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Blanket peatland carbon dynamics: palaeoenvironmental contributions to the land use debate

by

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ABSTRACT

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Blanket peatlands are the most efficient UK terrestrial habitat for carbon storage and their protection is essential to meet greenhouse gas reduction targets. ‘Carbon conscious’ management seeks to manage the ecosystem specifically to maintain or enhance future carbon sequestration. Uncertainty over the impact of land use on blanket peatland carbon sequestration hampers attempts to manage the ecosystem in a sustainable manner. This thesis addresses both the poor understanding of the relationship between land use and carbon accumulation and the lack of palaeoenvironmental records which assess the integrity of Welsh blanket peatlands.

A suite of dating techniques are used to reconstruct carbon accumulation rates across a range of peatlands. Data are combined with plant macrofossil results to offer insight into how land use has affected peatland development. Reported results emphasize that management cannot be considered a constant, linear forcing on blanket peatland development. An alternative method for examining the impacts of land management is presented, namely focussing on the cumulative expression of land management as manifested in vegetation records. Results suggest *Sphagnum*-dominated sites exhibit the most favourable conditions for carbon accumulation and the expansion of *Molinia caerulea* should be of concern to a carbon conscious peatland manager. Future directions for research are also discussed in light of the demonstrated fixation of the land use debate on comparatively recent anthropogenic impacts (i.e. post-AD 1850).

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Academic Thesis: Declaration Of Authorship

I, Robert Charles Collier, declare that this thesis and the work presented in it are my own and has been generated by me as the result of my own original research.

BLANKET PEATLAND CARBON DYNAMICS: PALAEOENVIRONMENTAL CONTRIBUTIONS TO THE LAND USE DEBATE

I confirm that:

1. This work was done wholly or mainly while in candidature for a research degree at this University;
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3. Where I have consulted the published work of others, this is always clearly attributed;
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5. I have acknowledged all main sources of help;
6. Where the thesis is based on work done by myself jointly with others, I have made clear exactly what was done by others and what I have contributed myself;
7. None of this work has been published before submission:

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"Uncertainty is a very good thing: it's the beginning of an investigation, and the investigation should never end"

-Tim Crouch, 2010

Definitions and Abbreviations

^{241}Am	Americium-241 (radioisotope)
^{14}C	Carbon-14 (radioisotope)
^{210}Pb	Lead-210 (radioisotope)
^{137}Cs	Caesium-137 (radioisotope)
^{241}Pu	Plutonium-241 (radioisotope)
^{226}Ra	Radium-226 (radioisotope)
^{222}Rn	Radon-222 (radioisotope)
^{238}U	Uranium-238 (radioisotope)
ACT	Acrotelm/catotelm transition
*a*DNA	Ancient DNA
AD	Anno domini
ANOVA	Analysis of variance
ARCA	Actual rate of carbon accumulation
BC	Before Christ
BD	Bulk density
Bq	Becquerel
Brexit	Anticipated withdrawal of the UK from the European Union
BRP	Bryniau Pica
BSW	Bog surface wetness
C	Carbon
C/N	Carbon/nitrogen ratio
ca.	Circa
CA	Correspondence analysis
CAP	Common Agricultural Policy
CAR	Carbon accumulation rate
CEH	Centre for Ecology and Hydrology
CH_4	Methane
CIC	Constant initial concentration
CO_2	Carbon dioxide
CRS	Constant rate of supply
DCA	Detrended correspondence analysis
DIC	Dissolved inorganic carbon
DOC	Dissolved organic carbon
DT	Dunn test
EU	European Union
GHG	Greenhouse gas
Glastir	Welsh government agri-environment scheme
GMEP	Glastir Monitoring and Evaluation Programme

HFRO	Hill Farming Research Organisation
K	Potassium
KW	Kruskal-Wallis test
LIA	Little Ice Age
LOESS	Locally weighted smoothing
LORCA	Long-term apparent rate of carbon accumulation
LVW	Lake Vyrnwy
m asl	Metres above sea level
MCA	Medieval Climate Anomaly
MIG	Migneint
N	Nitrogen
NEE	Net ecosystem CO_2 exchange
nMDS	non-Metric multidimensional scaling
NNR	National Nature Reserve
NPP	Net primary productivity
NT	National Trust
NVC	National Vegetation Classification
OS	Ordnance Survey
P	Phosphorous
PAR	Peat accumulation rate
PAT	Peat Accumulation Type
PCA	Principle components analysis
PES	Payment for ecosystem service
PLY	Plynlimmon
POC	Particulate organic carbon
r	Linear correlation coefficient
r^2	Linear regression
RERCA	Recent apparent rate of carbon accumulation
S. s.	*Sphagnum* section
SAC	Special Area of Conservation
SCP	Spheroidal carbonaceous particle
SD	Standard deviation
SoR	Start of record
SPA	Special Protection Area
sp.	Unspecified species
SSSI	Site of Special Scientific Interest
UKCP09	UK Climate Projections 2009
undiff.	Undifferentiated
UOM	Unidentifiable organic matter
WTD	Water table depth

Chapter 1 Introduction

1.1 Project rationale

Greenhouse gases (GHG) such as carbon dioxide (CO_2) and methane (CH_4) may alter the Earth system energy balance due to their radiative forcing capacity (Stocker *et al*. 2013). A widespread scientific consensus attributes a large component of the recent warming of the Earth's climate to the increase in atmospheric concentrations of GHGs since the Industrial Era (IPCC, 2013). New climatic conditions present an array of socio-economic challenges for human society (IPCC, 2014). There is therefore great interest in the carbon cycle and its response to perturbations. Of particular interest is the manner in which human activities alter how carbon cycles through the terrestrial environment and is exchanged with other earth system components (e.g. the atmosphere). Land use may reinforce climate change through the degradation of terrestrial carbon stores (Ciais *et al*. 2013). There is therefore a need to manage these carbon stores more appropriately to limit carbon emissions and mitigate against climatic change (Smith *et al*. 2014a).

Peatlands account for approximately one third of the global terrestrial carbon store (547 Gt) despite covering only *ca.* 3% of the Earth's land surface (Gorham, 1991; Holden, 2005a). Peatlands remove CO_2 from the atmosphere as plants that grow on or near the surface photosynthesize. Some carbon is released from the peatland to the atmosphere as CH_4 or CO_2. Most sequestered atmospheric carbon is stored within the peatland as plant remains undergo minimal decomposition following deposition as peat (Charman, 2002). Peatlands have in this way had a net cooling impact on the Earth system through the Holocene (Frolking and Roulet, 2007; Limpens *et al*. 2008). There is uncertainty as to whether peatlands will remain as carbon sinks through the 21st Century since peatland functioning can be altered by a range of drivers such as climate change and land management (Curtis *et al*. 2014; McVeigh *et al*. 2014). If peatlands are damaged then carbon accumulation may cease and previously sequestered carbon could be released to the atmosphere. Damaged peatlands currently emit *ca*. 10% of the global CO_2 emissions associated with land use activity (Smith *et al*. 2014a), thereby reinforcing climate change.

Blanket peatlands are the principal British peatland type by area (*ca*. 2,208,533 ha) (JNCC, 2011). They are the UK's most efficient terrestrial habitat type for carbon storage and are estimated to store *ca*. 3.2 Bt of carbon (Dunn and Freeman, 2011). Failure to recognise the importance of blanket peatland ecosystem services (e.g. carbon accumulation) has resulted in their progressive degradation by market driven land management such as afforestation and livestock rearing (Maltby, 2010). Concern surrounding the integrity of the blanket peatland resource has led to increased interest in their sustainable management (HM Government, 2011; Welsh Government, 2013). A particular rationale for sustainable management is to ensure blanket peatlands continue to function as carbon sinks to mitigate against climate change (DEFRA, 2009; Environmental Audit Committee, 2016).

A shift from intensive exploitation of peatlands to less intensive management practices has primarily been funded under Pillar 2 of the European Union (EU) Common Agricultural Policy (CAP). CAP subsidies paid under 'agri-environment' schemes compensate farmers for adopting less profitable land use techniques (e.g. lower sheep stocking densities) on the premise they are 'environmentally sensitive' (Reed *et al*. 2009). The EU LIFE+ programme also granted >£30 million for direct peatland restoration projects (e.g. ditch blocking) (IEMA, 2016). An alternative land management system to replace such subsidy based approaches may be implemented by recognising the supply of previously unacknowledged environmental goods (Maltby, 2010). The notion that land managers may be paid for supplying environmental services has been postulated for some time (Engel *et al*. 2008; European Commission, 2011; Reed, 2010). The Welsh Government's 'Glastir' scheme attempts to address the lack of market recognition for such services (e.g. carbon accumulation) by compensating land managers who supply them (Welsh Government, 2016). Whilst the science underpinning Glastir is currently weak, the recognition of the value of services such as carbon accumulation is an important step toward sustainable peatland management.

Glastir is currently funded by the Welsh Government and the EU. The primary financial source for promoting sustainable peatland management in Wales, and at a wider scale across the UK, will be lost following the UK's anticipated withdrawal from the EU, hereafter termed 'Brexit'. The UK Government has guaranteed continued subsidy provision until the end of the current CAP commitment period (AD 2020) (Orford and Henderson, 2016). No plan is in place to replace CAP funds beyond this deadline. A new land management system must support

Britain's farmers given the importance of CAP in maintaining upland agriculture (Reed *et al.* 2009). Uncertainty surrounding post-Brexit UK land management and environmental laws (Environmental Audit Committee, 2017) emphasises the importance of ensuring blanket peatlands are protected from future unsustainable exploitation.

EU-exit presents an opportunity to change environmental management. A 'payment-by-results' model that compensates land managers for the sustainable supply of ecosystem services (e.g. carbon accumulation) is a viable alternative to the current system (Garrod *et al.* 2017; Reed, 2016). A lack of understanding as to how land use affects carbon accumulation hampers efforts to manage blanket peatlands appropriately and the chances of adopting a sustainable 'payment-by-results' model. This thesis seeks to address this lack of understanding of the relationship between land management and blanket peatland carbon accumulation.

The peatlands of the British Isles have been found to act as palaeoenvironmental archives (Barber *et al.* 2000; Blundell *et al.* 2008; Langdon and Barber, 2005; Mauquoy *et al.* 2002). The majority of such investigations use ombrotrophic raised bogs to detect a climatic signal in proxy records (Charman, 2002). Blanket peatlands have been relatively under-examined using palaeoenvironmental methods given their reduced peat accumulation rates, perceived poor preservation of environmental records and complex topographical patterns can confound the relationship between peatland functioning and climate (Blundell *et al.* 2016; Chambers *et al.* 2012).

Palaeoenvironmental studies have been used to explicitly examine the impacts of human actions of blanket peatland sites (Blundell and Holden, 2015; Chambers and Daniell, 2011; McCarroll, 2014). Welsh blanket peatlands have been historically managed under a range of uses such as livestock grazing and afforestation. These sites offer a valuable opportunity to explore carbon accumulation and land management interactions and to evaluate whether palaeoenvironmental analyses may contribute to sustainable land management initiatives.

1.2 Project aim

The primary aim of this thesis is to characterise the impact of land management on recent (*ca.* last 150 years) blanket peatland carbon accumulation.

1.3 Thesis structure

Chapter 1 has outlined the project rationale and the aim of this research project. Chapter 2 reviews the literature on the impacts of peatland land management and generates research questions to be used to achieve the project aim. Chapter 3 details fieldwork, laboratory work and statistical methodologies. Chapter 4 presents a detailed overview of examined study sites. Chapter 5 details chronological results and the process behind forming each age/depth model. Chapter 6 details the reconstructed vegetation development of each site. Chapter 7 presents reconstructed carbon accumulation records for each site. Chapter 8 presents a discussion of all results to determine the principle driving forces behind changes in recent Welsh blanket peatland carbon accumulation. The extent that this understanding can be applied to contemporary peatland management is also assessed. Chapter 9 presents the conclusions drawn from this thesis and outlines future research questions that must be addressed.

1.4 Notes on terminology

1.4.1 Peatland classification

Peat is defined as "*partially decomposed remains of plants with over 65% organic matter (dry weight basis) and less than 20-35% inorganic content*" (Charman, 2002, page 4). There is a vast literature surrounding 'peatland' environments and use of terminology is not always clear and consistent. 'Mire' and 'peatland' suggest differing conditions within the European lexicon, where the former details a site actively accumulating peat and the latter applies also to systems that may not exhibit active peat accumulation at present (Charman, 2002). For simplicity, this thesis follows North American nomenclature and uses 'mire' and 'peatland' interchangeably to describe a site regardless of if it is actively accumulating peat.

Differentiation between peatland types occurs principally according to hydromorphological criteria. Ombrotrophic peatlands (bogs) receive moisture inputs solely from atmospheric processes. Minerotrophic peatlands (fens) receive moisture from the atmosphere, surface run-off and groundwater inputs. This key difference in moisture source gives rise to differing characteristics between bogs and fens (Table 1.1).

Table 1.1: General characteristics of bogs and fens (Lindsay, 1995).

Characteristic	**Bog**	**Fen**
Geographical distribution	Moist and boreal climates	World-wide, moist locales
Abundance	Less numerous than fens	Numerous
Surface topography	Convex or raised	Concave or flat
Peat depth	Principally deep (>1m)	Principally shallow
pH	3.5-4.5	4.0-9.0
Nutrient source	Atmospheric deposition	Groundwater and atmosphere
Productivity	Low	Low/high
Decomposition	Low	Relatively high
Floristic diversity	Low	Low/high

This thesis follows Lindsay *et al.* (1988) and defines criteria for blanket peatland formation as areas with a slope of 0-5°, annual minimum of 1,000mm rainfall, minimum of 160 days rainfall ≥1mm and where the warmest month mean temperature is ≤15°C. These environmental requirements constrain blanket peatlands primarily to those gently sloping and glacially smoothed upland regions in the British Isles with an oceanic climate (Clement, 2005). Variable hydrological conditions mean differing sections of blanket peatlands may be closer in form to raised bogs or minerotrophic fens (Charman, 2002). Hydromorphological variability creates much abiotic and biotic variation both within and between blanket peatlands. One can describe such variation in terms of blanket peatland ‘elements’ at differing spatial scales. These elements in ascending order of scale are (Lindsay, 2010):

- Vegetation- Fine scale changes in vegetation communities.
- Nanotope- Small-scale structure changes (e.g. hummocks, hollows, lawns and ridges). The term has historically been interchangeable with ‘microform’.

- Microtope- Patterns of nanotopes across a peatland (e.g. 'hummock-hollow' complexes).
- Mesotope- Multiple microtopes which are described according to their position in the landscape. An example is the 'valleyside mire' which occurs at the base of a slope and is near to a watercourse (Figure 1.1).
- Macrotope- Assemblage of adjacent mesotopes that are hydrologically linked within a landscape to form a 'blanket peatland'.

This thesis exclusively applies the term 'blanket bog' to areas of a blanket peatland that are ombrotrophic (UK Biodiversity Group, 1999). The term blanket bog therefore applies to the 'watershed mire' and 'watershed-valleyside mire' mesotopes (Figure 1.1). Blanket peatlands differ from other upland habitats but are often included under the 'moorland' moniker. Moorland habitats range from blanket bogs through to dry heaths as peat is not explicitly required for a 'moorland' classification, merely acidic, base deficient upland soils (≥250m asl) (Holden *et al*. 2007a). Many biophysical differences exist between these habitats (Shaw *et al*. 1996; Martin *et al*. 2013); hence, when considering the impacts of management it is imprecise to classify blanket peatlands as moorlands and this thesis does not refer to moorland habitats other than when identifying the shortcomings of other work.

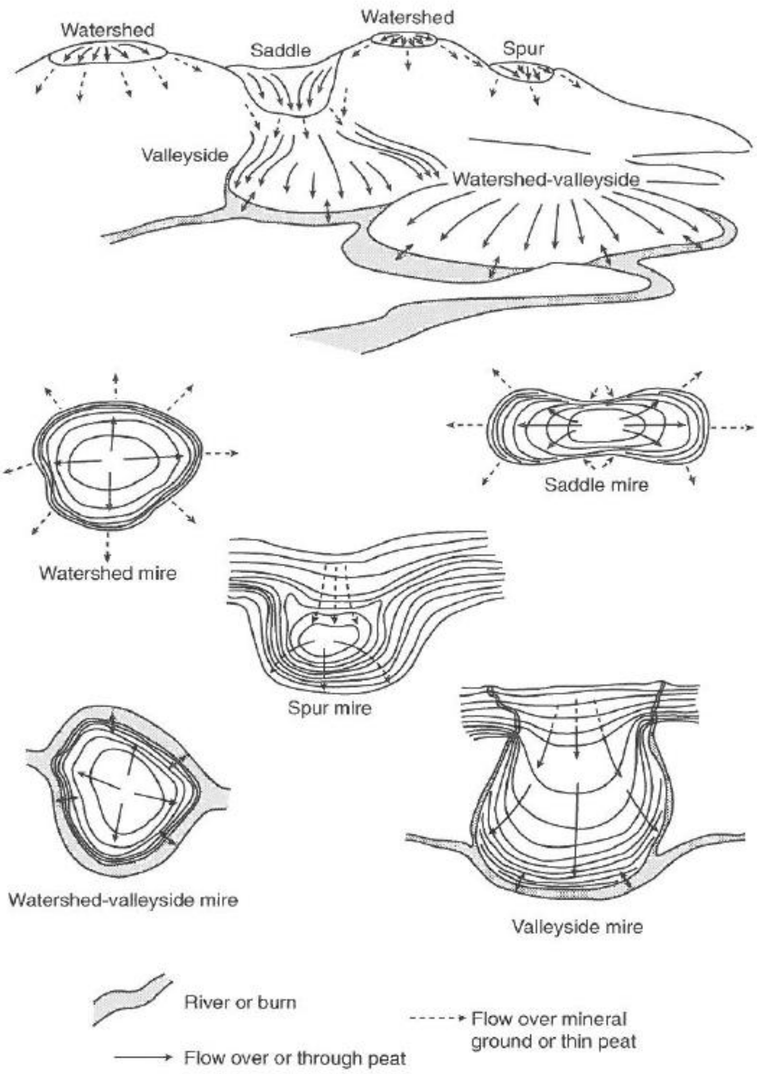


Figure 1.1: Blanket peatland mesotope categories (Lindsay, 1995).

1.4.2 Intact condition

The terms ‘natural’, ‘near-natural’ and ‘semi-natural’ are used when describing peatlands or uplands more broadly (e.g. Littlewood *et al.* 2010; Palmer *et al.* 2004; Thompson *et al.* 1995). General references to blanket peatlands as ‘natural’ systems are inappropriate as the anthropogenic activities of firing and grazing were the primary drivers of clearing British uplands (Evans *et al.* 2007). This thesis uses the term ‘intact’ to identify a bog that is actively accumulating peat and is expected to have not been substantially disturbed post-initiation. ‘Integrity’ describes system condition, in that an intact bog has high integrity. ‘Degraded’ signifies a decline in system integrity (*sensu* Shepherd *et al.* 2013) that negatively affects the ability of the system to supply ecosystem services (*sensu* Millennium Ecosystem Assessment, 2005).

1.4.3 National Vegetation Classification

Ecologists classify environments according to surface vegetation using the National Vegetation Classification (NVC) system (Rodwell, 1991; Table 1.2). The system operates at a coarse spatial scale and classifies environments primarily at the mesotope level (Lindsay, 2010). The NVC system was developed using contemporary ecological data. Peatlands are however systems that have accumulated over millennial timescales. As NVC ‘constant’ species have been determined according to short-term (decadal scale) biophysical processes, many blanket peatland NVC communities are not recorded in the palaeoecological record (Tallis, 1998). The NVC system also does not recognise blanket peatland ‘no-analogue’ communities, past combinations of extant species which no longer appear together (*sensu* Williams and Jackson, 2007). Many contemporary peatland vegetation communities are therefore atypical/degraded when compared with long-term peatland development. Such a weakness undermines contemporary ecological principles in that modern processes may operate markedly differently to system states several thousand years ago (*sensu* Seddon *et al*. 2014).

Table 1.2: NVC types (excluding sub-communities) of primary relevance (Rodwell, 1991).

Code	**Title**	**Constant species**
M1	*Sphagnum auriculatum* bog pool	*Eriophorum angustifolium*, *Menyanthes trifoliata*, *S. auriculatum*, *Sphagnum cuspidatum*
M2	*Sphagnum cuspidatum/recurvum* bog pool	*Erica tetralix*, *E. angustifolium*, *Drosera rotundifolia*, *S. cuspidatum/recurvum*
M3	*Eriophorum angustifolium* bog pool	*E. angustifolium*
M17	*Scirpus cespitosus-Eriophorum vaginatum* blanket bog	*Calluna vulgaris*, *E. tetralix*, *E. angustifolium*, *E. vaginatum*, *Molinia caerulea*, *Narthecium ossifragum*, *Potentilla erecta*, *S. cespitosus*, *Sphagnum capillifolium*, *S. papillosum*
M18	*Erica tetralix-Sphagnum papillosum* raised and blanket bog	*C. vulgaris*, *E. tetralix*, *E. angustifolium*, *E. vaginatum*, *S. capillifolium*, *S. papillosum*, *S. tenellum*, *Odontoschisma sphagni*
M19	*Calluna vulgaris-E. vaginatum* blanket bog	*C. vulgaris*, *E. angustifolium*, *E. vaginatum*, *Rubus chamaemorus*, *Pleurozium schreberi*, *S. capillifolium*
M20	*E. vaginatum* blanket and raised bog	*E. angustifolium*, *E. vaginatum*

Chapter 2 Literature review

2.1 Introduction

This chapter reviews existing evidence concerning the impact of land management on blanket peatland carbon accumulation. General information concerning blanket peatlands and the process of carbon accumulation is discussed first (Section 2.2) before considering whether the practices of burning (Section 2.3), drainage (Section 2.4), forestry (Section 2.5) and grazing (Section 2.6) may alter carbon accumulation. The historic use of these practices across the British Isles (Holden *et al*. 2007a) justifies their selection given the primary aim of this thesis is to characterise the impact of land management on carbon accumulation (Section 1.2). As will be discussed, some of these land uses are applied in conjunction (e.g. peatlands which are drained before a grazing regime is applied). It is necessary to consider these four land uses as somewhat discrete entities for this literature review to evaluate the evidence base concerning anthropogenic impacts on blanket peatlands.

The lack of published evidence regarding management impacts on carbon accumulation necessitates a largely theoretical basis for this review. Each section considers how a practice may alter four controls on peatland carbon cycling, namely '*plant community structure*', '*water table position*', '*peat matrix characteristics*' (adapted from '*peat temperature*') and '*chemical composition of plant tissues and peat*' (Blodau, 2002; Holden, 2005a). This chapter also assesses the responses of other habitats (e.g. raised bogs and fens) to land use in boreal and temperate regions to compensate for the paucity of such blanket peatland literature (as do similar reviews (Glaves *et al*. 2013; Lindsay, 2010)). Robust understanding of management will only be yielded through study on blanket peatlands, hence comparisons with other peatland types are tentative. Data recovered from blanket peatlands must be assessed according to site-specific characteristics as responses to land use in degraded areas (e.g. the southern Pennines) may not be comparable to *Sphagnum*-rich locales (Grant *et al*. 2012). Previous Welsh peatland palaeoenvironmental investigations are next discussed (Section 2.7) before assembled understanding is used to develop research questions which will address the thesis aim (Section 2.8). Issues and shortcomings with previous work are incrementally highlighted through this chapter. Where appropriate, references are made to this chapter's conclusions (Section 2.8) which highlight how this thesis will address these shortcomings.

2.2 Blanket peatland carbon dynamics

Blanket peatland habitats are the focus of this thesis. The habitat exhibits much variability in form across the British Isles given the various development pathways which each specific peatland has undergone (Lindsay, 1995). As such, this first section of the literature review provides the reader with an overview of blanket peatland ecosystems (Section 2.2.1) and peatland carbon dynamics (Sections 2.2.2, 2.2.3 and 2.2.4). This understanding is necessary for subsequent discussions which consider land management impacts in detail.

2.2.1 Blanket peatland ecosystems

This first sub-section presents an initial background to the blanket peatland habitat. This sub-section also introduces the debate over how differing types of land use affect blanket peatland development, hereafter termed the 'blanket peatland land use/management debate'.

2.2.1.1 Blanket peatland formation and development

Peat is the partially or fully decomposed remains of vegetation that once grew on, or near to, the surface of a peatland (Whittow, 2000). Peat may form where organic productivity exceeds decomposition, often following waterlogging of a substrate (Charman, 2002). Blanket peatland initiation is therefore a manifestation of a hydrological change that caused soil saturation. Moore and Bellamy (1973) proposed the controls of climate, geochemistry and hydrology as acting as 'templates' which control peatland formation and morphology. Charman (2002) expanded on this template, detailing additional 'external' factors that determine whether peat may form at any given site (Figure 2.1). The climate must provide adequate moisture to suppress decomposition through waterlogging for at least part of the year. Geological strata and topography must in combination permit the water table to remain at or above the land surface without excessive runoff losses. Biogeographical factors affect the relative productivity and decomposition rates of the flora that initiate a peatland. Finally, human activities may be key in forcing hydrological conditions so that peat may form (Charman, 2002). These 'external' factors initially affect the functioning of 'internal' processes of peatland development (Figure 2.1). Over time, 'internal' processes may come to dominate peatland functioning to the extent ‘external’ processes have limited impact on future development (Charman, 2002). Northern peatlands are non-linear systems given their sensitivity to initial conditions (e.g. fen-bog transition), step-change threshold tendencies (e.g. post-glacial expansion) and magnitudes of change which fail to match magnitude of drivers (e.g. vegetation succession) (Belyea, 2009).

Blanket peatlands occupy relatively warm and wet maritime conditions (≥1,000mm annual rainfall, mean annual temperature >-1°C, mean temperature of warmest month ≤15°C) of the northern peatland (>45°N) climatic envelope (Gallego-Sala and Prentice, 2012; McVeigh *et al.* 2014). Climatic controls primarily constrain blanket peatlands to the oceanic west and north of the British Isles (Sottocornola *et al.* 2009), though they are present towards the centre and east of the region (Figure 2.2; Tallis, 1997). There is debate regarding the relative importance of climatic and human factors in British blanket peat initiation. Highland (Tipping, 2008) and Faroe Island (Lawson *et al.* 2007) peats formed in the cool and wet early Holocene, which pre-dates notable human activity. Records from more southerly localities indicate a human agency in peat initiation through firing of the landscape and subsequent waterlogging *ca.* 3,100-1,150 BC (Caseldine and Hatton, 1993; Moore, 1993) in Exmoor (Merryfield and Moore, 1974), South- (Chambers, 1983), Mid- (Chambers, 1982a) and North Wales (Moore, 1973), the Pennines (Tallis, 1998) and Western Ireland (Huang, 2002). Human agency in initiating blanket peatlands was likely a temporally heterogeneous process rather than a single event. Smith and Cloutman (1988) demonstrated this whereby sixteen South-Wales blanket peat profiles within an 8ha area all appeared to have been initiated by anthropogenic firing, but over a period ranging *ca.* 6,050-2,050 BC. Such spatially and temporally variable development complicates attempts to establish systematic blanket peatland responses to external pressures (e.g. humans) (*sensu* Dice, 2009).

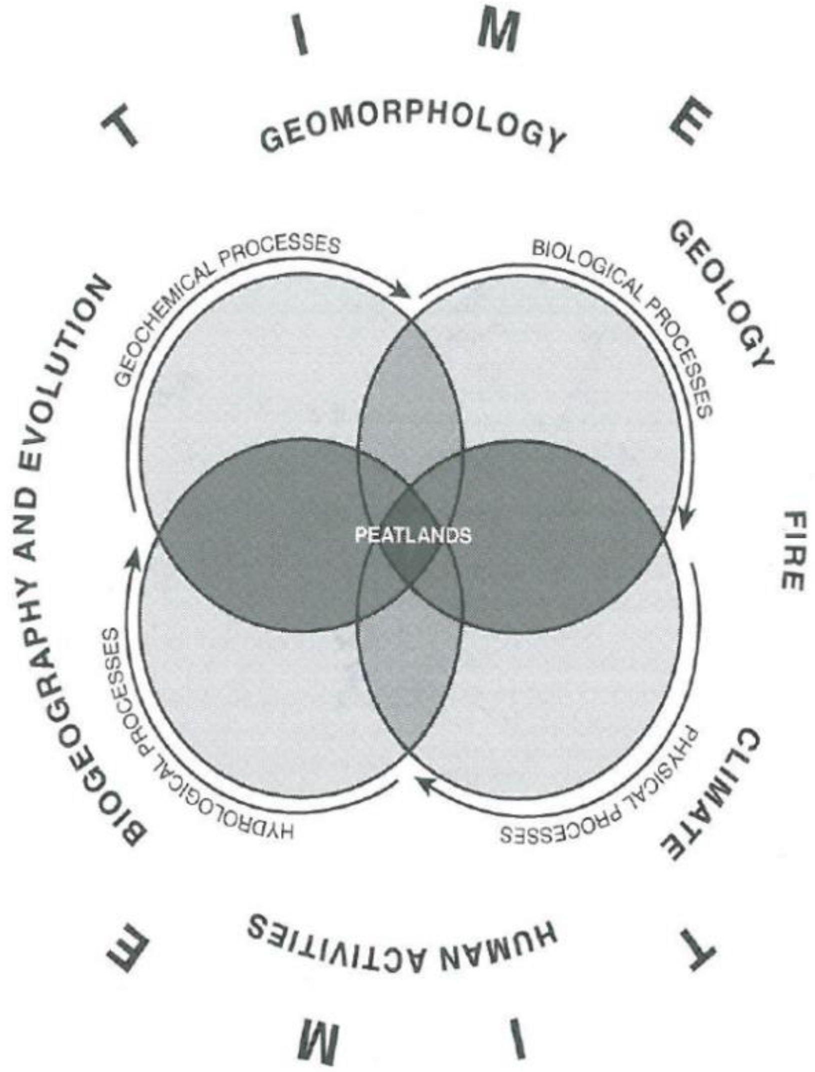


Figure 2.1: Conceptual model detailing the main factors influencing peatlands. The four interlocking circles detail ‘internal’ factors with 'external' factors surrounding these. All factors operate under the influence of time, hence its external position encompassing the model. The resultant ‘peatland’ is a product of all factors, hence its central location (Charman, 2002).

Figure 2.2: Distribution of blanket peat within Great Britain (Holden, 2005b).

Models of peat accumulation often separate peatland vertical structure into two functional layers. Basic separation is required as the fluctuating water table regulates the differing biophysical processes that occur within a peatland (Table 2.1). The upper layers of the profile (acrotelm) are never permanently submerged whilst the lower layers (catotelm) are permanently saturated, the boundary between the two is theoretically assignable where the water table reaches its deepest annual point (Charman, 2002). Practical separation between acrotelm and catotelm is not always easy and Ingram and Bragg (1984) proposed the term ‘haplotelmic’ for sites where the aerated acrotelm has been lost. In these cases, the catotelm is the only component of the peatland which remains and the haplotelmic form likely represent the impacts of degradation (Charman, 2002). Separating a peatland into only two components has been criticised as it assumes all ‘internal’ processes operate in two states either side of a discrete boundary (Belyea and Baird, 2006). Peatland ecohydrological variability suggests the use of multiple asynchronous boundaries may be a more appropriate way to describe peatland biophysical functioning (e.g. oxic/anoxic zone, high/low decay zone, mobile/immobile zone) (Morris *et al*. 2011). Such a scheme would acknowledge horizontal variation in vertical peatland functioning, so called ‘hot-spots’ of accelerated ecohydrological processes and mass and energy transfers (e.g. peat pipes, upwelling zones) (Morris *et al*. 2011). For simplicity, this thesis uses the ‘acrotelm’ and ‘catotelm’ terms to represent different functional components of blanket peatlands. A peat depth of ≥50cm is often set as the threshold for ‘deep’ blanket peat (Bain *et al*. 2011; Worrall *et al*. 2013) though shallower thresholds (*ca*. 30cm deep) have been suggested (Lindsay, 2010).

Table 2.1: Primary characteristics of peatland functioning and how they differ between the acrotelm (upper layer) and catotelm (lower layer) (Charman, 2002).

Characteristic	**Acrotelm**	**Catotelm**
Water table	Fluctuating	Absent
Water content	Variable	Constant
Aeration	Periodically aerobic	Anaerobic
Microbial activity	High, both aerobic and anaerobic forms	Low, anaerobic form only
Water movement	Relatively fast	Very slow
Energy and matter exchange	Rapid	Slow

2.2.1.2 Blanket peatlands and land management

Blanket peatlands in the British Isles have been subjected to land management largely unmatched by other northern peatlands (Worrall and Clay, 2012). Intensive management has occurred primarily due to the proximity of population centres to blanket peatlands (e.g. proximity of South Pennine peats to the industrial centres of Manchester and Sheffield) (Bragg and Tallis, 2001). Intensive land use at the detriment of ecosystem integrity (Evans *et al*. 2014a; Worrall *et al*. 2009) has resulted in >80% of contemporary UK peatlands being in a degraded state (Littlewood *et al*. 2010). Mismanagement of blanket peatlands may force a bog towards the status of dry heath, as Figure 2.3 shows the successional linkages that differing land uses may induce (Shaw *et al*. 1996).

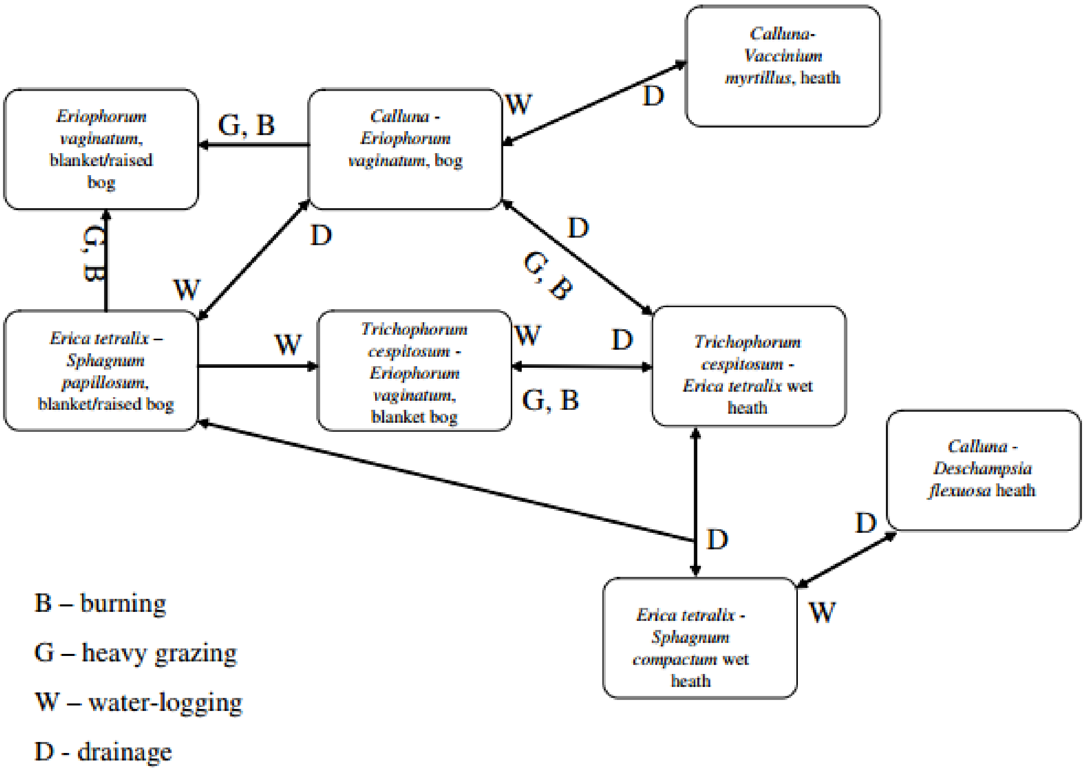


Figure 2.3: Simplified conceptual diagram detailing successional stages between bog and heath communities under differing management (re-drawn version (Gray and Levy, 2009) of original diagram (Thompson et al. 1995)).

Intensive peatland management is permissible given UK land ownership rights currently permit the maximisation of legal economic returns regardless of environmental cost (Burton *et al*. 2008; Hanley *et al*. 1999). Ecologically sensitive management techniques are often perceived as a threat to livelihoods given they require alterations to established methods of farming (e.g. reducing sheep stocking levels). Such concerns are especially pertinent considering the financial instability of upland agriculture (Holden *et al*. 2007a; NFU, 2015a; Reed *et al*. 2009). There must therefore be some economic incentive for land users to adopt more sustainable management approaches (McCarroll *et al*. 2016a; Wynne-Jones, 2013a) as markets primarily dictate agricultural techniques (DEFRA, 2010). Common Agricultural Policy (CAP) agri-environment schemes subsidise farmers undertaking sustainable management practices such as reduced sheep numbers per hectare (Martin *et al*. 2013). Restoration projects also attempt to return degraded sites to a favourable state (Parry *et al*. 2014; Wallage *et al*. 2006). A review of restoration projects found "*full recovery of blanket bog and its functions*" has yet to be observed (Shepherd *et al*. 2013, page 32). Such uncertainty surrounding the success rate, and costs of, restoration projects has hindered their uptake (Moxey and Moran, 2014).

Funding for restoration and sustainable management schemes primarily comes from corporate backing (e.g. corporate responsibility), institutional grants (e.g. EU LIFE+) or subsidy (e.g. agri-environment payments) rather than directly from the land manager (IEMA, 2016). Engagement from land managers is limited in the sense they are not willing to invest their own money and/or risk profits for the sake of sustainable peatland management. Wider adoption of sustainable practices may be encouraged if schemes offer direct financial incentives to land managers (Moxey, 2016).

