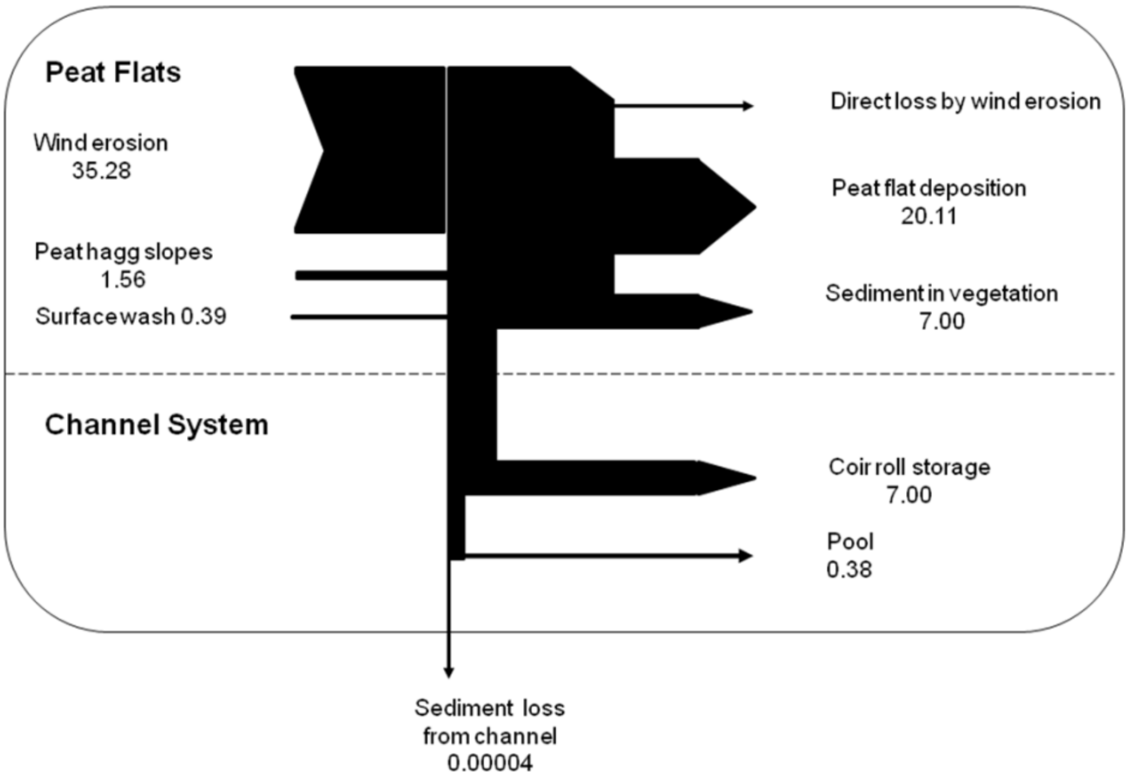


Figure 8.10 – (a) *Sediment budget created using data collected by Baynes (2012).*
(b) *Sediment budget created from data collected during the present study*

8.4 The effectiveness of current restoration measures at Flow Moss

The final research question to be addressed during this study was 'How have restoration methods at Flow Moss impacted on sediment yields and carbon loss via erosion?' This study aimed to assess whether restoration measures implemented at Flow Moss have proven successful in terms of reducing erosion and sediment transfer and reducing the area of bare peat through re-vegetation. The 7 ha site was fenced off in April 2010 in an attempt to reduce erosion from sheep grazing, and alongside this brash and vegetation seeds were spread (November and December 2010 and April 2011) in an attempt to reduce the area of bare peat. Figure 6.27 in Section 6.3 showed the mapped area of bare peat at Flow Moss, and it is possible to see that the bare peat area has reduced substantially. The area of bare peat reduced in area by 14% between January 2007 and April 2011 and 12% between April 2011 and April 2014. Overall, the bare peat area has reduced by 22% between January 2007 and April 2014. Installation of coir roll dams in February 2013 have aided further trapping of peat material on site (Figure 8.3 **(a)**) although much of this trapping occurs in the natural pool systems and by increasing vegetation .

As long as sediment export by fluvial action from the site remains low the key issue for the restoration of Flow Moss is the stabilisation and re-vegetation of the bare peat areas. This is a significant challenge due to the high levels of geomorphic activity recorded at the site which severely limits the effectiveness of simple brash spreading.

8.5 Issues for future study

This project has produced an annual sediment budget which focused on sediment lost from the terrestrial system at Flow Moss. As with all field-based studies, there were areas of the methodology which could be improved. Firstly, future studies could be expanded to monitor the flux of DOC leaving the site and directly measure the flux of carbon lost and sequestered through gaseous processes to provide a complete carbon budget without having to estimate carbon loss and sequestration. Secondly, the study could be expanded to cover a larger area of the actively eroding site. The present study focused on the area of bare peat at the central and northern end of the site and it is likely that the erosion dynamics encountered at the southern, more vegetated part of the site would be different to the results recorded here. Increasing the area of monitoring would assist in removing some of the uncertainty in scaling up measurements. In addition, an extended period of monitoring is necessary to ascertain the longer-term effectiveness of restoration measures implemented at Flow Moss. The success of peatland restoration can only been seen over long time scales (Evans and Warburton, 2007; Waddington *et al*., 2011) and it is unlikely that the full impact of the restoration measures will have been identified during this study. Thirdly, further monitoring of peatlands using TLS is needed to remove some of the limitations encountered during this study. Fixed objects with a flat reference surface should be used in an attempt to provide planes to be used during the registration process.

Possibly the greatest uncertainty in the restoration of peatlands is how these systems will respond to future climate change. Rowson *et al*. (2013) discuss the importance of quantifying how peatland ecosystems will respond to environmental variables in an attempt to gain information relating to how projected changes in climate may affect peat carbon cycles. One possible way to gain an understanding of how peatlands may respond to future climate change could be by looking at rates of past erosion. This study has attempted to use metals concentrations of peat cores as a proxy indicator of periods of deposition and erosion (Chapter 5, Section 2). This could be established by obtaining dates from the peat which could be used to confirm the state of environmental conditions when deposition or erosion were most active. The C/N ratio of the peat samples could also be used to show decomposition within the peat and to assist with the cross reference of the core samples. Once the cores have been cross referenced, metal peak concentrations from the peat cores could be used alongside historical mine and smelting records to estimate approximately when the peat was deposited. The past environmental record could then be used to identify environmental conditions at this time and validate models of peat erosion under future climate scenarios. Finally, extending similar monitoring studies of blanket bogs to other locations will allow the comparison of erosion dynamics and mechanisms of physical processes to be compared to establish if processes identified at Flow Moss are typical of an upland blanket peat bog.

8.6 Chapter summary

This chapter has discussed the data presented in Chapters Five, Six and Seven. The results indicate that the carbon store at Flow Moss is currently relatively stable. Figure 8.5 shows projections of rates of re-vegetation and carbon loss/sequestration for Flow Moss and suggests that if the rate of re-vegetation of the bare peat remains the same, Flow Moss could cease to act as a net source of carbon and become a net sink within 20 years. Nonetheless, this could change, especially given the uncertainties associated with future climate scenarios. The comparison of data collected by Baynes (2012) in 2010 and data collected during this study (2013/14) indicate that restoration practices implemented at Flow Moss have shown a measured level of success. Particulate peat yields collected in the sack traps in streams draining the site have shown a reduction of 98% between April 2010 and March 2014. When comparing changes in surface elevation of the bare peat flats however, over a similar period, the variability recorded in the present study (Figure 8.6 and 8.7) is far greater. This emphasises the fickle nature of the inter-annual variability of small catchment sediment budgets. Furthermore, when comparing sediment budgets constructed using data from the two studies, there is a large amount of variability. This suggests that for a true representation of erosion dynamics, monitoring over far longer timescales is necessary to attempt to distinguish true changes from inter-annual variability.

9. Conclusion

The aim of this research was to implement an extended monitoring project to assess the impact of peat bog restoration in mitigating carbon loss by upland erosion. More specifically, the project has estimated the size of the carbon store at Flow Moss (North Pennines, UK), identified the main drivers and pathways through which sediment and carbon are leaving the site, and as such indirectly investigated the effectiveness of restoration methods in reducing peat loss through erosion. This chapter outlines the main conclusions reached during the study.

9.1 Research questions and objectives: key findings

This research aimed to answer three key research questions (Section 1.1):

1) How can peatland carbon stores be accurately assessed, and what is the local carbon store at Flow Moss?

In order to answer this question, research objectives one and two (Chapter 1.1) were fulfilled. These objectives related to the subsurface properties of the peat. In Chapter 5.3.1, the size of the Flow Moss carbon reservoir was estimated to be 4004 (±0.03) tonnes. The results from the synthetic transect spacing experiment (Section 5.3.2) showed that sampling strategy has a profound impact on estimated peat depths and survey design is vitally important in the assessment of peat resources. The results demonstrated that depth surveys undertaken using a GPR intensive transect approach provide data of a far higher spatial resolution than that which could be collected using a manual depth probe. Furthermore, when using the high resolution GPR peat depth model to simulate a 100 m depth probe survey, the results showed a massive underestimate of both peat volume and maximum peat depth, with peat volume being under estimated by 27%.

Chapter 8 discussed the calculation of a simple Flow Moss carbon budget and showed that currently, the terrestrial store of carbon at Flow Moss is relatively stable. The estimated carbon budget suggests that approximately two tonnes of carbon are currently lost from Flow Moss each year. Although this is small, implementing successful restoration methods are still of great importance because if the current area of bare peat is successfully re-vegetated, Flow Moss will cease to act as a carbon source and become a net sink. Figure 8.5 in the discussion Chapter predicts that as vegetation cover increases, the amount of carbon lost through erosion will continue to decrease, until a balance point is reached between vegetation cover and carbon emissions in approximately 20 years. Subsequently, carbon sequestration at Flow Moss will continue to increase, reaching 1 t C yr^{-1} by approximately 2040. These predictions could be impacted by future climate change with increasing temperatures and decreasing rainfall resulting in greater amounts of carbon loss due to increased microbial activity.

2) What are dominant processes driving erosion at Flow Moss?

The above research question was answered using Objectives 4 and 5 (Chapter 1.1), which were to construct a sediment budget for Flow Moss and monitor environmental conditions which occurred during monitoring. Results reported in Chapters 6, 7 and 8 can be used to answer this research question. The sediment budget constructed for Flow Moss indicates that the bare peat flats are currently geomorphologically very active and sediment transfer is predominantly driven by high intensity rainfall and high wind speed events. Aeolian processes are the main method of sediment transport on the bare peat flats, however particle detachment by rainfall, frost heave and surface desiccation result in greater yields of sediment being available for transport. The majority of sediment is re-distributed on the bare peat flats and in the adjacent vegetation and very little is lost from the site. During this study, TLS was used to monitor changes in surface height between repeat surveys. The results were compared to those recorded using pole transects and showed a positive correlation between the two data sets (R^2 = 0.74). This indicates that TLS can provide a suitable alternative method for quantifying changes in peat surface height over a spatial scale far greater than that which can be achieved using point measurements. Nonetheless, there are limitations to the methods that need to be taken into account, and one of the greatest issues encountered during this study was that of whether the small scale changes in the height of the peat surface were within the detectable limits of the TLS system.

3) How have restoration methods at Flow Moss impacted on sediment yield and carbon loss via erosion?

To answer this research question, two research objectives were defined (Chapter 1.1). These objectives were to construct a sediment budget for Flow Moss and undertake surveys of change in vegetation cover.

Results from the sediment budget indicate that a large decrease (98%) has occurred in the yield of sediment being lost from the site through the channel system. Baynes (2012) estimate that 0.0018 t a^{-1} of sediment was being lost through the channel system, while this study found the value to be 0.000036 t a^{-1}. Although these values are small, the relatively large decrease in the peat lost through the channel system suggests re-vegetation is having an impact on sediment transfer, and more recently, the coir rolls used to dam the channel systems are starting to have an impact.