2.2.1.3 An ecosystem services framework to promote sustainable management

Sustainable management is a key component of environmental governance (Millennium Ecosystem Assessment, 2005; United Nations, 2015) as previous failures to recognise the value of environmental services (McInnes *et al*. 2017) have led to widespread ecosystem degradation (HM Government, 2011; Natural Capital Committee, 2015; Sukhdev *et al*. 2010). 'Natural capital' represents the biophysical components of the Earth system (e.g. a peatland) within which 'ecosystem processes' occur (e.g. peat accumulation). These processes provide a range of 'ecosystem services' (e.g. carbon accumulation) that deliver 'benefits' (e.g. regulation of the climate) which promote 'human wellbeing' (Figure 2.4). The ecosystem services concept proposes that biophysical resources are as expendable as other forms of economic capital (e.g. financial, human) and that degradation may jeopardise future ecosystem service supply (Alexander *et al*. 2016). 'Payment for ecosystem service' (PES) schemes aim to address the failure of markets to value these services (Dunn, 2011; Robertson, 2012). There are concerns that assigning ecosystem services a standardised monetary value oversimplifies complex biophysical processes which interact to produce services (Gómez-Baggethun and Ruiz-Pérez, 2011; Kosoy and Corbera, 2010; Peterson *et al*. 2010). Others criticise the ethics of monetising nature (McCauley, 2006; Ridder, 2008) as it proliferates an anthropocentric world view (Silvertown, 2015). Economic arguments do not negate non-monetary arguments (e.g. ethics); hence, PES approaches simply provide additional evidence in a decision-making process (Schröter *et al*. 2014). An ecosystem service framework can therefore support a land management initiative (de Groot *et al*. 2012; Díaz *et al*. 2015; Jax *et al*. 2013; TEEB, 2010) if a service (e.g. carbon accumulation) can be quantified.

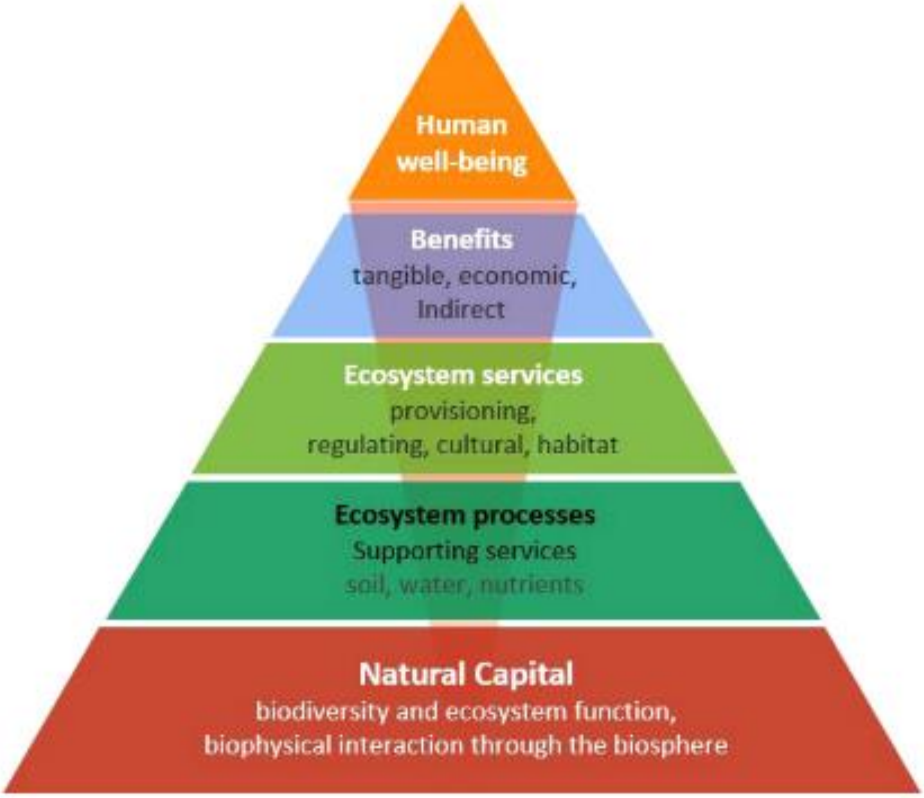


Figure 2.4: Schematic diagram showing how ‘human well-being’ and the 'natural capital' of the biosphere are related. From this capital 'ecosystem processes' supply 'ecosystem services'. These services provide many 'benefits' to society that promote 'human well-being’. The internal pyramid shows that increased human well-being can provide benefits to the natural world, and as such, human actions can promote the restoration of natural capital (Alexander *et al*. 2016).

Carbon accumulation is a key climate regulation service (Abson *et al*. 2010) and blanket peatlands are the UK’s largest and most efficient terrestrial carbon store (Dunn and Freeman, 2011). Continued carbon sequestration is vital to achieve the target of the UK Climate Change Act of 80% greenhouse gas (GHG) reductions by AD 2050 relative to AD 1990 levels (HM Government, 2009, 2011; Moxey and Moran, 2014). A recent amendment to Article 3.4 of the Kyoto Protocol allows Annex I countries to report GHG emission reductions arising from peatland restoration during the agreement’s second commitment period (ending AD 2017-20) (UNFCC, 2011; Hughes *et al*. 2014). Appropriate management of agricultural activity is a key component of Government policy (Warner *et al*. 2017) given 10%

of UK AD 2015 GHG emissions arose within the sector (DBEIS, 2017). Peatland carbon stores are liable to change given factors can both inhibit future carbon accumulation and release previously assimilated carbon (Blodau, 2002; Laiho, 2006). The UK Government may not meet GHG emission targets if mismanagement reduces peatland carbon sequestration (Ostle *et al.* 2009; Ramchunder *et al.* 2009) and loss of 5% of the *ca.* 3.2 Bt peatland carbon store would equal UK annual GHG emissions (Bain *et al.* 2011). In light of such concerns, the adoption of 'carbon conscious' management practices are called for to conserve existing deposits and maintain or enhance future sequestration (Dunn and Freeman, 2011; Hughes *et al.* 2014).

PES approaches require those who benefit from a service to pay those who provide the service (Guerry *et al.* 2015). Carbon accumulation is difficult to exchange as a service as it is both 'non-excludable' (none can be excluded from the provided benefit) and 'non-rival' (use by one person does not affect the use of another) (Dunn, 2011; Goldman *et al.* 2007). The Peatland Code (IUCN, 2015, 2017a) is an initial peatland PES designed primarily to promote restoration of damaged sites (Reed *et al.* 2017). Carbon store benefits of restoration can be reported under the scheme (Smyth *et al.* 2015) but cannot be traded in a market environment (IEMA, 2016). Government funding is therefore still required to promote carbon conscious management (Bain *et al.* 2012; Bonn *et al.* 2014; Dunn, 2011) until such PES approaches are more widely adopted and developed.

'Glastir' is the Welsh land management program offering payments for the supply of traditionally non-tradeable environmental services whilst "*reducing carbon and greenhouse gas emissions*" from land use (Welsh Government, 2016, page 7). Glastir follows an 'input' payment model calculated as 'income foregone' as opposed to payments for quantified supply of a service (Wynne-Jones, 2013b). Input models provide a weak link between payment and ecosystem service (Reed *et al.* 2014). An 'output' PES approach pays land managers specifically for a supplied service (e.g. kg carbon sequestered per m^2). The Welsh Government (Wynne-Jones, 2013b) and European Commission (2011) seek to adopt output focussed payment models. Such a model may promote carbon conscious peatland management by appealing to the desires of the farming community to provide services rather than receive subsidies (Wynne-Jones, 2013a). The Welsh Government is due to receive €1.96 billion for distribution to land managers during the current CAP commitment period of AD 2016-2020 (Orford and Henderson, 2016). A desire for public agricultural spending to be administered in a payment-by-results model (Helm, 2016; Reed, 2016) could be reflected in a PES framework to ensure public money is spent on the delivery of public goods.

Whilst many consider peatland management regimes to either initiate or reinforce existing losses of carbon (e.g. Armentano and Menges, 1986; Dawson and Smith, 2007; Janssens *et al*. 2005; Worrall *et al*. 2010), no systematic data currently supports this notion. Science must robustly demonstrate the benefits of ecosystem services if arguing for their consideration in land management decisions (Daily *et al*. 2009; Ruffo and Kareiva, 2009). The promotion of carbon conscious peatland management therefore requires quantitative understanding of how carbon accumulation relates to land use, hence the stated aim of this thesis (Section 1.2).

2.2.1.4 Palaeoenvironmental contributions to land management policy

Ecosystem management is determined according to the information available to those developing management plans (Jeffers *et al*. 2015). Ecological datasets spanning fewer than fifty years (Davies *et al*. 2014) are often used to devise overly-precise conservation and land management targets (Hiers *et al*. 2016). Using such information can promote the construction of somewhat static, unnatural and bounded conditions when considered against the long-term development of an ecosystem (Pellatt *et al*. 2015). Peatland restoration targets are often devised using National Vegetation Classifications and offer little-to-no empirical evidence as to previous site conditions (Chambers and Daniell, 2011). Palaeoecology can generate longer datasets to quantify how atypical current conditions are in the context of natural variability (Brewer *et al*. 2012; Chambers *et al*. 2017; Jackson and Hobbs, 2009; Pellatt *et al*. 2015; Talbot *et al*. 2010; Willis and Birks, 2006), provide empirical targets for restoration (Clarke and Lynch, 2016; Davies and Bunting, 2010) and quantify ecosystem service supply over long time-scales (Jeffers *et al*. 2015; Pearson *et al*. 2015; Prager *et al*. 2016).

2.2.2 The carbon accumulation process

The impact of climatic (Charman *et al*. 2013; Loisel and Yu, 2013a; Yu *et al*. 2010) and autogenic (Loisel and Yu, 2013b, c; Quillet *et al*. 2013; Shiller *et al*. 2014) controls on carbon accumulation are increasingly understood. There is a lack of understanding of the impacts of local and regional factors (e.g. human activity) on carbon dynamics. This project aims to address the lack of understanding regarding the impact of land management on blanket peatland carbon accumulation. The information below is presented so the reader may understand why peatlands accumulate carbon and to act as a foundation upon which subsequent discussions around carbon accumulation may build.

Peatland carbon accumulation is a function of net primary productivity (NPP) and decomposition, the former required to be greater for net carbon accumulation (Figure 2.5). Any driver altering NPP or decomposition may alter carbon dynamics.

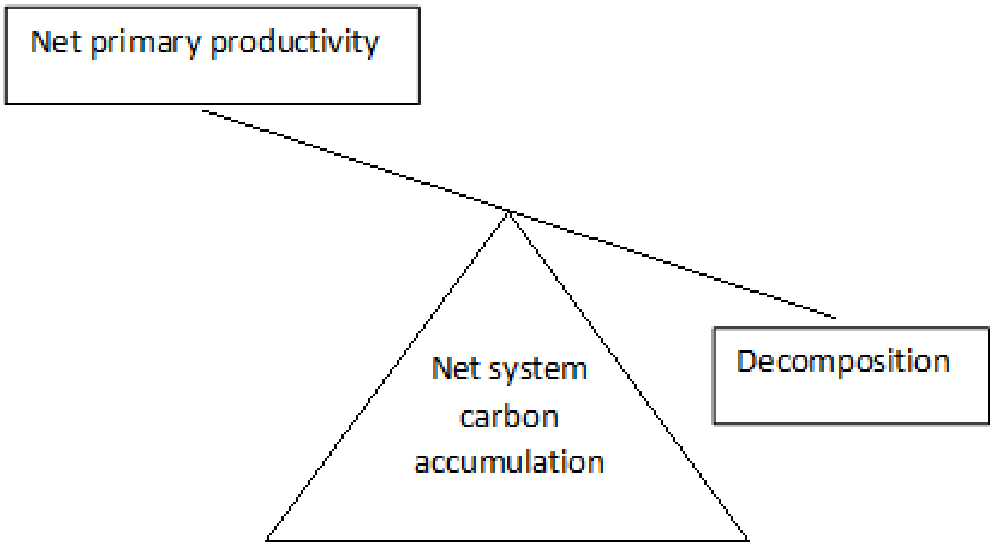


Figure 2.5: Conceptual model detailing an intact peatland in net carbon accumulation mode. The raising of decomposition at the expense of NPP may force a net carbon loss.

NPP is determined by the productivity of live peatland vegetation as plants fix atmospheric carbon through photosynthesis. Plant productivity varies between peatland type and between geographic regions where the cooler and shorter boreal and sub-arctic growing seasons limit NPP (Lund *et al*. 2010). Only 2-16% of annual productivity is passed into the catotelm as the rest is lost to decomposition (Tolonen and Turunen, 1996). The living plants of a peatland often only contain 1-1.5% of peatland carbon with the remaining 98.5-99% retained in the peat matrix (Blodau, 2002). Decomposition alters the constitution of complex organic matter (e.g. plant litter and root exudates) into simpler, lower molecular weight forms (Figure 2.6) that microbes can assimilate (Conant *et al*. 2011; Laiho, 2006; Thormann, 2006). Decomposition is greatest in the aerated acrotelm and continues at a lower rate in the catotelm. Peat becomes increasingly decomposed with depth (Beer and Blodau, 2007; Moore *et al*. 2007), reducing the peat pore size and hydraulic conductivity of the catotelm relative to the acrotelm

(Clark *et al*. 2008; Holden and Burt, 2003; Morris *et al*. 2015a). Clymo (1984) suggested that a 'limit to growth' would be reached as net decay in a catotelm of increasing thickness would match the rate of litter addition at some point in bog development, assuming constant acrotelm litter supply and decomposition rates (Figure 2.7). Carbon accumulation is greater in bogs (raised and blanket) than other peatlands (Lindroth *et al*. 2007; Tuittila *et al*. 2013; Turunen *et al*. 2002) due to low rates of decomposition (Lang *et al*. 2009; Thormann *et al*. 1999).

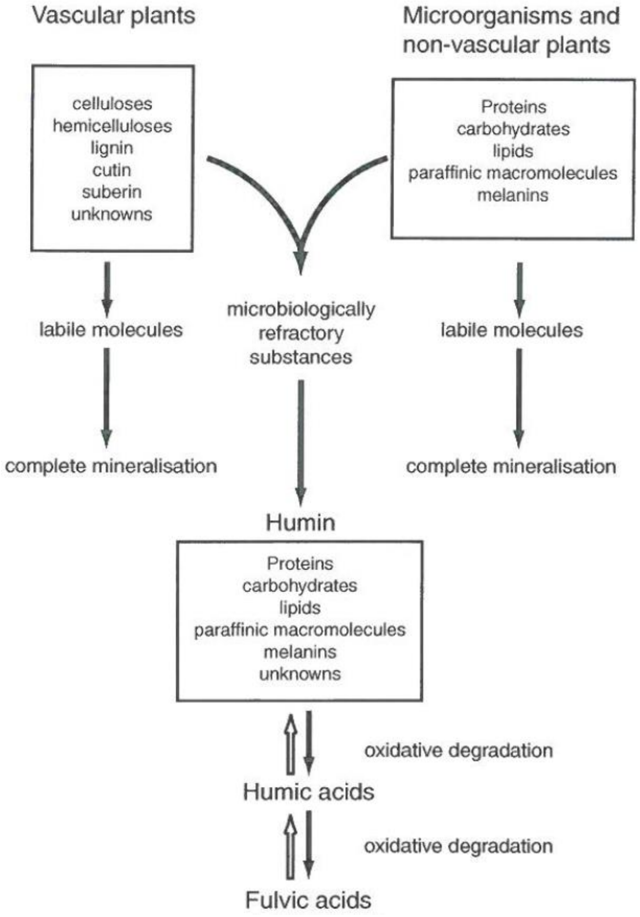


Figure 2.6: Schematic diagram showing pathways by which microbes degrade plant matter to humic and fulvic acids (re-drawn version (Charman, 2002) of original diagram (Hatcher and Spiker, 1988)).

Soil carbon cycling often follows kinetic theory, assuming decomposition should increase alongside temperature given non-limiting substrate availability and enzyme activity (Aerts, 2006). Peatlands do not follow such theory (Blodau, 2002). The primary peatland decomposers of bacteria and fungi (Clymo, 1965) release extracellular enzymes (e.g. β-glucosidase, chitinase) that promote the decomposition of organic material (Sinsabaugh, 2010). Phenolic substances are 'recalcitrant' (resistant to decay) secondary plant metabolites that directly inhibit the activity of these enzymes (Freeman *et al*. 2004a, b; 2012). The concentration of phenolic compounds is greater in bogs than other peatland types (Min *et al.* 2015; Williams *et al.* 2000). Phenol oxidase is the name given to an assortment of extracellular enzymes capable of oxidising phenolic substances (Pind *et al.* 1994). The largely anaerobic conditions of intact peatlands limit phenol oxidase activity (Jassey *et al.* 2011) because it uses oxygen as an electron receptor (Freeman *et al.* 2004b). Oxygen availability thereby indirectly reduces decomposition, given hydrolase enzymes could function in anaerobic peatlands were it not for the accumulation of phenolic substances (Fenner *et al.* 2005a). Freeman *et al.* (2001) proposed that it is the oxygen constraint on phenol oxidase that is the main inhibitor of decomposition in peatlands, hence drought or drainage may cause substantial carbon mineralisation (‘enzymic latch’ hypothesis).

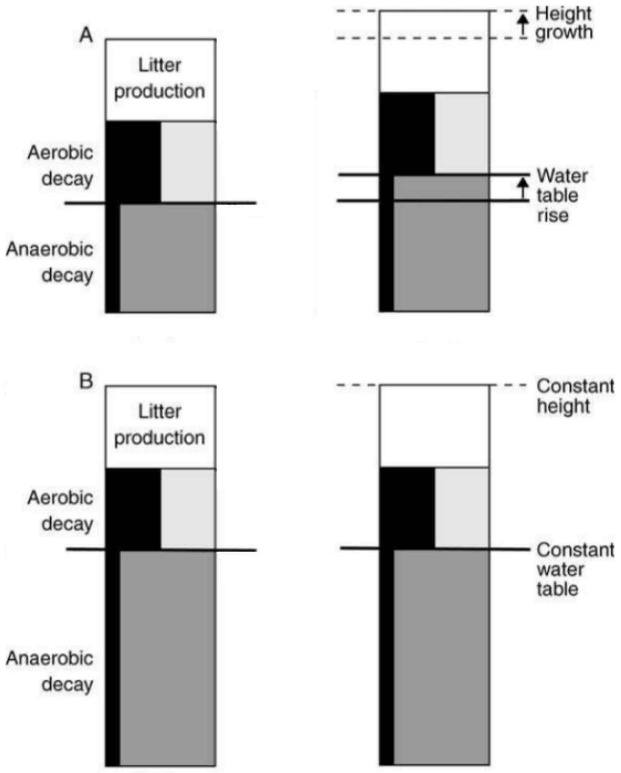


Figure 2.7: Conceptual model detailing a 'limit to growth'. Constant input (white) to the acrotelm (medium grey) is added annually, and a constant of this is lost to aerobic decomposition (black). Constant anaerobic decay of catotelm material also occurs. The water table rise matches bog growth, maintaining constant acrotelm thickness. (A) an early stage of bog development, where litter inputs exceed decay losses. (B) in later stages of bog development, the ‘limit’ is reached as inputs equal losses (adapted from Belyea and Baird (2006)).

Carbon exists in peatlands in four main forms, carbon dioxide (CO_2), methane (CH_4), dissolved organic carbon (DOC) and particulate organic carbon (POC) (Figure 2.8). CO_2 flux from peatlands is measured to track productivity (photosynthetically fixed CO_2) and decomposition in the form of ecosystem respiration (CO_2 lost through plant (autotrophic) and microbial respiration (heterotrophic)) (Lafleur, 2009; Pries *et al*. 2015). These two measures together determine the net ecosystem CO_2 exchange (NEE) (Koehler *et al*. 2011; Waddington and Roulet, 2000). CH_4 is a product of anaerobic decomposition, hence its production is consigned primarily to the catotelm. Methane is often oxidised to CO_2 in the aerated acrotelm (Basiliko *et al*. 2004; Bubier and Moore, 1994), though vascular plant aerenchyma tissue allow methane to bypass aerated substrate (Smith *et al*. 2003a). DOC is the general term applied to both high (e.g. phenolics) and low (e.g. amino acids) molecular weight organic compounds suspended in solution and $<0.45\mu m$ in size (Fenner *et al*. 2004, Kolka *et al*. 2008). DOC originates in peatlands through plant exudation and as a component of the decomposition process (Holden, 2005a). The implications for carbon store integrity therefore differ according to the age of DOC, namely, does DOC originate from ‘young’ or ‘old’ carbon (Fenner *et al*. 2004). Results suggest DOC is primarily derived from recently fixed carbon in intact blanket peats and so from the early stages of decomposition/exudation (Tipping *et al*. 2010). High DOC concentrations and export may therefore not indicate abnormal alterations to biogeochemical cycling (‘predisposed release concept’ (Freeman *et al*. 2004a)), provided contemporary carbon accumulation is still occurring (Evans *et al*. 2014b). POC is the flux which represents the physical loss of peat and the carbon it contains (i.e. previously sequestered catotelm carbon) during erosion (Evans and Lindsay, 2010; Francis, 1990; Pawson *et al*. 2008).

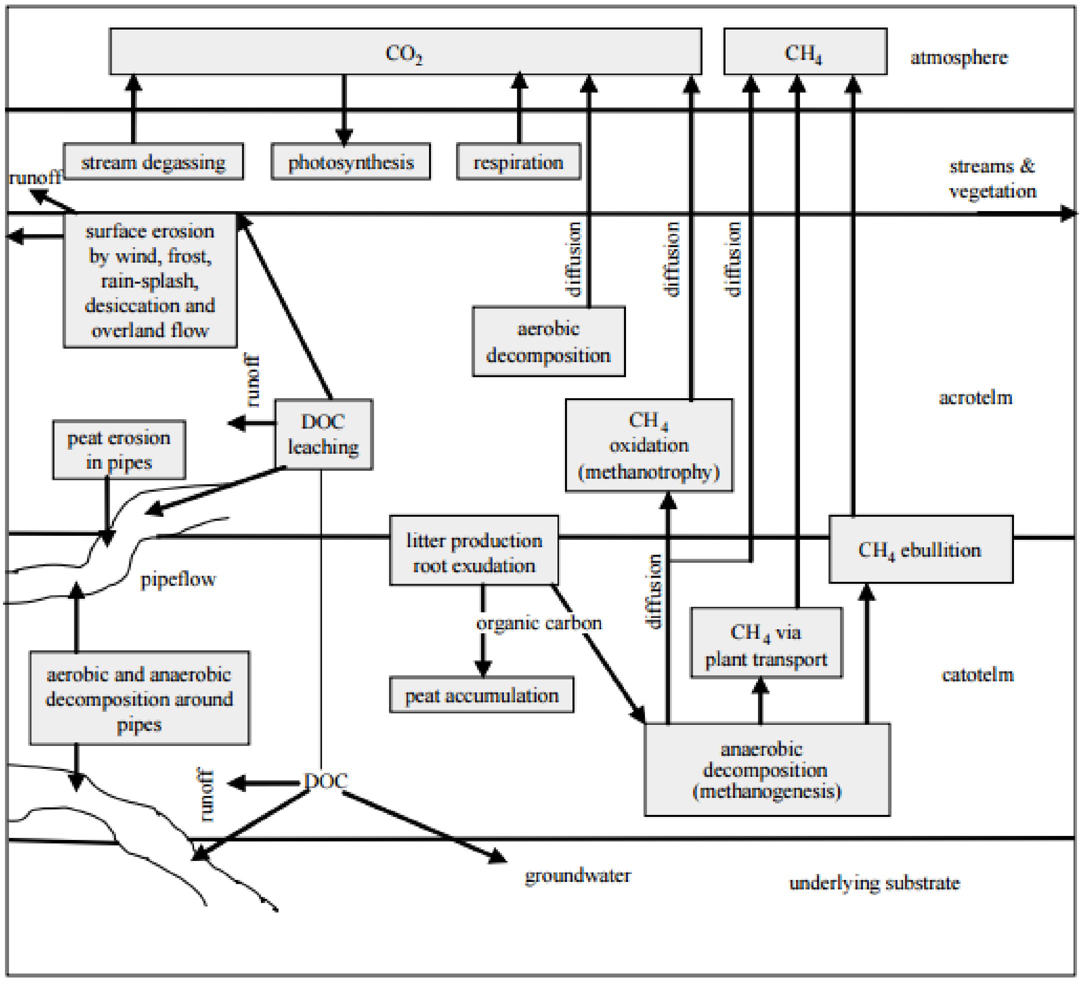


Figure 2.8: Conceptual diagram detailing components of blanket peatland carbon cycling. Abbreviations detailed in Section 2.2.2 (Holden, 2005a).

2.2.3 Measuring peatland carbon dynamics

Peatland carbon accumulation is a complex process that varies in time and space. A variety of methodologies exist to examine current changes in the carbon balance (‘contemporary monitoring’), to predict changes using mathematical formulae (‘modelling’) and to reconstruct past carbon dynamics (‘palaeoenvironmental approaches’) (Figure 2.9). The information below considers each of these methodologies in turn and both 1) shows why palaeoenvironmental approaches are an acceptable approach for this thesis to understand how land use affects carbon dynamics and 2) presents caveats concerning the use of palaeoenvironmental approaches which this thesis will address.

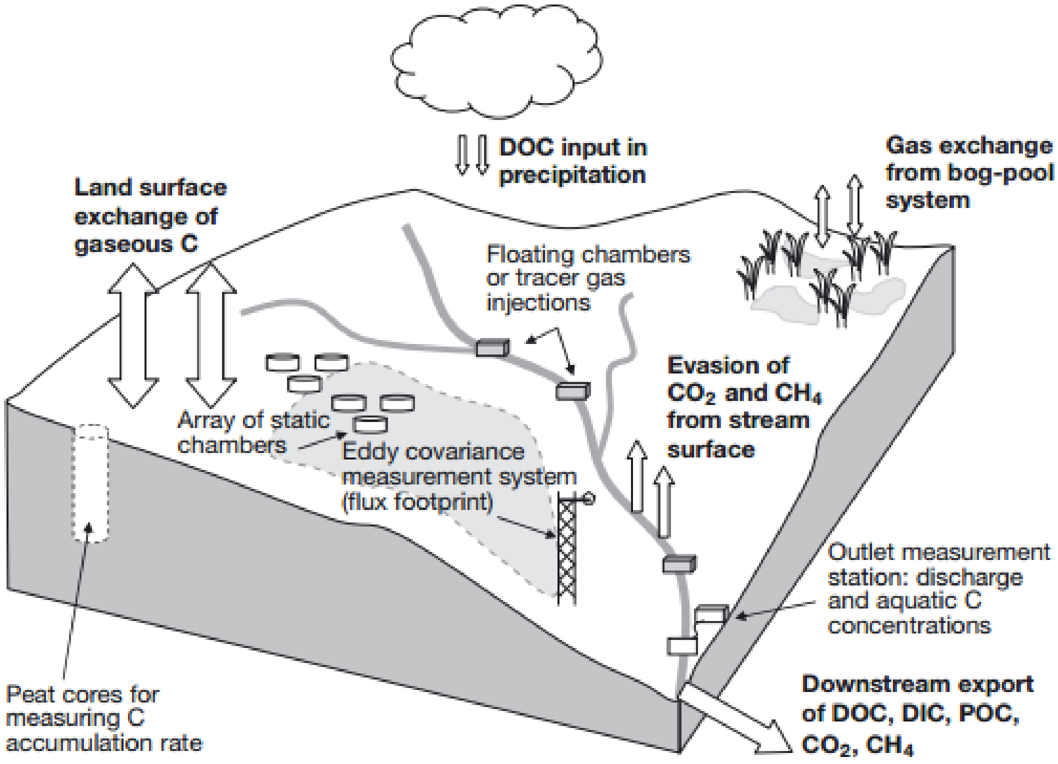


Figure 2.9: Diagram detailing methods for investigating blanket peatland carbon dynamics. Arrow size represents magnitude of flux. DIC signifies dissolved inorganic carbon. Other abbreviations detailed in Section 2.2.2 (Billett *et al*. 2010).

Contemporary carbon fluxes (e.g. CO_2, CH_4) can be assessed *in situ* using a variety of experimental methodologies (Billett *et al*. 2010). Flux measurements require high temporal frequency and duration to characterise sub- and inter-annual emission variability (Delarue *et al*. 2015; Dinsmore *et al*. 2010, 2013; Evans and Lindsay, 2010; Limpens *et al*. 2008). Bubier *et al*. (2007) and Billett *et al*. (2010) recommend a minimum study duration of five-years. The longest currently published (AD 2017) flux record is a fifteen-year study from the Mer Bleue peatland, Canada (Humphreys *et al*. 2014; Roulet *et al*. 2007). Even long-term flux studies (e.g. eight-years (Dorrepaal *et al*. 2009)) may not be sufficient to characterise decadal-scale changes in carbon dynamics, let along centennial-scale ones (Page and Baird, 2016). Multiple within-site measurements are also required to understand the spatial variability of carbon accumulation (Chapman and Thurlow, 1996; Frogbrook *et al*. 2009; Malmer and Wallén, 1999; Waddington and Roulet, 1996). Focussing on the response of a single flux to a perturbation does not present a complete understanding of changes in peatland carbon dynamics (Rowson *et al*. 2010; Worrall and Burt, 2005). One flux may increase (e.g. CO_2) whilst others decrease (e.g. CH_4 and DOC) in response to a perturbation such as water table drawdown (Gažovic *et al*. 2013). Inclusion of all fluxes calculates a 'carbon budget' that determines whether a system is net accumulating or emitting carbon (Koehler *et al*. 2011).

There is a lack of flux data recording 'before and after' changes in peatland management given the expense and logistical constraints with performing such long-term studies (Kareksela *et al*. 2015). Chronosequence flux studies examine multiple sites along a management continuum (i.e. at different periods within a rotational burning cycle) (e.g. Clay *et al*. 2012, 2015) to attempt to address the lack of 'before and after' studies. Chronosequence approaches assume each site would develop in the same manner were it not for the impact of a given variable (Collins and Adams, 1983; Johnson and Miyanishi, 2008). Blanket peatland heterogeneity undermines such an assumption (Glaves *et al*. 2013; Turner *et al*. 2013); hence it cannot be certain whether observed differences are arising due to the impacts of the management type of interest or through intrinsic site-specific differences.

Models use simplified representations of reality to study peat accumulation over long periods. Clymo (1984) used constant productivity and decomposition mechanics to propose a theoretical point where net peatland decay would equal production, thereby reaching a 'limit' to peat and carbon accumulation (Figure 2.7). Empirical peat accumulation data agree reasonably well with such outputs (Charman, 2002). 'Constant' input and decay mechanics have been criticised as inappropriate for real-world environments (Belyea and Baird, 2006; Kilian *et al*. 2000). Recent modelling incorporates autogenic feedbacks alongside external forcings to simulate peatland development. Model outputs can be compared with palaeoecological data for validation (Belyea and Malmer, 2004; Frolking and Roulet, 2007; Quillet *et al*. 2013, 2015; Spahni *et al*. 2013; Tuittila *et al*. 2013; Wu and Blodau, 2013). Most published models (as of AD 2017) have limited applicability in the blanket peatland land use debate because of their complicated and high cost formulation, alongside uncertainties in modelling anthropogenic forcings in detail (e.g. Frolking *et al*. 2010, 2014; Kurnianto *et al*. 2015). A recent study suggests anthropogenic impacts (in this case drainage) may be incorporated into detailed models (Young *et al*. 2017). These results showed some agreement with *in situ* studies (e.g. 'self-rewetting' as proposed by

Williamson *et al.* (2017)), but these results require further testing against as yet undeveloped models which incorporate anthropogenic forcings.

Peat cores can be used to reconstruct net carbon accumulation rates over decadal-millennial timescales (Billett *et al.* 2010). Such palaeoenvironmental approaches assess cumulative impacts of past changes (Kareksela *et al.* 2015) and integrate all carbon budget components (Turunen, 2003). Studies using multiple within-site cores demonstrate internal variability in carbon accumulation records (Ireland *et al.* 2013; Loisel and Garneau, 2010; Ohlson and Dahlberg, 1991; Ohlson and Økland, 1998; Watson *et al.* 2015). True characterisation of the carbon dynamics of a peatland may therefore not be achievable using a single record. As such, this thesis will use multiple cores to assess within-peatland variability in carbon dynamics (Section 2.8).

Palaeoenvironmental rates of carbon accumulation can be calculated according to a variety of techniques, namely long-term apparent rate of carbon accumulation (LORCA), carbon accumulation rate (CAR), recent apparent rate of carbon accumulation (RERCA) and actual rate of carbon accumulation (ARCA). ‘Apparent’ carbon accumulation rates do not account for carbon lost to decomposition but are instead measures of carbon in the peat at the time of core extraction. Apparent rates determined for peat that accumulated at a point in time (Time A) prior to a perturbation occurring (Time B) may underestimate the ‘true’ rate of carbon accumulation if the perturbation promoted decomposition of peat sequestered at Time A (Frolking *et al.* 2014). Acrotelm peat at the time of core extraction has undergone much less decomposition relative to deeper substrates. Surface layers therefore produce greater carbon accumulation rates than older sections of a core (Beilman *et al.* 2009; Billett *et al.* 2010; Earle *et al.* 2003). Such recently accumulated peat is often not compared with older portions of the peat profile to avoid misleading conclusions regarding recent changes in carbon accumulation (Garneau *et al.* 2014; Magnan and Garneau, 2014; Roulet *et al.* 2007; Tolonen *et al.* 1992; van der Linden *et al.* 2014). This thesis will attempt to determine where the acrotelm/catotelm transition occurs to better understand how rates of acrotelm carbon accumulation relate to carbon sequestered into the catotelm (Section 2.8).

LORCA is calculated by dividing the mass of the peat deposit by its basal age (Lindsay, 2010). Inter-site comparisons are limited unless peats are of comparable thickness (Yu *et al.* 2010) or age (Turunen, 2003) as only then could relative differences in LORCA be identified. Studies using LORCA fail to account for variable peat accumulation rates (PAR), treating the system as a 'black box' in assuming all peatlands follow similar post-initiation development trajectories (e.g. Gorham *et al*. 2012; Korhola *et al*. 1995; Nakatsubo *et al*. 2015). CAR is a revision of LORCA that accounts for variable PAR between dated horizons (Mäkilä, 1997). There can be confusion regarding the use of CAR and LORCA in the literature. Some (e.g. Garneau *et al*. 2014; Turunen *et al*. 2004) use LORCA to describe carbon accumulation calculated using a variable PAR. Their method is therefore closer to CAR (*sensu* Mäkilä, 1997). A dataset composed entirely of Finnish sites (Mäkilä, 2011; Mäkilä and Goslar, 2008) suggested intact bogs exhibited greater average CAR for the last 300 years whilst drained sites occupy the lower end of the spectrum (Figure 2.10).

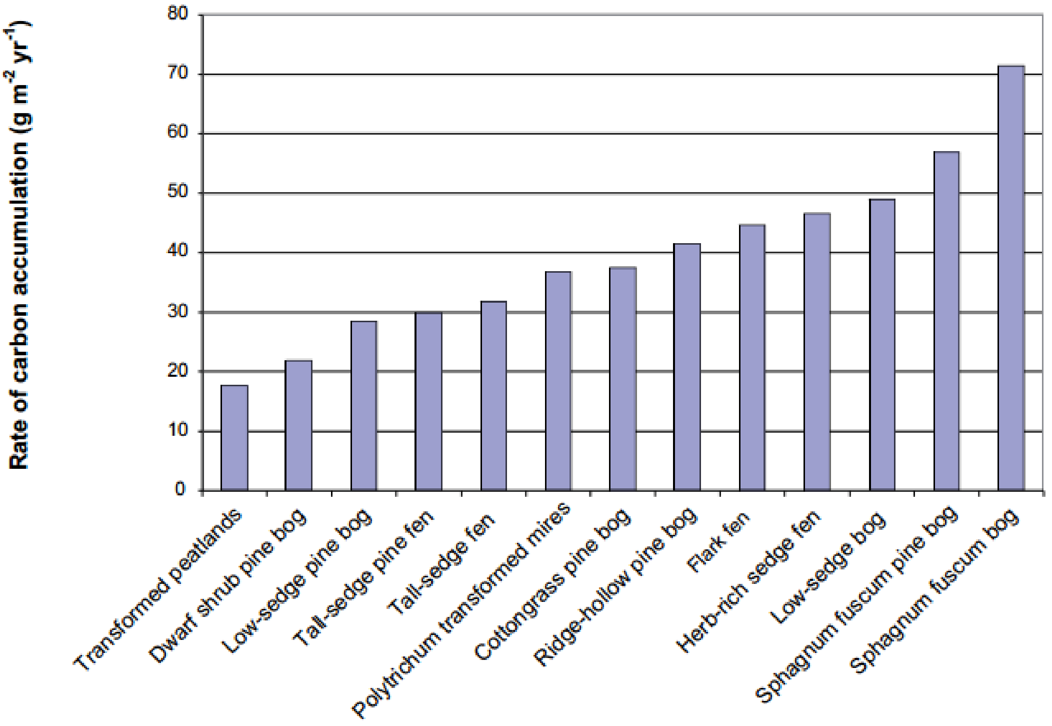


Figure 2.10: Details average CAR for those measured increments <300 years old, classed according to site type as detailed in the source document (Mäkilä, 2011).

RERCA is similar to LORCA as a constant PAR is calculated by dividing the peat mass above a dated point by the time elapsed since that dated point (Lindsay, 2010). RERCA often exceeds CAR/LORCA with values ranging from 11.8-380 g C m^{-2} yr^{-1} across differing peatland types and regions (Bao *et al*. 2015; Lamarre *et al*. 2012; Ohlson and Økland, 1998; Tolonen *et al*. 1992; Turunen *et al*. 2004). There is confusion regarding the use of CAR and RERCA. Some (e.g. Turner *et al*. 2014) refer to RERCA when calculating carbon accumulation using differing PARs. In this way, the approach is interchangeable with CAR (*sensu* Mäkilä, 1997). RERCA provides the best palaeoenvironmental estimates of contemporary peatland carbon accumulation given the technique's focus on recent timeframes. Values from the Moor House National Nature Reserve (NNR) found values of 20-125 g C m^{-2} yr^{-1} (Hardie *et al*. 2007). Billett *et al*. (2010) used data from four blanket peatlands to suggest carbon accumulation post-AD 1970 (56-72 g C m^{-2} yr^{-1}) may have declined relative to carbon accumulation rates *ca*. 150 years ago (35-209 g C m^{-2} yr^{-1}).

ARCA represents the balance between material sequestered to the catotelm and substrate lost to decomposition (Lindsay, 2010). ARCA often produces lower values than other carbon accumulation measures, typically recorded as 70% of LORCA (Korhola *et al*. 1995). 'Apparent' methodologies are more commonly published given complexities of using ARCA decay coefficients. Accounting for ongoing decay can produce a more accurate view of peatland functioning. Wang *et al*. (2015) demonstrated this where the inclusion of decay coefficients suggested post-AD 1960 CAR increased 1.6x times relative to Holocene average CAR rather than the apparent fourfold increase where decay coefficients weren't used.

2.2.4 Controls on carbon accumulation

This sub-section discusses how the identified four controls of '*plant community structure*', '*water table position*', '*peat matrix characteristics*' (adapted from '*peat temperature*') and '*chemical composition of plant tissues and peat*' (Blodau, 2002; Holden, 2005a) may affect blanket peatland carbon accumulation. Understanding presented below is used in later sections of this literature review to explore how specific land uses may affect carbon accumulation if they affect one of these four controls.

2.2.4.1 Plant community structure

Peatland management has an acknowledged capacity to alter vegetation and there are concerns this may alter carbon accumulation (Bain *et al*. 2011; Ward *et al*. 2007) as vegetation structure, litter quality and root dynamics all directly influence carbon cycling (Armstrong *et al*. 2012, 2015; Belyea and Malmer, 2004; Heijmans *et al*. 2002, 2008; Limpens *et al*. 2008; Ward *et al*. 2015). The tolerance of vegetation to biotic and abiotic variables (Figure 2.11) alongside inter/intra-specific competition determine plant community structure (Dorrepaal *et al*. 2005; Sottocornola *et al*. 2009). Studies primarily designate peatland vegetation according to functional groups; bryophytes (primarily *Sphagnum*), sub-shrubs and graminoids (Breeuwer *et al*. 2009; Kuiper *et al*. 2014).

Sphagnum mosses are principal peatland constituents often declared ‘ecological engineers’ as they manipulate their environment to encourage their survival (Hájek *et al*. 2009; Laing *et al*. 2014; van Breemen, 1995). Recalcitrant litter ensures *Sphagnum* often constitute a substantial component of peat matrices (Heijmans *et al*. 2001; Robroek *et al*. 2007a; Turetsky, 2003) and mean the genus is often deemed as playing a key role in long-term carbon accumulation (Heijmans *et al*. 2002; Limpens *et al*. 2012). The position of a *Sphagnum* species along the microtopographical gradient (Figure 2.11) has implications for carbon accumulation. Bog hollow *Sphagnum* species typically exhibit faster growth rates than hummock *Sphagnum* as they assign resources primarily as metabolic carbohydrates (Jiroušek *et al*. 2015; Turetsky *et al*. 2008). Metabolic carbohydrates confer greater productivity but are preferential for decomposition (Gunnarsson, 2005; McNeil and Waddington, 2003; Robroek *et al*. 2007a). Hummock affiliated *Sphagnum* species assign resources as recalcitrant structural carbohydrates and exhibit lower decomposition rates than hollow *Sphagnum* species (Belyea, 1996; Clymo, 1965; Johnson and Damman, 1993; Moore *et al*. 2007; Rochefort *et al*. 1990). Hummocks are thereby often perceived to be larger carbon sinks than hollows (Byrne and Farrell, 2005; Laine *et al*. 2007; Moore *et al*. 2015).

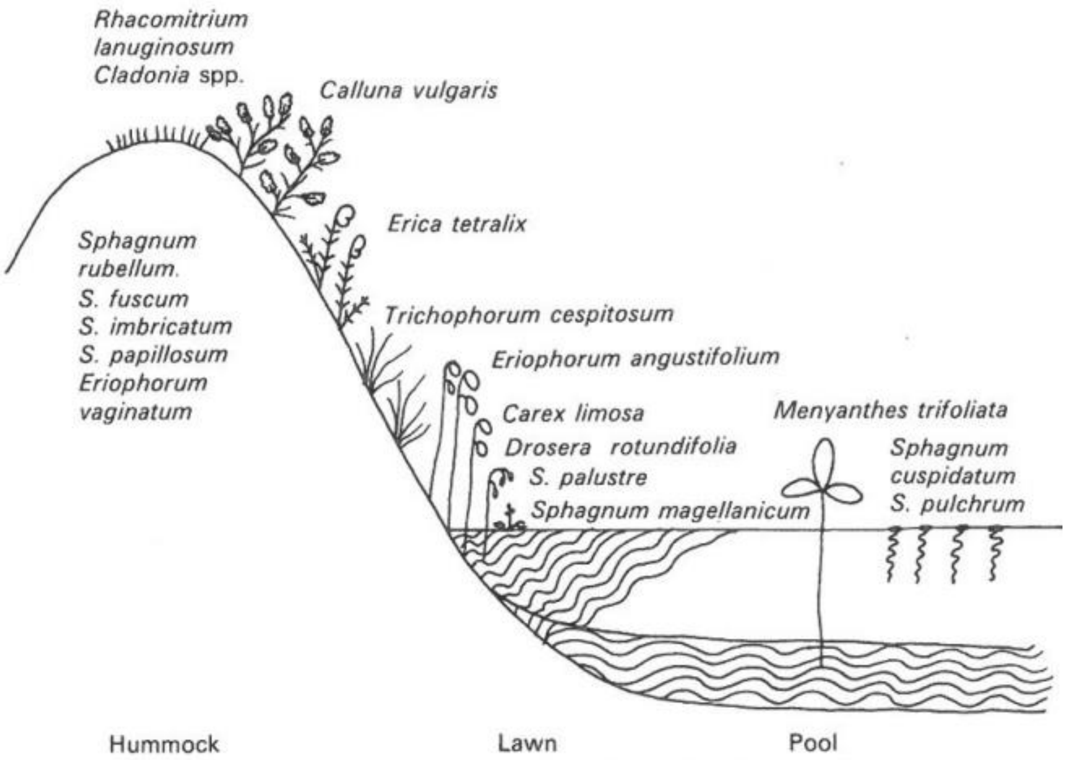


Figure 2.11: Generalised raised bog vegetation affinity to water table along a hummock-pool gradient (Birks and Birks, 2004).

Vascular plants exhibit greater NPP than bryophytes (Blodau, 2002). The majority of vascular NPP fails to be sequestered to the catotelm as decomposers favour nutrient enriched vascular tissue (Heijmans *et al*. 2002; Laiho, 2006). Whilst *Calluna vulgaris* and *Eriophorum vaginatum* NPP was 10x that of *Sphagnum*, *Sphagnum* litter provided 28.5 g m^{-2} of the 30 g m^{-2} annually sequestered to the catotelm in Wallén (1992). Exudates and secretions from plant roots provide labile matter for CO_2 and CH_4 production (Del Giudice and Lindo, 2017; Dunn *et al*. 2016; Heijmans *et al*. 2001; Leroy *et al*. 2017; Wallén, 1992) and stimulate the decomposition of recalcitrant substrates by 'priming' microbial communities (Dieleman *et al*. 2017; Gogo *et al*, 2011; Kuzyakov, 2002, 2010; Robroek *et al*. 2016). Vascular plants also stimulate evapotranspiration and water table drawdown (Fenner *et al*. 2007a; Wendel *et al*. 2011). Many peatlands therefore exhibit a reduction in carbon accumulation, or switch to a net carbon source, following the replacement of *Sphagnum* with vascular plants (Fenner *et al*. 2007a, b; Heijmans *et al*. 2002; Nichols *et al*. 2014; Silvola *et al*. 1996).

Graminoid expansion and a loss of biodiversity is a key peatland conservation concern (Anderson *et al*. 2006; Chambers *et al*. 2007a, b, 2013) and nutrient rich graminoid tissue is preferential for decomposers (McNamara *et al*. 2008; Trinder *et al*. 2008). Monocot roots penetrate deeper into the peatland than shrub roots (Moore *et al*. 2002), supplying oxygen and labile matter for decomposers (Hardie *et al*. 2009). Whilst many associate sub-shrub assemblages with slower carbon cycling relative to graminoid assemblages (Hobbie, 1996; Ward *et al*. 2007, 2009), others disagree. No blanket peatlands dominated by *C. vulgaris* were net sinks in Dixon *et al*. (2015) given the shrub's tendency to promote respiration and water table depth (WTD) increases. Armstrong *et al*. (2012) recommended *C. vulgaris* removal to promote carbon accumulation given the greater WTD and *in situ* DOC concentrations recorded relative to *Sphagnum* and *Molinia caerulea* plots.

Floristic communities indicate environmental change through their integration of short/long-term responses to peatland developmental controls (e.g. WTD). Their response to environmental drivers suggests vegetation may act as reliable proxies for GHG dynamics (Couwenberg *et al*. 2011; Karu *et al*. 2014). Such a proxy method is of value to peatland GHG management, being easier to implement than soil property mapping (Garnett *et al*. 2001) or ecosystem modelling (Audet *et al*. 2013). Bubier *et al*. (1995) demonstrated peatland vegetation acts as a proxy for methane emissions. Differing vegetation types display characteristic CH_4 emission values (Couwenberg *et al*. 2011; Gray *et al*. 2013; Kao-Kniffen *et al*. 2010) given the interdependence of flora and CH_4 flux with WTD (Bubier and Moore, 1994). Inter/intra-specific competition and GHG dynamics responding faster to perturbations weaken such vegetation based proxy methods (Audet *et al*. 2013; Dias *et al*. 2010). The applicability of a proxy method in wider peatland management may also be limited if vegetation/carbon relationships are site specific (Bubier *et al*. 1995). This thesis will explore whether vegetation type may be a useful indicator of carbon dynamics for Welsh blanket peatlands in addition to investigations around the impacts of specific land use types on carbon accumulation (Section 2.8).

2.2.4.2 Water table position

Water tables and the processes they regulate vary through time (Laine *et al*. 2007; Worrall *et al*. 2006) and space (Chapman and Thurlow, 1992; Lewis *et al*. 2012), necessitating multiple measurements to characterise WTD. Changes in the position of the saturated boundary determine the relative strengths of aerobic and anaerobic processes (Fenner *et al*. 2005b; Waddington and Price, 2000). Anoxia induced below the water table may be the main inhibitor of decomposition in peatlands (Kettunen, 2003; Laine *et al*. 2009; Philben *et al*. 2014) given aerobic peat decomposition rates are approximately fifty times greater than anaerobic rates (Clymo, 1983). Falling water tables expose previously submerged peat to aeration, stimulating decomposition (Brake *et al*. 1999; Pind *et al*. 1994) and the loss of carbon sequestrated to the catotelm (Evans *et al*. 2014b). Whilst water tables are deeper under hummocks than hollows, the former often display greater carbon accumulation (Malmer and Wallén, 1999; Tolonen *et al*. 1992; Turunen *et al*. 2004). Substrate quality is key in determining how labile carbon is under falling water tables (Straková *et al*. 2011). Only the decomposition of labile litter types (Hogg, 1993; Lindsay, 2010) and the upper reaches of the catotelm may increase if WTD increases (Chimner and Cooper, 2003; Hogg *et al*. 1992). Pre-perturbation WTD thereby regulates the carbon cycling response to water table drawdown through determining antecedent vegetation and so litter type exposed to aerated decomposition.

2.2.4.3 Peat matrix characteristics

Loss of surface vegetation exposes bare peat, leading to desiccation as substrate moisture content declines as evaporation increases (Holden *et al*. 2001). Desiccated peat is vulnerable to erosion by the elements (Foulds and Warburton, 2007) and land management (e.g. sheep disturbing the surface) (Ostle *et al*. 2009). Substrate loss represents a loss of previously sequestered carbon and a reduced capacity for future carbon accumulation as the unstable surface is difficult for vegetation to recolonize (Ramchunder *et al*. 2009; Salonen, 1992). Revegetation of bare peat is recommended to conserve existing carbon deposits and promote future carbon accumulation (Dixon *et al*. 2015; Holden *et al*. 2008; Worrall and Rowson, 2008; Worrall *et al.* 2011a). Severe erosion can promote the formation of erosion channels through the peat (Figure 2.12). There are multiple explanations as to why erosion complexes form, ranging from their expectance as development end-points (Conway, 1954; Johnson, 1957; Pearsall, 1956) to that human activity both initiates (Ellis and Tallis, 2001; Mackay and Tallis, 1996) or accelerates a ‘naturally’ occurring trend (Bradshaw and McGee, 1988).

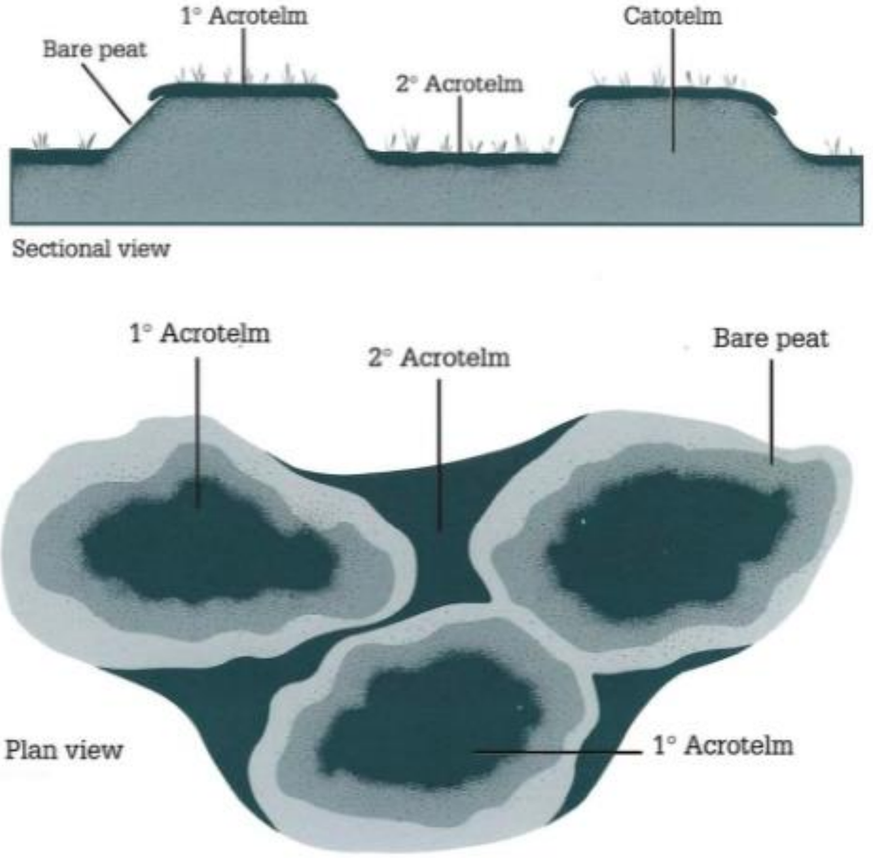


Figure 2.12: Schematic diagram displaying secondary order erosion complex in sectional and plan view. Sides of hags (‘bare peat’) signify the exposed catotelm of the original surface (‘1° acrotelm’), with a secondary acrotelm (2°) forming on previously bare peat. Primary acrotelm (1°) may still be accumulating (Lindsay, 1995).

Temperature regulates most soil biogeochemical processes (Biederbeck and Campbell, 1973; Lloyd and Taylor, 1994). Increases in temperature can promote microbial activity and so increase decomposition rates (Bergman *et al.* 1999; Briones *et al.* 2014; Brown *et al.* 2015a). Moore and Dalva (1993) demonstrate this where extracted cores displayed increased CO_2 (x2.4) and CH_4 (x6.6) emissions at 23°C relative to 10°C. Temperature driven increases in decomposition may primarily affect labile substrates (Briones *et al.* 2014; Conant *et al.* 2011; Hogg *et al.* 1992) and only affect recalcitrant matter if fresh labile matter is available as this would stimulate microbial activity (Carrera *et al.* 2015).

2.2.4.4 Chemical composition of plant tissues and peat

Atmospheric pollution can alter habitat functioning through the provision of major macronutrients (nitrogen (N), phosphorous (P) and potassium (K)) and other elements (e.g. titanium) (Bragazza *et al.* 2005; Fowler *et al.* 2005; Malmer and Wallén, 2005). Low nutrient bog environments are sensitive to atmospheric nutrient loading given their ombrotrophic nature (Porter *et al.* 2012; Woodin and Farmer, 1993; Woodin *et al.* 1991). Persistent upland clouds with high precipitation frequency render blanket peatlands particularly vulnerable to nutrient loading (Britton *et al.* 2005; Kirkham, 2001). A 'critical loading limit' is the maximum exposure to a pollutant a habitat can withstand without the pollutant degrading the habitat's functioning (Nilsson and Grennfelt, 1988). The blanket bog critical loading limit is 0.5-1 g N m^{-1} yr^{-1} (5-10 kg N ha^{-1} yr^{-1}) (Bobbink and Roelofs, 1995). Variance within this range may be expected but has not been adequately examined to date (AD 2017).

N loading affects carbon cycling both directly and indirectly (Currey *et al.* 2010). Microbial communities favour plant litter with high nutrient concentrations for decomposition (Aerts and Chapin III, 2000; Hogg *et al.* 1994; Limpens and Berendse, 2003; Lund *et al.* 2009; Scheffer *et al.* 2001). N loading also promotes extracellular enzyme activity via inhibiting *Sphagnum* phenolic production (Bragazza and Freeman, 2007) and promoting phenol oxidase activity (Bragazza *et al.* 2006, 2012). Nutrient loading indirectly affects carbon accumulation by altering peatland vegetation assemblages. *Sphagnum* thrives under N loads below the critical loading limit where it intercepts nutrients from atmospheric fallout, limiting nutrient availability for other plant types (Granath *et al.* 2009, 2012; Malmer *et al.* 2003). At N loads exceeding the critical limit, the inability of *Sphagnum* to regulate its nutrient uptake leads to tissue N content reaching toxic levels as P limits productivity (Berendse *et al.* 2001; Lamers *et* al. 2000; Limpens *et al.* 2004; Tomassen *et al.* 2003). *Sphagnum* displacement by N pollution promotes the growth of vascular species that were previously nutrient limited, this posing negative implications for carbon accumulation (Section 2.2.4.1; Basiliko *et al.* 2006; Bragazza *et al.* 2004, 2005; Gunnarsson and Rydin, 2000; Malmer and Wallén, 2004; Nordbakken *et al.* 2003; Wu and Blodau, 2015).

Local factors can alter peatland nutrient loading in addition to diffuse pollution sources. N and P are associated with small soil particles vulnerable to wind entrainment. Any forcing increasing particle susceptibility to erosion and redistribution can therefore alter local nutrient regimes (Field *et al.* 2010). Dust deposition from localized landscape clearances likely caused the wide-scale decline of *Sphagnum austinii* and its replacement with other *Sphagnum* species or vascular plants (Hughes *et al*. 2007, 2008; McClymont *et al*. 2008; Swindles *et al*. 2015a; Turner *et al*. 2014; van Geel and Middeldorp, 1988). Dust inputs altered nutrient regimes in an American kettle peatland, prompting a change from a *Sphagnum* to vascular plant dominated system and increasing rates of peat decomposition (Ireland and Booth, 2012). Nutrient leaching from tephra deposition on a Japanese bog shifted the site towards fen-like vegetation as *Carex* displaced an *S. magellanicum*/*S. papillosum* community, with an associated decline in CAR to 9 g C m^{-2} yr^{-1} relative to the 27 g C m^{-2} yr^{-1} pre-eruption average (Hughes *et al*. 2013). The system exhibited resilience with CAR increasing to an average of 67 g C m^{-2} yr^{-1} (peaking at >100 g C m^{-2} yr^{-1}) for *ca.* 300-years alongside *S. papillosum* re-emergence.

2.2.5 Section summary

This section has provided a background to the blanket peatland land use debate and demonstrated how four controls may influence carbon cycling. The following sections build upon this initial understanding and discuss how selected management practices (Section 2.1) affect carbon cycling through altering these four controls.

2.3 Impacts of burning on carbon accumulation

Anthropogenic firing in the uplands of the British Isles dates from the Mesolithic (Froyd, 2006; Innes and Blackford, 2003). The prevalence of burning has increased over the last *ca*. 200 years through rotational burning to regulate vegetation structure (Harris *et al*. 2011). Limited quantitative information is available concerning the impact of rotational burning on blanket peatlands (Davies *et al*. 2016a; Sutherland *et al*. 2006) despite concerns regarding the effect this practice may have on peatland functioning (CCC, 2015; Dawson *et al*. 2002). This section discusses the impact of rotational burning on blanket peatlands and assesses the way these impacts may affect carbon accumulation.

2.3.1 Background to the use of burning in blanket peatland management

Fire ignition occurs when the temperature of an item rises above a specific point in the presence of oxygen. Fire intensity is a measure of energy/heat production at the flaming front. Fire speed (duration of time an area is subjected to a given intensity) regulates impacts as slow moving hot fires result in greater damage than faster moving cooler burns (Glaves *et al*. 2013). Rotational burning uses 'cool burns' which fail to ignite the main peat body (i.e. <100°C at 1cm peat depth) or remove surface vegetation and litter (Gray and Levy, 2009). It is perhaps debateable whether this standard can be consistently achieved as multiple real-world (Brown *et al.* 2015a; Yallop and Clutterbuck, 2009) and experimental studies (Lee *et al.* 2013; Rawes and Hobbs, 1979) record 'bare peat' where rotational burning is practiced.

The primary rationale for burning on shooting estates is to control the growth of *C. vulgaris* for red grouse (*Lagopus lagopus scotica*). Differing areas (≤2 hectares in size) are burned on an 8-25 year return-interval to create a mosaic of *C. vulgaris* of differing ages (Ramchunder *et al*. 2013). Younger *C. vulgaris* stands offer greater nutrient concentrations for food whilst mature stands provide shelter for grouse chicks (Lance, 1983). Responses to firing differ markedly between blanket bogs and heathlands, yet many regulatory authorities do not acknowledge differing habitat types in upland management policy (Thacker *et al*. 2014; Yallop *et al*. 2006a, b). On actively accumulating bogs heather should not reach the degenerative phase of its growth cycle at it requires continual sprouting of new shoots to avoid burial beneath the peat surface (Forrest, 1971; MacDonald *et al*. 1995; Wallén, 1983). Rationale for using rotational burning to maintain *C. vulgaris* vigour on heathlands does not therefore translate to blanket peatland management (Holden *et al*. 2007a; Tucker, 2003). The European Commission (2017) currently questions the legality of burning on active UK blanket bogs protected under the EU Habitats Directive. Grouse shooting is worth nearly £100 million to the UK economy per annum and supports the equivalent of >2,500 jobs in deprived rural areas (BASC, 2014). Grouse moors are currently subsidised at £56 per hectare under the Common Agricultural Policy (HM Government, 2014). Subsidies are contentious (Monbiot, 2014, 2016) given grouse moors are owned by some of the wealthiest members of British society (Avery, 2015; Monbiot, 2013). There is therefore substantial financial and political interest associated with the use of managed firing.

The DEFRA (2007) voluntary burn code presumes against managed burning on blanket bog. It is naïve to assume it no longer occurs (Ramchunder *et al.* 2013; Yallop *et al*. 2006a, b) as an approximate five-fold increase in firing incidence since the AD 1940s across English uplands occurred primarily on blanket bogs

(Thacker *et al.* 2014). Douglas *et al.* (2015) also suggest that rotational burning increased between AD 2000 and 2010 on deep peat (Figure 2.13). These studies rely on aerial and/or satellite imagery to identify fires, and the findings of Douglas *et al.* (2015) have been challenged on the ability of MODIS imagery to detect rotational burning (Davies *et al.* 2016b; Douglas *et al.* 2016a).

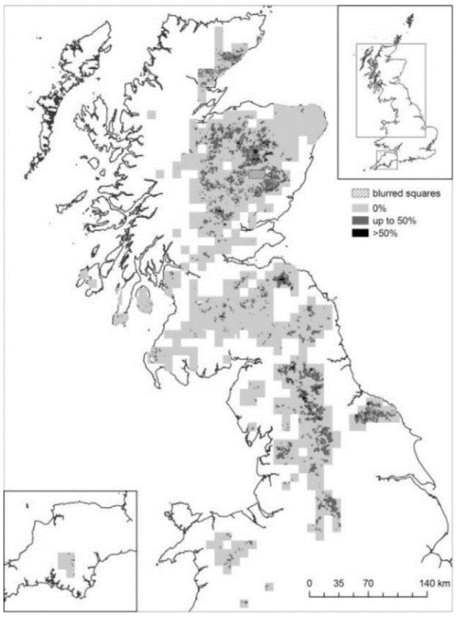


Figure 2.13: Percentage area of *C. vulgaris* dominated upland burned within 1-km grid squares. A third of squares in England and Scotland overlie deep peat (Douglas *et al.* 2015).

The majority of blanket bog burning research originates from the experimental plots at Hard Hill in the Moor House NNR, North Pennines. Established in AD 1954 the experiments focus on the impacts of burning and grazing and their interaction. Four blocks (A-D) (90x60m) comprised of six adjoining sub-plots (30x30m) are positioned along an altitudinal gradient (600-632m asl) (Figure 2.14). All blocks were initially burned, then each sub-plot managed according to treatments of no burning, 10 and 20-year rotational burning and grazed or non-grazed conditions for each firing condition (Alday *et al.* 2015). Grazing pressure has remained ≤0.3 ewes ha^{-1} through the experiment (Rawes and Welch, 1969; Alday *et al.* 2015). The area is classed as M19b, *Empetrum nigrum* spp. *nigrum* sub community of *C. vulgaris- E. vaginatum* blanket mire (Gray and Levy, 2009), with several pre-established ‘reference’ plots available in the vicinity for further comparison. The capacity of these sites to contribute to the debate on peatland management practices is criticised on several grounds. Flaws in the experiment have arisen as trampling from intensive surveying and hydraulic conductivity between sub-plots may mean observed differences (e.g. vegetation community change) are not occurring solely due to differing treatments (Holden *et al*. 2012; Lindsay, 2010). A site-specific lack of *Sphagnum* coverage (Grant *et al*. 2012) and atypical limestone substrate (Gray and Levy, 2009) also suggest results may not extrapolate to other blanket peatlands. Finally, such well-managed small-scale environments may not be true analogues for commercial activities practiced by a wide variety of land managers over heterogeneous landscapes. One should therefore be cautious if applying conclusions and understanding from Hard Hill to other blanket peatlands (Glaves *et al.* 2013; Grant *et al.* 2012).

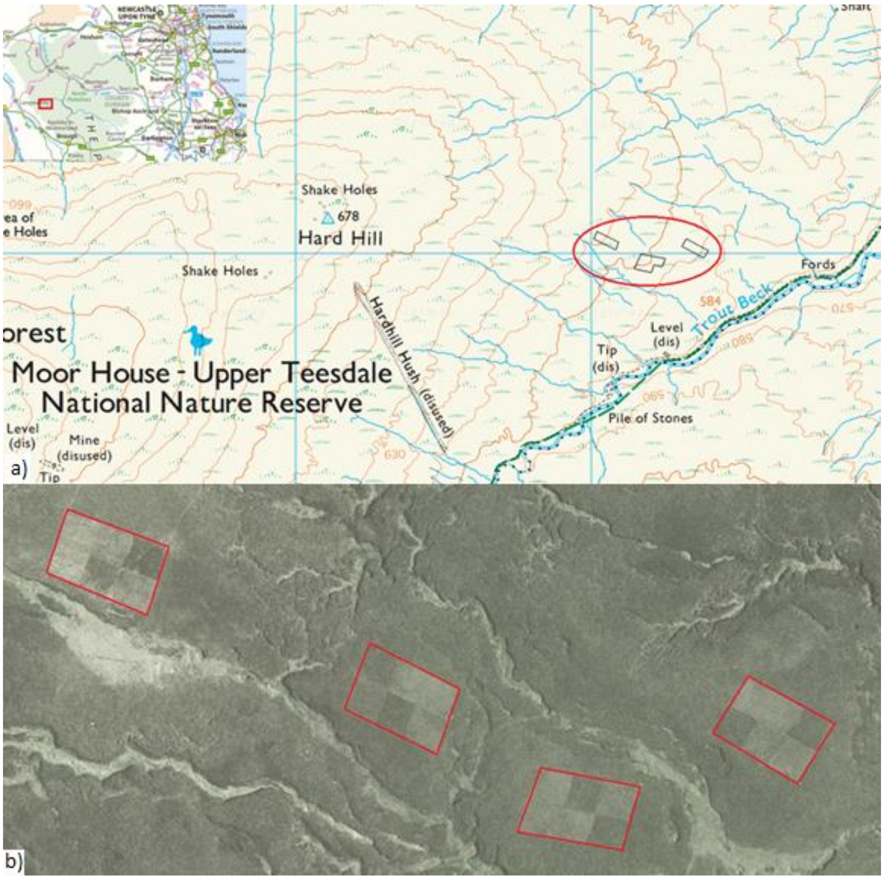


Figure 2.14: a) Location of the Hard Hill experimental plots (the map only acknowledges three blocks) (Edina, 2016). b) Aerial image of the plots (Google, 2016). Note the visible within-block boundaries between each of the six treatments.

2.3.2 The effect of burning on plant community structure

Both firing return-interval (Reed et al. 2010; Worrall and Adamson, 2008) and pre-burn vegetation community (Glaves et al. 2013; Hobbs, 1984) affect the response of vegetation to burning. Return-intervals <10-15 years encourage a graminoid dominated community (Grant *et al.* 2012; Radley, 1965; Stewart *et al.* 2004) as *Eriophorum* sp. and *M. caerulea* grow rapidly following burning due to increased nutrient availability (Anderson *et al.* 2006) and protection from the heat offered by their tussock forms (Shaw *et al.* 1996). Under longer rotations (≥20 years) *C. vulgaris* may recover to dominate at the expense of bryophytes and graminoids if other pressures (i.e. grazing) are excluded (Alday *et al.* 2015; Hobbs, 1984).

There is a tendency to consider fire to have a detrimental impact on *Sphagnum* growth (Bain *et al.* 2011; Glaves *et al.* 2013; Tucker, 2003). Noble *et al.* (2017) recently showed that *Sphagnum* is negatively associated with rotationally burned sites on a national scale but did not taxonomically differentiate within the genus. Harris *et al.* (2011) found *Sphagnum* failed to recover across a range of Peak District sites following burning, though this may be due to the 'marginal' nature of the sites for blanket peat growth (*sensu* Bragg and Tallis, 2001). Feather mosses often outcompete *Sphagnum* following fires in boreal peatlands, which may reduce carbon accumulation given the reduced productivity and greater decomposability of the former (Turetsky *et al.* 2010). The loss of *Sphagnum papillosum* from Keighley Moor and its replacement by an atypical *C. vulgaris* and *E. vaginatum* community is deemed a result of increased firing frequency for grouse management in the 20th Century. Prior to this, a range of *Sphagnum* genera had dominated the area from AD 500-1900 (Blundell and Holden, 2015). Overbeck (1975, cited in Hughes *et al.* (2008)) suggests that firing and/or subsequent charcoal deposition may have contributed to the loss of *S. austinii* from German bogs. Pigott and Pigott (1963) also implicate burning in the decline of *S. austinii* in the Yorkshire Dales.

It is not simple to conclude that fire has a detrimental impact on *Sphagnum* growth as variable responses to burning have been recorded across the genera (Eddy *et al.* 1969). *Sphagnum capillifolium* recovered to a near pre-fire coverage <10 years after a severe wildfire at Glasson Moss NNR (Lindsay and Ross, 1994). *S. capillifolium* also responds well under managed burning at the Hard Hill sites where it dominated the '10-year rotation' plots alongside *Eriophorum* sp., whereas *C. vulgaris* and hypnoid mosses dominated the '20-year' rotation plots (Lee *et al.* 2013). Results from Abbeyknockmoy, Cors Caron (Hughes *et al.* 2008), Borth (Slater and Slater, 1978) and Walton Moss (Mauquoy *et al.* 2002) suggest no relationship between fire frequency and *S. austinii* abundance. Sub-arctic palaeoecological records show *Sphagnum fuscum* (Kuhry, 1994) and *Sphagnum* undiff. (Pitkänen *et al.* 1999) recovered following severe fires in a few decades.

Increased abundance of graminoids at the expense of *Sphagnum* would likely reduce carbon accumulation given the increase in labile litter supply and aeration of peat by roots (Section 2.2.4.1). Shortening fire return intervals reduced carbon accumulation in Canadian bogs with it taking *ca*. 13 years for net carbon accumulation to start again (Wieder *et al*. 2009). The '10-year rotation' plots at Hard Hill (surveyed nine years post-burn) demonstrated declining aboveground and near-surface carbon stocks (56% and 60% respectively) relative to unburned controls, though no associated change was detected on deeper peat (Ward *et al*. 2007). Decline of carbon-rich shrubs (51%) and bryophyte (91%) biomass alongside increased *E. vaginatum* biomass (88%) was the key change in reducing carbon storage. Burning implementation on a 10-year basis equated to a 25.5g C m^{-2} yr^{-1} reduction in long-term carbon accumulation (Ward *et al*. 2007). Orwin and Ostle (2012) also found reduced biomass in '10-year' plots where *E. vaginatum* dominated relative to '20-year' burning and 'no burn' plots.

Lee *et al*. (2013) believe the *E. vaginatum* dominated '10-year burn rotation' Hard Hill plots are favourable for peat accumulation relative to the '20-year rotation' *C. vulgaris* dominated plots because *E. vaginatum* is a 'favourable' species for peat accumulation under the NVC system (M17-20; Table 1.2; Rodwell, 1991). The designation of *E. vaginatum* as 'favourable' for peat accumulation occurs primarily as its leaf and rhizome fibres are decay resistant, and the fibrous nature of *Eriophorum* peat retains water more efficiently than *Sphagnum* peat (Hughes *et al*. 2000). Both Ostle *et al*. (2012) and Ward *et al*. (2007) suggest that an *E. vaginatum* dominated community may reduce carbon accumulation relative to a mixed bryophyte and *Calluna* community. Peat accumulation and carbon accumulation may therefore not always follow the same trajectory (i.e. as the former increases, the latter declines) if the peat forming flora have a low carbon content (e.g. *E. vaginatum* as in Ward *et al*. (2007)).

New vegetative growth following firing may promote carbon fixation relative to unburned, older vegetation. The increased productivity of recovering vegetation reduced the carbon source size of burned Hard Hill plots (117.8 g C m^{-2} yr^{-1}) relative to unburned conditions (source of 156.7 g C m^{-2} yr^{-1}) (Clay *et al*. 2010). Ward *et al*. (2007) found increased photosynthesis under '10-year burn' plots, though this finding must be set against the reduction in carbon storage under burning. Alday *et al*. (2015) recommended '20-year burn' rotations to improve peatland carbon storage given the increased height and biomass of *C. vulgaris* relative to those under '10-year burn' intervals. Alday *et al*. failed to distinguish between blanket bog and heathlands, assessing their results under the ‘moorland’ moniker and provided no detailed comment on the implications of burning for long-term carbon sequestration. One must consider that whilst vascular plants would be expected to have increased productivity, their contribution to long-term peat and carbon accumulation is less in most cases than less productive *Sphagnum* (Lindsay, 2010).

2.3.3 The effect of burning on water table position

Results from the Hard Hill plots suggest rotational firing reduces WTD (Worrall and Adamson, 2008) which may reduce respiration (Clay *et al.* 2010). Worrall *et al.* (2007a) attribute reduced WTD under burned plots to mature *C. vulgaris* in unburned plots increasing WTD through evapotranspiration. Clay *et al.* (2009a) present the same rationale to explain reduced WTD and also found reduced hydraulic conductivity in the surface layers of burned sites. Hydraulic conductivity of peat decreases as decomposition increases (Morris *et al.* 2015a). If firing did increase decomposition then a lowering of the peat surface may occur which would reduce WTD. This critique is merely observational as Clay *et al.* (2009a) present no bulk density (BD) or humification metrics that may indicate an increase in decomposition. The critique does demonstrate why a reduced WTD may not necessarily increase carbon accumulation. Reduced hydraulic conductivity would also be detrimental for *Sphagnum* growth (Section 2.3.2) given the genus is dependent on passive capillary action to avoid desiccation (Thompson and Waddington, 2008). Reduced WTD under rotational burning is also recorded in the Peak District (Worrall *et al.* 2013).

A survey of five unburnt and five rotationally burned blanket peat catchments across the Pennines found burning significantly lowered water tables (5.3cm mean) due to increased peat temperatures (Brown *et al.* 2015a) promoting evaporation (Holden *et al.* 2015). Sites <2 years post-burn exhibited the greatest WTD whilst sites >10 years post-burn displayed the shallowest water tables. Holden *et al.* (2015) suggest increased WTD from rotational burning may reduce carbon accumulation by promoting decomposition. Peat temperature cannot reach 100°C until its water content evaporates (Tucker, 2003). Peat with a high moisture content (x5 dry weight) and >80cm deep only lost surface vegetation and litter during a wildfire on Rosedale Moor whilst shallower peat with a lower moisture content was either charred or completely consumed (Maltby *et al.* 1990). There are concerns that managed burns on drained blanket peatlands may promote carbon loss greater than fires on undrained bogs given the reduced moisture content of drained sites increases peat vulnerability to combustion (Brown *et al*. 2015b). Whilst results supporting these concerns are yet to be presented from a blanket peatland, drainage of a boreal fen increased wildfire carbon loss 9x relative to undrained conditions (Turetsky *et al.* 2011).