The results displayed in Section 6.3 indicate that since 2007, the area of bare peat at Flow Moss has reduced by 2977 m^2, or 21.6%. Restoration attempts at Flow Moss began in 2010 and between 2011 and 2014; the area of bare peat has reduced by 2000 $m^{2,}$ or 14.5%. This demonstrates that attempts to re-establish vegetation cover have been partly successful; despite there still being a large area of bare geomorphologically active peat which needs re-vegetating.

9.2 Conclusions

The main conclusions of this study are:

- Erosion processes at Flow Moss are driven by the environmental conditions preceding the erosion event. Wind speed is a key driver of erosion at Flow Moss, however erosion though aeolian processes is limited by sediment availability. Most erosion occurs when high wind speed events coincide with high intensity rainfall or frost heave events which are responsible for disturbing the bare peat surface and promoting sediment entrainment.
- The sediment budget outlined in Chapter 8 highlights several key processes. Firstly, the majority of sediment (35.28 tonnes) is transported by aeolian processes. However, it is estimated that 20.11

tonnes of this peat is re-deposited on the bare peat flats, or within the surrounding vegetation and only a small fraction is actually leaving the site. The amount of POC leaving the site is very small (0.000036 tonnes) and this occurs through small, ephemeral channels located at the catchment outlet.

- The results outlined in this study indicate that TLS is a suitable tool for monitoring changes in elevation of the peat surface over areas far larger than those which could be monitored using fixed point measurements. Nevertheless, there are limitations to the method that need to be taken into account, particularly in terms of detection limits and accounting for error in the TLS data.
- Currently, the terrestrial store of carbon at Flow Moss is estimated to be 4004 tonnes. This is estimated from a high resolution GPR survey which covered the entire Flow Moss site in a network of transects spaced two meters apart. The synthetic transect spacing experiment conducted as part of this study indicates that sampling strategy used in peat depth estimation has a great impact on the peat volume estimates, with a discrepancy of 27% between peat volumes estimated from the data collected at a 2 m spacing, and the volumes estimated from the model created from data on a 100 m grid (aiming to replicate a manual peat probe depth survey).
- The carbon budget estimated for Flow Moss suggests approximately 1.6 tonnes or 0.05% of the carbon store is being lost each year, indicating that Flow Moss is a net source of *C*. Since restoration and monitoring began in 2010, there has been a decrease in the small

volume of peat lost by fluvial transport through the channel system. This would indicate that the restoration practices which have been implemented have had a positive effect. However, the area of bare peat at Flow Moss has only reduced in size by 2966 m^2 or 22% since January 2007, with a reduction of 997 m^2 or 12% occurring since restoration began. This suggests future efforts should be targeted at re-vegetating the bare peat flats.

The restoration of peatlands requires the co-operation of land owners and stakeholders who may all have different goals and objectives. Perhaps one of the greatest challenges for future peatland restoration and monitoring projects will be encouraging landowners, stakeholders and policy makers to find a balance where peatlands are protected and restored without impacting the land use needs of stakeholders in the local areas.

Appendix A

A1. Post processing of TLS data

TLS data collected during each field visit were loaded into RiScan projects and the point clouds merged and registered using fixed tiepoint locations for which the dGPS coordinates were known. At this stage, the point clouds were cleaned to remove erroneous points such as ‘air shots’ which may be caused by dust particles or raindrops. RiScan’s Multistation Adjustment (MSA) tool was used to improve the registration between the six point clouds (one collected from each scan station). During this process, the ‘prepare data’ function was used. This process identifies flat surface patches within the point cloud and assigns each of these a point indicating the centre of gravity of all the points within a defined area. The MSA tool matches these points by iteratively modifying the position and orientation of each scan position until the error is below the user defined threshold (Riegl, RiScan Pro Manual, 2005). Once this stage in processing was achieved, the remaining processing was implemented using two different workflows (Figure A1). The two different workflows were used to establish whether registering data in different ways impacted on the change detection results. During the MSA process, an error value is provided. This value is representative of the standard deviation of the distance between the plane patches which the MSA process is attempting to match. However, the parameters used when implementing the plane patch filter may affect the MSA value and a false level of accuracy may be achieved. MSA values were produced when the six point clouds for each epoch were registered together and when the data sets from different epochs were registered. The MSA values are displayed in Table A1.

The point clouds were filtered using RiScan's in built 'Octree' filtering. This filter uses an Octree structure based on a cube which is divided into 8 equally sized cubes which are divided iteratively. The extension of the base cube is input (i.e. the resolution at which the data are to be filtered) and after the generation of Octree filtering, one cube contains a single point, which is the centre of gravity of all the points which were previously located within this cube (RiScan Pro manual. 2005). Once the data had been processed in RiScan, polydata files were created and exported at three different resolutions, (20 cm, 2 cm and unfiltered) as ASCII files of *x,y,z* data points. These resolutions were selected as DEMs filtered at 20 cm and 2 cm have previously been used by Grayson *et al*. (2012) in a study utilising TLS for peatland monitoring. Unfiltered data were exported and later gridded at 1 cm for comparison of results obtained at 2 cm resolution.

Once the data had been exported the files were loaded into Envi (Exelis Visual Information Solutions, Boulder, Colorado, version 5) and the 'rasterize point data' function used to create rasters from the exported ASCII files. The 20 cm and 2 cm data were gridded at 20 cm and 2 cm respectively and the unfiltered data were gridded at a 1 cm resolution. The point density of the point cloud was found to be 0.8 per cm^2, therefore gridding the data at a resolution of less than one cm could have led to over sampling of the data and thus a false result when calculating volume differences between DEMs.

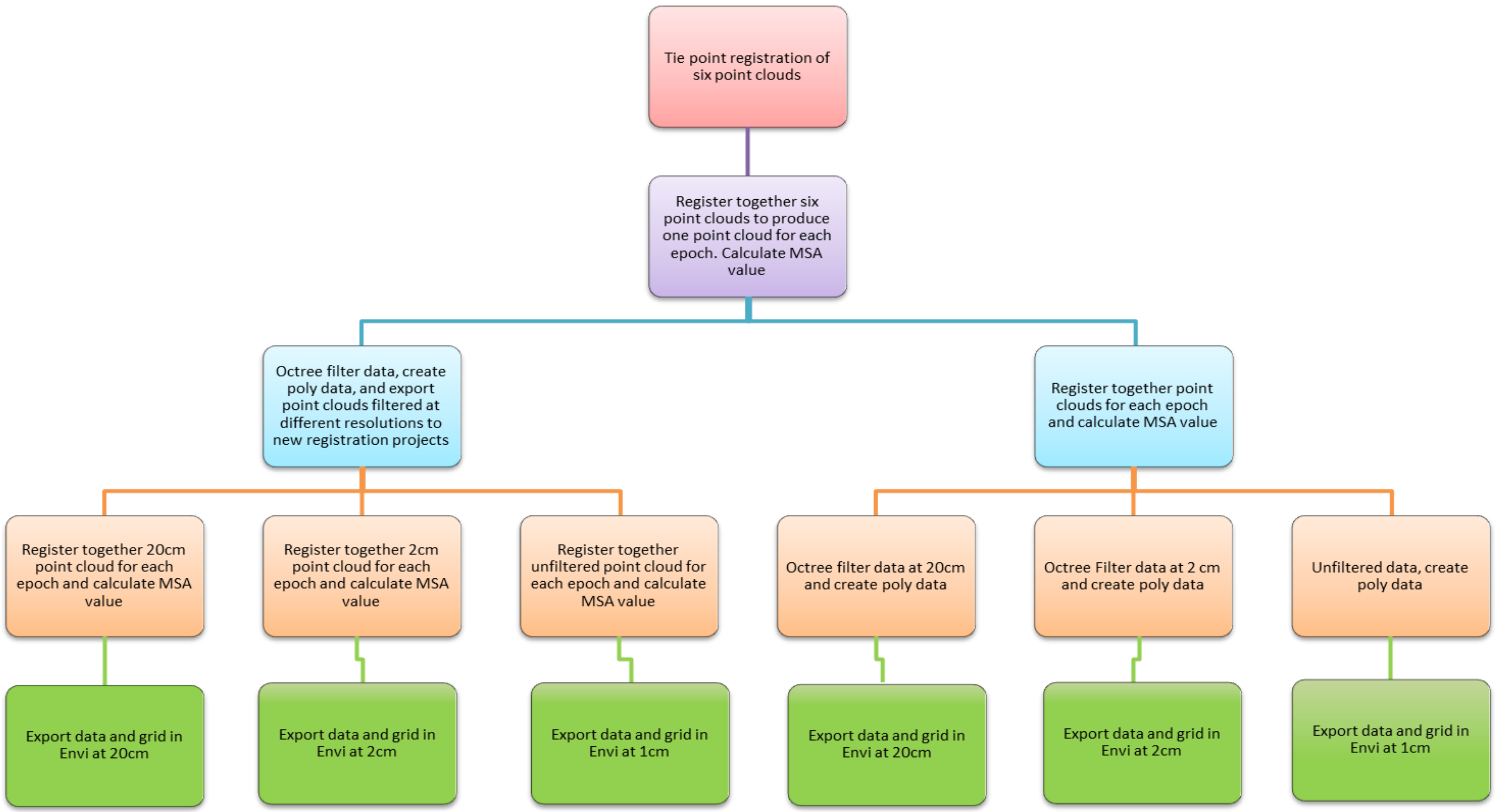


***Figure A1**– A flow chart outlining the work Flow Process in RiScan*

Table A1 - *Error values for the target registration, MSA for each epoch and MSA values when epochs of different resolutions were registered together*

	Target registration (m)	**MSA within month**	**Overall MSA 0.2m (m)**	**Overall MSA 0.02m (m)**	**Overall MSA nf (m)**	**Overall MSA all data sets (m)**
06/06/2013						
1	0.0126	0.0083	0.0067	0.0048	0.0068	0.0071
2	0.0155					
3	0.0141					
4	0.016					
5	0.0096					
6	0.0126					
08/08/2013						
1	0.018	0.007	0.0067	0.0048	0.0068	0.0071
2	0.0176					
3	0.0181					
4	0.0089					
5	0.0132					
6	0.0159					
19/08/2013						
1	0.0189	0.0076	0.0067	0.0048	0.0068	0.0071
2	0.0159					
3	0.0193					
4	0.0174					
5	0.0186					
6	0.0192					
16/09/2013						
1	0.0179	0.0072	0.0067	0.0048	0.0068	0.0071
2	0.0165					
3	0.0173					
4	0.0143					
5	0.0132					
6	0.0162					
10/10/2013						
1	0.0169	0.0082	0.0067	0.0048	0.0068	0.0071
2	0.0167					
3	0.0182					
4	0.0114					
5	0.0185					
6	0.018					

24/10/2013						
1	0.0189	0.0069	0.0067	0.0048	0.0068	0.0071
2	0.0176					
3	0.0201					
4	0.0248					
5	0.0264					
6	0.0237					
08/11/2013						
1	0.0182	0.0063	0.0067	0.0048	0.0068	0.0071
2	0.0187					
3	0.0154					
4	0.0139					
5	0.0188					
6	0.02					
25/11/2013						
1	0.0192	0.0062	0.0067	0.0048	0.0068	0.0071
2	0.0196					
3	0.0199					
4	0.014					
5	0.0146					
6	0.0136					
04/04/2013						
1	0.0159	0.007	0.0067	0.0048	0.0068	0.0071
2	0.0185					
3	0.0187					
4	0.0133					
5	0.0195					
6	0.0158					

Once DEMs had been created, the earlier DEM was subtracted from the later DEM (e.g. 25th November 2013 subtracted from 4th March 2014) to create a DEM of Difference (DoD) identifying differences in surface heights which had occurred between the two epochs.