2.3.4 The effect of burning on peat matrix characteristics

Peat temperatures can increase both during a fire (Turetsky *et al.* 2015) and in the years following burning if surface vegetation is completely removed ('clean burn') as this exposes peat to increased irradiance (Brown *et al*. 2015a; Orwin and Ostle, 2012). Greater peat temperatures during/following firing may increase decomposition rates (Mitchell and McDonald, 1992; Ramchunder *et al.* 2013). Peat ignition during a fire will cause the combustion of previously assimilated carbon (Davies *et al*. 2013; Frank *et al*. 2015; Radley, 1965; Turetsky *et al*. 2015) and leave a surface crust which is vulnerable to erosion once the fire has finished (Ramchunder *et al.* 2013; Tallis, 1987; Yallop *et al*. 2010).

The relationship between burning and DOC is unclear. The notion that burning increases DOC export from blanket peatland catchments or increases DOC pore water concentration is both supported (Clutterbuck and Yallop, 2010; Holden *et al.* 2014; Grayson *et al*. 2012; Mitchell and McDonald, 1995; Ramchunder *et al*. 2013; Yallop and Clutterbuck, 2009; Yallop *et al.* 2010, 2012) and refuted (Chapman *et al*. 2010; Clay *et al.* 2009a, b, 2012; Ward *et al*. 2007; Worrall *et al.* 2007a, b, 2013a). Differing DOC responses to rotational burning at the plot scale (e.g. Clay *et al.* 2009b) to that at the catchment scale (e.g. Yallop and Clutterbuck, 2009) pose a difficulty in determining a consistent response (Evans *et al*. 2017; Davies *et al*. 2016a; Holden *et al*. 2012).

LORCA was nearly reduced by half in Finnish peatlands which had been burned (9.1 ± 1 g m^{-2} yr^{-1}) relative to unburned sites (17.7 ± 0.6 g m^{-2} yr^{-1}) (Pitkänen *et al*. 1999). Tolonen *et al*. (1992) recorded a similar result in a burned boreal bog. Wildfire frequency and carbon accumulation were negatively correlated in two Canadian raised bogs (Kuhry, 1994). Fires may also have reduced carbon accumulation across a series of Canadian peatlands (Camill *et al.* 2009) though climatic controls may confound the association. Shiller *et al.* (2014) suggest burning on a Canadian raised bog may have increased CAR though do acknowledge charcoal may be influencing the BD values used to calculate CAR. Surface cores recovered from the Hard Hill experimental plots suggest burning may reduce blanket peatland RERCA. Garnett *et al.* (2000) used Spheroidal Carbonaceous Particle (SCP) analysis to compare PARs following the late-20th Century 'take-off' in SCP values. Their results suggest '10-year' rotational burning may have reduced carbon sequestration by 73 g C m^{-2} yr^{-1} relative to unburned conditions. The validity of these results is undermined by the use of substandard BD calculations, an arbitrary SCP date and constant carbon concentration. Billett *et al.* (2010) expanded on this data to produce mid-20th Century carbon accumulation values of 1) 'grazed' 115.4 g C m^{-2} yr^{-1}, 2) 'grazed and burned' 64.9 g C m^{-2} yr^{-1} and 3) 'unburned and ungrazed' 98.4 g C m^{-2} yr^{-1} for the Hard Hill plots. These results, albeit acknowledging uncertainty, support the notion that differing management regimes can influence carbon dynamics, and that rotational burning may reduce carbon accumulation.

2.3.5 The effect of burning on the chemical composition of plant tissues and peat

Burning may increase nutrient availability if ash is incorporated into the peat matrix (Davies *et al.* 2008) or if oxidation occurs because of burning (Ramchunder *et al.* 2009). P is often a limiting nutrient on plant growth in blanket bogs. *E. vaginatum* exhibits effective post-burn growth in part due to the increase in P availability following firing (Holden *et al.* 2007a). Deposited ash is vulnerable to erosion (Fisher, 2006; Kinako and Gimingham, 1980; Maltby *et al.* 1990), particularly if the burn produced a hydrophobic crust (Shaw *et al.* 1996). A Nova Scotian raised bog recorded increased N mineralization and reduced carbon accumulation following a fire (Charman *et al.* 2015). Nutrients may be lost in the smoke that rises during a fire (Allen, 1964; Evans and Allen, 1971). Weak evidence for N loss under burning is exhibited in a Peak District chronosequence (Rosenburgh *et al.* 2013) and no difference in peat N concentrations were recorded between burned and non-burned sites in Brown *et al.* (2014).

2.3.6 Section summary

Rotational burning does not appear to affect blanket bog functioning in a consistent manner, likely due to responses varying according to burning intensity and site-specific characteristics. Despite uncertainty caused by this variability, DEFRA (2007) and many researchers suggest discontinuing burning for the benefit of blanket bog integrity (Holden *et al.* 2007a; Shaw *et al.* 1996; Tucker, 2003) given firing has the mechanistic capacity to influence carbon accumulation

(Dawson and Smith, 2007; Gray and Levy, 2009). Quantifying the impacts of firing on carbon accumulation is though required to change land management policy. Quantitative understanding of such impacts is still limited (Orwin and Ostle, 2012; Tucker, 2003) and drawn primarily from the arguably unrepresentative site of Hard Hill (Glaves *et al* 2013). This thesis will attempt to improve quantitative understanding of the impact of firing on blanket peatland carbon accumulation (Section 2.8). Cores will be recovered from multiple ‘non-experimental’ sites to assess the variability of responses in carbon dynamics and vegetation change to firing. Dating methods will be used to attempt to detect whether peat has been lost during previous phases of firing and charcoal analysis will be used to identify changing patterns of fire use over time (i.e. archaeological, historic, modern).

2.4 Impacts of drainage on carbon accumulation

Blanket peatlands are a wetland habitat, hence efforts to alter hydrological conditions may be expected to alter peatland development. High peatland water tables are key in allowing the accumulation of organic matter due to the anoxic conditions they create. Concerns regarding the integrity of drained peatland carbon stores are therefore understandable (Rowson *et al*. 2010; Sakovets and Germanova, 1992). This section discusses the impacts of drainage on blanket peatlands and assesses the way these impacts may affect carbon accumulation.

2.4.1 Background to the use of drains in blanket peatland management

Drain/ditch/grip installation has affected >1.5 million ha of British blanket bog (Parry *et al*. 2014). Most drains were installed following the Hill Farming Act (AD 1946) in an effort to increase upland productivity under the post-World War II ‘Feed-Britain’ mentality (Holden *et al*. 2004; Reed *et al*. 2009). No evidence at the time suggested increased WTD would increase upland productivity (Armstrong *et al*. 2009; Stewart and Lance, 1983). Post-World War II ditches were typically dug 50cm deep and wide at intervals of 20m (Figure 2.15) and connected to an outlet stream in a herringbone form (Armstrong *et al*. 2009). Installation of ditches was encouraged by CAP subsidies, with subsidies finally halted in the AD 1980s (Ramchunder *et al*. 2009). The primary mode of UK blanket peat restoration is a desire to raise water levels through ditch blocking (Carroll *et al*. 2011). Site-specific factors may alter responses to drain blocking and the success of a restoration project (Armstrong *et al*. 2010; Worrall *et al*. 2007b).

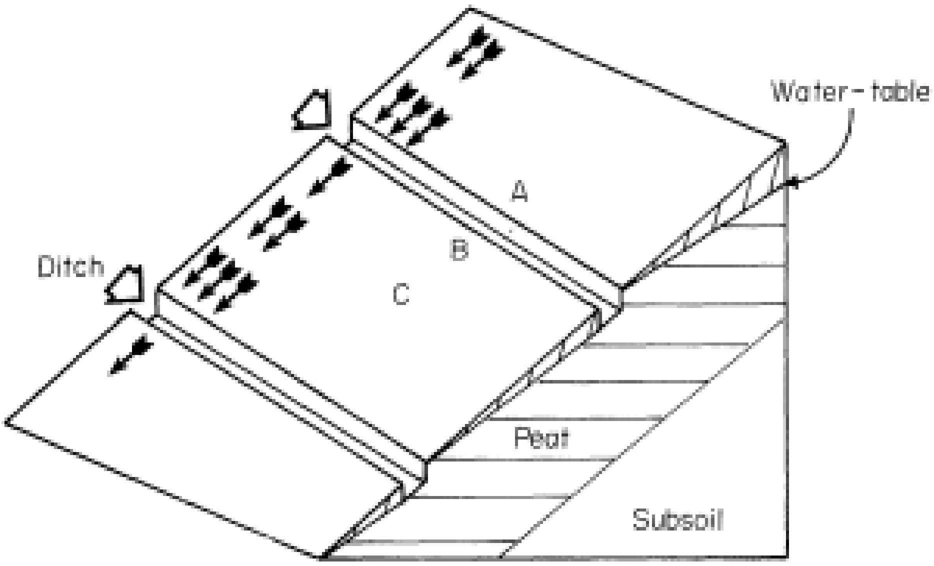


Figure 2.15: Schematic diagram detailing expected response to ditching. Arrows represent direction and extent of through peat water flow. Successful drainage exhibits increased WTD at (b) relative to (a) or (c) (Coulson *et al.* 1990).

2.4.2 The effect of drainage on plant community structure

A greater WTD makes it harder for *Sphagnum* to obtain moisture for photosynthesis through capillary action (Robroek *et al.* 2007b). Bellamy *et al.* (2012) recorded few *Sphagnum* within 0.5m of a drain in Forsinard but greater numbers 2.5m away. Drainage in a Polish peatland displaced *Sphagnum* from the palaeoecological record (Glina *et al.* 2017). Stivrins *et al.* (2017) suggest that drainage caused a change in *Sphagnum* species composition in cores recovered *ca.* 1km from drains. They only present macrofossil evidence in support so it is uncertain whether some other driver may have affected *Sphagnum* cover (e.g. pollution) so far from a drain. *Sphagnum* is often replaced by vascular plants (Kilian *et al.* 2000) and/or brown/feather mosses following drainage (Laine *et al.* 1995a; Straková *et al.* 2010). Aerated substrate favours *C. vulgaris* establishment (Ratcliffe, 1959; Stewart and Lance, 1991) although *Calluna* may be outcompeted by graminoids if intense grazing accompanies drainage (Coulson *et al.* 1990).

Hummock *Sphagnum* have a greater resistance to desiccation than hollow or lawn forms as their greater growth density encourages moisture retention and capillary efficiency (Bragazza, 2008; Gerdol, 1995; Hayward and Clymo, 1982; Weston *et al.* 2015). Cores from two Irish raised bogs subjected to experimental water table drawdowns showed no detrimental impact to capitulum biomass for hummock

Sphagnum whilst wetter affiliated *S. magellanicum* exhibited reduced capitula biomass (Robroek *et al*. 2007a). Drainage on a Finnish bog caused hummock affiliated *Sphagnum* to displace hollow *Sphagnum* over a 30-year period (Komulainen *et al*. 1999). Results from Mer Bleue further demonstrate the persistence of *Sphagnum* following drainage where an *S. capillifolium* lawn exhibited greater PARs than other areas of the bog following drainage (Talbot *et al*. 2010).

2.4.3 The effect of drainage on water table position

Drains are installed parallel to slope contours to dissect through peat water flow lines (Ramchunder *et al*. 2009). Greatest WTD increases occur on downslope edges as drains intercept water that would have percolated through this section of peat (Figure 2.15). The ‘effectiveness’ of drain installation may not always be recorded accurately. The loss of peat interstitial water following drainage causes unsupported peat to collapse (Holden *et al*. 2017; Lindsay, 2010; Williamson *et al*. 2017). The collapsing peat surface therefore effectively ‘chases’ the water table vertically down (Figure 2.16). One cannot truly consider the impacts of drainage to have been minimal without measuring WTD changes according to the pre-disturbance mire height (Lindsay, 2010).

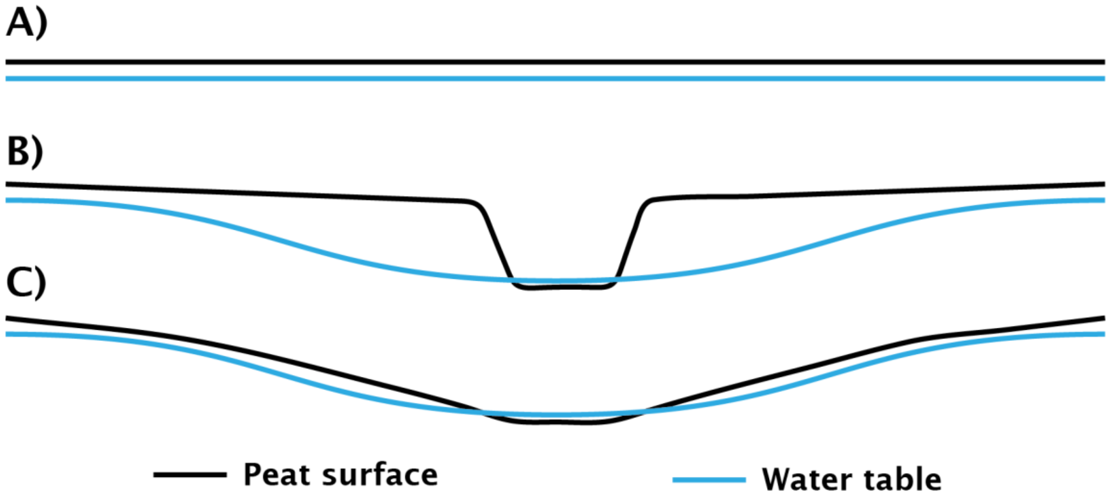


Figure 2.16: Schematic diagram demonstrating the ‘effectiveness’ of ditching. A) indicates pre-drainage conditions, B) indicates ditch installation whilst C) indicates sometime after drainage where peat surrounding the drain has collapsed (adapted from Williamson *et al*. (2017)).

Successful drain installations (i.e. those which increase WTD) have been recorded (Coulson *et al*. 1990; Wilson *et al*. 2010). Peatland ecohydrological feedbacks may though limit the effectiveness of drain installation in causing water table drawdowns. The hydraulic conductivity of peat declines as the substrate dries and decomposes. Any increases in decomposition due to initial water table drawdowns may therefore limit both atmospheric (Morris *et al*. 2015a; Waddington *et al*. 2015) and lateral moisture losses (Young *et al*. 2017). The high rainfall and cool temperatures that favour blanket bog initiation may also limit drainage effectiveness. Grips at Moor House (>450m asl) did not significantly affect WTD, but 10cm water table drawdowns were recorded downslope of drains at Waskerley bog (<450m asl) (Coulson *et al*. 1990). A survey of Norwegian bogs showed a 30m ditch-interval was required to promote a 30cm water table drawdown where June-September rainfall was 20mm, but a 7.5m ditch spacing was required for the same WTD increase if rainfall reached 700mm (Braekke, 1983). Antecedent hydrological conditions may also regulate ditching effectiveness (Armstrong *et al*. 2009) as ditch installation in 'wetter' sites did not significantly affect WTD at Lake Vyrnwy (Carroll *et al*. 2011) or the North Pennines (Stewart and Lance, 1991). These collected findings must be considered against the 'pre-disturbance mire height' critique of WTD investigations presented above.

Aerobic decomposition occurs faster than anaerobic decomposition. Drainage is therefore expected to increase loss of carbon as CO_2 (Moore and Knowles, 1989; Scanlon and Moore, 2000) and so reduce the integrity of the carbon store (Janssens *et al*. 2005; Nieveen *et al*. 2005; Pitkänen *et al*. 2012). Decomposition rates following drainage at Waskerley were 70% greater than at the Moor House site where drainage was less effective (Coulson *et al*. 1990). A large study of Finnish peatlands found increased decomposition and reduced PARs at drained sites (Pitkänen *et al*. 2012). Freeman et al. (2001a) attributed a fourfold increase in CO_2 emissions following aeration to increased phenol oxidase activity. Mäkilä (2011) found drainage reduced carbon accumulation by an average of 10.8 g C m^{-2} yr^{-1} across a variety of peatland types due to increased decomposition. A transect performed from an undrained to drained area of a boreal peatland deemed ditching to have reduced carbon accumulation by 35 g C m^{-2} yr^{-1} (Laine and Minkkinen, 1996). Aeration increases methane oxidation, with an estimated reduction in efflux from 60 to 40 kg CH_4 ha^{-1} yr^{-1} following drainage recorded on the Migneint (Cooper *et al*. 2014).

2.4.4 The effect of drainage on peat matrix characteristics

Loss of interstitial water as drainage occurs causes the bog surface to subside and peat to compact. A variety of drained peatlands show increased near-surface peat BD relative to undrained areas (Silins and Rothwell, 1998; Sundström *et al.* 2000; Wells and Williams, 1996). Desiccation causes macropores and cracks to form in the peat matrix. These can widen into soil pipes which create localized increases in decomposition and act as subterranean conduits for DOC and POC loss (Holden, 2005a). Ground penetrating radar surveys have shown that catchments that have undergone artificial drainage have significantly greater soil pipe density (Holden, 2005b). Holden (2006) recommends blocking older drains first for carbon conscious management as greater POC losses occurred in older drains. Drain blocking may restore substrate characteristics (e.g. BD) to some degree (Haapalehto *et al.* 2010; Wallage, 2008). Ditch installation exposes sections of the catotelm to the atmosphere. Drains continually erode downward after formation (Holden *et al.* 2004), though topography regulates self-scouring behaviour to slopes >4° (Holden *et al.* 2007b). Sediment export (POC losses) is often greater from drained than intact/restored catchments (Holden *et al.* 2007b; Ramchunder *et al.* 2012). 18.7% of Oughtershaw Moss catchment sediment originated from ditches draining only 7.3% of the blanket peatland (Holden *et al.* 2007a). Grips also act as gaseous flux hotspots. 95.8% of CH_4 and 21.6% of CO_2 emissions originated from drains at Moor House despite ditches covering only 9.3% of the area (McNamara *et al.* 2008).

The quality of newly exposed peat will affect the response of the carbon cycle to drainage (Straková *et al.* 2010, 2011). The catotelm has little fresh carbon, meaning the recalcitrant substrate does not provide enough energy to sustain microbial activity even if the peat is exposed to aerated decomposition (Lindsay, 2010). Chimner and Cooper (2003) demonstrated this where the majority of increased CO_2 emissions from a fen occurred following initial water table lowering (0-5cm below the surface), with continual WTD increases (6-10 and 11-40cm) only causing minor emission increases. Addition of labile organic matter (i.e. 'fresh' carbon) (*sensu* Fontaine *et al.* 2007) alongside drainage could increase decomposition activity in the catotelm (Fenner and Freeman, 2011; Gogo *et al.* 2011). Walker *et al* (2016) recently presented evidence that vascular plant exudates may 'prime' microbial communities in blanket peat. It seems likely therefore that drained peatlands may be vulnerable to carbon losses primarily if they are dominated by vascular plants, semi-regardless of the recalcitrance of the peat itself.

The relationship between drainage and DOC is unclear. The notion that drainage increases DOC export or peat pore water concentration is supported (Armstrong *et al.* 2010; Wallage, 2008) and refuted (Grayson *et al.* 2012; Moore, 1987). DOC response to drain blocking may be scale dependent and vary between the drain, outlet stream and the catchment outflow (Turner *et al.* 2013; Wilson *et al.* 2011a). Evans *et al.* (2014b) found drainage increased the age of carbon in DOC for a range of peatland types. The authors found that drained blanket peatlands still exhibited largely <50-year DOC source ages. A Peak District blanket bog was a notable exception where the intensive management history may be incorporating catotelm carbon (>2,000 years old) as DOC (Evans *et al.* 2014b). A reduction in DOC concentration following restoration may represent its increasing use as a methanogenic substrate as WTD declines (e.g. Fenner *et al.* 2005a), as opposed to reduced peat decomposition (i.e. the target restoration seeks to achieve). Few DOC focussed studies acknowledge the potential for this, instead focussing solely on DOC flux and negating other pathways of peatland carbon loss.

2.4.5 The effect of drainage on the chemical composition of plant tissues and peat

Successful drainage increases nutrient availability through peat oxidation (Cannell *et al.* 1993; Dias *et al.* 2010), particularly carbon-bound N and sulphur (Holden *et al.* 2004). Peat N, P, K, calcium and iron concentrations all significantly increased following drainage in a Newfoundland bog, primarily because of an increase in BD (Wells and Williams, 1996). Williams and Wheatley (1988) found mean peat N content increased 1.5x times following a 50cm water table drawdown. Nutrient availability may not necessarily increase following drainage given mineralization depends on substrate quality in addition to aeration (Laiho *et al.* 1999). Variability in nutrient cycling response is epitomized by Williams (1974) where cores recovered from an Irish blanket bog showed reduced N mineralization when WTD increased to 18cm, but a WTD increase of 34cm increased N mineralization. Nutrients may also be lost via flushing through ditch networks in solute form (Sallantaus, 1992).

2.4.6 Section summary

Drainage seems likely to reduce blanket peatland carbon accumulation though site-specific conditions (e.g. antecedent saturation, substrate quality) will affect water table response to drainage (Armstrong *et al.* 2009) and restoration (Armstrong *et al.* 2010; Price *et al.* 2003). Laiho (2006) questions whether a consistent carbon cycling response to water table drawdown can ever be identified given inter-/within-site variability. More work is clearly therefore needed to examine site-specific factors which determine the response of peatlands to drainage. This thesis will attempt to improve quantitative understanding of the impact of drainage on blanket peatland carbon accumulation (Section 2.8). Cores will be recovered from multiple blanket peatland sites to assess the variability in responses in carbon dynamics to drainage. Plant macrofossil analysis will also be used to determine the response of peatland vegetation to drainage.

2.5 Impacts of forestry on carbon accumulation

Forming a forest where there previously was none substantially alters a blanket peatland. A consistent impact of afforestation on peatland carbon dynamics is yet to be established (Maljanen *et al*. 2010; Vanguelova *et al*. 2016). The debate regarding impacts centres on whether carbon assimilated by growing trees can offset losses through peat decomposition (Figure 2.17). Results suggest forestry may provide a net increase (Cannell *et al*. 1993; Minkkinen *et al*. 2002; Worrall *et al*. 2010) or net decrease (Armentano and Menges, 1986; Morison, 2012; Silvola, 1986) to carbon storage. Most work originates from Scandinavian mires (Silvola *et al*. 1996) and hence has a limited applicability to the blanket peatland management debate. This section discusses the impacts of forestry on blanket peatlands and assesses the way these impacts may affect carbon accumulation. The term ‘forestry’ is hereafter used to represent the activity of silviculture, namely the growing and cultivation of trees (O’Hara, 2015).

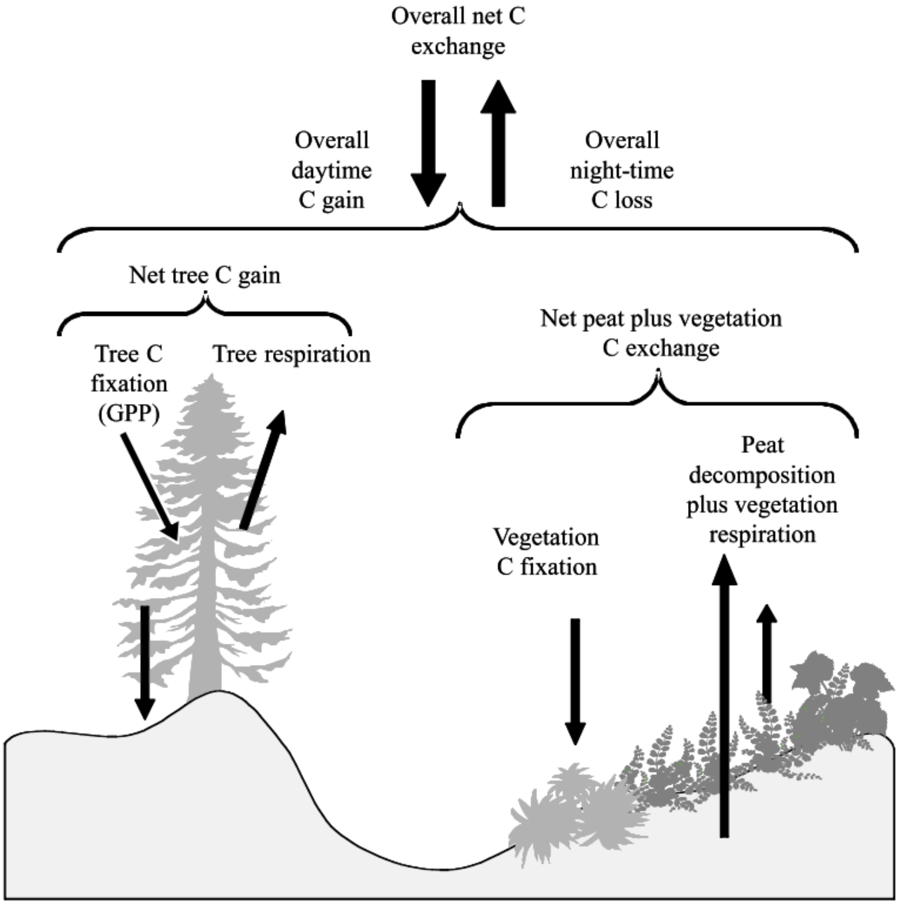


Figure 2.17: Schematic diagram representing main components of the carbon cycle in afforested peatlands (Hargreaves *et al*. 2003).

2.5.1 Background to the use of forestry in blanket peatland management

Conifer plantations were created on large areas of UK blanket bog from AD 1950-80 as part of the expansion of the timber industry (Wellock *et al*. 2011). Drainage must occur prior to planting (Brocklehurst, 2005) as conifer seedlings do not germinate properly on saturated ground (McVean, 1963; Moir *et al*. 2010). Plantations are rotated on a 40-100 year cycle where felling of dead trees allows planting of new seedlings (Worrall *et al.* 2011b). The Forestry Commission has discouraged new plantations since the AD 1990s owing to worries of the adverse impacts on peatland functioning (Reed *et al*. 2009) and new plantations are no longer permitted on peat ≥50cm deep (Worrall et al. 2011b). Forestry is different to the ‘natural colonisation’ of bogs by trees following a water table drawdown caused by artificial drainage or climate (Edvardsson *et al*. 2015a, b; Heijmans *et al*. 2013).

2.5.2 The effect of forestry on plant community structure

Tree installation disturbs the pre-existing vegetation cover of a peatland and the subsequent tree growth alters light availability. The shaded environment, alongside tree roots, encourage water table drawdowns and nutrient cycling and present harsh conditions for *Sphagnum* growth. 97% of *Pinus sylvestris* trees on a boreal bog grew from areas once dominated by *Sphagnum rubellum* (Ohlson *et al*. 2001). The plantation canopy favours shade affiliated mosses (e.g. *Dicranium*, *Hylocomium* and *Pleurozium* sp.) (Hájek *et al*. 2009; Lachance and Lavoie, 2004; Leonard *et al*. 2017; Pellerin and Lavoie, 2003; Turetsky *et al*. 2010; Waddington *et al*. 2015). Such mosses do not accumulate carbon as efficiently as *Sphagnum* (Kangas *et al*. 2014). *Sphagnum* can persist following afforestation as *S. papillosum* occurs on Scottish plantation floors (Hargreaves *et al*. 2003) and *Sphagnum* in boreal forests can outcompete feather mosses if tree canopy coverage is not complete (Fenton *et al*. 2007).

Installation of drains prior to planting promotes peat decomposition (Ohlson *et al*. 2001) and tree litterfall and root exudation further stimulates aerobic microbial activity (Domisch *et al*. 1998; Minkkinen *et al*. 1999). The response of peatland carbon cycling to afforestation depends on the tree species (Maljanen *et al*. 2010), yield class (m^3 wood ha^{-1} yr^{-1}), plantation spacing and fell age (Morison *et al*. 2010). *Pinus sitchensis* is the main species on afforested peat in Wales and is typically grown at yield class 10 on peat >50cm deep (Vanguelova *et al*. 2012). Carbon sequestration in trees declines and then halts as trees reach maturity, meaning without stock rotation all afforested sites eventually become a net carbon source (Morison, 2012; Natural England, 2010). Standard plantation management is to fell soon after tree annual productivity declines, typically 40-70 years according to species and local conditions (Vanguelova *et al*. 2012). Management for yield class 10 *P. sitchensis* on deep peat is for 1.7m spacing, which yields 100 t C ha^{-1} when felled at 62 years of age (Figure 2.18).

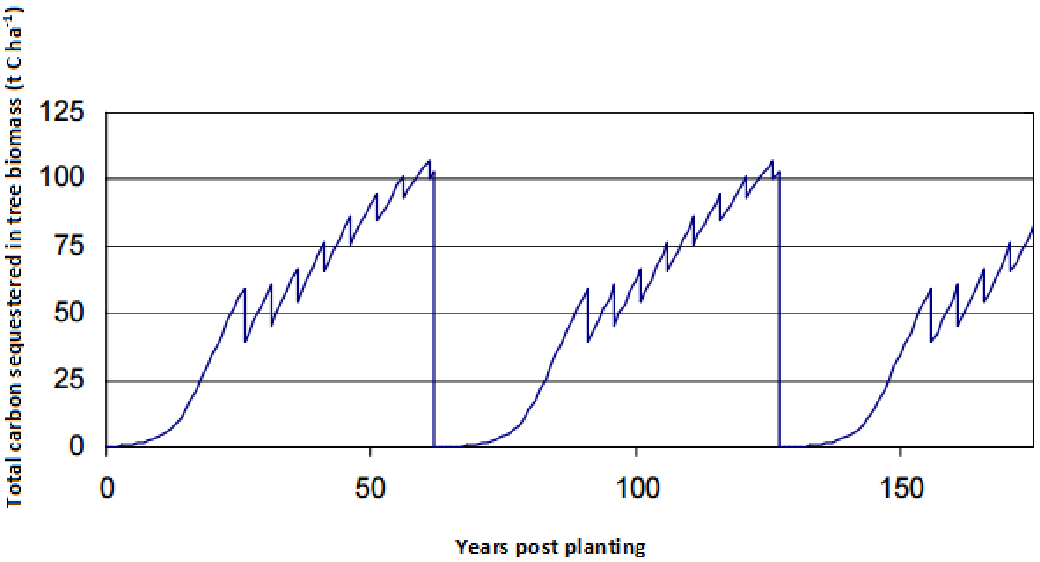


Figure 2.18: Modelled carbon accumulation in yield class 10 *P. sitchensis* plantation over plantation cycles (62-year felling regime). There is assumed to be no impact on the peat which may affect productivity of between rotations (adapted from Vanguelova *et al.* (2012)).

Whether carbon sequestered in trees can offset carbon lost through peat decomposition determines the net impact of afforestation on a site's carbon budget. Afforestation in parts of the Piitsonsuo mire caused an average change from a net sink (25 g C m^{-2} yr^{-1}) to a net source (250 g C m^{-2} yr^{-1}) (Silvola, 1986). Simola *et al.* (2012) estimated afforestation had caused a *ca.* 150 g C m^{-2} yr^{-1} loss across a range of Finnish peatland types. Their results were based on comparisons of carbon inventories conducted during coring on the same peatlands in the AD 1980s. Simola *et al.*'s (2012) methodology was unorthodox as they used BD and ash % profiles to 'match' peat cores to previous survey locations to assess change over time. Despite associated recovery error, a coarse BD sampling resolution (10cm homogenized depth increments) and the use of estimated BD values for the AD 1980s survey, results do provide a peat core derived estimate of the negative effect of afforestation on carbon accumulation. Pitkänen *et al.* (2013) subsequently suggested afforestation promoted carbon losses of 131 ± 28 g C m^{-2} yr^{-1} when cores from an intact section of a Finnish bog were compared to those recovered within a plantation.

Others suggest the alternate, in that afforestation increases net carbon storage. Minkkinen *et al.* (2002) deemed the carbon sequestered in planted trees to have reduced net Finnish peatland radiative forcing potential (-3m W m^{-2}). Their study does not account for changes in surface vegetation following afforestation, despite their calculations assuming carbon sequestration in the peat would increase post-afforestation. Krüger *et al.* (2016) found that inventory approaches (i.e. using peat cores) for determining whether forestry constituted a net carbon gain or loss in the peat may depend on the method used to calculate the change. They recommended calculating stock above a temporal horizon as the most applicable and robust approach. This is the approach used in this thesis.

Hargreaves *et al.* (2003) measured NEE at intact, drained and mature plantation blanket peatlands. They determined that whilst drainage would cause a 25 t C km^{-2} yr^{-1} sink to become a 200-400 t C km^{-2} yr^{-1} source, afforestation would result in a return to a 300 t C km^{-2} yr^{-1} sink. Hargreaves *et al.* therefore deemed forestry to offer a net benefit to blanket peatland carbon storage. Lindsay (2010) criticises their site selection in that Auchencorth Moss ('intact' site) had been extensively drained and mined for peat extraction (Dinsmore *et al.* 2010) and the trees where CO_2 measurements occurred at Mindork Moss ('mature plantation') were 26-years old, not yet halfway to their 65-year felling age. These uncertainties mean that intact blanket peatland carbon accumulation was underestimated by *ca.* 0.15-0.45 t C ha^{-1} yr^{-1} and 'mature' forestry carbon accumulation was overestimated by an uncertain amount. The oxidative effect of the trees was also underestimated relative to a 'mature' forest as Lindsay (2010) showed that continuing the model past its 26-year culmination induces an oxidative loss rate *ca.* 1-3 t C ha^{-1} yr^{-1} greater than Hargreaves *et al.* (2003) accounted for. More detailed *in situ* studies of how afforestation affects peatland carbon dynamics (i.e. palaeoenvironmental techniques employed in this thesis) are therefore required for future testing against modelled impacts of carbon loss.

2.5.3 The effect of forestry on water table position

Drainage in preparation for planting promotes microbial activity in the aerated substrate (Wellock *et al.* 2011) thereby increasing CO_2 efflux and reducing CH_4 efflux (Cannell *et al.* 1993; Hargreaves *et al.* 2003). No significant differences in CO_2 emissions were recorded in the Cloosh Forest between intact and afforested sites as the recalcitrance of deeper peat confined increased microbial activity to peat <10cm deep (Byrne and Farrell, 2005). Other researchers have also found afforestation may only promote decomposition in the upper 10-15cm peat (Brake *et al.* 1999; Maljanen *et al.* 2010; Minkkinen *et al.* 1999). Root uptake and rainfall interception promote further water table drawdowns as the trees grows

(Edvardsson *et al.* 2015a; Heijmans *et al.* 2013; McHaffie *et al.* 2002). WTD can therefore be greater under plantations relative to surrounding blanket peat even if both have been artificially drained in a similar manner (i.e. drain depth and width). Plantations can alter the hydrological conditions of the surrounding peatland. Peat water content and BD sampled along a transect at Bad á Cheo from blanket bog to a plantation were significantly correlated with proximity to the forest (Pyatt *et al.* 1992; Shotbolt *et al.* 1998). A regression model suggested de-watering and subsidence of neighbouring substrate might be limited to ≤30m but observations of subsidence >40m from the plantation suggest the effect may extend further (Lindsay, 2010).

There is mixed evidence regarding the impact of restoration (‘clearfelling’) on WTD. Reduced evapotranspiration through reduced root uptake and rainfall interception would be expected to cause water tables to rise following clearfelling. Results from Caithness (Anderson, 1999), Finland (Päivänen, 1974), the Forth Valley (Anderson, 2010), Norway (Päivänen, 1980) and Ontario (Berry and Jeglum, 1988) support this notion. Seven out of eight peatlands surveyed in the St. Lawrence Lowlands exhibited a reduced WTD following clearfelling. The depth to water table on a single bog did increase which Dubé *et al.* (1995) attributed to increased evaporation from the exposed peat surface. Clearfelling at Horse Hill Moss failed to reduce WTD, attributed to the forestry having altered the hydraulic conditions of the peatland to the degree that restoration may not be possible (Clothier, 1995 (in Anderson (2001))). If restoration is successful, then a decline in WTD should reduce heterotrophic respiration of peat. A 30% reduction in CO_2 efflux upon clearfelling recorded by Byrne and Farrell (2005) was though attributable primarily to lost living root respiration rather than a decline in heterotrophic activity. Post-restoration CO_2 efflux may increase given the abundance of roots, stumps and logging residue which undergo decomposition following clearfelling (Mäkiranta *et al.* 2010).

2.5.4 The effect of forestry on peat matrix characteristics

Drain installation in preparation for forestry causes the peat to subside (Section 2.4.4) and the weight of growing trees causes additional subsidence (Anderson, 2001; Minkkinen *et al.* 1999). Drains in a Norwegian bog ‘lost’ 75% of their depth after 34-years as the peat subsided under the weight of trees (Braekke, 1983). Afforestation at Bad á Cheo caused 60-80cm peat subsidence relative to a pre-established datum and increased BD relative to adjoining peat (Shotbolt *et al.*

1998). Oxidative losses are likely given CO_2 efflux was 90% greater from within the forestry relative to surrounding bog (Chapman and Thurlow, 1992), though the study does not distinguish between autotrophic or heterotrophic respiration. Peat BD and carbon concentration was significantly higher in forested Finnish fens relative to intact sites (Minkkinen and Laine, 1998a). Minkkinen and Laine (1998b) subsequently compared these data with government records of peat depth recovered prior to drainage and concluded that afforestation had increased the carbon store of the peat by 5.9 ± 14.4 kg m^{-2}. Their conclusion is questionable as the increase in carbon concentration under forestry likely occurred due to carbon being stored in a lower substrate volume (due to subsidence) rather than an increase in carbon sequestration. Cannell and Dewar (1995) suggest that root and litter production may increase the peat carbon stock relative to unplanted conditions, though they used modelled peat carbon losses (Cannell *et al.* 1993) which failed to account for DOC and POC fluxes during stock rotations. Antecedent conditions likely affect the degree to which peat subsides following afforestation as areas of Norwegian bogs where pre-forestry BD was high showed little evidence of post-forestry subsidence (Braekke, 1983).

2.5.5 The effect of forestry on the chemical composition of plant tissues and peat

N and P limitation inhibit tree growth on intact bogs. Foresters aim to stimulate nutrient availability through fertilizer application and aeration (Parry *et al.* 2014). The application of fertilizer inhibits *Sphagnum* growth given the genus thrives in low nutrient environments (Holden *et al.* 2011). Continued water table drawdowns through evapotranspiration promote nutrient mineralisation (Edvardsson *et al.* 2015a; Heijmans *et al.* 2013). Litterfall and root exudation provide further nutrient inputs to surface peat layers as the forest develops (Braekke, 1987; Laiho *et al.* 1999; Sundström *et al.* 2000) thereby promoting bryophyte coverage of the forest floor (Minkkinen and Laine, 1998b). Miller *et al.* (1996) attributed a reduction in peat N concentration at an afforested blanket peatland in Caithness to nutrients lost by flushing in drainage ditches.