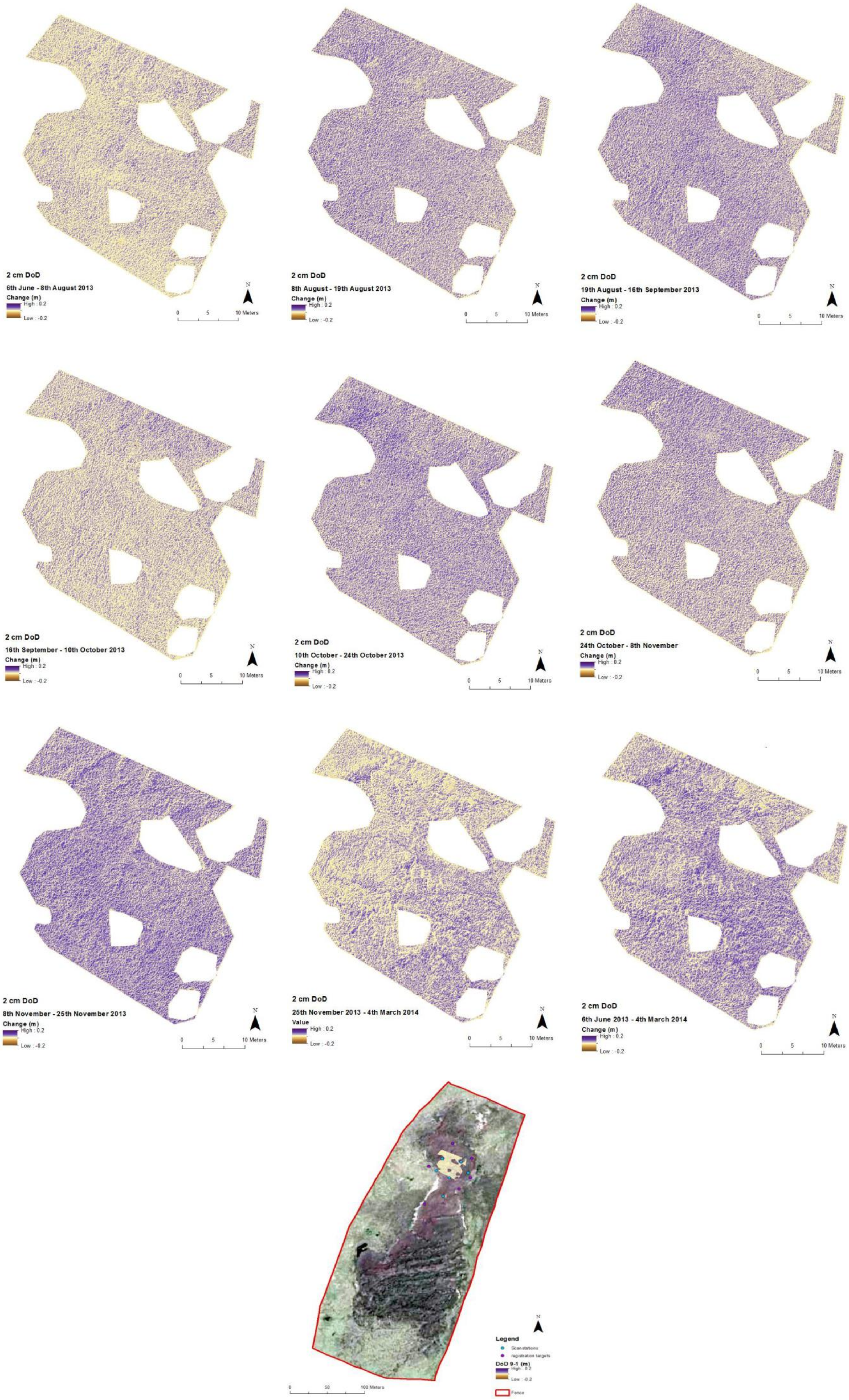


Figure A2 – *DoDs at a 2 cm resolution with areas of vegetation and peat haggs masked out.*

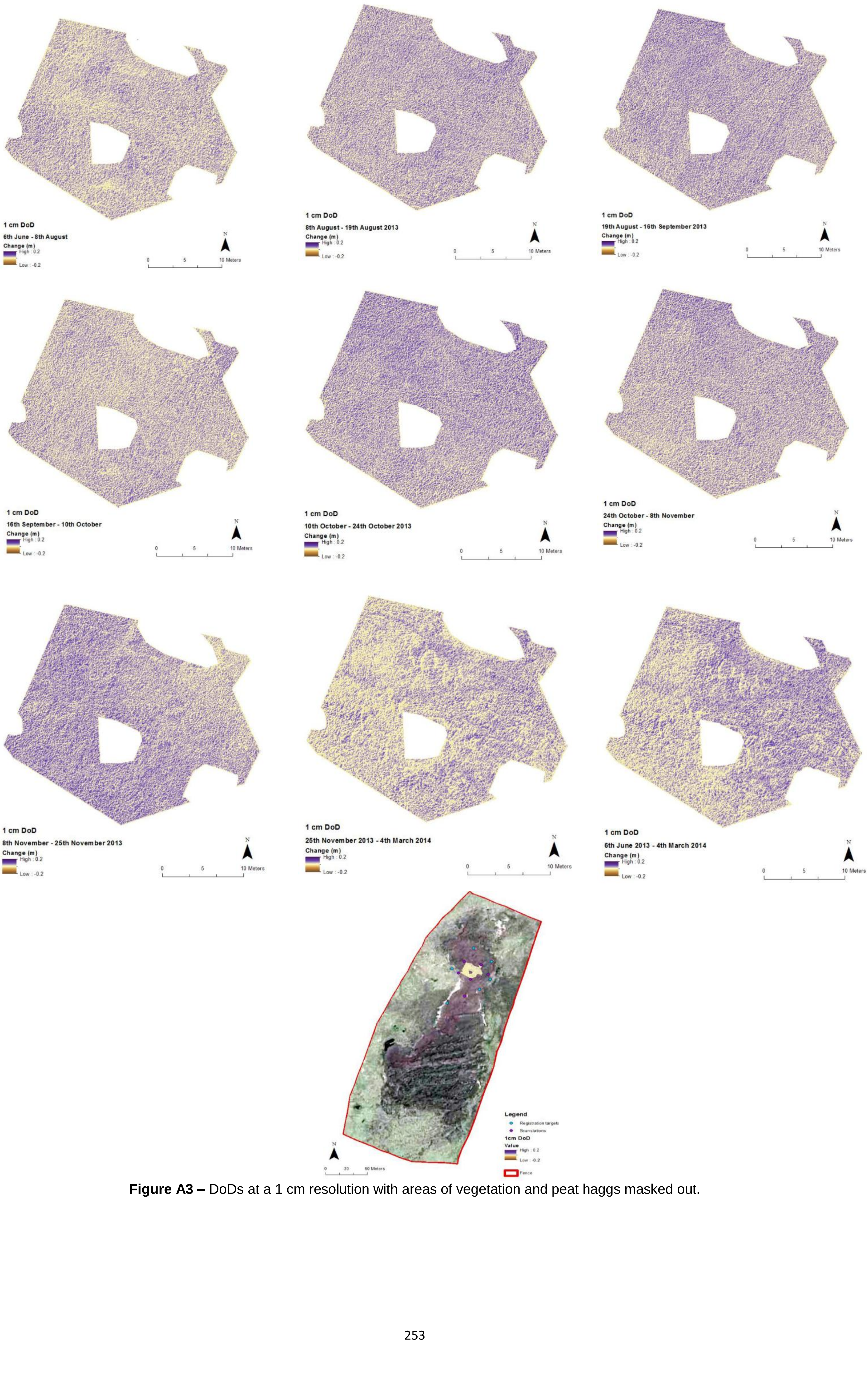


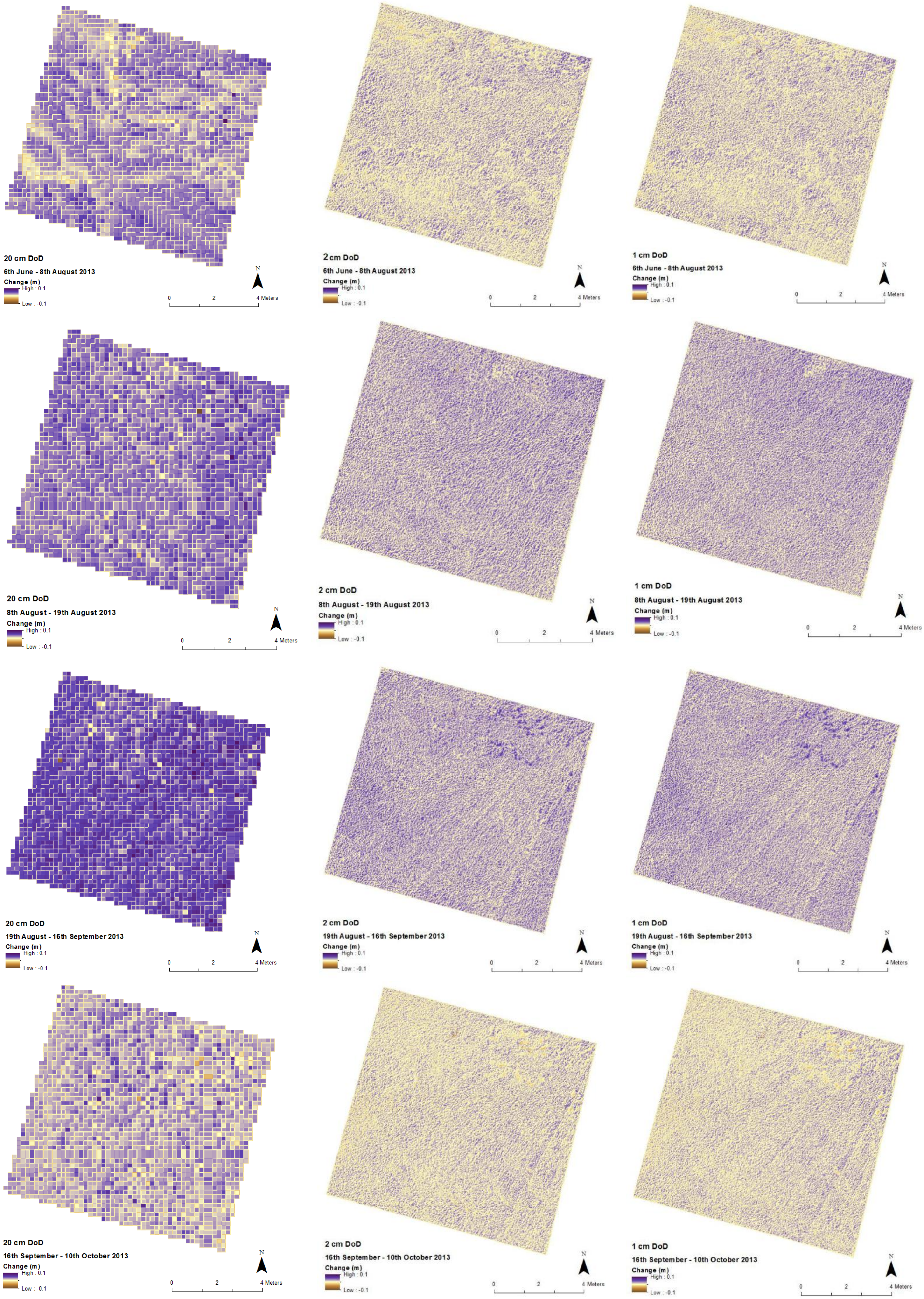
Figure A3 – DoDs at a 1 cm resolution with areas of vegetation and peat haggs masked out.

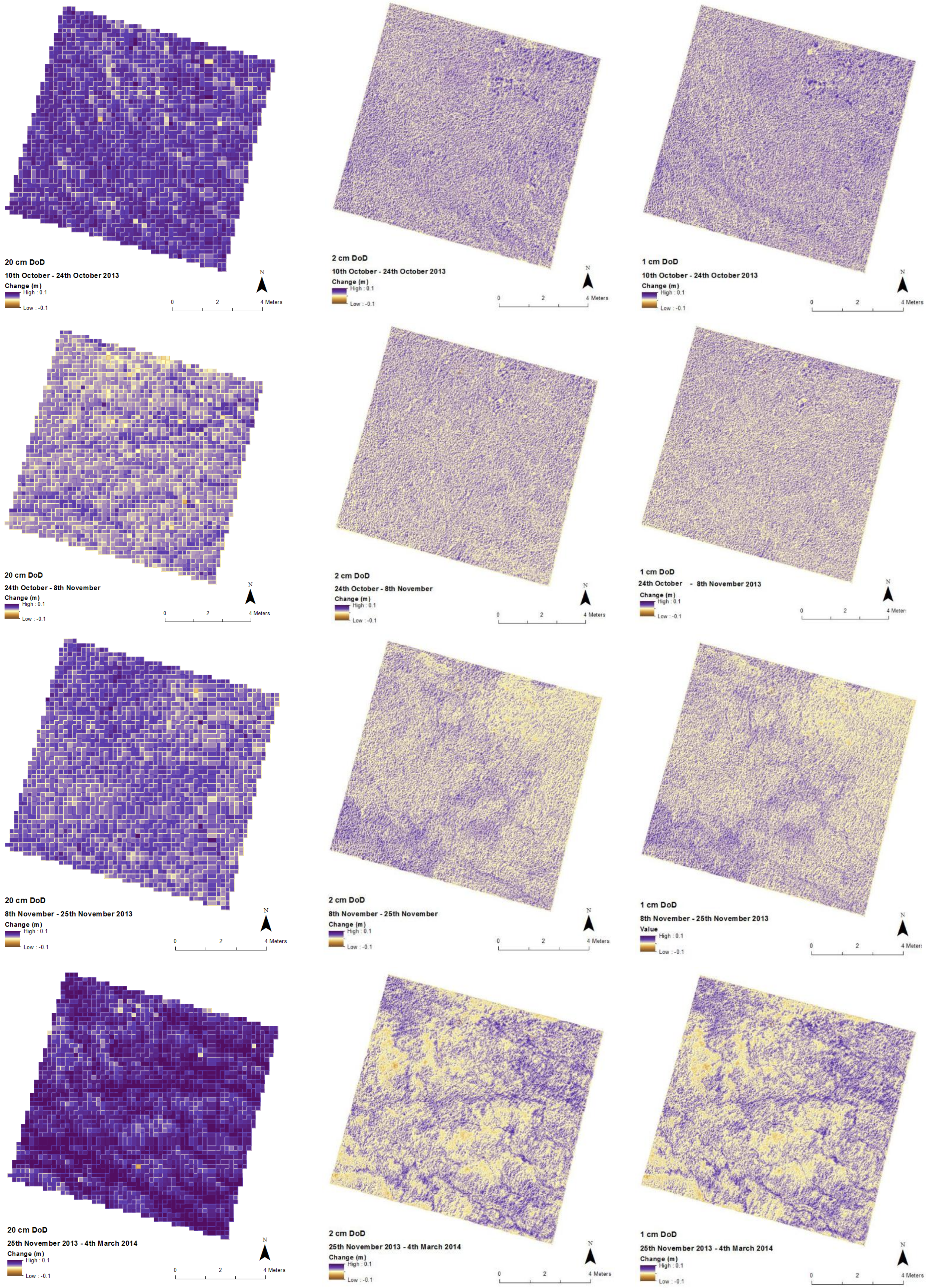
A2. Comparison of different filtering resolutions and MSA

To assess the impact of point cloud filtering on the volume differences calculated from the TLS data, a sample area of the TLS point cloud was filtered and gridded at three different resolutions and used to calculate a volumetric change and changes in surface height occurring between epochs. The selected area was 96 m^2 and purposely chosen as it was an area of bare peat known to contain no vegetation, monitoring equipment or peat haggs. The DEMs gridded at a 20 cm, 2 cm and 1 cm resolution were used to create DoDs of this area. These are shown in Figure A4.

Figure A4 shows a large difference in detail of the DoDs created from data filtered at 20 cm and DoDs created from data filtered at 2 cm and 1cm. As would be expected, the data created from DEMs of a higher resolution show much more detail and it is possible to see areas of micro-topography forming on the peat surface (e.g. the DoD produced from DEMs created with data captured on 25th November 2013 subtracted from the DEM created from data captured on 4th March 2014).

During point cloud processing in RiScan, MSA values were produced; the MSA values were considered the limits of the detectable change using TLS. Values that fall within the ±MSA value (shown in Table 6.2) were considered erroneous and removed from the DoDs and the difference values re-calculated. The DoDs with the error threshold implemented are shown in Figure A5.


20 cm DoD
6th June - 8th August 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
6th June - 8th August 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
1 cm DoD
6th June - 8th August 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
20 cm DoD
8th August - 19th August 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
8th August - 19th August 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
1 cm DoD
8th August - 19th August 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
20 cm DoD
19th August - 16th September 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
19th August - 16th September 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
1 cm DoD
19th August - 16th September 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
20 cm DoD
16th September - 10th October 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
16th September - 10th October 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
1 cm DoD
16th September - 10th October 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N



20 cm DoD
10th October - 24th October 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
10th October - 24th October 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
1 cm DoD
10th October - 24th October 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
20 cm DoD
24th October - 8th November
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
24th October - 8th November
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
1 cm DoD
24th October - 8th November 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
20 cm DoD
8th November - 25th November 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
8th November - 25th November
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
1 cm DoD
8th November - 25th November 2013
Value
High : 0.1
Low : -0.1
0 2 4 Meters
N
20 cm DoD
25th November 2013 - 4th March 2014
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
25th November 2013 - 4th March 2014
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
1 cm DoD
25th November 2013 - 4th March 2014
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N