2.5.6 Section summary

There is no consensus as to whether afforestation provides a net benefit or loss to the peatland carbon sink. The need to drain a peatland in preparation for plantation disturbs a system that would likely continue accumulating carbon if left in an intact state. Any carbon accumulation 'gains' for planting must therefore first offset losses associated with interrupted carbon accumulation. Whilst modelling of the carbon balance of the above-ground component is robust, knowledge of below-ground changes is limited with much uncertainty associated with oxidative loss rates and changes in peat carbon storage. This thesis will contribute to an improved understanding of the impact of forestry on blanket peatland carbon accumulation (Section 2.8). Multiple cores will be recovered from within an artificially forested blanket peatland to assess the variability in impacts on carbon accumulation. These 'below ground' values will be compared against estimates of 'above ground' carbon accumulation to determine whether forestry has had a net gain or loss in terms of carbon accumulation at the site.

2.6 Impacts of grazing on carbon accumulation

Overgrazing, "*the result of environmentally unsustainable livestock practices brought about through overstocking*" (Brown *et al*. 2001, page 56), has a variety of negative impacts on upland habitats including erosion and biodiversity loss (Martin *et al*. 2013). The impact of grazing on peatland carbon dynamics is though largely uncertain. This section discusses the impacts of grazing on blanket peatlands and assesses the way these impacts may affect carbon accumulation. This thesis does not distinguish between 'grazing' (consumption of forbs and grasses) and 'browsing' (consumption of trees and shrubs) herbivory (Tanentzap and Coomes, 2012). As such, 'grazing' is hereafter used to refer to general herbivory on peatland flora. The approaches outlined here consider the impacts of herbivores on the peatland. A holistic approach to carbon conscious management should consider GHG fluxes associated with the grazers themselves (Worrall and Clay, 2012).

2.6.1 Background to the use of grazing in blanket peatland management

Upland grazing has long been a component of the Welsh economy (Winchester and Straughton, 2009). Sheep became the most abundant Welsh livestock *ca*. AD 1500-1700 following a medieval dependence on cattle farming (Hughes, 1973). The progressive enclosure of Welsh uplands, adoption of sheep breeds able to persist on the hills through winter and increased stocking levels all intensified the impact of grazing *ca*. AD 1760-1830 (Hughes *et al*. 1973). Subsidies offered for sheep on a 'per-headage' basis under the Hill Farming Act continued to increase stocking post-World War II (Worrall *et al*. 2011b). CAP payments promoted sheep numbers during the AD 1950s-90s (Smith *et al*. 2003b; Thompson *et al*. 1995) which resulted in a 71% rise in sheep numbers in Wales from AD 1974-1998 (ECOSSE, 2007). These institutional policies failed to appreciate the ecological sensitivity of the uplands. An assortment of CAP initiatives sought to reduce upland stocking levels from their mid-AD 1990s peak (Worrall *et al*. 2011b). Agri-environment schemes often require a grazing regime of ≤1.5 ewes ha^{-1} to qualify for payments (Smith *et al*. 2003b). Total sheep and lamb numbers in Wales fell from 11,505,214 in AD 1998 to 8,237,737 in AD 2009, though had increased to 9,503,977 by AD 2015 (Welsh Government, 2015).

The EU CAP classes the majority of Welsh agricultural land as 'Less Favoured Areas' (LFA) (Figure 2.19). LFA are "*areas where farming is handicapped by geography, topography or climate*" (European Commission, 2015, page 19). Reductions in the size of the upland agricultural workforce followed the late-AD 1990s decline in upland sheep populations (Reed *et al*. 2009). Most British upland farms would be economically unviable without CAP subsidies. The level of investment is "*disproportionately large*" (Institute for European Environmental Policy, 2009, page 3) considering the number of jobs and produce provided. Upland farming is strategically important given rural communities often rely on the sector for employment (Reed *et al*. 2009; 2014). The agricultural community view farming as lying "*at the heart of the uplands of England and Wales*" (NFU, 2015b, page 3) as hill farmers self-identify as suppliers of produce and stewards of the environment (Burgess *et al*. 2000; Burton *et al*. 2008).

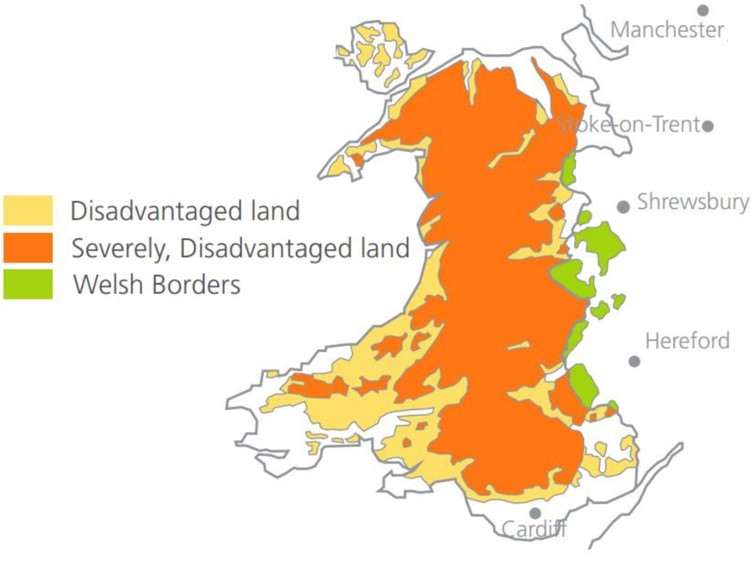


Figure 2.19: Map detailing the extent of 'Less Favoured Areas' ('disadvantaged' and 'severely disadvantaged') agricultural land in Wales (NFU, 2015b).

Grazing pressure is a function of the number of grazers and their movement through the environment (Alday *et al*. 2015). This pressure is measured as a 'stocking level' which is defined as the number of animals per unit area. Land managers use the metric to set grazing regimes (e.g. ≤1 ewes ha^{-1}) as overstocking can lead to overgrazing (Pakeman *et al*. 2003; Thompson *et al*. 1995). The metric oversimplifies livestock-landscape interactions and may detrimentally affect the environment (Critchley *et al*. 2008). Grazers exert a heterogeneous pressure across an area as they are attracted first to flora that they favour (i.e. palatable and nutritious plants) and then consume less favourable flora near to these favoured sources (Grant *et al*. 1976; Shaw *et al*. 1996). Martin *et al*. (2013) suggest plot-scale studies that attempt to investigate the impact of grazing (e.g. Hard Hill (Section 2.3.1)) cannot account for such complex grazer-vegetation interactions. The impact of a given stocking density thus varies according to antecedent vegetation community structure (Holden *et al*. 2007a; Hulme *et al*. 2002; Ross *et al*. 2016). The desire to set a nationwide standard 'livestock per unit area' measure that can avoid habitat degradation may therefore be a misguided approach to British blanket bog management and lead to unsustainable grazing regimes. The notion of 'standardization' is also the antithesis of the adaptive approach of farmers to upland grazing management (Wynne, 1992). Sheep stocking levels which supposedly avoid blanket bog degradation range from ≤1 ewes ha^{-1} (Grant *et al*. 1985; Hulme and Birnie, 1997; Welch and Rawes, 1966) to ≤0.5 ewes ha^{-1} (DEFRA, 2008; Rawes and Williams, 1973; Shaw *et al*. 1996) according to differing sources.

Difficulties quantifying herbivore presence/abundance and impact are compounded when using palaeoenvironmental techniques. Late- and post-Medieval stocking records primarily aggregate livestock numbers at diocese or parish level (Hughes *et al*. 1973; The National Archives, 1988). Where livestock data are refined to farm-level, single values relate to total farmed area (The National Archives, 1943). Upland agri-environment schemes are administered as a stocking regime within 'polygons' which are multiple hectares in size and include variable habitat types. The stocking regime of a polygon may not apply to a blanket bog if livestock actively avoid that habitat because of the vegetation cover and/or WTD, behaviour demonstrated by *in situ* research (Grant and Armstrong, 1993; Holland *et al*. 2010; Scottish Natural Heritage, 1996). Differing agricultural records can also produce contradictory impressions of stocking intensity at a given site. AgCensus data suggests large swathes of the Migneint blanket bog were managed according to a stocking regime of 6-7 ewes ha^{-1} (Bonn *et al*. 2010), far above recommended levels for this habitat (DEFRA, 2008). AgCensus data is calculated as a ward level aggregated value and does not acknowledge variations for individual land areas. Cells apparently subject to a 6-7 ewe ha^{-1} stocking regime (AgCensus) were instead managed at ≤0.5 ewes ha^{-1} when assessed at a holding level using agri-environment data (Bonn *et al*. 2010), thereby demonstrating discrepancies which can exist between differing land use records.

Anecdotal evidence may provide insights into previous animal behaviour. Engaging with land managers in this way and using 'on the ground' experience may improve upland land use decision making (Reed, 2008; Reed *et al*. 2006). Anecdotal evidence is though subject to the bias and fallibility of human recollection (Engelhardt, 1999). Vested interests may also affect the reporting of information (Greenhalgh *et al*. 2014; Schacter *et al*. 1993). Anecdotal evidence is therefore not preferable in devising land management approaches (Dicks *et al*. 2014; POST, 2011; Sutherland *et al*. 2004). Fungal spore evidence (e.g. *Sporormiella*) has been used to infer changes in herbivore dynamics (Gill *et al*. 2013; Mighall *et al*. 2013), though no reliable quantitative link between spores and herbivore numbers has been presented to date (AD 2017) which could be used to reconstruct grazing pressure in this thesis.

There are therefore no precise means for palaeoenvironmental investigations to reconstruct where animals congregated/moved through the landscape using documentary or anecdotal evidence, despite such behavioural patterns being important in determining herbivore impacts (Hester and Baillie, 1998; Oom *et al*. 2008; Ross *et al*. 2016). These concerns do not mean the effects of grazing may not be identifiable in palaeoenvironmental profiles, but attempts to link peatland development to quantitative stocking regimes may produce erroneous conclusions. Spurious attempts to attach causality to a change in peatland integrity may hinder attempts to determine sustainable grazing regimes (*sensu* Davies *et al*. 2016a).

2.6.2 The effect of grazing on plant community structure

Livestock select plants for consumption according to nutritional value, digestibility and spatial distribution (Hulme *et al*. 2002; Fraser *et al*. 2009; Thomson and Simpson, 2006) at a given time of year (Williams *et al*. 2010, 2012). Herbivores preferentially graze graminoids with high nutritional value (*M. caerulea*, *Scirpus cespitosus*) from late spring through summer (Fraser *et al*. 2009; Milligan *et al*. 2004). When these forms senesce in late autumn herbivory switches to less nutritious vegetation (*C. vulgaris*, *E. vaginatum*, *Erica tetralix*) (Grant *et al*. 1976). Species of livestock vary in their body size and morphology, their digestive tract characteristics and the way they consume vegetation (Fraser *et al*. 2009). Cattle graze a variety of vegetation (e.g. *E. vaginatum*, *Juncus effusus*, *M. caerulea* and *Nardus stricta*) as their requirement for large quantities of food and their consumption method (the tongue grabs and pulls vegetation) does not allow for highly selective grazing (Martin *et al*. 2013). Sheep are more selective grazers than cattle, favouring *C. vulgaris* and 'palatable grasses' as their smaller mouths and narrow jaws allow discrimination in plant choice (Holland *et al*. 2010). Within-species variation in feeding strategy also occurs. The replacement of wethers with more selective grazing ewes due to consumer preference for lamb encouraged the spread of *N. stricta* during the 20th Century (Ball *et al*. 1981; Roberts, 1959).

Many upland studies document the expansion of graminoids and the decline of *Calluna* because of overgrazing (Anderson and Yalden, 1981; Britton *et al*. 2005; Cooper *et al*. 1997; Hope *et al*. 1998; Medina-Roldán *et al*. 2012; Palmer *et al*. 2004; Rawes and Williams, 1973; Sansom, 1999; Walker and Elias, 1989). A complete canopy of *C. vulgaris* can suppress graminoid growth (Alonso *et al*. 2001). The disruption of the canopy by grazing leads to *Calluna* being outcompeted for nutrients and light by faster growing graminoids (Hartley and Amos, 1999; Hartley and Mitchell, 2005; Oom *et al*. 2008). Proponents of hill farming refute the notion that grazing detrimentally affects heather and instead advocate that 'undergrazing' permits the expansion of bracken (*Pteridium*) at the expense of *Calluna* (NFU, 2015b). Evidence supporting the concept of undergrazing is often anecdotal and originates from the agricultural community (e.g. Joyce, 2012; Silcock *et al*. 2012). It is unlikely that adverse impacts to blanket peatlands would arise because of too little grazing (Holland *et al*. 2010) and the AD 2009 Countryside Survey found no evidence suggesting upland stocking reductions allowed the expansion of bracken (Martin *et al*. 2013).

DEFRA (2008) recommend blanket peatland stocking levels of ≤0.5 ewes ha^{-1} to avoid overgrazing. *C. vulgaris* cover declined following an increase in stocking from ≤0.5 to >0.55 ewes ha^{-1} at the Hill Farming Research Organisation's (HFRO) blanket bog test site in Argyll (Grant *et al*. 1976). An increase from <0.4 to 3.4 ewes ha^{-1} promoted *E. vaginatum* and bare peat alongside a decline in *C. vulgaris* and *Sphagnum papillosum* in the Moor House NNR (Rawes and Welch, 1969). Stocking level declines on the Kinder Estate from 2.5 to <0.5 ewes ha^{-1} permitted a decline in graminoids and a recovery of *C. vulgaris* (Anderson and Radford, 1994), though a lack of controls mean this may be in part attributable to changes in the wider environment. Replacing sheep with cattle has been suggested as a method to promote a widespread return to mixed *C. vulgaris/Sphagnum* blanket bog vegetation communities (Anderson *et al*. 2006; Grant *et al*. 1985, 1987) given the less selective nature of cattle grazing.

Sheep stocking and graminoid cover are not always positively related. An experiment on the HFRO blanket bog test site recorded declines of *E. vaginatum* alongside *C. vulgaris* as stocking rate increased (Grant *et al*. 1985). A 15-year cessation of grazing at Butterburn flow did not significantly alter vegetation communities (Smith *et al*. 2003b) though initial stocking (0.65 ewes ha^{-1}) may not have been a sufficient level to 'degrade' the antecedent vegetation. The Hard Hill experimental plots have been stocked at <0.3 ewes ha^{-1} for their duration and this grazing has not substantially altered vegetation assemblages (Lee *et al*. 2013) or affected *C. vulgaris* height or biomass (Alday *et al*. 2015). Rawes and Hobbs (1979) likewise consider stocking of <0.5 ewes ha^{1} to allow a stable vegetation community. ≤0.5 ewes ha^{-1} may in cases degrade vegetation, as at Moor House where exclusion of even such a low stocking level promoted *C. vulgaris* cover (Rawes, 1983). The aforementioned Rawes and Welch (1969) also recorded increased abundance of *C. vulgaris* and *S. papillosum* where the previous regime of <0.4 ewes ha^{-1} was removed. Inter-site differences determine how sensitive a bog is to a grazing regime and multiple outcomes to the same stocking level should be expected (Adamson and Kahl, 2003; Martin *et al*. 2013).

Land managers have used fire to structure peatland vegetation communities for grazing herbivores as well as for grouse. Burns for sheep management aim to promote graminoid growth early in the year (Lindsay, 2010) and *C. vulgaris* in the autumn months (Brown *et al*. 2015a). Firing on a blanket bog at Glenamoy promoted graminoid coverage over five years, improving sheep performance (15% greater lamb births, 30% greater lamb growth, 32% greater liveweight of total sheep) relative to unburned controls (Lance, 1983). The DEFRA (2007) burn code explicitly recommends against burning in the uplands alongside 'intense' grazing pressure (no specific stocking metric is provided). The graminoid growth a burn promotes (Section 2.3.2) would attract grazers to a site when it is most vulnerable to physical damage (Anderson *et al*. 2006; Chapman *et al*. 2009), likely leading to the exposure of bare peat (Pearsall, 1941). Rawes and Hobbs (1979) therefore recommended stocking <0.1 ewes ha^{-1} following a fire.

Direct herbivory on *Sphagnum* is uncommon given its lack of palatable tissue. Pressure from grazing occurs directly through trampling and indirectly through altered interactions with other vegetation (Shaw *et al.* 1996). A national scale inventory of *Sphagnum* sp. cover suggests the genus is negatively associated with livestock presence (Noble *et al.* 2017). Elevated sheep stocking is implicated in the declines in *Sphagnum* alongside the expansion of *E. vaginatum* at Moor House (Eddy *et al.* 1969) and *N. stricta* on Shetland (Birnie and Hulme, 1990), though these data are based on habitat surveys over time as opposed to enclosure/exclosure experiments. Sheep grazing is also implicated alongside pollution as causing the decline in *Sphagnum* across the south Pennines (Tallis, 1987). *Sphagnum* coverage can increase if grazing ceases (Rawes and Hobbs, 1979; Rawes and Welch, 1969). Chapman and Rose (1991) recorded the opposite at Coom Rigg Moss where re-surveyed quadrats showed the cessation of grazing and burning initiated a *Sphagnum* decline and an *Eriophorum* sp. expansion. Chapman and Rose (1991) do not overtly recommend grazing to promote *Sphagnum* coverage as the *Sphagnum* decline may be an ongoing response to the mid-20th Century drainage of the system.

2.6.3 The effect of grazing on water table position

Few studies address the capacity for herbivores to alter WTD. Grazing alters WTD indirectly given drainage of blanket peatlands was undertaken to promote productivity for grazing livestock (Section 2.4.1) The Hard Hill experimental plots did record a reduced WTD in grazed plots relative to ungrazed plots (Worrall *et al.* 2007a). Clay *et al.* (2010) suggest the reduced CO_2 efflux they recorded in the same grazed plots may be because a reduced WTD reduced aerobic decomposition. Whilst Clay *et al.* (2010) suppose grazing may therefore create a favourable environment for carbon accumulation, any compaction of the peat surface from sheep trampling may ‘lower’ the surface and so reduce WTD. Such an occurrence would not favour carbon accumulation.

2.6.4 The effect of grazing on peat matrix characteristics

Herbivore trampling damage is determined both by peatland characteristics (i.e. BD, vegetation coverage) and by livestock hoof-to-weight ratio (ECOSSE, 2007; Shaw *et al.* 1996). Clay and Worrall (2013) recovered twelve peat cores from Atkinson Moss and used a weight dropped from a height of 50cm to simulate sheep ‘trampling’ at ‘high’ (daily) and ‘low’ (once a week) intensity. Simulated trampling reduced photosynthesis by up to 75%, increase BD (0-10cm deep) and damaged the *Sphagnum* surface layer. The only association in CO_2 fluxes was between the presence or absence of trampling, not according to trampling intensity (Clay and Worrall, 2013). Frogbrook *et al.* (2009) implicate trampling by sheep as increasing near-surface BD (<30cm deep) on Plynlimmon. Livestock tracks concentrate trampling effects to narrow paths through a peatland. Tracks in Teesdale and Wharfedale exhibited increased BD and reduced rainfall infiltration rates relative to surrounding peat (Holden *et al.* 2007a). The disruption of the surface layer by trampling can expose bare peat and encourage POC losses (Birnie and Hulme, 1990; Dawson *et al.* 2002; Watson and O'Hare, 1979), with elevated sheep stocking levels increasing the incidence of bare peat (Rawes and Hobbs, 1979; Rawes and Welch, 1969) and erosion rates (Anderson and Yalden, 1981; Shimwell, 1974) at multiple sites. Sheep may be able to initiate erosion/hagg and gully complexes (Section 2.2.4.3) directly (Simpson *et al.* 1998; Evans, 2005; Parry *et al.* 2014) or act in tandem with another driver to form the complex (Pearsall, 1941). Others suggest sheep can only reinforce existing complexes (Lindsay, 2010; Mackay and Tallis, 1996; Ratcliffe, 1959).

The cores Garnett *et al.* (2000) recovered from Hard Hill suggest grazing may not detrimentally affect carbon accumulation. SCP determined PARs were not significantly different between ungrazed and grazed plots. Billett *et al.* (2010) calculated RERCA of 1) 'grazed' 115.4 g C m^{-2} yr^{-1} and 2) 'unburned and ungrazed' 98.4 g C m^{-2} yr^{-1}. Despite aforementioned concerns regarding methodology (Section 2.3.4) these results support the notion that light grazing (<0.3 ewes ha^{-1}) may not adversely affect blanket peatland functioning.

2.6.5 The effect of grazing on the chemical composition of plant tissues and peat

Few studies address the capacity for herbivores to alter peatland chemistry. Defecation and urine increase nutrient availability at a very fine scale (Medina-Roldán *et al.* 2012). These inputs may promote nutrient affiliated *Juncus* sp. (Shaw *et al.* 1996) and cause isolated increases in decomposition (Shepherd, 2001). Herbivores primarily alter nutrient regimes indirectly by promoting the expansion of graminoids (e.g. *E. vaginatum*, *M. caerulea*) given such litter is labile and nutrient rich relative to *Sphagnum* litter (Ward *et al.* 2007).

2.6.6 Section summary

Previous research has not detected a systematic impact of herbivory on blanket peatland carbon accumulation. Any impact would likely vary according to the degree of grazing intensity, this being variable according to antecedent peatland characteristics (e.g. vegetation community composition). Appropriate management to avoid overgrazing would consider the impacts of herbivores using a flexible, site-by-site approach. The tendency of herbivore management to operate according to large-scale processes (Gordon *et al*. 2004) means such an approach may be hard to implement. Low-intensity grazing may result in favourable conditions for carbon accumulation, but data are required to test this notion beyond the confines of the Hard Hill plots. This thesis will attempt to improve quantitative understanding of the impact of grazing on blanket peatland carbon accumulation (Section 2.8). Cores will be recovered from multiple peatlands where grazing has occurred, and which have not been part of experimental conditions akin to Hard Hill. Plant macrofossil evidence will also be used to determine whether the impact of grazing can be detected using palaeoenvironmental records.

2.7 Environmental change in the Welsh uplands

There is a paucity of palaeoenvironmental records characterising Welsh blanket peatland development. Table 2.2 details existing (AD 2017) investigations of Welsh upland peatlands with locations given in Figure 2.20. All are blanket peatlands, save for Bryn y Castell (Mighall and Chambers, 1995), Cefn Glas (Smith and Green, 1995), Moel y Gerddi (Chambers *et al*. 1989), Cwm Cywion and Llyn Llydaw (Ince, 1983) which are all basin mires ≥300m asl. Studies were compiled primarily using the Payne *et al*. (2016) peatland palaeoecology database. Several additional sources were also included (Buckley and Walker, 2001; Chambers *et al*. 1989; Ince, 1983; Moore, 1968; Moore and Chater, 1969; Wiltshire and Moore, 1983). Twenty-eight of the twenty-nine studies examined pollen, the only one which didn’t (Mighall *et al*. 2002) was from a site where the lead author had already published pollen work (Mighall and Chambers, 1993). Twenty-five reported stratigraphy, nine assessed charcoal and five examined plant macrofossils. None examined testate amoebae. The relevance of this focus on pollen evidence is discussed below.

Table 2.2: Details of previous Welsh upland peat palaeoenvironmental investigations. Numbers relate to sites in Figure 2.20. S= Stratigraphy, M= Macrofossils, C= Charcoal and P= Pollen.

No.	Site(s)	S	M	C	P	Source
1	Migneint	X		X	X	Blackford (1990)
2	Moel Sych	X	X		X	Bostock (1980)
3	Bryniau Pica	X			X	Buckley and Walker (2001)
4	Brecon Beacons	X			X	Chambers (1982a)
5	Cefn Gwernffrwd	X			X	Chambers (1982b)
6	Coed Taf	X			X	Chambers (1983a)
7	Cefn Gwernffrwd	X			X	Chambers (1983b)
8	Moel y Gerddi	X		X	X	Chambers *et al*. (1989)
9	Hirwaun, Mynydd Llangatwg	X	X	X	X	Chambers *et al*. (2007a)
10	Drygarn Fawr		X	X	X	Chambers *et al*. (2007b)
11	Twyn Mwyalchod		X		X	Chambers *et al*. (2013)
12	Migneint		X	X	X	Ellis and Tallis (2001)
13	Mynydd Hiraethog	X		X	X	Grant (2007)
14	Llys-y-coed	X		X	X	Grant (2009)

Table 2.2 (cont.)

15	Banc Llyn Mawr	X		X	Grant (2012)
16	Cwm Cywion, Llyn Llydaw	X		X	Ince (1983)
17	Cefn Mawr	X	X	X	Lascelles (1995)
18	Copa Hill	X		X	Mighall and Chambers (1993)
19	Bryn y Castell	X		X	Mighall and Chambers (1995)
20	Copa Hill	X			Mighall *et al*. (2002)
21	Plynlimmon	X	X	X	Mighall *et al*. (2013)
22	Plynlimmon	X		X	Moore (1968)
23	Elan Valley, Llyn Gynon			X	Moore and Chater (1969)
24	Pen y Gader-Fawr, Ty Isaf, Waun Fach	X		X	Price (1981)
25	Waun-Fignen-Felen	X		X	Smith and Cloutman (1988)
26	Cefn Glas	X		X	Smith and Green (1995)
27	Glaslyn	X		X	Taylor (1973)
28	Carneddau	X		X	Walker (1993)
29	Nantybeddau, Pwll-nant-ddu	X		X	Wiltshire and Moore (1983)
30	Aber Valley	X	X	X	Woodbridge *et al*. (2012)

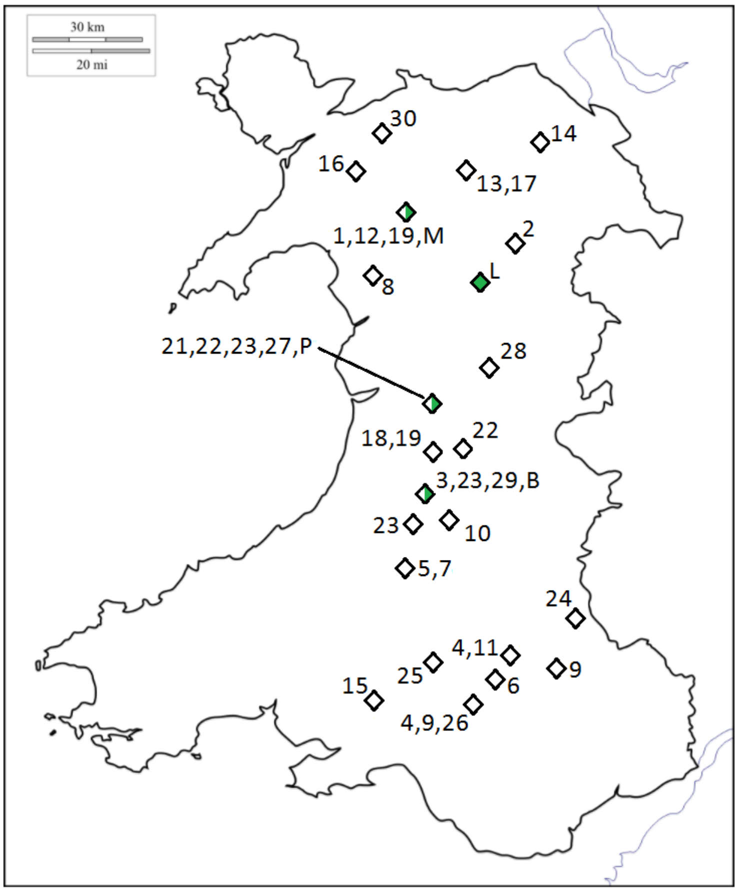


Figure 2.20: Locations of Welsh upland peatland palaeoenvironmental investigations. Numbers refer to previous work as detailed in Table 2.2, whilst letters refer to sites in this thesis (B=Bryniau Pica, L=Lake Vyrnwy, M=Migneint and P=Plynlimmon; Chapter 4). Open diamonds indicate previous work, half green and half open diamonds indicate areas subject to previous work and studied in this thesis, whilst solid green diamonds indicate sites solely investigated in this thesis. Base map source: www.d-maps.com.

Figure 2.21 summarises environmental records for human impacts and climatic change which are expressed in upland peatland records from Mid and North-Wales (>52°N), given the relevance of these regions to this thesis (Chapter 4). Several sources report a pollen signal of increasing anthropogenic deforestation post-AD 0 (Buckley and Walker, 2001; Chambers *et al*. 2007b; Mighall and Chambers, 1995; Woodbridge *et al*. 2012) due to the adoption of ploughing technology (Lascelles, 1995) and the Roman occupation of Wales *ca.* AD 48-400 (Bostock, 1980; Walker, 1993). Roman withdrawal (AD 400) led to agricultural instability and upland abandonment as tribal feuding occurred (Moore, 1968). Evidence for woodland regeneration, a decline in upland pastoralism and a more transient human signal is recorded in several pollen profiles *ca.* AD 400-1000

(Grant, 2009; Moore, 1968). Declines in upland occupancy may be linked to the concurrent climatic deterioration expressed in macrofossil records around this time (Blackford, 1990; Bostock, 1980). The warm Medieval climate anomaly (MCA) (*ca.* AD 950-1350; Lamb, 1965) prompted upland settlement and cereal cultivation (Bostock, 1980; Grant, 2007, 2009; Mighall and Chambers, 1995; Mighall *et al*. 2013). Upland grazing during this period (*ca.* AD 950-1350) was likely related to the expansion of monastic influence (Moore and Chater, 1969). Declines in upland activity *ca*. AD 1300-1450 are associated with the black death, Glyndŵr's rebellion (Grant, 2009; Moore, 1968) and the onset of the Little Ice Age (LIA), though the timing and nature of the LIA are contentious (Diaz *et al*. 2011; Jones and Mann, 2004). Upland activity again intensified *ca*. AD 1550 with the re-expansion of sheep grazing (Mighall *et al*. 2013) which continued over the coming centuries (Ellis and Tallis, 2001). Sheep production increasing during the Napoleonic Wars (AD 1803-1815) (Bostock, 1980) alongside upland enclosure and cereal cultivation (Moore and Chater, 1969).

Isolating the effect of a single driver on biotic patterns requires consideration of the many different drivers which could affect ecosystem development (Birks, 1986). Classifying the relative importance of climatic, human and autogenic drivers of change can be challenging in late-Holocene peat deposits (Swindles *et al*. 2012; Turner *et al*. 2014; van der Linden and van Geel, 2006). The majority of Welsh upland peatland records published to date (AD 2017) used pollen as the primary evidence of environmental change (Table 2.2). There are problems when reconciling the spatial scale (i.e. local vs regional) that a pollen signal represents (Fægri and Iversen, 1989; Sugita, 2007a, b). Such uncertainties complicate interpretations that can be made about the development of vegetation surrounding the deposit or the development of the deposit itself. It is therefore unsurprising that pollen and plant macrofossil records from the same peatland can suggest differing signals of environmental change (Chambers *et al*. 2013; McCarroll *et al*. 2017). Reliance on an arguably unrepresentative proxy means understanding of long-term Welsh upland peatland development is coarse. Highlighting this overreliance on pollen analysis is not intended as a damming critique of previous work given many studies in Table 2.2 were primarily interested in landscape scale change. The lack of plant macrofossil records is instead stressed to emphasize the limits of our understanding surrounding long-term changes in peatland integrity rather than our relatively better understanding of landscape scale changes in the wider Welsh environment. This study will address this issue by using a technique which assesses the state of the peatlands themselves (plant macrofossil analysis) rather than using techniques which assess environmental change primarily at the landscape level (i.e. pollen).

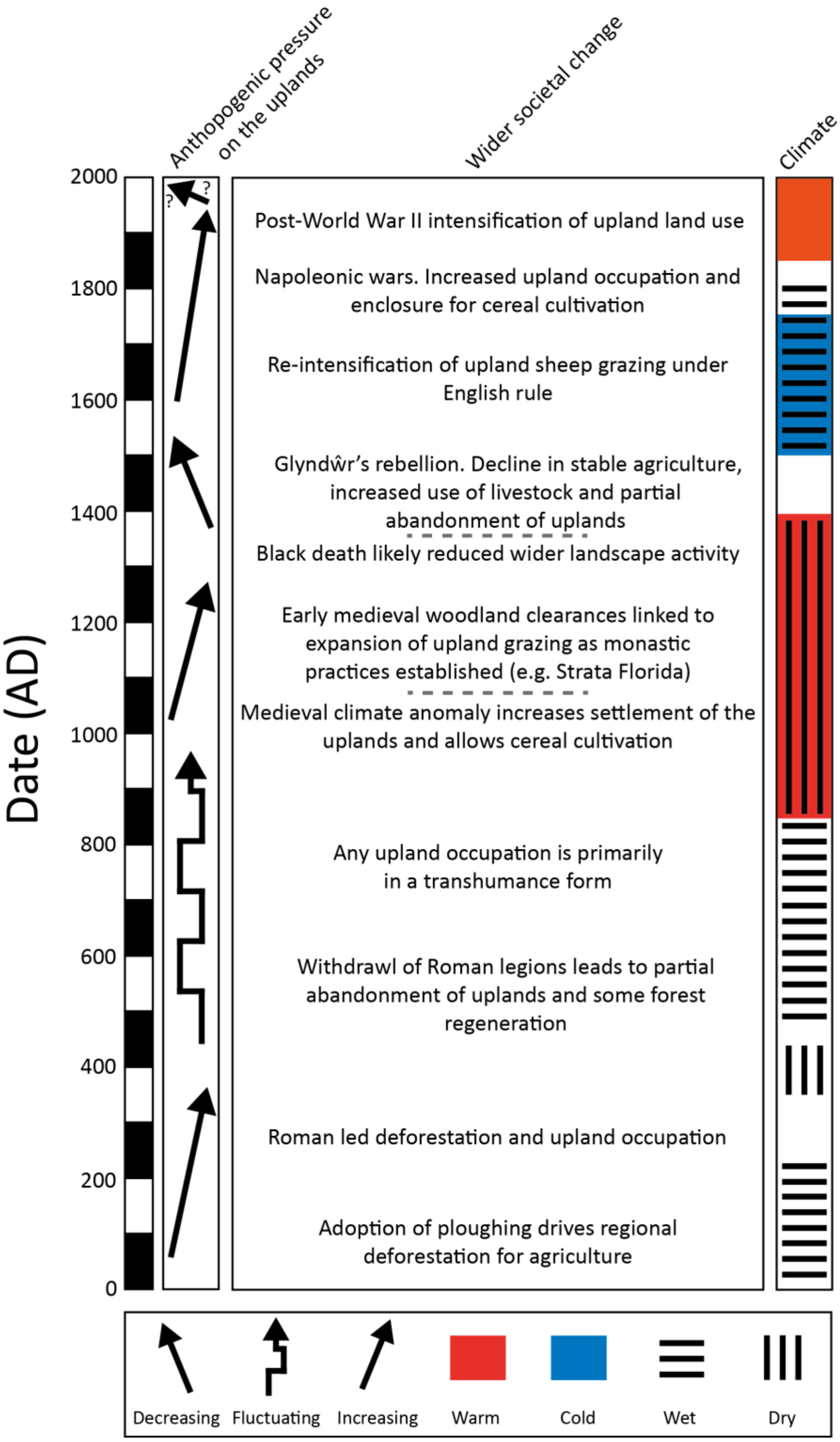


Figure 2.21: Regional palaeoenvironmental synthesis for anthropogenic and climatic changes, as expressed in mid- and north-Wales upland peatland records (>52°N). '?' refers to presumed decline in pressure from late 20th Century agri-environment schemes (Section 2.6.1).

2.8 Conclusions of the literature review

This chapter has demonstrated that the land use types of burning, drainage, forestry and grazing have affected the development of blanket peatlands across the British Isles. Previous work has not reliably identified the effect of these land uses on carbon accumulation. Hence, the aim of this thesis is to characterise the impact of land management on recent (*ca*. last 150 years) blanket peatland carbon accumulation. Shortcomings of previous work have been incrementally highlighted through this chapter. These shortcomings have formed the basis for developing three research questions which will be used to address the thesis aim (see below). Each research question is accompanied by a rationale for its development and a commentary on how this thesis will address the question.

2.8.1 Research question one: Do differing land use practices affect blanket peatland carbon accumulation in a characteristic manner?

How land use influences carbon accumulation is largely uncertain and understanding of impacts are primarily theoretical. Few researchers have quantified carbon dynamics in relation to land management and those that have report largely equivocal evidence (e.g. Clay *et al*. 2009a; Ward *et al*. 2007). Many comment on the negative implications for carbon accumulation if land use alters peatland characteristics (e.g. WTD) but present no measured carbon accumulation rates to support their assertions (e.g. Brown *et al*. 2015a; Holden *et al*. 2015). This thesis must therefore quantitatively examine the impact of land use as opposed to theoretically implicating change.

To address this question, cores will be recovered from several blanket peatlands which have been managed according to the four land uses selected for investigation. The degree to which each land use type produces a characteristic rate of carbon accumulation may then be assessed according to results generated from recovered cores. This thesis will provide new insight into the relationship between carbon accumulation rates and land use from ‘non-experimental’ blanket peat sites, reducing the need for peatland researchers to rely on experimental locations (e.g. Hard Hill) for such investigations.

2.8.2 Research question two: Is it appropriate to treat peatland management regimes as discrete entities, or is it more appropriate

to recognise that multiple environmental drivers are responsible for peatland development?