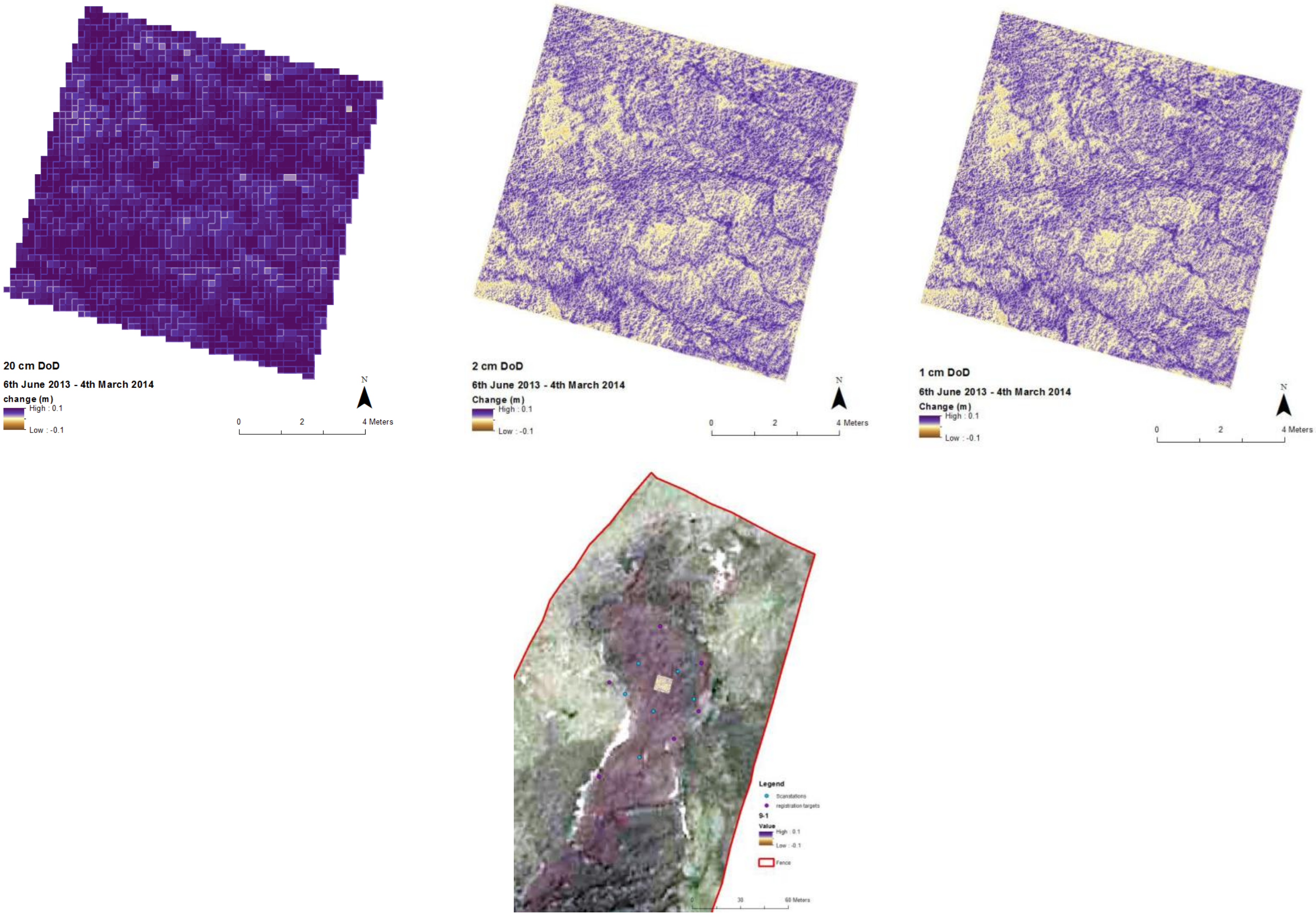
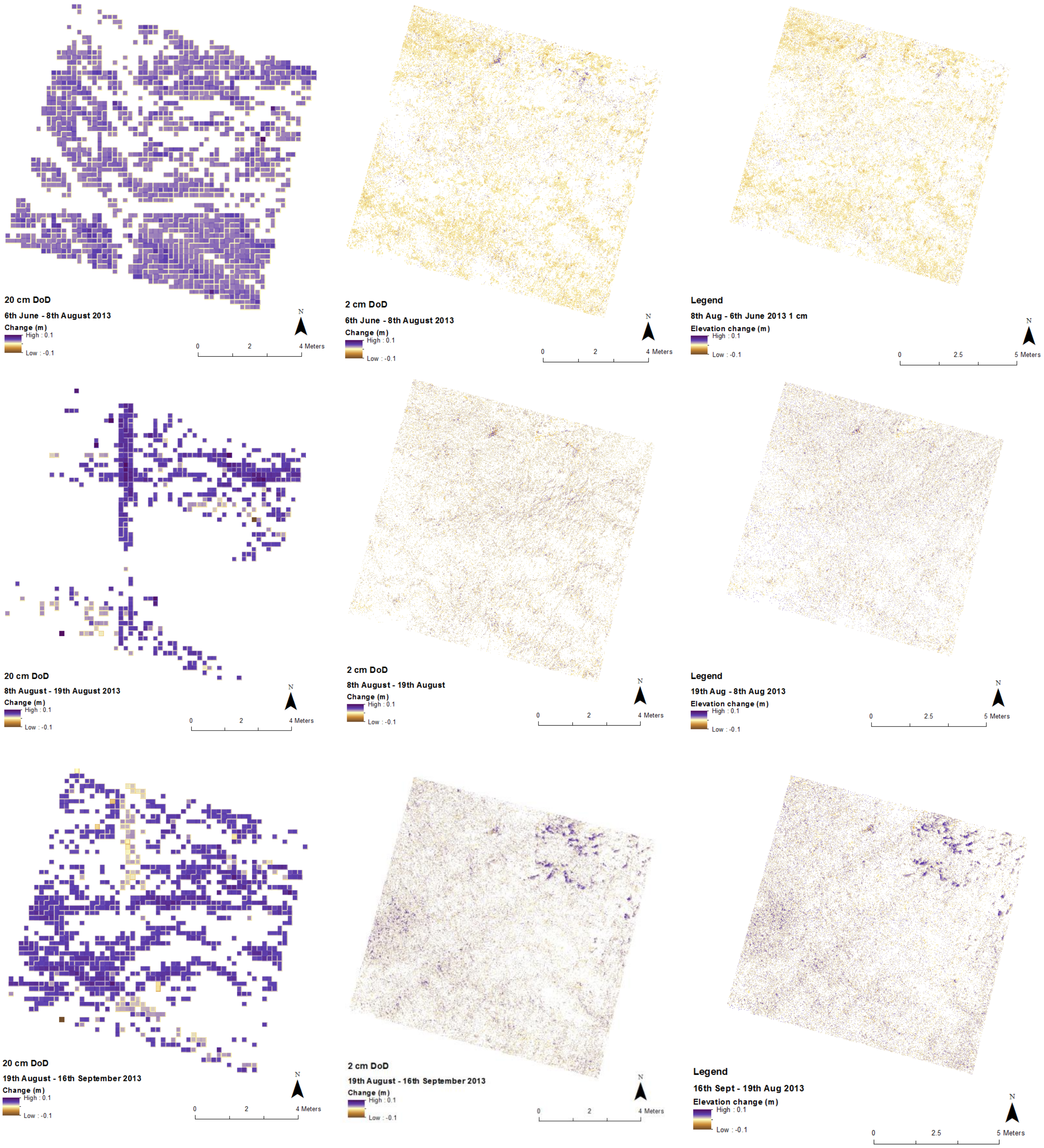
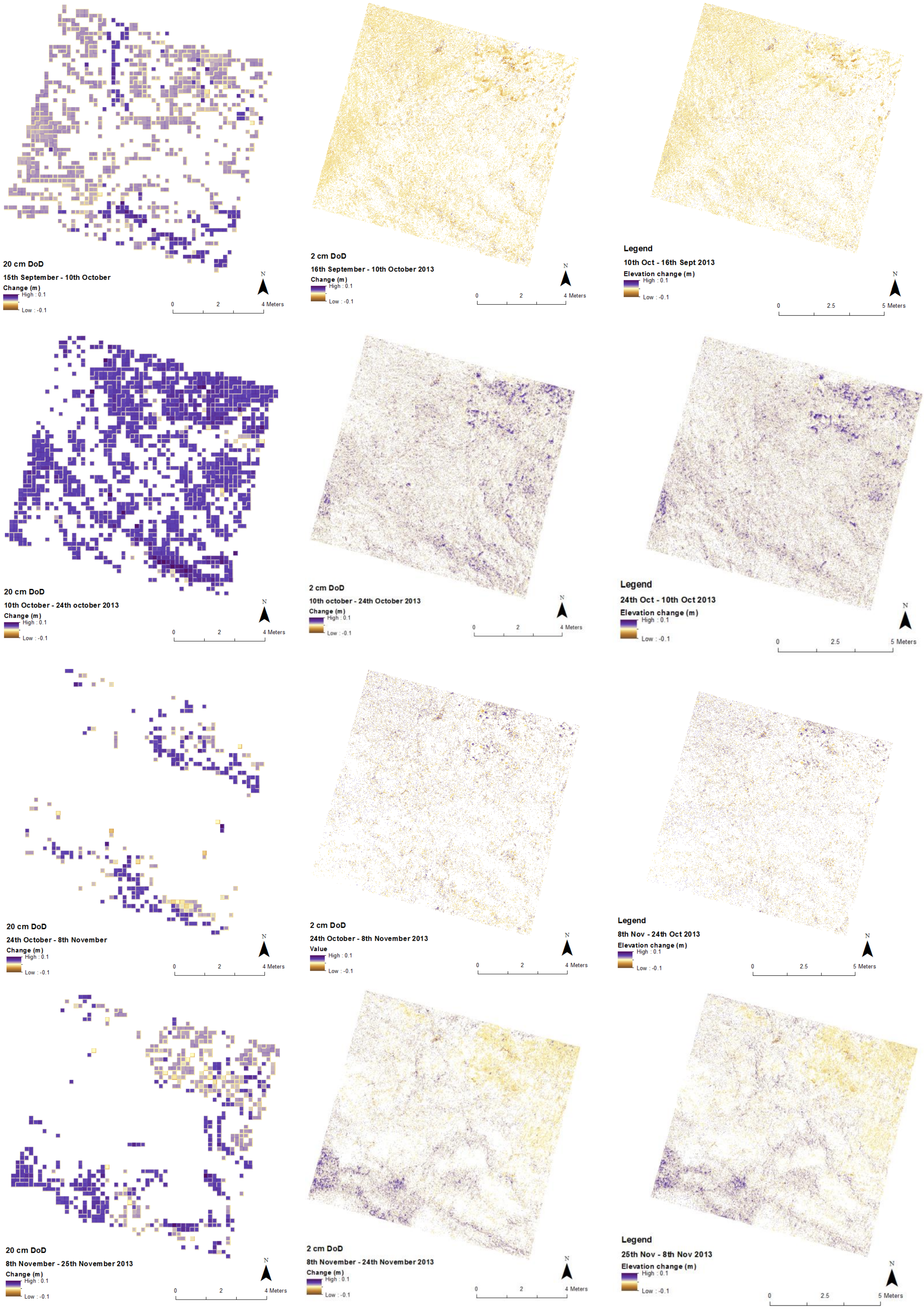


Figure A4 – *DoDs of the same area of bare peat created from DEMs gridded at 20cm, 2cm and 1cm.*

The change in surface height values calculated from the DoDs including error and the DoDs with the error removed are shown in Chapter 6 (Figure 6.24). This indicates that removing values of +/-0.00071m (the MSA value produced during post processing) has the biggest effect on changes in surface height calculated using the 1cm DoD. However, the DoD created from data estimated using a 20 cm resolution produce a change in surface height far greater than the DoDs from data filtered at the 2 cm and 1 cm. Removing erroneous data has the least effect on the DoDs gridded at a 2 cm resolution.

During processing of the TLS data, two different workflows were used (Figure A1). It was established during post processing that registering the point clouds in different projects produced different results to point clouds registered together in one project and then filtered. Figure A5 shows the DoDs created from DEMs produced from TLS data which had been filtered at different resolutions and then registered together in different RiScan projects. The DoDs in Figure A5 have had the error values of +/- 0.0071 m removed. When visually comparing Figure A4 and Figure A5, a clear difference can be identified. Figure A5 shows that when the point clouds were registered within the same project, more values fell within the +/- 0.0071 m error threshold and so were removed. This would suggest that the order of processing of the point clouds may have an impact on the change detection values generated from the TLS data.


20 cm DoD
6th June - 8th August 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
6th June - 8th August 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
Legend
8th Aug - 6th June 2013 1 cm
Elevation change (m)
High : 0.1
Low : -0.1
0 2.5 5 Meters
N
20 cm DoD
8th August - 19th August 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
8th August - 19th August
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
Legend
19th Aug - 8th Aug 2013
Elevation change (m)
High : 0.1
Low : -0.1
0 2.5 5 Meters
N
20 cm DoD
19th August - 16th September 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
19th August - 16th September 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
Legend
16th Sept - 19th Aug 2013
Elevation change (m)
High : 0.1
Low : -0.1
0 2.5 5 Meters
N


20 cm DoD
15th September - 10th October
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
16th September - 10th October 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
Legend
10th Oct - 16th Sept 2013
Elevation change (m)
High : 0.1
Low : -0.1
0 2.5 5 Meters
N
20 cm DoD
10th October - 24th october 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
10th october - 24th October 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
Legend
24th Oct - 10th Oct 2013
Elevation change (m)
High : 0.1
Low : -0.1
0 2.5 5 Meters
N
20 cm DoD
24th October - 8th November
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
24th October - 8th November 2013
Value
High : 0.1
Low : -0.1
0 2 4 Meters
N
Legend
8th Nov - 24th Oct 2013
Elevation change (m)
High : 0.1
Low : -0.1
0 2.5 5 Meters
N
20 cm DoD
8th November - 25th November 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
2 cm DoD
8th November - 24th November 2013
Change (m)
High : 0.1
Low : -0.1
0 2 4 Meters
N
Legend
25th Nov - 8th Nov 2013
Elevation change (m)
High : 0.1
Low : -0.1
0 2.5 5 Meters
N

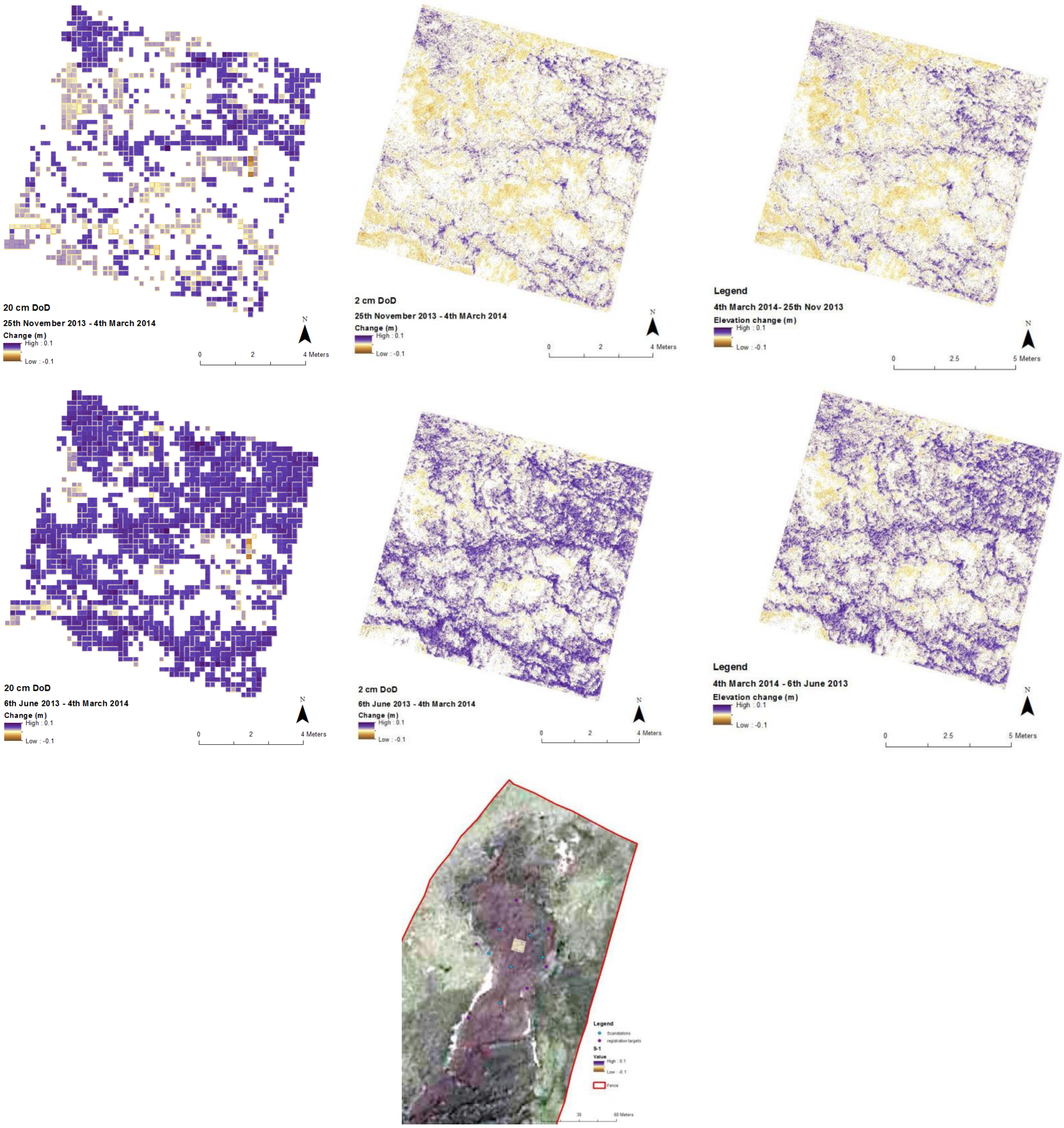


Figure A5 – *DoDs created from DEMs projects registered in the same RiScan project gridded at 20 cm, 2 cm and 1 cm. Values within +/- 0.0071m (highest MSA value) have been considered erroneous and removed from the DoD.*

To confirm this, the calculated change in surface height values for the DoDs produced from data processed using the two different processing work flows are shown in Figure A6. Data labelled as ‘1’ (e.g. 20 cm_1) are those where MSA registration occurred in different projects, while data labelled as ‘2’ were registered in the same project as the 1cm data. This shows that change in surface height values estimated using point clouds registered in different projects would appear to produce a higher estimate of change than those registered within the same project.

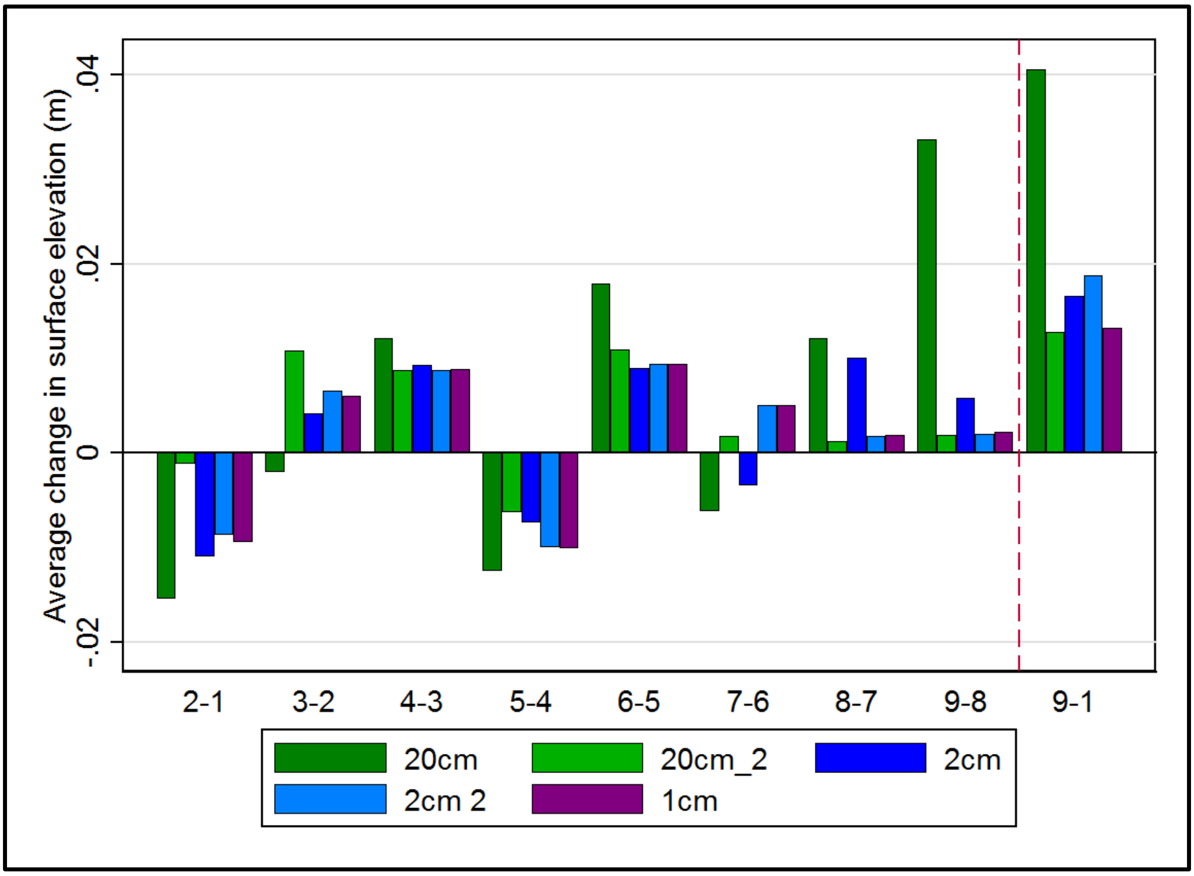


Figure A6 - *A bar chart showing the change in surface elevations calculated using DoDs filtered at three different resolutions and registered in different RiScan projects. The maximum error value of +/- 0.0071 m has been removed. Data labelled as ‘2’ were registered in the same project.*

The observed differences are most likely due to slight variations in the matching of the point clouds when using the MSA function in RiScan pro. When point clouds of different resolutions were registered in separate projects, the MSA values for the data gridded at 2 cm came out lowest at 0.00048 m, while the data gridded at 20 cm and 1 cm had lower MSA values. When the point clouds filtered at different resolutions were registered together in the same RiScan project an MSA value of 0.00071 m was achieved. Although this error value is higher than the value achieved when the 2 cm filtered point clouds are registered separately, by registering the different resolution point clouds in the same project, it means that all data sets have a consistent value for registration error. Therefore, it was decided that the data sets which had been registered together prior to filtering would be used for the change detection in this study; however this limits the size of the point cloud that can be filtered and registered in RiScan. Furthermore, as discussed below (Figure A7) the given MSA value should be used with caution.

Figure A6 demonstrates that the resolution at which the data were filtered and the order of processing affects the values of change calculated using TLS data. This is most apparent in the DoDs created from data at a 20 cm resolution. This would suggest that point cloud filtering and registration can impact on change detection results. The value produced when using RiScan's Multistation adjustment (MSA) provides an indication of how well two point clouds are matched, and in this study the MSA value was used as an error threshold to mask out values of change considered to lie outside the detectable limits of the TLS system. Nonetheless, during the initial post processing of the TLS point clouds, on several occasions, the MSA value was found to be small, suggesting the point clouds were well matched, however when the alignment of the point clouds was visibly checked a clear height offset was observed. Figure A7 shows one of the most extreme examples of an offset between the point clouds found during this study. The two point clouds were registered together and an MSA value of 0.0077 m provided. However when the alignment of the point clouds was visibly checked, a clear height offset between the datasets was observed.

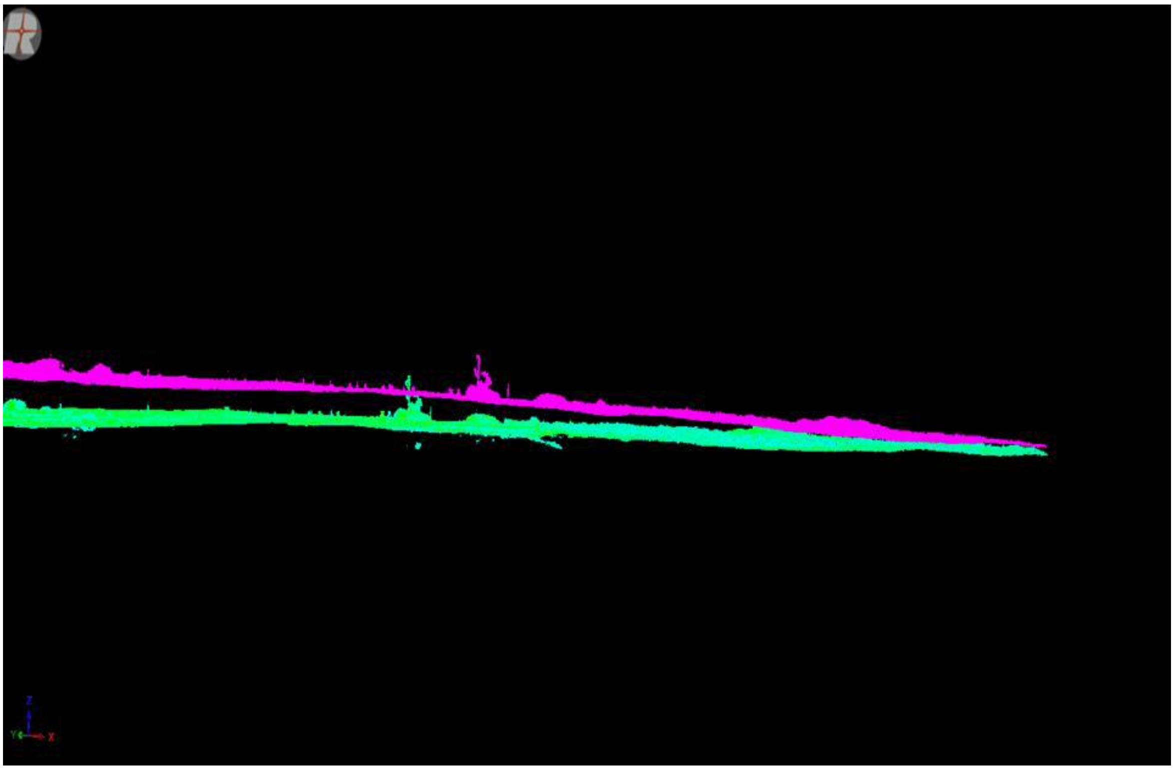


Figure A7– *A screenshot from RiScan showing a clear offset between the point clouds, yet the given MSA value was 0.0077 m.*

This would suggest that the MSA value being produced during the registration process could be providing a false level of registration accuracy, and while some areas of the point cloud may be matched well (reducing the

MSA value), there could be a large offset in other areas. The MSA offset was found to result from settings used from the plane patch filter implemented prior to MSA registration. When the post plane patch filter point clouds were examined it was found that a large number of the plane patch points were located on the vegetated area of the peat, where it is known large changes in height will have occurred between scans (Figure A8 (a)). To overcome this plane patch filter was run again with the filter settings changed and the minimum number of points per plane increased (Figure A8 (b)). The aim was to filter the data with settings that allowed as many plane patch points to be located within known fixed points within the point cloud (such as the fence) and as few as possible on areas where there was known to be a large amount of change (e.g. vegetation).