Blanket peatlands are affected by a variety of internal and external forcings as they develop over time. A consistent theme presented through this literature review has been the uncertainty surrounding the impact of differing land use types on peatland development. Uncertainty does not only relate to the impact of land management on carbon dynamics (see above), but relates also to the uncertain impact of management on all the aspects of peatland systems (e.g. plant community structure, WTD) given a consistent response to management is rarely recorded. It is this variability which creates uncertainty as to how land use affects blanket peatlands. It may be flawed to suggest all blanket peatlands will respond to land use in an identical manner given heterogeneous geological settings, initiation mechanisms, vegetation community composition and prior management histories. Peatlands are seldom subject to a single management practice (Davies *et al*. 2013) and understanding of how practices interact is far from satisfactory (Evans *et al*. 2014b).

This thesis has, thus far, considered how each of the four chosen land use types affect blanket peatlands in relative isolation. Separating land uses into discrete entities was done to aid the readers' understanding of the evidence base (Section 2.1), but also to emphasize the degree to which researchers tend to consider management types in isolation. This thesis will address this research question by 1) assessing whether discrete management signals are recorded using palaeoenvironmental techniques and 2) by evaluating whether other indicators of peatland state may be used to explore the relationship between land use and peatland carbon dynamics. The potential use of vegetation communities as indicators of peatland condition was raised in Section 2.2.4.1. This thesis will explore the potential that vegetation community composition may be used to assess both 1) the degree to which a peatland may have been disturbed by human activity and 2) whether peatlands which exhibit similar patterns of vegetation development also exhibit similar rates of carbon accumulation.

2.8.3 Research question three: Did late-19th and 20th Century management impacts affect blanket peatland integrity to a greater degree than pre-AD 1850 impacts?

Beyond their role in initiation (Section 2.2.1.1), humans have been active near blanket peatlands through their ongoing development. Burning and grazing indicators are recorded in Welsh peatlands since *ca*. 4,000 BC (Bostock, 1980; Woodbridge *et al*. 2012) and an increase in the abundance of such indicators occurs post-AD 0 (Mighall and Chambers, 1993, 1995). Blanket peatlands are therefore relevant to the ‘Anthropocene’ debate. The debate centres around attempts to define the time where humans became the dominant forcing of planetary development (Crutzen, 2002; Malm and Hornborg, 2014; Steffen *et al*. 2007, 2011). Whilst many researchers endorse *ca*. AD 1950 as the threshold for the Anthropocene (Dean *et al*. 2014; Steffen *et al*. 2015; Zalasiewicz *et al*. 2014), the long-history of human interactions with blanket peatlands mean humans may have been the dominant forcing of peatland development much earlier than AD 1950. Late-19th and 20th Century peatland management is typically assumed to have increased pressure on these habitats relative to earlier periods (Harris *et al*. 2011; Holden *et al*. 2007a). If pre-AD 1850 land use had more of an impact on peatland integrity, then these antecedent conditions may control peatland response to subsequent land use. The lack of records monitoring Welsh blanket peatland integrity (Section 2.7) means there is uncertainty as to whether the effect of pre-AD 1850 land use lingers to this day.

To address this question, this thesis will present palaeoenvironmental evidence which explicitly monitors changes in peatland state (plant macrofossil analysis) as opposed to wider landscape scale change (e.g. pollen analysis). Chronological controls will be used to date peat accumulation since *ca*. AD 1850 (i.e. during the supposed ‘increase’ in anthropogenic pressure on these systems) to assess the relative proportion of pre- and post-industrial human disturbance recorded at each study site.

Chapter 3 Methods

3.1 Introduction

This thesis aims to characterise the impact of land management on blanket peatland carbon accumulation. The study design and methodological approaches detailed within this chapter are designed to address this aim. Sampling strategy (Section 3.2), sample recovery (Section 3.3), laboratory (Section 3.4) and statistical approaches (Sample 3.4.5) are presented in turn alongside a background to the technique and a justification for its selection.

3.2 Sampling strategy

Previous research has noted that carbon dynamics can vary at fine spatial scales within a peatland (Section 2.2.3). It is therefore questionable whether the standard palaeoecological approach of recovering a single core from a peatland would be truly representative of changes across a site. To improve the replicability of results it was decided that multiple cores would be recovered across several blanket peatland macrotopes (*sensu* Section 1.4.1). A spatially nested design (Figure 3.1) was selected to explore the spatial scale at which carbon accumulation varied across Welsh blanket peatlands in response to land use change. The terms ‘site’, ‘sub-site’ and ‘sampling point’ are used exclusively hereafter according to their position illustrated in Figure 3.1. ‘Sites’ are blanket peatland macrotopes selected for study. ‘Sub-sites’ are areas within ‘sites’ where multiple cores were recovered at individual ‘sampling points’.

This thesis formed part of the Glastir Monitoring and Evaluation Programme (GMEP) conducted by the Centre for Ecology and Hydrology (CEH). The GMEP warranted that at least forty-eight cores be recovered in this project. This number was increased to fifty during fieldwork planning to improve replicability. GMEP funding had been allocated for ten cores to undergo radiometric dating (Section 3.4.3). These ten cores are hereafter termed ‘master cores’. The remaining forty cores are hereafter termed ‘replicate cores’. The allocation of master and replicate cores alongside site selection rationale are discussed in Chapter 4.

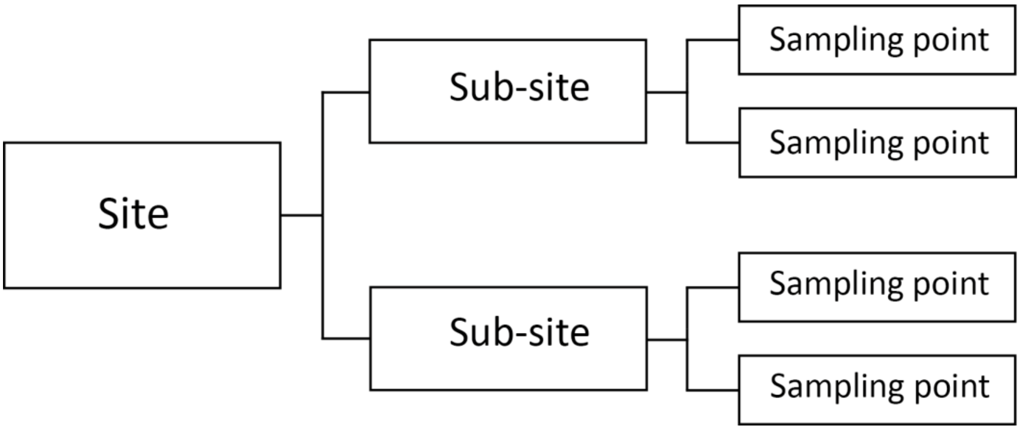


Figure 3.1: Schematic diagram detailing nested approach to core recovery. Number of 'sub-sites and 'sampling points' are arbitrary for illustrative purposes.

3.3 Sample recovery and preparation

Selecting the point in a peatland where a core will be recovered from is an important component of a palaeoenvironmental study. Palaeoecological investigations typically recover cores from the deepest area of a lawn microtope (Section 1.4.1) given the increased likelihood of recovering a record sensitive to previous climatic fluctuations (Barber *et al*. 1998; de Vleeschouwer *et al*. 2010). Many degraded peatlands exhibit a loss of hummock-hollow complexes (Chambers *et al*. 2017; Lindsay, 2010) meaning delineation between microtope types is not always possible. Most sites in this thesis did not exhibit traditional microtopography given they were selected to examine management impacts (Chapter 4).

Each sub-site was extensively field walked prior to sampling. Sampling points were selected to capture variation in contemporary vegetation communities given plants are known to respond to the impacts of land use and to affect carbon accumulation (Section 2.2.4.1). Where possible, cores were recovered from active *Sphagnum* lawns or from 'smaller level' *Sphagnum* patches (*sensu* Amesbury, 2008). Monocot tussocks were specifically avoided given sampling may have meant subsequent analysis was primarily assessing the plant rather than the peat matrix. Sampling points were assessed for a peat depth of at least 50cm using a specifically designed peat cutter (de Vleeschouwer *et al*. 2010; Lageard *et al*. 1994) with a *ca*. 1m long blade (Figure 3.2). Samples were recovered using a 20x10x50cm (width, depth, height) sharpened monolith tin for the master cores and a 10x10x50cm tin for the replicate cores. The tin was gradually eased into the peat using a clean serrated knife to cut into unconsolidated live vegetation. Additional cuts were made with the peat cutter once the tin was *ca*. 15cm deep and could freely stand. The tin was eased into the peat until the enclosed top was flush with the surface of the live vegetation. A *ca*. 55-60cm deep excavation hole was dug behind the monolith and a cut was made across the bottom of the tin. A final cut was made with the re-cleaned peat cutter down the open face of the tin before the sample was recovered. Peat density is an important determinant of rates of carbon accumulation (Section 3.4.2). As such, care was taken during monolith tin insertion and recovery to avoid peat compaction. Each monolith tin was cleaned prior to recovery of subsequent samples to avoid cross-contamination (de Vleeschouwer *et al*. 2010). Recovered samples were transferred to 8mm thick corrugated plastic open-face boxes of the same dimensions as the used monolith tin. Boxes were wrapped in carbon stable plastic bags and sealed with electrical tape. Master cores were stored horizontally and in the dark at 4°C to limit fungal growth. Replicate cores were stored horizontally and in the dark in a cool room given limited cold storage availability. Cores were sampled in a clean laboratory. The outer edges of each core were manually removed using a spatula to avoid contamination of material displaced by coring. Stainless steel implements were cleaned using de-ionised water and blue roll between each subsample. Cores were archived for future work.

Figure 3.2: Peat cutter used in fieldwork. Individual is 188cm tall for reference.

3.4 Laboratory methods

This section describes methods used to analyse cores following their recovery in the field. Analytical approaches were selected because the data they produce could answer research questions outlined in Section 2.8. Peat bulk density, carbon content, radiometric dating and Spheroidal Carbonaceous Particle dating can be used to reconstruct carbon accumulation rates to address research question one (Section 2.8.1). Plant macrofossil analysis can be used to reconstruct vegetation change over time to answer research question two (Section 2.8.2) and the selected dating techniques can be used to establish an AD 1850 horizon in each core to answer research question three (Section 2.8.3).

3.4.1 Peat bulk density

Peat bulk density (BD) represents the mass of peat within a given volume and is typically expressed as g cm^3. The metric is an important component in determining the amount of carbon contained within a peatland (Chambers *et al.* 2010). A known volume of fresh peat was recovered at 1cm and 4cm contiguous increments for master and replicate cores respectively. Fresh samples were measured and cut manually using a ruler and Vernier callipers. This approach avoids sample compression which can occur when using volumetric samplers or the loss of material which can occur when using water displacement to calculate BD (Bengtsson and Enell, 1986). Each sample represented a 1cm thick horizon for the master cores (0cm corresponds to homogenized BD of 0-1cm depth) or a 4cm thick horizon for the replicates (0cm corresponds to homogenized BD of 0-4cm depth). Recovered samples were placed into pre-weighed crucibles which had been thoroughly cleaned and rinsed with de-ionised water (hereafter ‘cleaned’). Crucibles were re-weighed to determine the mass of the fresh sample and placed in an oven for 12 hours at 105°C. Crucibles were cooled to room temperature in a desiccator and re-weighed to determine dry peat mass. BD was determined using Equation 3.1, where BD is bulk density (g cm^3), DW is dry weight (g) and FV is fresh peat volume (cm^3):

$$BD = \frac{DW}{FV}$$ Equation 3.1

BD values were combined with peat accumulation rates (Sections 3.4.3 and 3.4.4) to determine rate of dry mass accumulation following Equation 3.2, where DMA is dry mass accumulation rate (g m^2 yr^{-1}) and PAR is peat accumulation rate (mm yr^{-1}):

$$DMA = BD \times PAR \times 1000 \qquad \text{Equation 3.2}$$

3.4.2 Peat carbon and nitrogen content

A set number of samples were allocated for carbon (C) and nitrogen (N) content analysis within the GMEP work package. Limits on sample numbers necessitated that master cores be analysed at 2cm resolution whilst replicate cores could only be analysed at 4cm resolution for C/N content. Each sample represented a 2cm thick horizon for master cores (0cm corresponds to homogenized 0-2cm depth) or a 4cm thick horizon for replicates (0cm corresponds to homogenized 0-4cm depth). Samples were placed into cleaned crucibles and placed in an oven for 12 hours at 105°C. Samples were cooled to room temperature in a desiccator and ground to a fine powder (<130 μm) using a pestle and mortar (Hughes *et al.* 2013). Milled samples were transferred to glass vials for transport to laboratories at CEH Lancaster. C and N content was determined using a Vario Elemental Analyser following CEH accredited methodologies. Quality control and duplicate sample testing variance were all within expected ranges which suggests the reported results were accurate (Keenan, P. pers. comm. 2014). Results were expressed as total element % of the provided peat sample. These values were combined with BD to determine C stocks and with dry mass accumulation rates (Section 3.4.1) to determine carbon accumulation rates following Equation 3.3, where CAR is carbon accumulation rate (g C m^2 yr^{-1}) and C is carbon (%):

$$CAR = \frac{DMA \times C}{100} \qquad \text{Equation 3.3}$$

3.4.3 Radiometric dating techniques

^{210}Pb dating has been extensively employed to date recent peat accumulation (*ca.* last 150 years) (Turetsky *et al.* 2004). The approach uses the radioactive decay chain of ^{238}U. ^{226}Ra decays to ^{222}Rn and a portion of atoms are released to the atmosphere where they decay to ^{210}Pb. ^{210}Pb nuclides are returned to the terrestrial environment by precipitation or dry deposition (Appleby, 2001). ‘Total’ ^{210}Pb activity in ombrotrophic peats is comprised of ‘unsupported’ and ‘supported’ ^{210}Pb activity. ‘Supported’ ^{210}Pb activity results from *in situ* decay of ^{226}Ra. ‘Unsupported’ ^{210}Pb activity is that activity which relates to atmospheric fallout and is measured as the portion of 'total' ^{210}Pb activity in excess of 'supported' ^{210}Pb activity. The chronological range of ^{210}Pb dating occurs where total ^{210}Pb activity reaches equilibrium with supported ^{210}Pb (i.e. ^{210}Pb from atmospheric fallout (unsupported activity) has completely decayed). Equilibrium between supported and unsupported activity typically occurs over a *ca.* 150-year period, this being the applicable timeframe of ^{210}Pb age/depth models (Appleby, 2001).

The age for a given peat depth is determined according to the degree of unsupported ^{210}Pb activity, with linear interpolation between dated points to construct the age/depth model. The constant initial concentration (CIC) and constant rate of supply (CRS) models are the two most commonly applied methods for ^{210}Pb dating of sedimentary systems (Appleby, 2001). The CIC model assumes that 1) the ^{210}Pb flux is principally driven by changes in wider sedimentary systems (e.g. large lakes) and 2) each portion of sediment has the same initial unsupported ^{210}Pb concentration per standard portion of dry mass (Le Roux and Marshall, 2010). These assumptions invalidate the use of the CIC model in peatlands because 1) ^{210}Pb flux is primarily determined by atmospheric deposition and 2) changes in peat accumulation rates affect the initial concentration of unsupported ^{210}Pb (Turetsky *et al.* 2004). The CRS model is appropriate for use in peatlands because it can accommodate variable sediment accumulation rates and differing initial unsupported ^{210}Pb concentrations (Appleby, 2008; Urban *et al.* 1990). The age of a particular depth (CRS model) is provided following Equation 3.4 (Appleby, 2001; Appleby and Oldfield, 1978), where T is years since deposition, λ is the decay constant for ^{210}Pb (0.03114), A_0 is the unsupported ^{210}Pb inventory below the specified depth and A is the total unsupported ^{210}Pb inventory:

$$T = \left(\frac{1}{\lambda}\right) \ln \left(\frac{A_0}{A}\right)$$

Equation 3.4

The CRS model assumes the supply of unsupported ^{210}Pb is constant, that ^{210}Pb is transferred rapidly to the substrate and that ^{210}Pb is immobile following deposition (Appleby, 2001). There is concern over the reliability of this last assumption, meaning that ^{210}Pb chronologies should be validated using independent markers (e.g. ^{137}Cs, ^{241}Am) to test for post-deposition ^{210}Pb migration (Abril, 2004; Oldfield *et al.* 1995). ^{137}Cs and ^{241}Am isotopes were released into the atmosphere because of aboveground nuclear weapons activity, hence changes in their fallout concentration profiles can be used as chrono-stratigraphic markers in sedimentary sequences (e.g. the AD 1986 Chernobyl disaster) (Turetsky *et al.* 2004). ^{137}Cs can be mobile in peat given there is a lack of clay to bind exchangeable ions (e.g. Cs+) (Gerdol *et al.* 1994; Shand *et al.* 1994). Such mobility potentially undermines its use as a temporal marker (Oldfield *et al.* 1995; Parry *et al.* 2013). ^{137}Cs can also be affected by vegetation uptake if vascular plants dominate a site given their extensive root networks may affect the mobility of the radioisotope (Turetsky *et al.* 2004). However, successful use of ^{137}Cs as an independent age marker has been recorded in ombrotrophic peats where the technique exhibits agreement with other chronological methods (Appleby, 2008; Appleby *et al.* 1997; Bao *et al.* 2015; Sanders *et al.* 1995). ^{241}Am is thought to be less mobile than ^{137}Cs in peats (Oldfield *et al.* 1995) though its behaviour is less understood relative to ^{137}Cs (Turetsky *et al.* 2004).

Only master cores underwent radiometric dating analysis. Following analysis for BD (Section 3.4.1) those same samples were transferred to carbon stable sample bags for transport to the Environmental Radiometric Facility at University College London. These samples included live vegetation layers (Olid *et al.* 2008). Samples were analysed under direct gamma assay in an ORTEC HPGe GWL well-type coaxial low background intrinsic germanium detector which was calibrated using samples with known gamma activity (Yang, H. pers. comm. 2014). Isotopes were detected by their gamma emissions at 46.5 keV (^{210}Pb), 295 and 352 keV (^{226}Ra), 662 keV (^{137}Cs) and 59.5 keV (^{241}Am) (Appleby, 2001; Appleby *et al.* 1986) with corrections for self-absorbing low energy gamma rays (Appleby *et al.* 1992).

3.4.4 Spheroidal Carbonaceous Particle analysis

Spheroidal Carbonaceous Particles (SCPs) are elemental carbon particulates formed via the incomplete combustion of coal and oil in industrial processes (Rose, 2008; Yang *et al.* 2001). Their presence in sedimentary records is considered indicative of industrial activity as SCPs have no natural source in the Quaternary (Swindles, 2010). Three features of SCP concentration profiles are identifiable as time markers, based on their correlation with changes in industrial activity (Rose *et al.* 1995). These features are (A) the start of the record (*ca.* AD 1850s), (B) a rapid increase in SCP concentration (*ca.* mid-AD 1950s) and (C) peak SCP concentration (*ca.* late-AD 1970s) (Figure 3.3). The dating of these features changes by geographic region due to variations in industrial activity. North and Central Wales (Chapter 4) have average dates for the three features of A) AD 1850 ± 25, B) AD 1955 ± 15 and C) AD 1976 ± 3 (Rose and Appleby, 2005). These values were used in this thesis. Whilst the beginning of the record (A) and the peak (C) are relatively easy to allocate to SCP profiles, the identification of the rapid increase in SCPs (B) can be hard to identify given ambiguity as to what constitutes a ‘rapid increase’. Rose *et al*. (1995) suggest identifying (B) as the depth where the extrapolated gradients of the slow increase in SCP concentrations and the rapid increase in SCP concentrations intercept, as illustrated in Figure 3.3. This is the method adopted in this thesis.

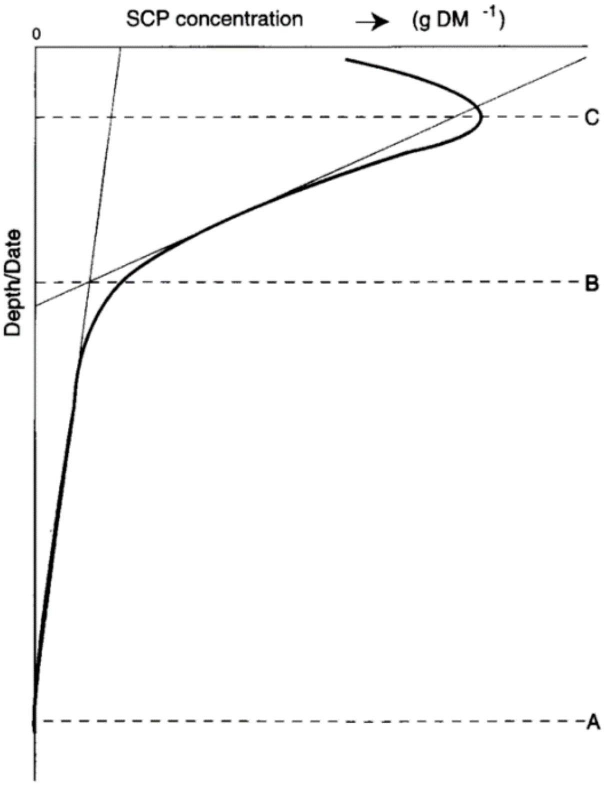


Figure 3.3: Schematic profile detailing an idealised SCP concentration curve. The location of the three dateable features (A, B and C) are indicated. The intercept of the pre- and post-B extrapolated gradients, as referred to in the text, is also indicated (Rose *et al*. 1995).

SCPs provide an alternative to radiometric dating methods where there are concerns over ^{210}Pb mobility (Yang *et al*. 2001). SCP mobility in peat is considered minimal (Swindles, 2010) due to the limited post-deposition movement of pollen grains which are of a similar size to SCPs (Clymo and Mackay, 1987). It is questionable whether the results of Clymo and Mackay (1987) are applicable when discussing SCP mobility given SCPs and pollen are composed of differing elements. Punning and Alliksaar (1997) have though shown minimal post-deposition movement of fly-ash particles (of which SCPs are a component) in peat, with <1% of washed out of a laboratory test over 241 days. As such, it does seem appropriate to consider post-deposition SCP movement to be minimal.

SCP dating assumes peat accumulation is broadly constant as a sudden increase in peat accumulation may 'dilute' SCP concentration per portion of dry mass (Garnett *et al.* 2000; Swindles, 2010). The magnitude of the change in particle concentrations, often increasing from <1,000 particles g DM^{-1} at the start of the record to >10,000 particles g DM^{-1} at the peak, suggests changes in sediment accumulation are unlikely to completely mask the three features of the SCP concentration curve (Parry, 2011). Unexpected characteristics of SCP concentration curves in lake sedimentary profiles (e.g. double peaks) may be inferred as evidence of post-deposition particle mobility or that concentrations have been diluted by variable sediment accumulation rates (Rose and Appleby, 2005). To counter concerns over downward mobility of SCPs one may consider the start of the record (A) to occur when SCPs are consistently recorded at every depth increment given their ubiquitous release during wide-scale industrial activity (Blundell, 2002; Clement, 2005). This thesis adopts such an approach and considers the start of the SCP record (A) to be the deepest SCP occurring sample above which SCPs are consistently recorded.

SCP extraction followed Rose (1990, 1994) with the addition of *Lycopodium* tablets to aid quantification (Hendon and Charman, 2004; Parry *et al.* 2013). 1cm thick samples were recovered at 2cm and 4cm non-contiguous intervals for master and replicate cores respectively. Fresh peat samples were transferred to cleaned crucibles and dried 12 hours at 105°C. Crucibles were cooled in a desiccator and a known mass (0.1-0.2 g) was transferred to polypropylene test tubes. 3ml concentrated HNO_3 was added and samples stood for 12 hours in a fume cupboard. A further 3ml of concentrated HNO_3 was added and the tubes were subjected to an 80-90°C hot bath for *ca.* 2 hours. Samples were centrifuged, decanted and rinsed twice. Two *Lycopodium* tablets of known particle concentration were added to each sample (Stockmarr, 1971) and dissolved using 10% HCl. Samples were centrifuged, decanted and rinsed with the resultant residue evaporated onto a microscope slide and mounted in Aquatex. SCPs and *Lycopodium* were counted under x400 magnification. SCPs were identified and counted following Rose (2008) against a constant frequency of 500 *Lycopodium*. *Lycopodium* concentration was determined following Equation 3.5, where LC is *Lycopodium* concentration (particles per g), N is the number of particles added and DW is dry sample weight (g):

$$LC = \frac{N}{DW}$$ Equation 3.5

SCP concentration was expressed using Equation 3.6, where CONC is SCP concentration (particles per g), RC is raw SCP count and LN is the constant number of *Lycopodium* counted on the slide:

$$CONC = \frac{LC \times RC}{LN}$$ Equation 3.6

3.4.5 Plant macrofossil analysis

Identification of plant macrofossils can be used to reconstruct past peatland vegetation communities (Mauquoy *et al.* 2010). One assumes the dominance of a species occurs due to the occurrence of ecological conditions which favour its growth and abundance (Charman, 2002). The ecological affinities of each taxon identified during analysis therefore allow inferences to be drawn regarding peatland development as vegetation changes can be interpreted as manifestations of the impacts of drivers such as climate, competition and land management. A uniformitarian approach is a precursor to palaeoenvironmental inferences in that one assumes taxa conform to their contemporary ecological tolerances. Uniformitarian principles offer a sufficiently robust approach to make inferences regarding past changes in environmental conditions (Mauquoy and van Geel, 2007). The approach is undermined if a species has altered its niche over centennial to millennial timescales (e.g. *Sphagnum austinii* (Barber, 1981; Stoneman, 1993)). Exact phytosociological reconstructions may not be possible, nor are reconstructions strictly comparable with neophytosociological surveys, given assemblages may be distorted by differential decomposition rates and remains from plants that overlie the sample of interest (e.g. younger monocotyledon roots penetrating older peat) (Mauquoy and van Geel, 2007). Plant macrofossil analyses are therefore partial expressions of prior vegetation community structure.

4cm^3 (1x1x4cm) subsamples were recovered at 2cm non-contiguous increments for macrofossil analysis. Samples were rinsed through a 125 µm sieve using a standardised volume of water (5 litres) which has been found to be effective in removing fine material and disaggregating litter within the matrix (Mauquoy *et al.* 2010). Samples were moved to Sterilin tubes prior to analysis using de-ionised water. Samples were poured into a clear plastic trough with sufficient de-ionised water added to float the remains (Mauquoy *et al*. 2010). Samples were identified under a low power (x10-x50) light microscope using reference literature (Atherton *et al*. 2010; Daniels and Eddy, 1985; Mauquoy and van Geel, 2007; Smith, 2004) and reference type material collections held at the University of Southampton Palaeoenvironmental Laboratory. Individual plant remains were mounted in Aqua-Mount and examined under x200-x400 magnification to aid in identification using cell structures. Vegetative macrofossils were quantified using an ordinal five-point abundance scale where 0 = absent, 1 = rare, 2 = occasional, 3 = frequent, 4 = common and 5 = abundant (Barber, 1981; Walker and Walker, 1961). Individual plant components (e.g. *Eriophorum vaginatum* epidermis, leaves, roots, seeds) were classified separately as pseudo-taxa to aid in the interpretation of results (Mauquoy *et al*. 2010). Charcoal and peat mites were also recorded according to the five-point scale.

3.5 Statistical methods for analysing palaeoecological data

Palaeoecological datasets represent complex biotic assemblages sampled at numerous points in space and time (Prentice, 1986). Changes in ecological community composition can be visually identified using stratigraphic diagrams. The size and complexity of palaeoecological datasets often limits the use of such an approach to offering little more than qualitative observation (Prentice, 1986). Ordination can be used to investigate complex multivariate data, such as palaeoecological community data. Ordination outputs are numerical representations of environmental gradients that occur in space and time.

Ordination orders ‘objects’ according to their numerical position in relation to multiple variables (e.g. environmental gradients). In the data used in this thesis ‘objects’ are taxa recorded within each sample (e.g. *S.* s*. Sphagnum*). Taxa can be displayed in a bi-plot according to their values on differing gradients. Similar taxa are ordinated close to each other whilst dissimilar species exhibit greater differences. In this way ordination techniques detect taxa which exhibit similar responses to variables within the ordination space. Taxa are often plotted according to their position of Axes one and two as these are the variables which exert the most control over taxa distribution. Ordination is used in palaeoecology to explore the response of species to one or more environmental gradients, these being controls on plant distribution such as temperature and hydrological conditions. ‘Indirect’ ordinations are used where the environmental gradients which control the distribution of plant communities are not directly known. Indirect ordinations are used to explore those latent gradients which are concealed within ecological community data given variables such as temperature are often not directly determinable over centennial-millennial timescales.

3.5.1 Detrended Correspondence Analysis

The most commonly employed indirect ordination is Detrended Correspondence Analysis (DCA) which is an ordination technique developed to circumnavigate the arch effect of Correspondence Analysis (Hill and Gauch, 1980; Jackson and Somers, 1991). DCA addresses the arch effect by dividing the first axis into segments and non-linear rescaling data within each segment along each further axis (Legendre and Legendre, 2012; Ter Braak and Prentice, 1988). The DCA relies on user interpretation to determine whether the detrending and rescaling have ordinated data according to some gradient which taxa may realistically respond to (Daley and Barber, 2012). The primary assumption of the DCA is that all taxa are responding in a unimodal manner to the inferred environmental variables. If an insufficient gradient length has been sampled then taxa may be responding in a linear manner, an occurrence which invalidates the use of a DCA. If the gradient is short then a linear model such as a Principal Components Analysis (PCA) should be used (van Bellen *et al*. 2016), though plant populations should be expected to respond in a non-linear manner to a range of variables if one has sampled a sufficient gradient length (Austin, 1976). The DCA Axis one eigenvalue can be used to determine the correct response model. Eigenvalues <0.2 imply the use of a linear model, 0.2-0.3 indicate either a linear or unimodal model and values >0.3 indicate a unimodal model is appropriate to explore the data. DCA works effectively where a species is recorded across much of the dataset and increases and decreases in abundance. Isolated occurrences of taxa may not return an accurate position on the gradient and may cause other taxa to cluster together if the rare taxa are ordinated at the extreme end of an axis. Rare taxa may therefore in cases violate the assumptions of the DCA, though this can be addressed by down-weighting their effect on the ordination or removing them entirely.

DCA has been used to indirectly reconstruct peatland environmental characteristics (Barber *et al*. 2000; Hughes *et al*. 2006; Langdon and Barber, 2005; Swindles *et al*. 2010). However, DCA is not always effective at detecting the environmental gradient one thinks is likely driving the system (Blundell and

Barber, 2005; Mauquoy *et al.* 2008) and it may not be the optimum ordination technique when applied to quantitative environmental reconstructions (e.g. bog surface wetness (BSW); Daley and Barber, 2012). DCA is criticised in that the resultant taxon distributions may be affected by the number of segments used when dividing the first axis (Ejrnæs, 2000; Tausch *et al.* 1995). In this way, proximal taxa may only be similar because they happen to be re-scaled in different segments during the initial detrending, hence their relative positions in the final ordination may change if a differing number of segments were used (Jackson and Somers, 1991).

3.5.2 non-Metric Multidimensional Scaling

non-Metric Multidimensional Scaling (nMDS) is a rank-based ordination method based on a dissimilarity matrix which can model multiple non-linear responses, rather than Gaussian as in DCA (Minchin, 1987). The impact of rare species is minimised in an nMDS because it is a rank-based approach and it can model alternative distance measures (e.g. Bray-Curtis dissimilarity) (Payne *et al.* 2015; Virtanen *et al.* 1999, 2006). The nMDS assesses the data against multiple dimensions, approximately equivalent to the axes used in a DCA. Subjectivity remains in the interpretation of nMDS plots as the technique passively places taxa and samples in ordination space. The output presents the dimension with the most variance as Axis one which represents the dominant gradient of environmental change across the dataset (Minchin, 1987).

nMDS presents objects according to their rank-dissimilarity, hence the magnitude of difference between objects is lost. The nMDS approach thereby differs from DCA bi-plots which show object distribution along axes according to SD units. nMDS assumes that maximising the rank dissimilarity and distances between objects is an accurate portrayal of between taxon/sample relationships (Legendre and Legendre, 2012). The ‘stress score’ of a model refers to the degree which every pairwise rank order dissimilarity value in the data matrix is preserved by the ordination rank distances, in that the lower the stress score the better the preservation of matrix dissimilarity values in the ordination (Kruskal, 1964). Stress scores >0.3 are typically deemed to represent an inaccurate model whilst scores below this threshold suggest the model is suitable for interpretation (Legendre and Legendre, 2012). Widespread use of nMDS has yet to occur in palaeoecology so it is hard to determine its suitability as an ordination technique relative to the more established DCA. Some do consider the nMDS a more statistically rigorous approach to quantitative peatland environmental reconstructions (Payne *et al*. 2015) whilst others find it replicates issues which occur during DCA tests (e.g. bi-partite divisions in Holocene length BSW reconstructions (Daley and Barber, 2012)). Interpretations of the relative suitability of each method for analysing contemporary ecological community data are also mixed. Differing authors consider either the DCA (Hill and Gauch, 1980; Ruokolainen and Salo, 2006) or nMDS (Manjarrés-Martínez *et al*. 2012; Minchin, 1987; Ramette, 2007) as more accurate in modelling taxa responses to latent environmental gradients.

3.5.3 Procrustes rotation

Robust comparisons of differing ordination outputs can be performed using Procrustes rotation (Simpson, 2013); this analysis is a method used to compare the multidimensional shape of ordinations. The rotation attempts to find a position of maximal superimposition by minimising the sum of squared differences between corresponding taxa (Peres-Neto and Jackson, 2001). The two tested datasets (i.e. DCA and nMDS outputs) are first re-scaled and centred to ensure a congruent orientation (Jackson, 2005). One re-configured dataset is set as reference whilst the other is iteratively rotated to attempt to find the minimal residual differences between equivalent points (Peres-Neto and Jackson, 2001). The resultant sum of squares and correlation statistic can be used to determine whether ordinations which are plotting on differing orientations and/or scales are ordinating sample and taxa data in a concordant manner. A low sum of squares value and significant correlation suggest the two ordinations are displaying similar directions of environmental change. In this way, ordinations of the same raw dataset which plot data according to inter-sample distances (e.g. DCA) and rank differences (e.g. nMDS) can be robustly compared.

3.5.4 The use of DCA and nMDS in this thesis

Both DCA and nMDS ordinations were used to investigate the dominant gradients of environmental change in the plant macrofossil datasets in this thesis. The use of multiple ordination techniques may improve the reliability of palaeoecological analyses as if the same taxa are positioned closely to each other in multiple ordinations it is likely that both techniques are detecting similar trajectories of change (Daley and Barber, 2012).

All palaeoecological statistical analyses were conducted in the ‘Vegan’ package (Oksanen *et al*. 2017) within the ‘R’ environment (R Core Team, 2012). Data were converted to presence/absence values for DCA, nMDS and/or PCA ordinations (Prentice, 1986) as five-point scale data (Section 3.4.5) were incompatible with the selected ordination methods (Mauquoy *et al*. 2010). Differing components of the same species were homogenized to a single taxon (e.g. *E. vaginatum* epidermis and spindles classed as *E. vaginatum*) to avoid differing components of the same species plotting at differing points within the ordination (Roland, 2012). Concerns have been raised regarding the overrepresentation of monocotyledon roots in macrofossil profiles (Roland, 2012) because of their high resistance to decay (Heal *et al*. 1978) and that they may grow into waterlogged substrate where decomposition is suppressed (Brown, 2006). Monocotyledon roots may also be younger than other material recorded at a given depth as roots penetrate older peat. No roots for any monocotyledon taxa were included in ordinations given these concerns. Ordinations were conducted with unidentifiable organic matter (UOM) included and excluded to explore the effect of this pseudo-taxon. UOM is produced through the degradation of plant remains and could represent any peatland vegetation type. Use of only presence/absence UOM data therefore means UOM is likely to display a multi-modal response to a range of environmental drivers, thereby violating the unimodal assumption of the DCA.

DCA and nMDS outputs are presented as bi-plots of Axes one and two taxa scores to assess for environmental gradients which may affect plant community composition. Detrending within DCAs followed the standard Oksanen *et al*. (2017) method using a five-segment smoothing window. The effect of rare species was minimised by down-weighting using a standard 1/5 fraction unless otherwise stated. Bray-Curtis dissimilarity was used as the nMDS distance metric as the technique does not assume shared absences in each sample equate to taxa with similar ecological requirements (Quinn and Keough, 2002). Comparisons between normalised DCA and nMDS outputs (Charman *et al*. 1999) were performed using Pearson Correlation Coefficients. DCA and nMDS outputs were also compared using the Procrustes rotation function within the ‘Vegan’ package (Oksanen *et al*. 2017) using permutation testing (*n* = 999).

Chapter 4 Site descriptions

4.1 Introduction

Site selection is a critical process if this thesis is to achieve its aim (Section 1.2) as cores must be recovered from peatlands which have been managed under a variety of land uses. This chapter presents the rationale underpinning site selection (Section 4.2) alongside a detailed description of each site which includes assessments of previous palaeoenvironmental research (Section 4.3). Use of terms ‘site’, ‘sub-site’ and ‘sampling point’ follow the rationale in Section 3.2. Sampling point codes are reported as ‘Site code – sub-site number – master/replicate indicator’ (e.g. BRP-1-M, BRP-1-REP-A, MIG-4-REP-D).

4.2 Site selection

The rationale behind site selection is underpinned by the aim of this thesis which is to characterise the impact of land management (burning, drainage, forestry and grazing) on carbon accumulation. Sites were selected according to their history of being managed by one of these land use types to address research question one (Section 2.8.1). It must be considered that recovered cores may not represent the effects of any one of these management types in isolation. The difficulty in identifying sites managed solely according to a single land use relates to research question two (Section 2.8.2) and is explored in later chapters.

Two sites were selected each from Mid- and North-Wales to explore geographic variation in blanket peatland responses to management. South-Wales (<52°N) was excluded as a study region given concerns that the degraded nature of South-Wales blanket peats (Section 4.3) may mean that records of environmental change may be insufficient to address the thesis aim. The number of master and replicate cores allocated to each site is detailed and justified in each study site’s description in Section 4.3.