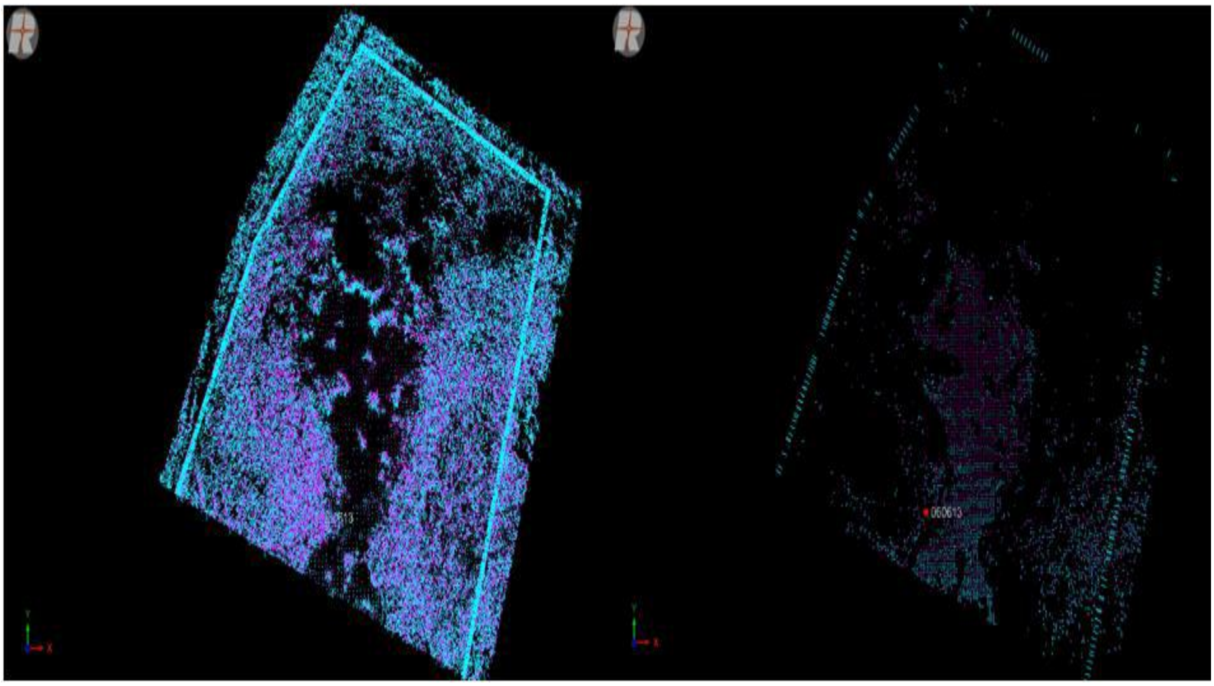


Figure A8– (a) *The plane patch filter run with settings which resulted in a large number of points in vegetation and* ***(b)*** *the filter settings were changed to create more points at fixed locations.*

The data displayed in Chapter 7 were produced from point clouds with as many points located on fixed objects as possible. When the data used displayed in Chapter 7 were registered an MSA value of 0.0071 m was produced and all point clouds thoroughly visually checked to ensure there was no offset between the point clouds.

A3. Recommendations

Based on the results found during this study, several recommendations can be made relating to the use of TLS for peatland monitoring:

- **Issues of registration error**

  One of the greatest limitations of the use of TLS for peatland monitoring stems from issues with registration during post processing. As shown in Figure A7, even when a very small MSA value is calculated, this may not be a true representation of the matching of the point clouds. Therefore, when registering point clouds together, they should be visibly checked to ensure there is no height offset in the data sets. Furthermore, more information is needed about the MSA process and the impact this will have on the detection of small scale changes (such as those which will occur in peatland surface height).

- **Fixed target locations**

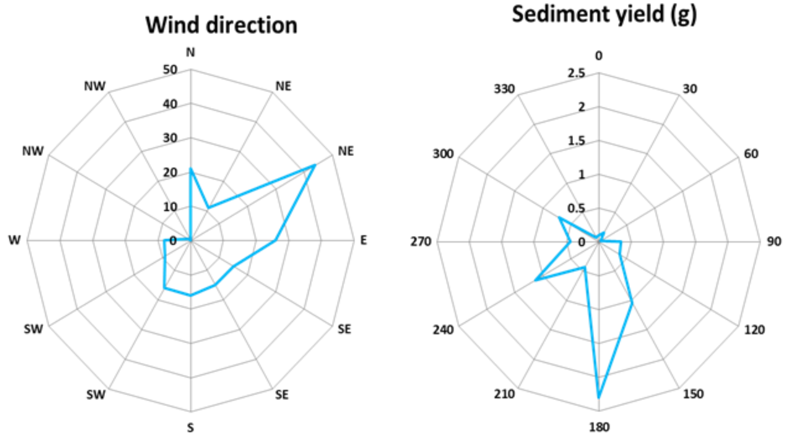
  During TLS surveys, fixed target locations are required. This could overcome two issues. Firstly, the addition of flat fixed objects could be located within the TLS survey area. These could be used during the registration process as points to which planes could be attributed in the plane patch filter. These could help to reduce errors which may occur during the MSA process. Furthermore, in this study, semi-permanent target locations were used, and these measured during every TLS survey to ensure no substantial movement had occurred. However, due to the nature of the peat surface, it is possible that very slight movements may have occurred between surveys. The addition of fixed targets and scan positons would reduce any errors associated with this.

Detectable limits of the TLS system

- Many of the changes in surface height recorded during this study occurred on a very small scale (changes of less than 1 cm), more research is necessary to identify the detectable limits of the TLS system for peatland monitoring.

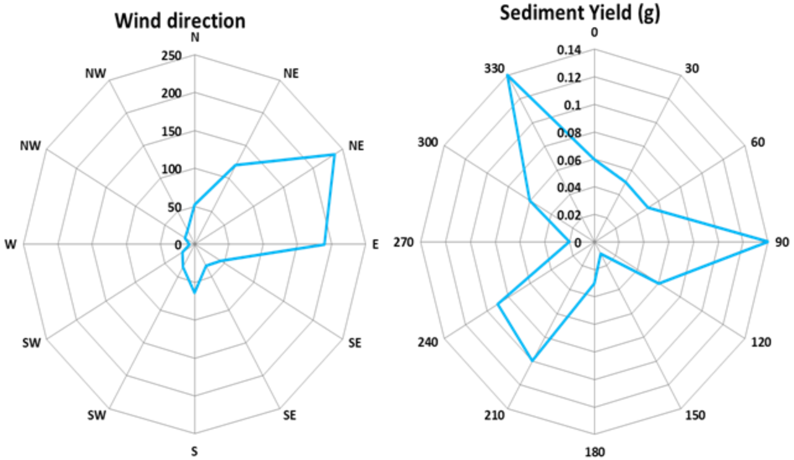
Appendix B

1st - 8th March 2013



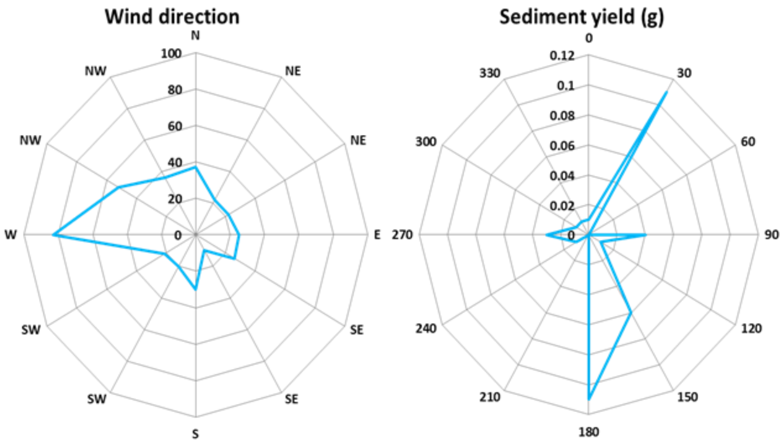
Mean wind direction: 125.6 ° Total sediment: 7.13g

8th March – 14th April 2013



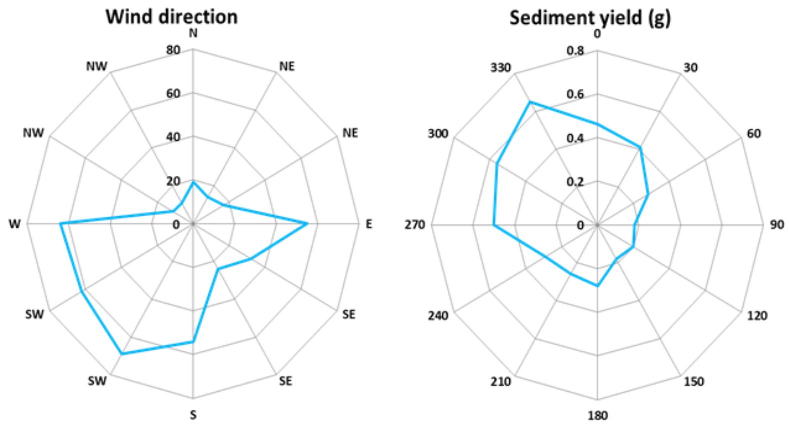
Mean wind direction: 107.8 ° Total sediment: 0.81g

21st May – 6th June 2013

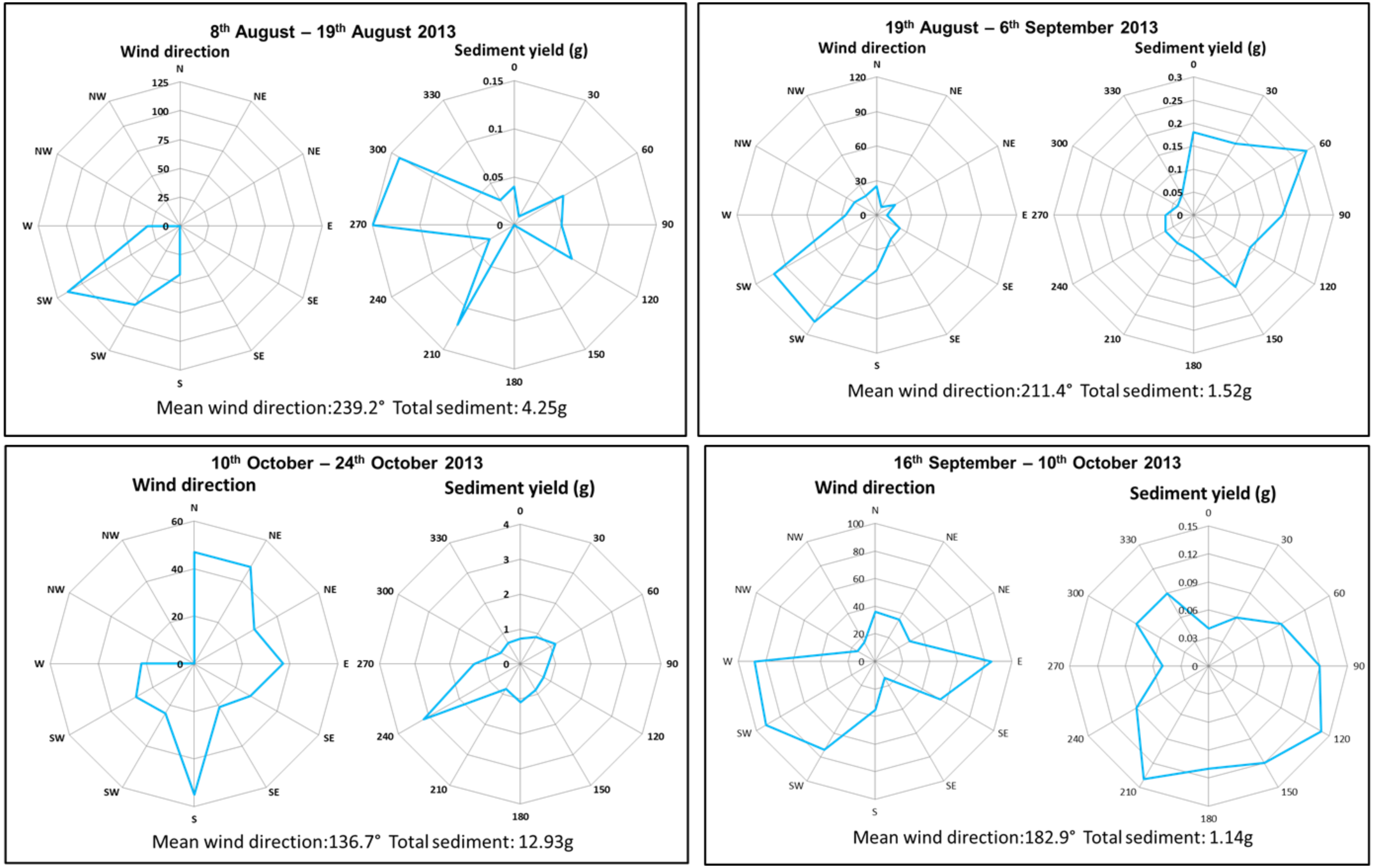


Mean wind direction: 205.7 ° Total sediment: 0.4g

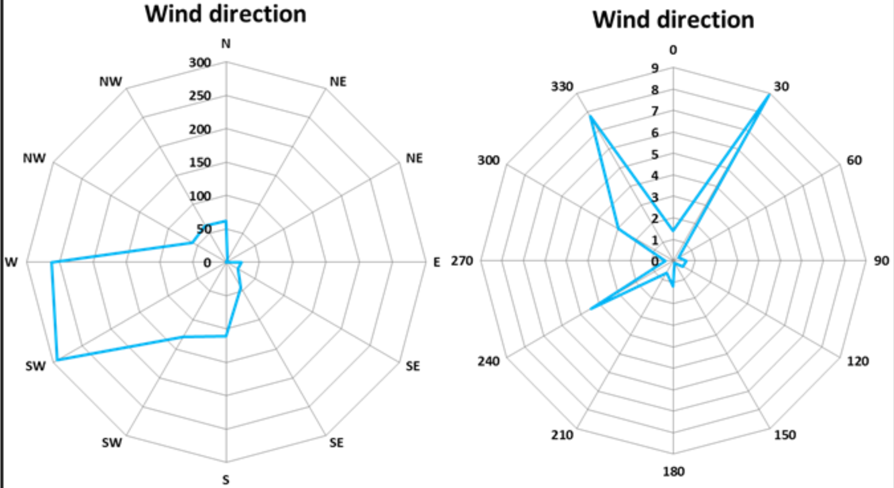
6th June – 24th June 2013



Mean wind direction:193.7 ° Total sediment: 4.25g

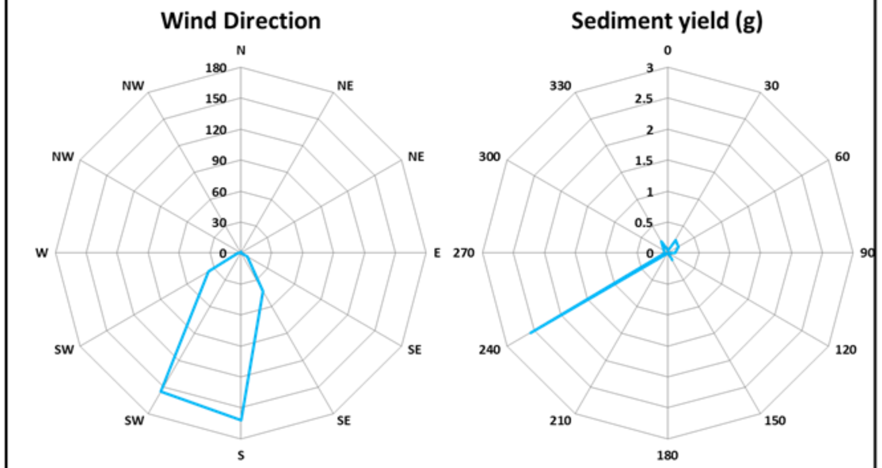

8th August – 19th August 2013
Wind direction
Sediment yield (g)
Mean wind direction:239.2° Total sediment: 4.25g
19th August – 6th September 2013
Wind direction
Sediment yield (g)
Mean wind direction:211.4° Total sediment: 1.52g
10th October – 24th October 2013
Wind direction
Sediment yield (g)
Mean wind direction:136.7° Total sediment: 12.93g
16th September – 10th October 2013
Wind direction
Sediment yield (g)
Mean wind direction:182.9° Total sediment: 1.14g

24th October – 8th November 2013



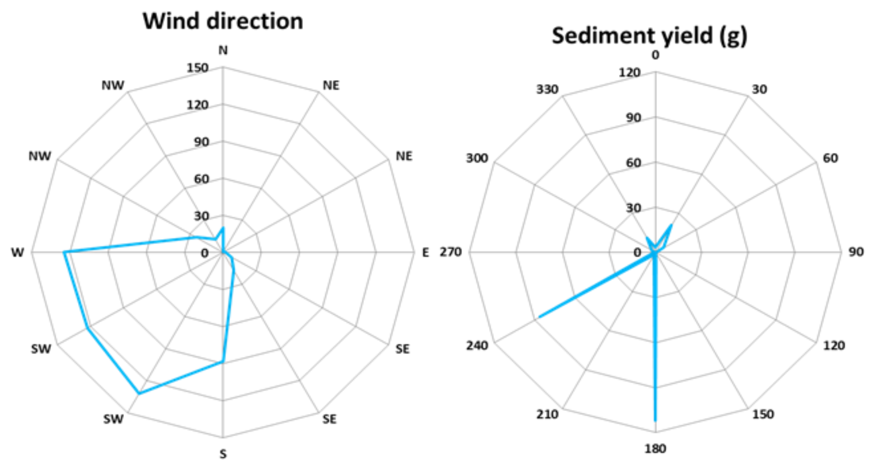
Mean wind direction:225.0° Total sediment: 29.25g

8th November – 25th November 2013



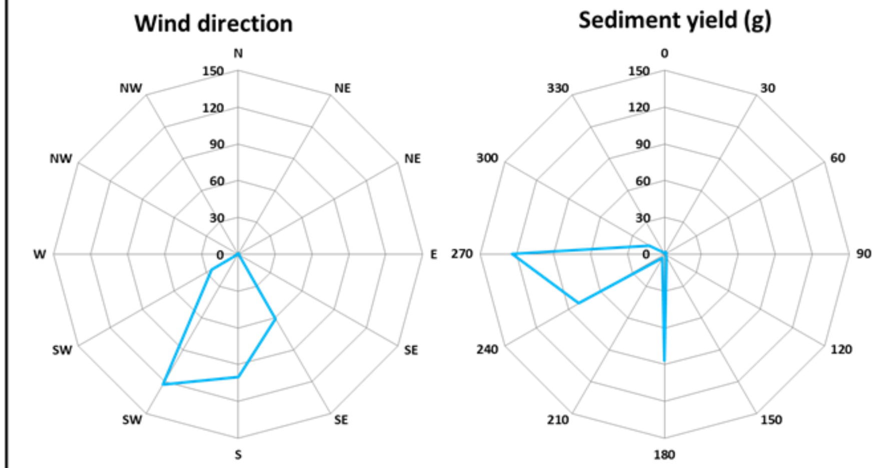
Mean wind direction:229.2° Total sediment: 3.63g

25th November – 17th December 2013



Mean wind direction: 238.5° Total sediment: 246.19g

17th December – 30th December 2013



Mean wind direction: 204.7° Total sediment: 319.71g

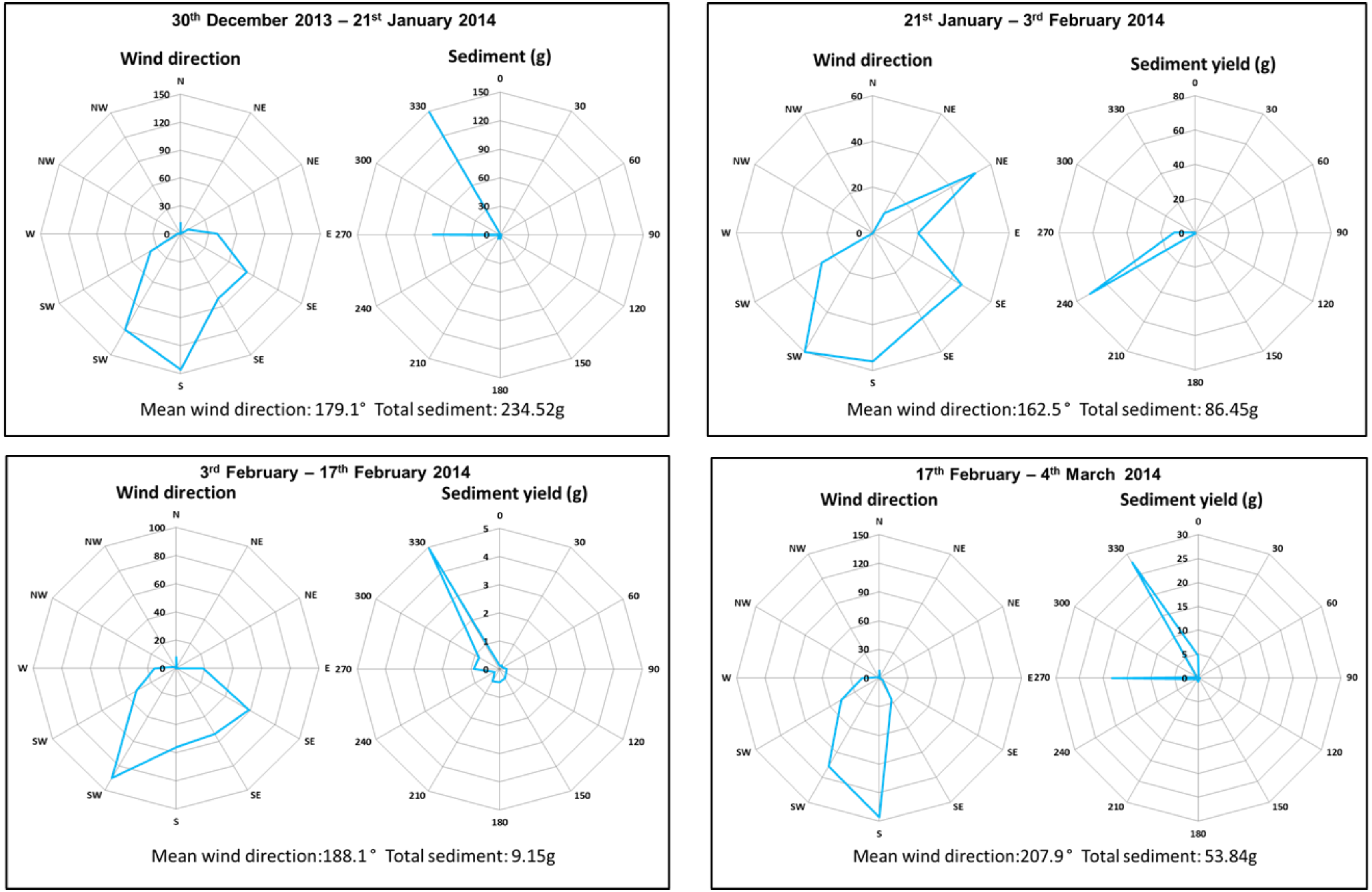


Figure B1 – *sediment yields and wind direction data*

Appendix C

The equations used to scale up the results to produce a sediment budget are shown below:

Firstly, yields of sediment lost through aeolian processes (y) was scaled up using the method shown in equation 8.1.

$$y = \left(\frac{(s \times 261.7)}{a1}\right) \times a2 \qquad \textbf{Eq. c.1}$$

Where: *s* = collected sediment

a1 = the area covered by the samplers

261.7 = scaling factor

a2 = 10777- the area of bare peat mapped during dGPS survey

The yield of peat stored in the pool (*y*) was calculated using equation 8.2:

$$y = (\Delta s \times a) \times BD \qquad \textbf{Eq. C.2}$$

Where: *Δ s* = mean change in surface height

a = area of the pool (124.23 m^2)

BD = Bulk density

A similar equation (equation 8.3) was used to calculate the yield of peat change recorded for the bare peat flats (*y*):

$$y = (\Delta s \times a) \times BD \quad \textbf{Eq. C.3}$$

Where: *Δ s* = mean change in surface height

a = area of the bare peat flats (10777)

BD = Bulk density

The yield of sediment lost through hydraulic processes was calculated by combining the total mass of sediment collected from the sack traps. Finally, the yield (*y*) of peat lost from the hagg slopes was calculated using equation 8.4.

$$y = ((\Delta s \times a1) \times a2) \times BD \ \textbf{Eq. C.4}$$

Where: *Δs* = change in surface height

a1 = area of monitored slopes

a2 = total area of exposed slopes digitised from the 2011 UAV image

BD = peat bulk density (77.62 kg m^{-3}*)*

References

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