Following from these design decisions, sites were selected which conformed to the following criteria:

1. Sites must be classified as blanket peatlands (UK Biodiversity Group, 1999; Section 1.4.1). There is no requirement for a sampling point to be

ombrotrophic given this thesis focuses on the development of the peatland itself rather than attempting to reconstruct climatic shifts.

2. Sampling points must have a peat coverage of at least 50cm so they may be classified as ‘deep peat’ according to current classifications (Section 2.2.1).
3. All sites must be located ≥250m asl to ensure sites are within ‘uplands’ as defined in land management decision making (Backshall, 2001; Backshall and Rebane, 2001; Holden *et al*. 2007a).
4. Where sites and sub-sites were specifically selected for their land use history, land management interventions must be known to a resolution of at least several decades. It can be difficult to resolve management histories, with the impact of this caveat explored in Chapter 8.

4.3 Location and description of study sites

The Welsh climate is a predominantly maritime one being wet and mild as the country is exposed to active Atlantic westerly weather systems making landfall (Met Office, 2016). Topographic variability induces substantial variation in climate characteristics between the uplands and lowlands. Welsh uplands are predisposed to the formation of blanket peat (Section 1.4.1) given the maritime climate causes annual precipitation for land ≥250m asl to exceed 1200 mm yr^{-1} (Yeo, 1997). Steep gradients restrict the formation of blanket peat in mountainous areas (e.g. the Carneddau, Glyderau and Snowdon mountain complexes). Blanket peat typically accumulates on the gentler sloping mountain complexes (e.g. the Berwyns) and upland plateaus (e.g. the Elenydd) that form part of the ‘central spine’ of the country (Yeo, 1997).

There are approximately 70,000 ha of blanket peat in Wales (Cris *et al*. 2011; JNCC, 2008) with most peat ≥50cm deep occurring in Mid- and North-Wales (>52°N) (Figure 4.1). Deep peat was previously more widespread in South-Wales, but mass erosion has completely removed some blanket peatland macrotopes, such as in parts of the Brecon Beacons (Reith, R. pers. comm. 2015). The relative lack of deep peat in South-Wales is in part due to human activity with pollution an important driver of peatland change (Yeo, 1997). Previous palaeoenvironmental research has addressed the degradation of South-Wales blanket peats (Chambers and Mauquoy, 1998). An issue of concern for Welsh peatland managers is the spread of the grass *Molinia caerulea* on South- and Mid-Welsh sites (Chambers and Daniell, 2011; Chambers *et al*. 2007a). The cause of this spread is debated.

The limited extent of *M. caerulea* in North-Wales may be related to the wetter and colder conditions relative to South- and Mid-Wales (Yeo, 1997). This argument does not centre on climate *per se* enabling *M. caerulea* expansion given its modelled climatic range spans the British Isles (Berry and Butt, 2001), but rather warmer temperatures permit the expansion of *M. caerulea* in combination with some other factor (e.g. firing, grazing pressure, nitrogen deposition) (Yeo, 1997).

The four designated sites were split equally between Mid- (Bryniau Pica and Plynlimmon) and North-Wales (Lake Vyrnwy and the Migneint) along a predominantly linear south-north transect (Figure 4.1). Reliable records of climatic change in the Mid- and North-Welsh uplands are complicated by a sparse network of weather stations located predominantly at altitudes ≤250m asl (Met Office, 2010). The exceptions are the Cwmystwyth (301m asl) and Lake Vyrnwy SAWS (360m asl) stations. In any case, it is uncertain if upland weather stations would be representative of climate at the study sites given spatial variability in Welsh upland climate characteristics (Yeo, 1997). The UK Climate Projections 2009 (UKCP09) gridded datasets provide an estimate for multiple climatic variables from AD 1910-2011 for each area of the British Isles according to a 5x5km grid (Met Office, 2017a). The data circumnavigate the issue of temporally variable and irregularly spaced weather stations and, of importance to this thesis, improve estimates of upland climate dynamics given difficulties with *in situ* monitoring (Jenkins *et al*. 2008). Monthly mean temperature and precipitation records (AD 1961-2010) were recovered for grids which overlay sites where cores were recovered (Table 4.1). Data were used to produce fifty-year average climatic characteristics for each study site (Figure 4.2; Table 4.2). Site descriptions of Bryniau Pica (Section 4.3.1), Lake Vyrnwy (Section 4.3.2), the Migneint (Section 4.3.3) and Plynlimmon (Section 4.3.4) are presented below.

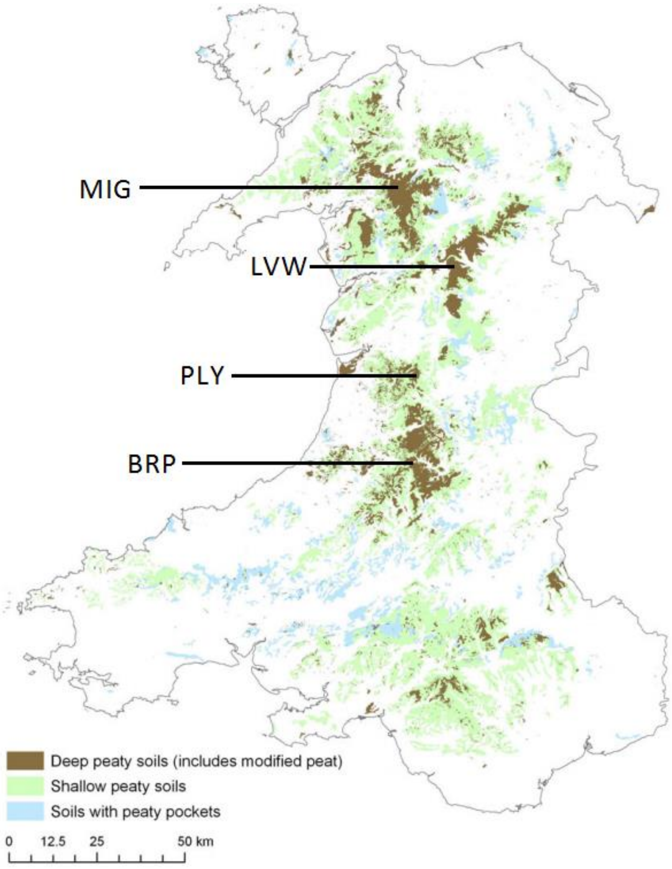


Figure 4.1: Location of study sites and their relation to the Welsh peatland resource. BRP= Bryniau Pica, LVW= Lake Vyrnwy, MIG= the Migneint and PLY= Plynlimmon. Base map source: Vanguelova *et al*. 2012.

Table 4.1: Easting (E) and northing (N) coordinates for the centre of each 5x5 km UKCP09 grid cell of interest in this study (Met Office, 2017b). Multiple grids were required for Lake Vyrwny and the Migneint given the geographical dispersion of cores at these sites (Sections 4.3.2 and 4.3.3 respectively).

Bryniau Pica		**Lake Vyrnwy**		**Migneint**		**Plynlimmon**	
E	**N**	**E**	**N**	**E**	**N**	**E**	**N**
282500	267500	292500	322500	277500	347500	282500	287500
		297500	322500	277500	342500		
				282500	347500		

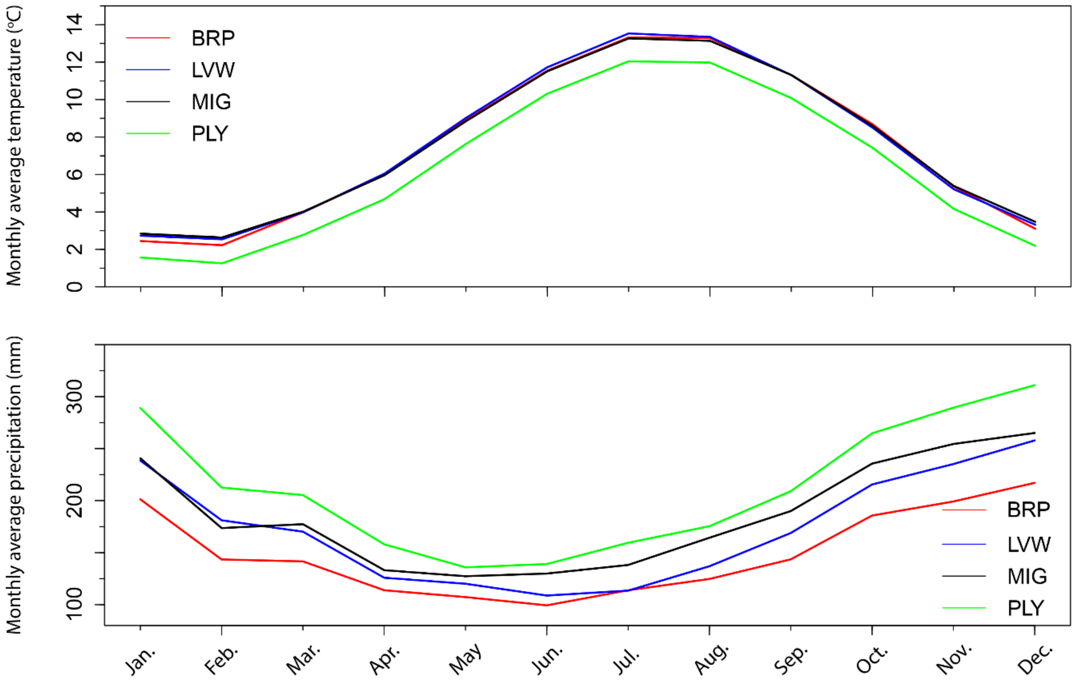


Figure 4.2: Modelled average AD 1961-2010 monthly temperature and precipitation data by site.

Table 4.2: Modelled average AD 1961-2010 climatic characteristics by site.

	Bryniau Pica	**Lake Vyrnwy**	**Migneint**	**Plynlimmon**
Average annual temperature (°C)	7.5	7.6	7.6	6.3
Coldest month (°C)	February (2.2)	February (2.5)	February (2.6)	February (1.3)
Warmest month (°C)	July (13.3)	July (13.5)	July (13.3)	July (12.0)
Average annual precipitation (mm)	1791.3	2072.9	2230.1	2549.5
Driest month (mm)	June (99.4)	June (108.8)	May (127.3)	May (135.9)
Wettest month (mm)	December (217.1)	December (257.9)	December (265.1)	December (311.1)

4.3.1 Bryniau Pica

BRP was selected for investigation because of the dominance of *M. caerulea* across the site (Figure 4.3), meaning BRP offers a valuable opportunity to explore the poorly understood effect of *M. caerulea* on peatland carbon accumulation. BRP is the only site within the 'South-Central moorland' biogeographical zone which Yeo (1997) viewed as important in representing the tendency for *M. caerulea* to dominate South-Wales peatlands. Whilst the annual temperature profile of BRP is not markedly different to the other sites it receives less annual precipitation (Figure 4.2; Table 4.2). *M. caerulea* expansion is a notable conservation concern elsewhere on the Elenydd Site of Special Scientific Interest (SSSI) with data from Drygarn Fawr *ca.* 8 km to the south-east indicating the genus began to increase in abundance *ca*. AD 1900 (Chambers *et al*. 2007b). The primary land use on the Elenydd is for sheep grazing with no substantial land use change having occurred at BRP over the last *ca*. fifty years save for fluctuations in stocking numbers (Baker, A. pers. comm. 2015). Records from the Elan Valley estate (Joyce, 2012) suggest mature sheep stocking fluctuated between 1-2.5 ewes ha^{-1} between AD 1878-2012 with peak stocking occurring during the AD 1980s and 1990s.

Sub-sites BRP-1 and BRP-2 were assigned to the main blanket bog complex (SN 8065, 8066) (Figure 4.3). Field walking demonstrated that *S.* s*. Sphagnum* patches persisted amongst *M. caerulea* tussocks in certain areas to the east of the bog where *M. caerulea* appeared less abundant (BRP-1). *M. caerulea* was more abundant to the west of the bog (BRP-2) with the *Sphagnum* patches smaller relative to those at BRP-1. The remaining sub-sites were assigned to shallower peat depths on the fringes of the main bog complex (BRP-3; BRP-4). No cores were recovered from actively degraded or eroding areas. Full BRP sampling point details are provided in Table 4.3.

Figure 4.3: Southward facing photo showing surface characteristics of main BRP watershed bog complex (licensed under Creative Commons, 2017).

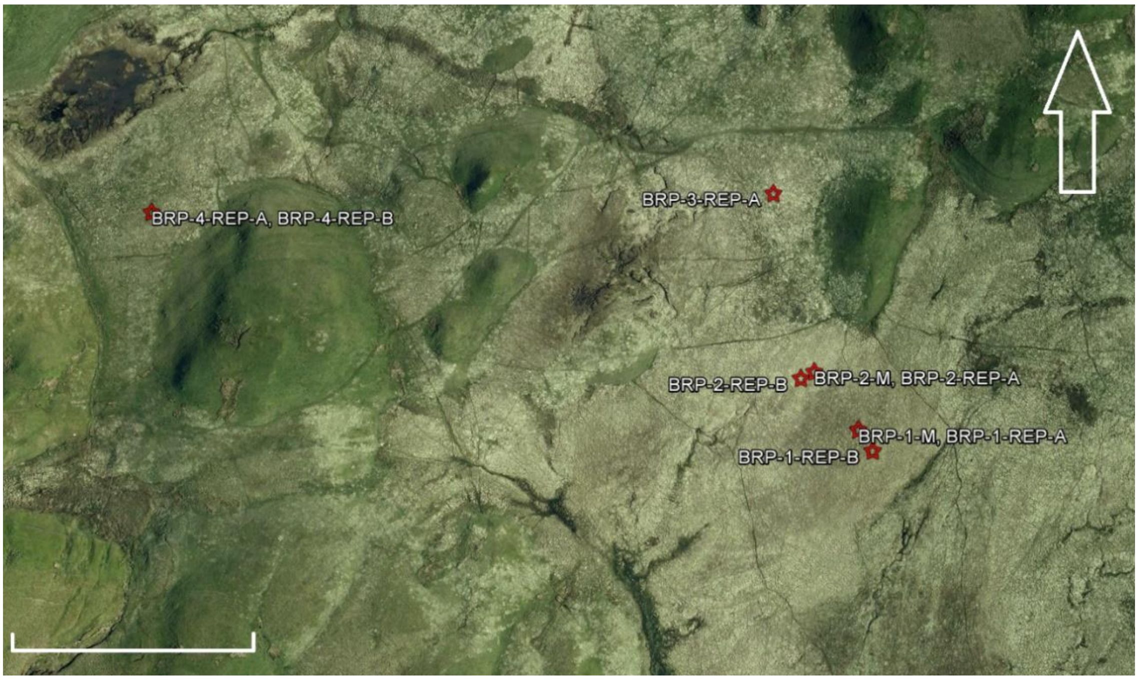


Figure 4.4: Aerial photograph of BRP sampling points (Table 4.3) (Google Earth- image date 31/5/2009). Scale bare indicates 200m, arrow indicates north.

Table 4.3: BRP sampling point details.

Sampling point	Latitude	Longitude	OS Grid Ref.	Altitude (m asl)	Surface vegetation
BRP-1-M	52.28033	-3.75556	SN 805 659	462	*S.* s. *Sphagnum* patch surrounded by *Molinia caerulea*.
BRP-1-REP-A	52.28031	-3.75547	SN 805 659	462	*S.* s. *Sphagnum* patch surrounded by *M. caerulea*.
BRP-1-REP-B	52.28014	-3.75531	SN 805 658	462	*S.* s. *Sphagnum* patch surrounded by *M. caerulea*. Some *Empetrum nigrum* in vicinity.
BRP-2-M	52.28078	-3.75609	SN 804 660	463	*S.* s. *Sphagnum* and *S.* s. *Acutifolia* patch intermixed with *M. caerulea*.
BRP-2-REP-A	52.28078	-3.75608	SN 804 660	463	*S.* s. *Sphagnum* and *S.* s. *Acutifolia* patch intermixed with *M. caerulea*.
BRP-2-REP-B	52.28072	-3.75626	SN 803 660	463	*Hypnum cupressiforme* layer proximal to *M. caerulea* tussock.
BRP-3-REP-A	52.28222	-3.75667	SN 803 663	469	*S.* s. *Acutifolia* and mixed brown moss patch surrounded by *Eriophorum vaginatum* and *M. caerulea*.

Table 4.3 (cont.)

BRP-4-REP-A	52.28200	-3.76514	SN 796 662	453	*S.* s. *Acutifolia* patch with *M. caerulea* and *Scirpus cespitosus*. *Vaccinium myrtillus* nearby.
BRP-4-REP-B	52.28197	-3.76508	SN 796 662	452	*S.* s. *Acutifolia* patch with *M. caerulea* and *S. cespitosus*. *V. myrtillus* nearby.

One previously published palaeoenvironmental record is available from BRP. Buckley and Walker (2001) recovered a 4.6m peat core close to BRP-2. They presented pollen and charcoal evidence which suggest Mesolithic people were present *ca*. 6,650 BC. Data suggest the site was actively used for cattle and sheep grazing during the 12th Century by Cistercian monks from the Strata Florida abbey *ca*. 6 km to the west. Data from a lake core from Llyn Gynon *ca.* 1.2km to the south-west suggest *M. caerulea* expansion and a decline in *Calluna* occurred *ca*. AD 1930 (Stevenson and Thompson, 1993). Another core from the same study at Llyn Hir *ca.* 2km to the north-west of BRP suggests the *C. vulgaris* decline occurred in the early 18th Century. Other peat cores from the Elenydd include Copa Hill *ca.* 10km north of BRP (Mighall and Chambers, 1993), the north shore of Llyn Gynon (Moore and Chater, 1969) and Esgair Nantybeddau and Pwll-nant-ddu, *ca*. 4 and 6.5km to the north-east of BRP respectively (Wiltshire and Moore, 1983). No absolute dates are provided in any of these studies, hence their inferences regarding environmental change are uncertain.

4.3.2 Lake Vyrnwy

The uplands which surround Lake Vyrnwy (LVW) form part of the Berwyn and South Clwyd Mountains Special Area of Conservation (SAC), the largest blanket peatland complex in Wales (CCW, 2008a). The LVW catchment was selected for investigation because of its historic use as a grouse estate from *ca*. AD 1800-1975 (Walker, M. pers. comm. 2015). Sheep grazing still occurred during this period with an expansion in stocking post-AD 1900 (Wilson *et al*. 2011b). Parts of the catchment were drained during the AD 1960s and 1970s at an average spacing of 45.1 ± 5.5m (Wilson *et al*. 2011a). Burning has been banned on the estate since AD 2003 with heather currently managed by cutting patterns which are visible in aerial imagery (Figure 4.5 and Figure 4.6). Current average sheep stocking within the catchment is <0.5 ewes ha^{-1} (Wilson *et al*. 2011a). *C. vulgaris* is abundant across much of the estate alongside *E. vaginatum*, with the *Calluna* dominance meaning large swathes of the blanket peat are considered in poor ecological condition (CCW, 2008a). *Calluna* typically dominates on the water shedding ridges (Figure 4.7) within the catchment whilst the slopes and saddles are more dominated by graminoids and *Sphagnum*. It is uncertain to what degree the effects of drainage, firing and/or grazing can be separated when attempting to explain contemporary *Calluna* dominance.

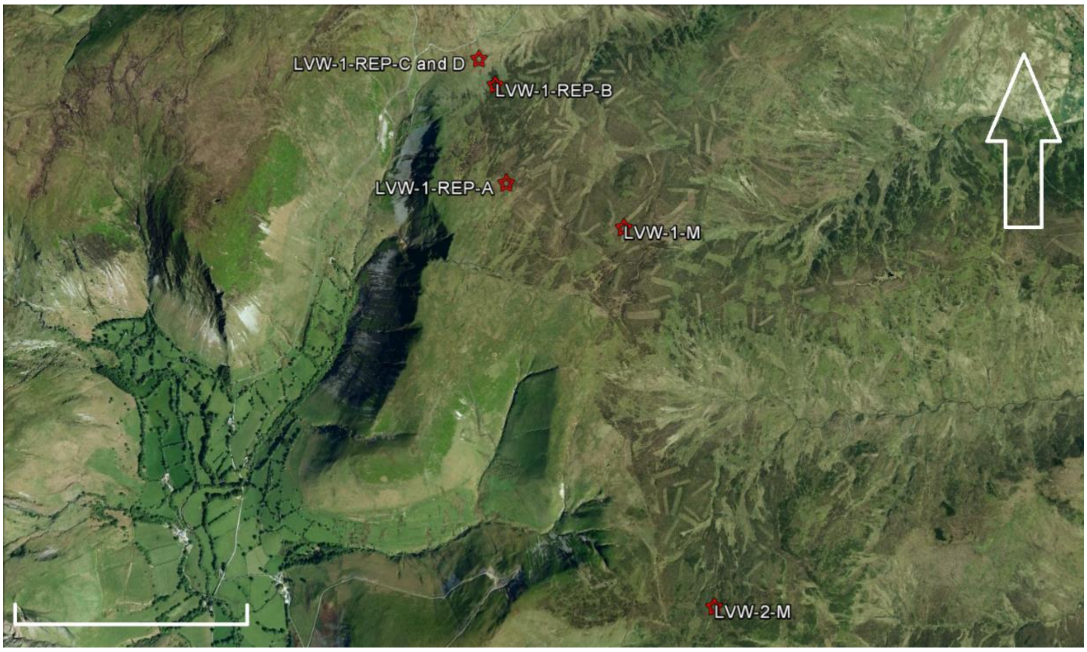


Figure 4.5: Aerial photograph of LVW sampling points within ‘Grid 1’ (Google Earth- image date 20/4/2009). Scale bare indicates 1,000m, arrow indicates north.

Figure 4.6: Aerial photograph of LVW sampling points within ‘Grid 2’ (Google Earth- image date 20/4/2009). Scale bare indicates 500m, arrow indicates north.

Figure 4.7: Northward facing photo from LVW-2 showing surface characteristics of LVW catchment.

Sub-sites LVW-1, LVW-2 and LVW-3 were located on water shedding ridges in three different LVW sub-catchments (the Eunant, North Hirddu and Nadroedd respectively) to record some of the variability in environmental change at differing points within the grouse estate. Four replicate cores were recovered from the LVW-1 sub-site to reflect the vegetation gradient from the ridge down into the saddles where *Sphagnum* patches became larger and *Calluna* was less abundant (Figure 4.5). LVW-2-M exhibited similar vegetation gradients to those at LVW-1-M. No replicates were recovered from LVW-2-M given visible off-road vehicle marks near the sub-site may have disturbed the surrounding peat. There were more topographical differences in the peatland at LVW-3-M relative to the other two sub-sites. Hummocky bedrock ridges and swales may mean peat is controlled by bedrock topography and not be sensitive to wider environmental changes. Local recovery of *Sphagnum* following ditch blocking activity (Walker, M. pers. comm. 2016) may suggest that peats at LVW-3-M are capable of recording land use. No replicates were recovered from LVW-3-M given uncertainty regarding morphometric controls on peatland development. Full LVW sampling point details are provided in Table 4.5. The average climatic characteristics of LVW (Figure 4.2; Table 4.2) were determined using two UKCP09 grid cells given the *ca*. 6.5km dispersion between western and eastern sub-sites. ‘Grid one’ refers to LVW-1 and LVW-2 whilst ‘grid two’ refers to LVW-3 (Figure 4.8; Table 4.4). Grid one has a cooler annual temperature and greater annual precipitation than grid two. The 5x5km UKCP09 grid cell for LVW-3-M encompasses most of the LVW reservoir (Met Office, 2017b). The water body and elevation range within the cell (*ca*. 250-588 m asl) may therefore be affecting modelled temperature and precipitation at LVW-3-M (*sensu* Prein and Gobiet, 2017). As such, it is uncertain whether LVW-1 and LVW-2 are subject to a differing climatic regime to LVW-3-M or whether the difference is an artefact of the UKCP09 dataset.

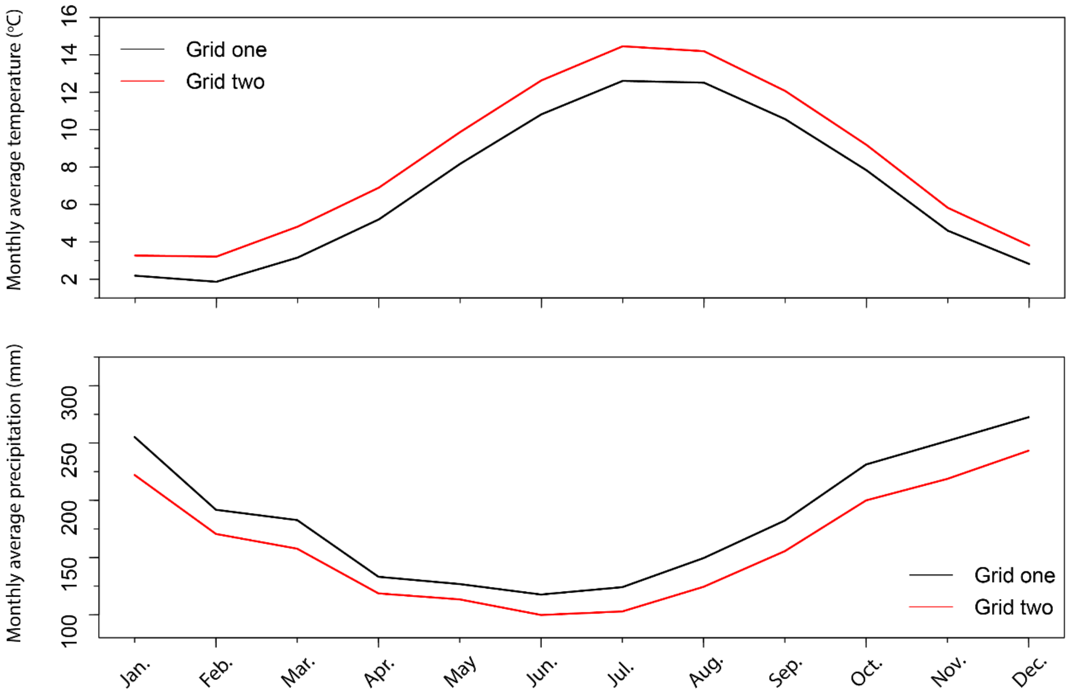


Figure 4.8: Modelled average LVW AD1961-2010 monthly temperature and precipitation data by UKCP09 grid.

Table 4.4: Summary of modelled average LVW AD 1961-2010 climatic characteristics by UKCP09 grid.

	Grid one	**Grid two**
Average annual temperature (°C)	6.9	8.4
Coldest month (°C)	February (1.9)	February (3.2)
Warmest month (°C)	July (12.6)	July (14.5)
Average annual precipitation (mm)	2218.7	1927.1
Driest month (mm)	June (117.7)	June (99.8)
Wettest month (mm)	December (272.6)	December (243.3)

Table 4.5: LVW sampling point details.

Sampling point	Latitude	Longitude	OS Grid Ref.	Altitude (m asl)	Surface vegetation
LVW-1-M	52.78508	-3.59683	SH 925 221	546	*S.* s. *Acutifolia* patch amongst dense *C. vulgaris* and *E. vaginatum* stand.
LVW-1-REP-A	52.78663	3.604917	SH 919 223	531	*S.* s. *Acutifolia* patch amongst dense *C. vulgaris* and *E. vaginatum* stand.
LVW-1-REP-B	52.79077	-3.60597	SH 920 228	493	Mixed brown moss patch amongst dense *C. vulgaris* and *V. myrtillus* stand.
LVW-1-REP-C	52.79181	3.607222	SH 918 228	484	Mixed *Sphagnum* lawn with *Eriophorum angustifolium* and *vaginatum*.
LVW-1-REP-D	52.79211	-3.60674	SH 919 229	482	*S.* s. *Sphagnum* lawn amongst *E. vaginatum*, *Erica tetralix*, *M. caerulea* and *Narthecium ossifragum*.
LVW-2-M	52.76997	-3.58948	SH 932 205	565	*S.* s. *Acutifolia* patch amongst dense *C. vulgaris* and *E. vaginatum* stand.

Table 4.5 (cont.)

LVW-3-M	52.79786	-3.50535	SH 986 234	498	*S.* s. *Acutifolia* patch amongst mixed *C. vulgaris*, *E. tetralix*, *E. vaginatum* and *V. myrtillus*.

No previous palaeoenvironmental investigations have been identified from the LVW catchment. The closest study is from a largely unpublished PhD thesis from the main Berwyn ridge *ca*. 12km to the north-east of LVW-3-M (Bostock, 1980). Portions of the data were published in Tallis (1995), though only the original source is considered here. Bostock (1980) assessed several peat cores and eroded faces which demonstrated a mixed vegetation history for the Moel Sych and Y Godor localities. Charcoal evidence suggests humans have been active in the Berwyn mountains since *ca.* 4,000 BC. Intense burning and grazing *ca*. AD 1215 (charcoal and pastoral pollen) coincided with a decline in *Sphagnum* spore abundance and its displacement from several macrofossil assemblages. *Calluna* has been present in the Berwyns since *ca*. 2,000 BC and has demonstrated several non-synchronous phases of decline and expansion (Bostock, 1980). Changes in apparent *Calluna* presence/abundance are not always related to changes in microscopic charcoal frequency. The assertion that *Calluna* dominance at LVW is likely related to late 19th-20th Century impacts of land management (CCW, 2008a) may therefore be misplaced.

4.3.3 The Migneint

The Migneint (MIG) is the second largest blanket peatland in Wales after the Berwyn and South Clwyd complex and was selected for investigation given all four land use types of interest have been practised across the site. All sub-sites were located within the 19,966 ha Migneint-Arenig-Dduallt SAC/Special Protection Area (SPA) (CCW, 2008b). The largest body of land within the SAC is the Ysbyty estate which was managed for both grouse shooting and sheep farming (Roberts, A. pers. comm. 2017) prior to the land being awarded to the National Trust (2017) in AD 1951. Drainage ditches were installed across much of the MIG *ca*. AD 1918-1939 and a more limited number were installed in the AD 1970s. Afforestation occurred on parts of the MIG as part of the mass planting of the uplands between

Betws-y-Coed and Dolgellau during the mid-AD 1960s to 1970s (Atherden, 1992). Numbers of sheep have declined within the Ysbyty estate from the relatively high levels of the late AD 1980s (Roberts, A. pers. comm. 2015).

There is notable variability in surface vegetation between sub-sites. *Sphagnum* lawns and some hummock-hollow topography persist on parts of the peatland whilst areas which have been drained are often dominated by *Calluna* (Figure 4.9). Most replicate cores in this thesis were assigned to the MIG because of the number of differing management histories initial research suggested had occurred across the site. MIG-1 was recovered from a relatively intact *Sphagnum* lawn at the head of the Nant y Brwyn catchment (Figure 4.10). MIG-2 was assigned to investigate the impacts of forestry at Blaenycwm with the master core recovered *ca*. 40m outside the forest. The master core for MIG-3 was recovered from a *Sphagnum* lawn to the west of Lyn Serw to contrast against drained replicates recovered from the eastern side (Figure 4.11). MIG-4 sampling points were on drained peat to the east of the Llyn Serw drains whilst MIG-5 sampling points were on drained peat to the east of Llyn Conwy. MIG-6 was assigned to a tenanted holding at Cefngwyn which was dominated by *M. caerulea* (Figure 4.12). Full MIG sampling point details are provided in Table 4.7.

Average climate characteristics for the MIG (Figure 4.2; Table 4.2) were calculated using three UKCP09 grid cells. ‘Grid one’ refers to MIG-1 and MIG-4, ‘grid two’ to MIG-2, MIG-2B, MIG-3 and MIG-4 and ‘grid three’ to MIG-6 (Figure 4.13; Table 4.6). The climatic regime at grid three is warmer and drier than grids one or two. The difference may be due to the lower altitude of MIG-6 which is *ca*. 85m asl lower than the next highest sub-site (Table 4.7). It is interesting to note the warmest and driest sub-site is the one dominated by *M. caerulea* given Yeo’s (1997) assertion that climatic factors may be responsible for the dominance of *Molinia*. Annual rainfall for grids one and two (2347.2 and 2246.9 mm yr^{-1} respectively) is within previously published MIG rainfall ranges (2200-2400 mm yr^{-1}) (Peacock *et al*. 2013). January and July temperatures for grids one and two are similar to the 2.2 and 12.8°C respective values reported by Ellis and Tallis (2001).

Several previous palaeoenvironmental studies were identified from the MIG. Blackford (1990) presented charcoal evidence from an eroding peat complex (SH 7642) *ca*. 1.2km west of MIG-3 which suggested human activity in the region occurred from *ca*. 1,800 BC. Ellis and Tallis (2001) presented a record from an eroding peat complex (SH 7744) *ca*. 2km west of Llyn Conwy. Their main conclusions were that burning increased in intensity *ca*. AD 600 and a subsequent increase in sheep grazing occurred *ca*. AD 1750, both of which may have contributed to the development of the erosion complex. Human activity post *ca*. AD 0 is also supported by a core from a valley mire at Bryn y Castell (SH 7242) *ca.* 5km west of MIG-2 (Mighall and Chambers, 1995). A final palaeoenvironmental record from the MIG is a lake core recovered from Llyn Conwy. Stevenson and Rhodes (2000) suggest firing during the last *ca*. 300-400 years has caused a decline in *Calluna* alongside an increase in *Poaceae*. Stevenson and Rhodes (2000) present the data as a ratio of *Calluna/Poaceae* pollen percentage abundance. The *Calluna* 'decline' may therefore be a methodological artefact related to *Poaceae* pollen productivity masking true landscape changes. This point is pertinent given field walking performed across the MIG suggests large swathes of *Calluna* persist at present.

Figure 4.9: Assorted images demonstrating variability in MIG surface vegetation. Clockwise from top left, afforested peat at Blaenycwm (MIG-2B), drained *C. vulgaris* peat to the east of Lyn Serw (MIG-4), *M. caerulea* dominated valleyside mire (MIG-6) and drained *C. vulgaris* peat to the east of Lyn Conwy (MIG-5).

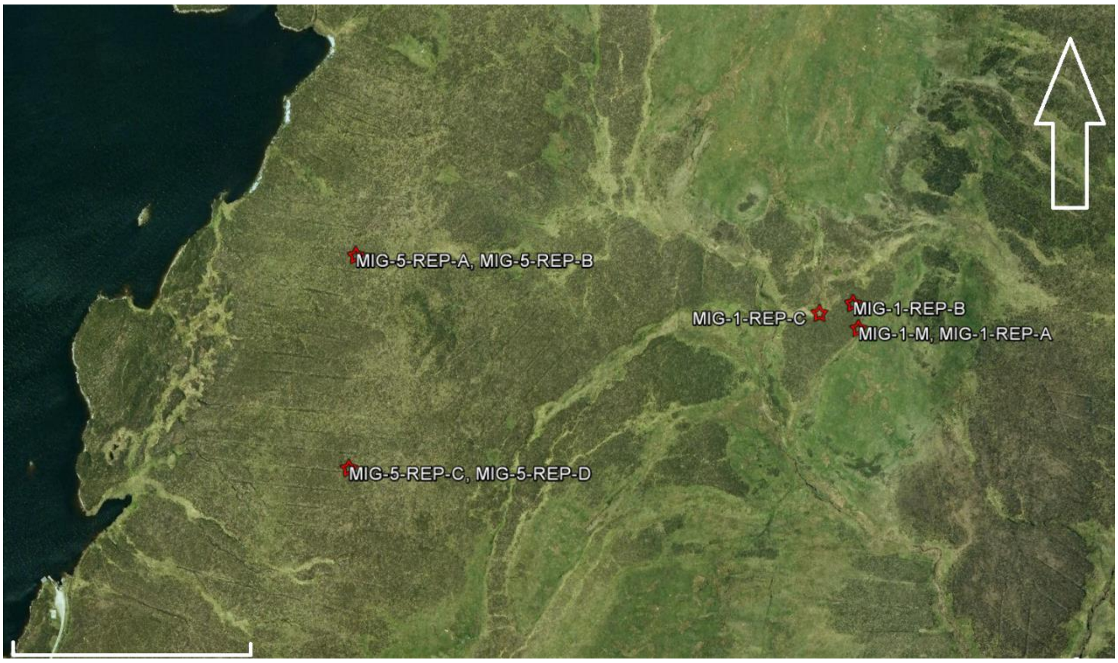


Figure 4.10: Aerial photograph of MIG sampling points within ‘Grid 1’ (Google Earth- image date 20/4/2009). Scale bare indicates 250m, arrow indicates north.

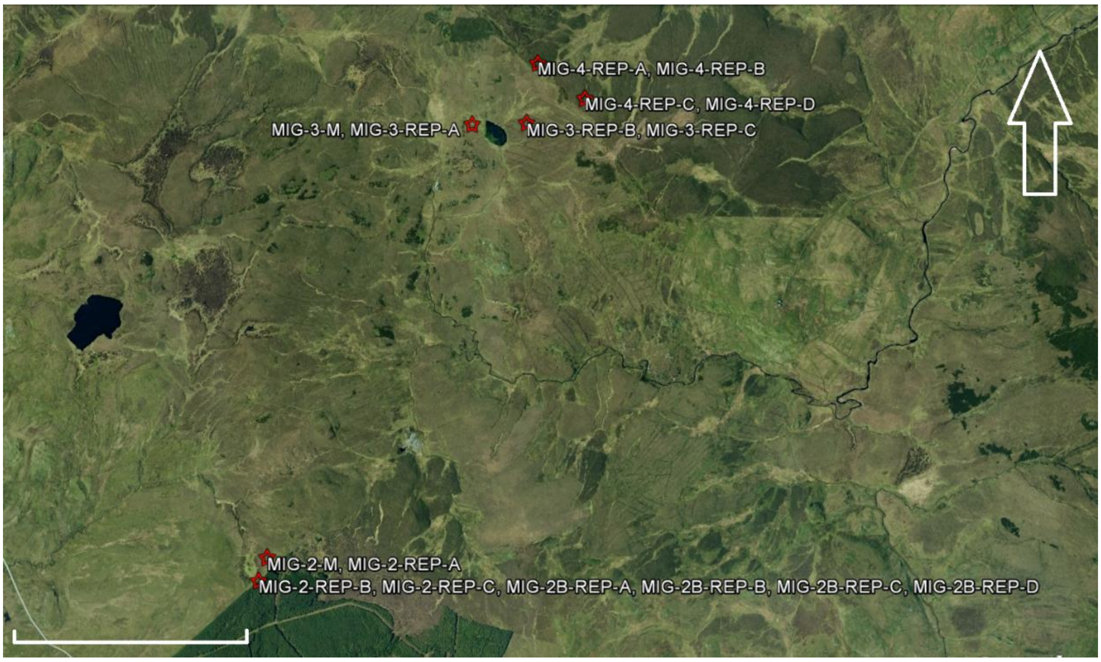


Figure 4.11: Aerial photograph of MIG sampling points within ‘Grid 2’ (Google Earth- image date 20/4/2009). Scale bare indicates 1,000m, arrow indicates north.

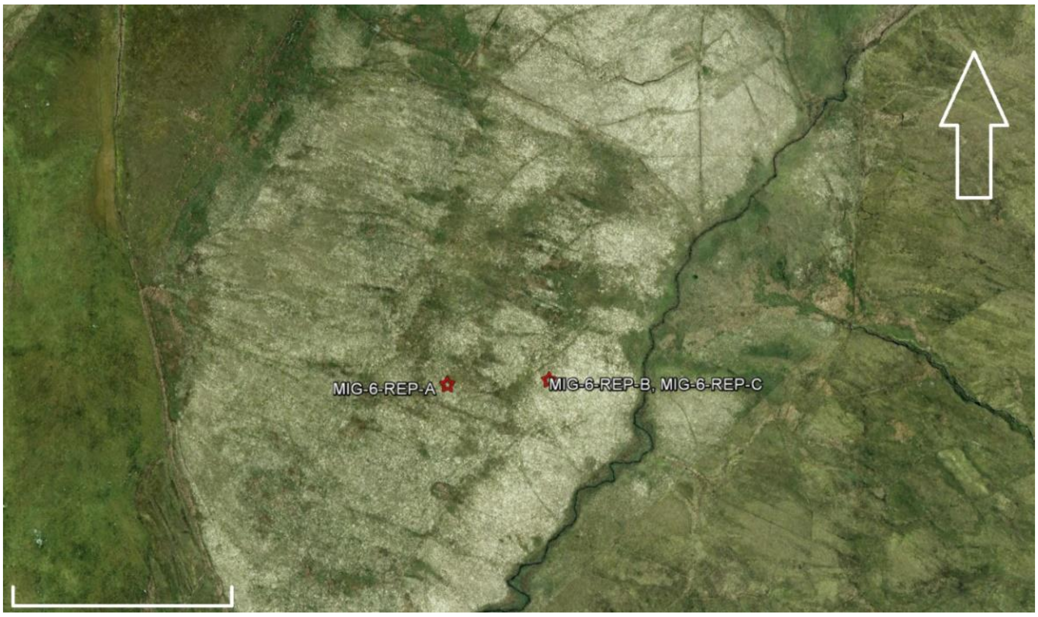


Figure 4.12: Aerial photograph of MIG sampling points within ‘Grid 3’ (Google Earth- image date 20/4/2009). Scale bare indicates 250m, arrow indicates north.

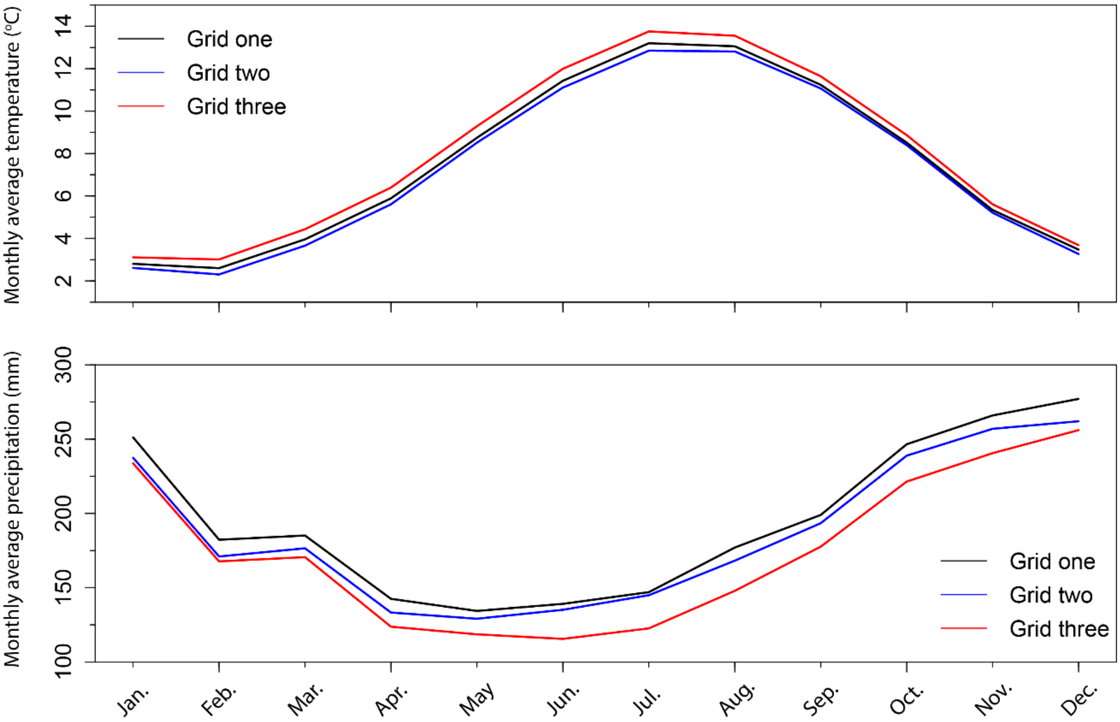


Figure 4.13: Modelled average MIG AD 1961-2010 monthly temperature and precipitation data by UKCP09 grid.

Table 4.6: Summary of modelled average MIG AD 1961-2010 climatic characteristics by UKCP09 grid.

	Grid one	Grid two	Grid three
Average annual temperature (°C)	7.5	7.3	7.9
Coldest month (°C)	February (2.6)	February (2.3)	February (3.0)
Warmest month (°C)	July (13.2)	July (12.8)	July (13.8)
Average annual precipitation (mm)	2347.2	2246.9	2096.1
Driest month (mm)	May (134.3)	May (129.1)	June (115.5)
Wettest month (mm)	December (277.1)	December (262.0)	December (256.1)

Table 4.7: MIG sampling point details.

Sampling point	Latitude	Longitude	OS Grid Ref.	Altitude (m asl)	Surface vegetation
MIG-1-M	52.997583	-3.804028	SH 790 461	452	*Sphagnum* lawn with *E. angustifolium* and *E. tetralix*.
MIG-1-REP-A	52.997664	-3.804061	SH 790 461	452	*Sphagnum* lawn with *E. vaginatum* and *Polytrichum*.

Table 4.7 (cont.)

MIG-1-REP-B	52.99788	-3.80414	SH 788 465	453	*S.* s. *Acutifolia* patch amongst *C. vulgaris* and *S. cespitosus*.
MIG-1-REP-C	52.99775	-3.80469	SH 788 464	453	*Sphagnum* lawn with *E. vaginatum* and *Polytrichum*.
MIG-2-M	52.95221	-3.83216	SH 770 411	489	*S.* s. *Acutifolia* patch amongst *C. vulgaris* and *E. vaginatum*.
MIG-2-REP-A	52.95235	-3.83217	SH 770 412	489	*S.* s. *Acutifolia* patch amongst *C. vulgaris* and *E. vaginatum*.
MIG-2-REP-B	52.95165	-3.83208	SH 770 410	489	Brown moss dominated patch in a furrow in the forestry.
MIG-2-REP-C	52.95165	-3.83208	SH 770 410	489	Brown moss dominated patch in a furrow in the forestry.
MIG-2B-REP-A	52.95149	-3.83263	SH 769 409	488	Mixed *S.* s. *Acutifolia* and brown moss patch on a ridge in the forestry.
MIG-2B-REP-B	52.95141	-3.83247	SH 769 409	488	Mixed *S.* s. *Acutifolia* and brown moss patch on a forestry ridge.
MIG-2B-REP-C	52.95142	-3.83226	SH 770 408	489	Brown moss dominated patch on a forestry ridge.

Table 4.7 (cont.)

MIG-2B-REP-D	52.95153	-3.83208	SH 770 400	489	Brown moss dominated patch in a furrow in the forestry.
MIG-3-M	52.96916	-3.82015	SH 779 429	454	*Sphagnum* lawn amongst *E. vaginatum* and *E. tetralix*.
MIG-3-REP-A	52.96918	-3.82015	SH 779 429	454	*Sphagnum* lawn amongst *E. vaginatum* and *E. tetralix*.
MIG-3-REP-B	52.9693	-3.81665	SH 781 429	459	*S.* s. *Acutifolia* patch amongst *C. vulgaris* and *E. vaginatum*.
MIG-3-REP-C	52.96923	-3.81625	SH 781 429	460	*S.* s. *Acutifolia* patch amongst *C. vulgaris*, *E. vaginatum* and *S. cespitosus*.
MIG-4-REP-A	52.97166	-3.81606	SH 783 432	461	*S.* s. *Acutifolia* patch amongst dense *C. vulgaris* canopy. Also *E. nigrum*, *E. vaginatum* and *S. cespitosus*.
MIG-4-REP-B	52.97162	-3.81601	SH 783 432	461	*S.* s. *Acutifolia* patch amongst dense *C. vulgaris* canopy. *Polytrichum* also within patch and *V. myrtillus* in vicinity.

Table 4.7 (cont.)

MIG-4-REP-C	52.97029	-3.81293	SH 785 431	465	*S.* s. *Acutifolia* and brown moss patch amongst *C. vulgaris*. *E. vaginatum* and *V. myrtillus.*
MIG-4-REP-D	52.97028	-3.81295	SH 785 431	465	*S.* s. *Acutifolia* patch amongst dense *C. vulgaris* and *E. vaginatum* stand.
MIG-5-REP-A	52.99816	-3.8119	SH 785 461	468	Mixed *S.* s. *Acutifolia* and brown moss patch amongst dense *C. vulgaris* canopy.
MIG-5-REP-B	52.99816	-3.8118	SH 785 461	468	Mixed *S.* s. *Acutifolia* and brown moss patch within *C. vulgaris* and *E. vaginatum* stand.
MIG-5-REP-C	52.99617	-3.81191	SH 785 458	473	Mixed *S.* s. *Acutifolia* and brown moss patch within *C. vulgaris* and *E. vaginatum* stand.
MIG-5-REP-D	52.99631	-3.81186	SH 785 458	473	Mixed *S.* s. *Acutifolia* and brown moss patch within *C. vulgaris* and *E. vaginatum* stand.
MIG-6-REP-A	52.9888	-3.7421	SH 831 449	368	Mixed brown moss patch proximal to *M. caerulea* tussock.

Table 4.7 (cont.)

MIG-6-REP-B	52.98881	-3.74039	SH 833 449	358	*S.* s. *Acutifolia* patch amongst dense *M. caerulea* coverage.
MIG-6-REP-C	52.98879	-3.74019	SH 833 449	358	Mixed *Sphagnum* patch amongst dense *M. caerulea* coverage.

4.3.4 Plynlimmon

Plynlimmon (PLY) (*Pumlumon* in Welsh) was selected for investigation given its history of grazing and the many eroded hag and gully systems that characterise the water shedding bogs (Yeo, 1997). The primary land use over the past several hundred years has been grazing and burning to control vegetation for sheep (Francis, 1990; Newson, 1976). Average stocking across PLY from AD 1902-2010 was estimated at 2.17 ewes ha^{-1} (Joyce, 2012). Agri-environment schemes mean average stocking is now *ca.* 0.5-0.7 ewes ha^{-1}. The east of the PLY massif is dominated by the Hafren forest plantation. Most planting occurred on mineral soils and peat <50cm deep (Newson, 1976), hence the summit blanket bog macrotopes are predominantly unaffected by afforestation. The modelled climate for PLY shows the site is cooler and subject to greater precipitation than other sites (Figure 4.2; Table 4.2). Modelled annual precipitation (2549.5 mm yr^{-1}) is in agreement with values reported by Hill and Neal (1997) within the Hafren forest (2518 mm yr^{-1}). Modelled average annual temperature (6.3°C) is cooler than Hill and Neal (1997) (7.3°C), which may be an effect of the sheltered nature of their weather stations within the Hafren forest relative to the exposed PLY summit.

Sub-sites PLY-1 and PLY-2 were located on peat near the sources of the Rivers Wye (SN 8086) and Severn (SN 8290) respectively. Peat at both sites exhibited secondary order erosion complexes (*sensu* Lindsay, 1995; Figure 2.12) with secondary acrotelm growing in between elevated hags (Figure 4.14). Vegetation at PLY-1 was characterised by *S.* s. *Acutifolia* patches amongst *E. vaginatum* tussocks. No replicates were recovered at this sub-site given the relative constancy of vegetation across the hags and the surrounding hillslopes were predominantly acid grassland (Figure 4.15). The erosion complex at PLY-2 was larger in extent than PLY-1 with stunted *C. vulgaris* growth on the tops of many hags. Peat close to the Severn Way walkway had eroded >1m from the top of the surrounding hags and no secondary acrotelm was visible. Cores were recovered away from the path to ensure that regular trampling by humans would not have affected the records. Five replicates were placed around the PLY-2 summit plateau according to observed variation in vegetation. Some *Sphagnum* lawns persisted away from the more intensely hagged areas of the summit whilst *Sphagnum* was near absent from some very dry areas which were dominated by *Calluna* (Figure 4.16). Full PLY sampling point details, including details of the vegetation cover where each replicate was recovered from, are provided in Table 4.8.

Figure 4.14: Images detailing surface characteristics at PLY sub-sites. Top photo taken looking east towards the PLY-1 saddle mire erosion complex, lower photo taken looking north from PLY-2 at the surrounding watershed mire erosion complex.

Figure 4.15: Aerial photograph of PLY sampling point PLY-1-M (Google Earth- image date 31/5/2009). Scale bare indicates 500m, arrow indicates north.

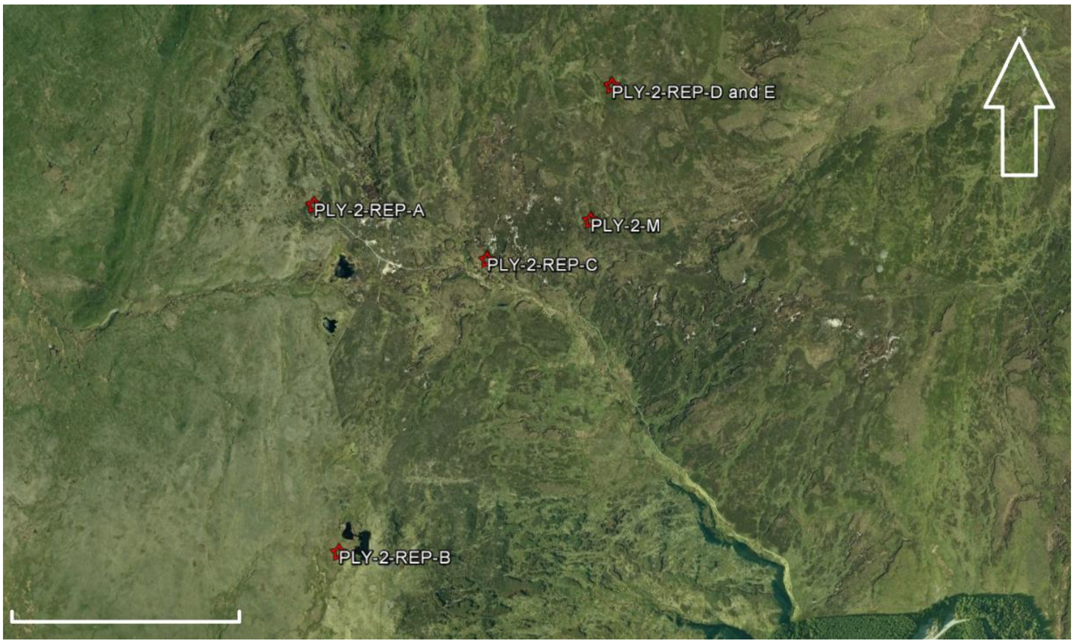


Figure 4.16: Aerial photograph of PLY sampling points at the source of the River Severn (Google Earth- image date 31/5/2009). Scale bare indicates 500m, arrow indicates north.

Table 4.8: PLY sampling point details.

Sampling point	**Latitude**	**Longitude**	**OS Grid Ref.**	**Altitude (m asl)**	**Surface vegetation**
PLY-1-M	52.46427	-3.75421	SN 809 865	540	*S.* s. *Acutifolia* patch with abundant *E. vaginatum*.
PLY-2-M	52.49543	-3.7332	SN 824 900	614	Mixed *S.* s. *Acutifolia* and brown moss patch amongst abundant *C. vulgaris* and *E. vaginatum*.
PLY-2-REP-A	52.49583	-3.74139	SN 818 901	613	Mixed brown moss patch surrounded by mixed *E. vaginatum*, *C vulgaris* and *S. cespitosus* assemblage.
PLY-2-REP-B	52.48925	-3.740833	SN 819 893	629	*S. papillosum* and *E. angustifolium* patch.
PLY-2-REP-C	52.49472	-3.73639	SN 822 899	610	Mixed *Sphagnum* lawn with *E. angustifolium* and *C. vulgaris*.
PLY-2-REP-D	52.498	-3.732639	SN 824 903	620	Mixed *Sphagnum* lawn with *E. angustifolium* and *C. vulgaris*.
PLY-2-REP-E	52.4985	-3.731056	SN 824 904	620	*P. schreberi* patch amongst dense *C. vulgaris* and *E. vaginatum*.

Two previous palaeoenvironmental records have been recovered from PLY. Moore (1968) presented pollen data from a core from SN 7985 *ca*. 1.3km south-west of PLY-1. He identified evidence of Neolithic deforestation, with more intense regional deforestation and grazing occurring during the Bronze Age and Roman periods. Dates for these changes were determined biostratigraphically. Mighall *et al*. (2013) recovered a core from SN 8289 *ca*. 500m south-east of PLY-2-M. Pollen and microscopic charcoal evidence suggests humans have been active at Plynlimmon since *ca*. 1,300 BC. Fungal spore evidence (*Sordaria*, *Sporormiella*) *ca*. AD 1515 is circumstantially linked to the documented increase in Welsh upland grazing intensity associated with monastic influence. Mighall *et al*. (2013) provide a radiocarbon dating framework and hence offer more robust timings of environmental change relative to Moore (1968). Mighall (2013) subsequently applied a ^{210}Pb model and additional pollen analyses to the same core. The refined chronology did not change any of the key conclusions of Mighall *et al*. (2013), though the additional pollen data indicates an increase in *Calluna* post *ca*. AD 1800 occurred alongside increased fire frequency and a decline in *Sphagnum* pollen (Mighall, 2013).

Chapter 5 Chronology

5.1 Introduction

Chronological techniques can be used to determine dateable horizons within a peat core. These horizons contextualise environmental changes recorded in cores and allow the rate of peat accumulation to be calculated. Peat accumulation rate affects carbon accumulation rate (Section 3.4.2). Accurately dating peat accumulation is therefore key for this thesis to achieve its aim of characterising the impact of land management on carbon accumulation.

The primary chronological consideration for this thesis was to use techniques which can accurately date peat which has accumulated over the last *ca*. 150 years, because this is the period of interest. There are concerns over the reliability of radiocarbon dating over this period as atmospheric ^{14}C has been diluted by industrial and nuclear activity (Le Roux and Marshall, 2010; Turetsky, 2004). As such, ^{210}Pb, ^{137}Cs and ^{241}Am analyses alongside Spheroidal Carbonaceous Particles (SCPs) were used to provide the necessary chronological control for this study (Sections 3.4.3 and 3.4.4). This chapter presents the results obtained from radiometric ^{210}Pb, ^{137}Cs and ^{241}Am analysis on the ten master cores, alongside the results of SCP analysis on all recovered cores. The first section considers analysis of the master cores (Section 5.2) and the next section discusses these results (Section 5.3). A third section details the results of analyses on the replicate cores (Section 5.4) before chapter conclusions are presented (Section 5.5).

5.2 Master core dating results

All sites have been subject to some form of management given the land use history of upland Wales (Section 2.7). To provide the most robust chronologies possible, all master cores were recovered from apparently actively accumulating areas of blanket peat and away from areas of obvious management (e.g. drains). Chronological results for each master core are considered sequentially below. Key features of the results (e.g. unsupported ^{210}Pb profiles, agreement between SCP and Constant Rate of Supply (CRS) models) are highlighted for each core, with these features discussed in detail in Section 5.3. Results figures are presented as Panels 1: total, supported and unsupported ^{210}Pb activity, 2: artificial fallout radionuclide activity and 3: SCP concentration with age markers A) AD 1850 ± 25,

B) AD 1955 ± 15 and C) AD 1976 ± 3. Age/depth models show the agreement between CRS models and independent age markers (^{137}Cs, ^{241}Am and SCPs). X-axis CRS errors detail chronological error for each modelled age. X-axis SCP errors signify errors of the dateable horizons (A) ± 25, B ± 15 and C) ± 3 years) whilst y-axis errors signify sampling error.

5.2.1 BRP-1-M

Total ^{210}Pb activity in BRP-1-M reaches an approximate equilibrium with supported ^{210}Pb activity *ca.* 24cm (Figure 5.1). Unsupported ^{210}Pb initially declines at 4cm, then increases to 15cm, suggesting an increased peat accumulation rate (PAR). Unsupported ^{210}Pb declines relatively uniformly below 15cm suggesting a constant PAR. There is a peak in the ^{137}Cs record at 15cm and the ^{241}Am record at 17cm (Figure 5.1). The isolated occurrences of SCPs at 24, 26 and 34cm were excluded (following Section 3.4.4), with the start of the record occurring at 20cm. Given the intercept at 20cm, the rapid increase (B) was assigned to 18cm as this was the next measured point. The non-monotonic variation in unsupported ^{210}Pb necessitates the use of the CRS model (Appleby, 2008). The model (Figure 5.2) produced reasonable agreement with the fallout radionuclide records, placing AD 1963 (nuclear weapons fallout) and AD 1986 (Chernobyl accident) around 17cm (^{241}Am peak) and 15cm (^{137}Cs peak) respectively. Whilst both SCP AD 1850 and 1976 markers are older than their equivalent depth in the CRS profile, AD 1955 shows agreement with the CRS model.

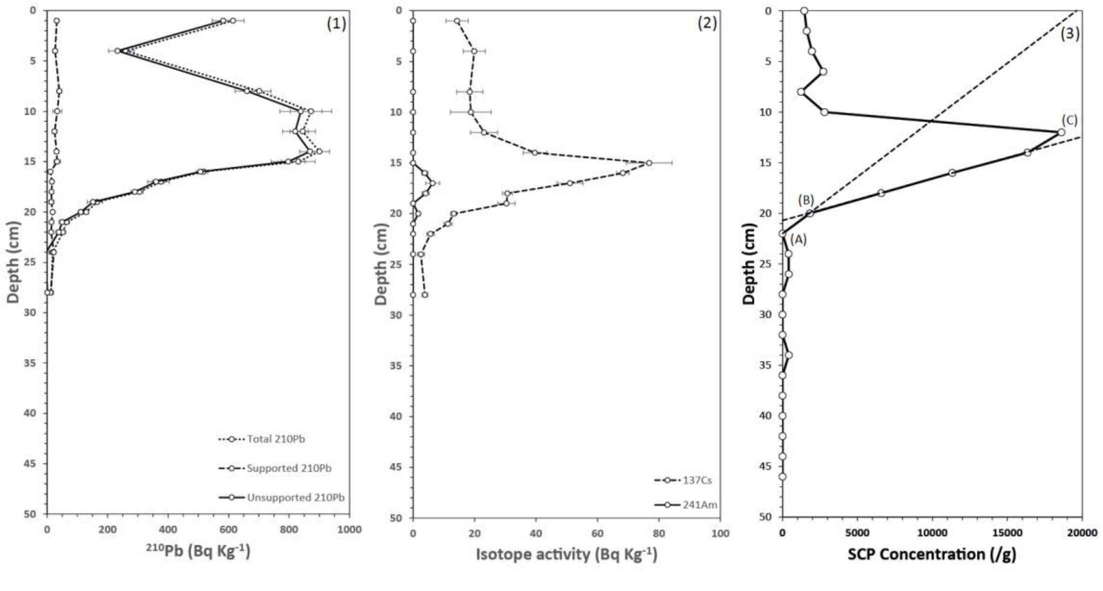


Figure 5.1:Dating results for core BRP-1-M.

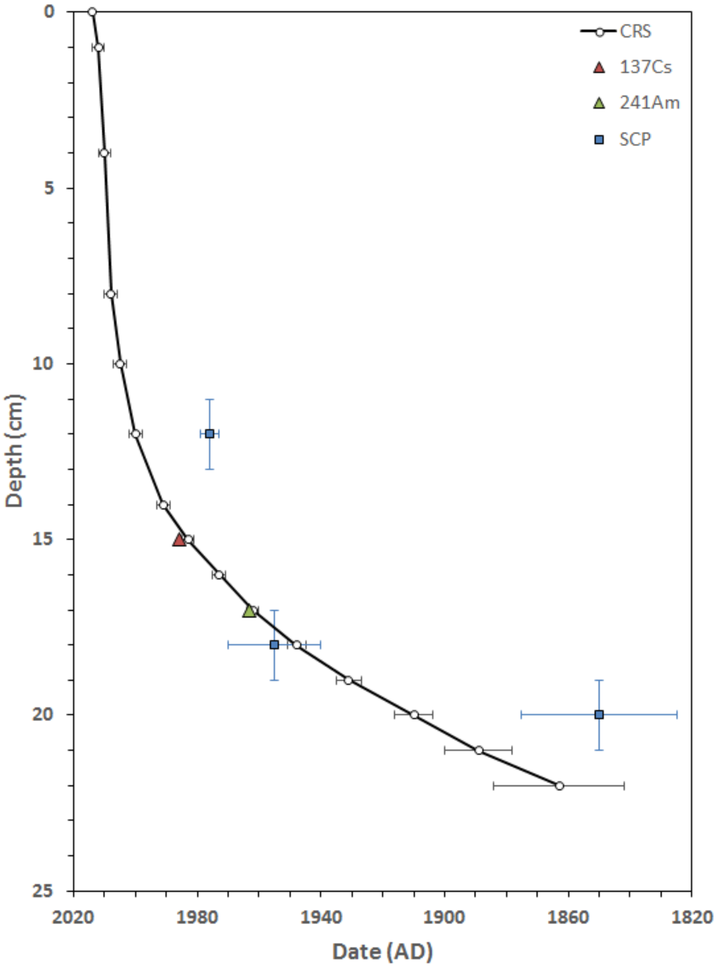


Figure 5.2: Comparison of CRS age/depth model with independent age markers for core BRP-1-M.

5.2.2 BRP-2-M

Total ^{210}Pb activity in BRP-2-M reaches an approximate equilibrium with supported ^{210}Pb activity *ca*. 36cm (Figure 5.3). Unsupported ^{210}Pb fluctuates in the upper 15cm of the core, before declining below this, suggesting PAR increased in the upper 15cm of the core relative to deeper peat. The unsupported ^{210}Pb decline below 24cm is relatively uniform. There is a clear peak in both the ^{137}Cs and ^{241}Am records at 18cm and an increase in ^{137}Cs activity at 24cm (Figure 5.3). The SCP concentration profile does not follow the 'typical' model (Section 3.4.5) and displays multiple peaks and troughs. Increased SCP concentrations 28-26cm were not deemed to indicate AD 1955 as the rise is not sustained. The pre-B and post-B intercept was recorded at 24cm as this is where the sustained increase in SCP concentrations occurs. The non-monotonic variation in unsupported ^{210}Pb necessitates the use of the CRS model. The determined model (Figure 5.4) agrees reasonably well with the fallout radionuclide records with AD 1986 placed just above the 17cm radionuclide peaks and the AD 1960's placing alongside the increase in ^{137}Cs activity at 24cm. Both SCP AD 1850 and 1976 markers are older than their equivalent depth in the CRS model whilst AD 1955 shows good agreement with the model.

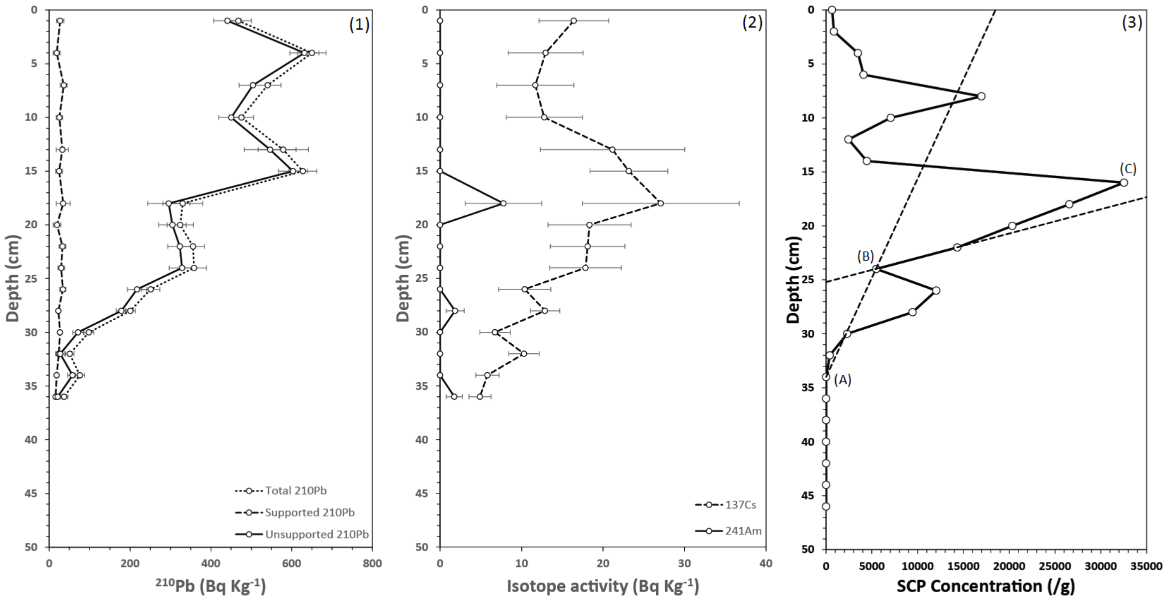


Figure 5.3: Dating results for core BRP-2-M.

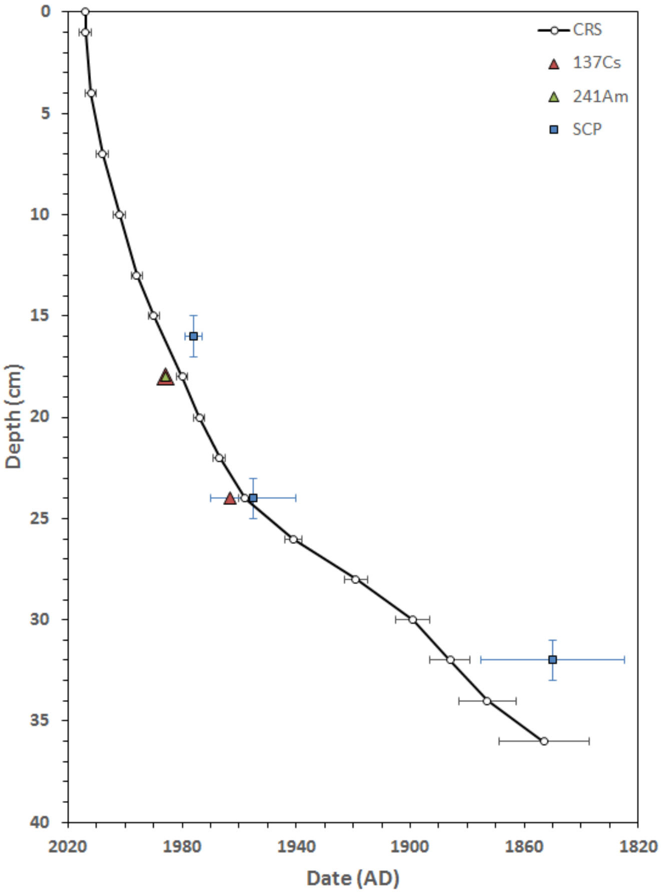


Figure 5.4: Comparison of CRS age/depth model with independent age markers for core BRP-2-M.

5.2.3 LVW-1-M

Total ^{210}Pb activity in LVW-1-M reaches approximate equilibrium with supported ^{210}Pb activity *ca.* 38cm (Figure 5.5). Unsupported ^{210}Pb increases from the core surface to a peak *ca.* 16cm, suggesting a rapid PAR in the upper core. Below 20cm unsupported ^{210}Pb declines relatively uniformly suggesting a constant PAR. There is a clear peak in the ^{137}Cs record at 24cm and another potential peak at 20cm (Figure 5.5). ^{241}Am activity is less pronounced and does not match the ^{137}Cs record. SCP concentrations follow the typical model, with A, B and C readily identifiable. The non-monotonic variation in unsupported ^{210}Pb necessitates the use of the CRS model. The determined model (Figure 5.6) agrees reasonably well with the ^{137}Cs record, placing AD 1986 *ca.* 21cm and AD 1963 *ca.* 25cm. These are offset slightly with the ^{137}Cs peaks at 20cm and 24cm respectively. Both AD 1850 and 1976 SCP markers show agreement with the CRS model, though AD 1955 is estimated as younger than the equivalent depth in the CRS model.

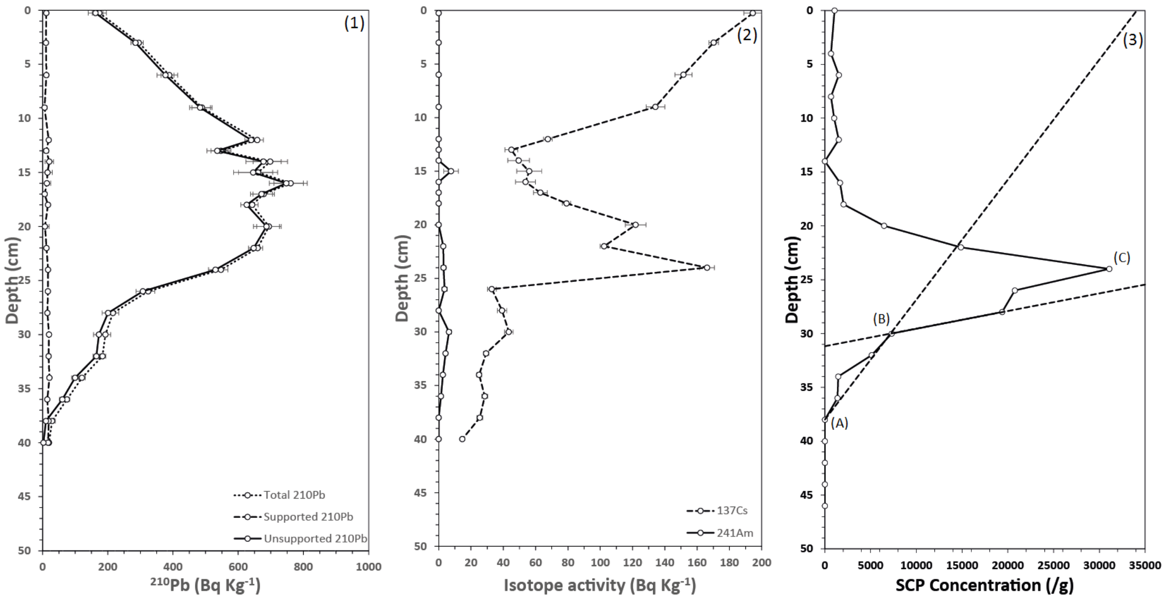


Figure 5.5: Dating results for core LVW-1-M.

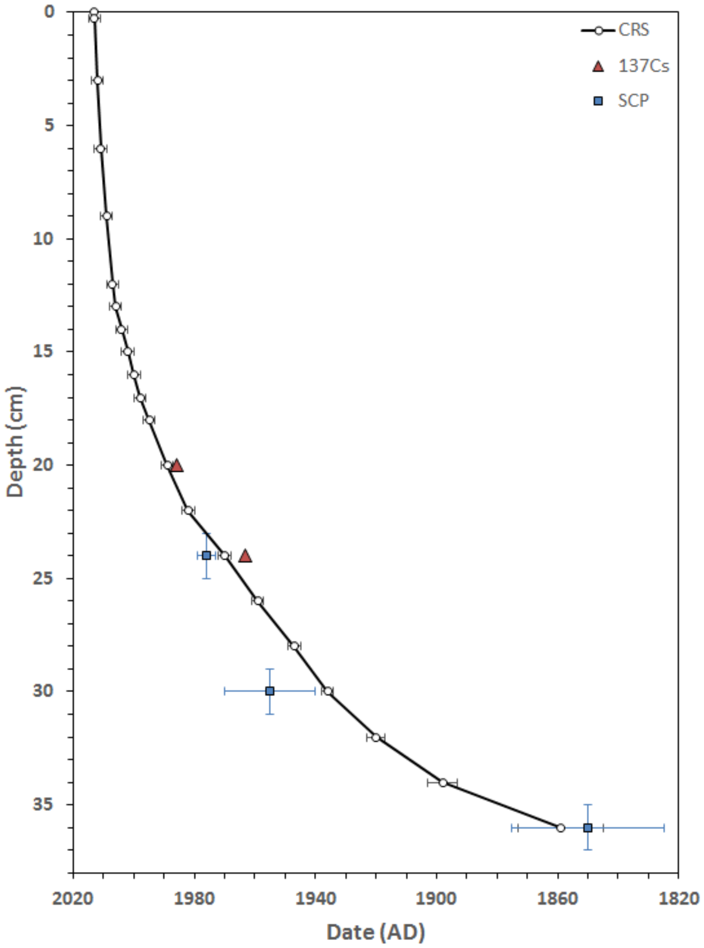


Figure 5.6: Comparison of CRS age/depth model with independent age markers for core LVW-1-M.

5.2.4 LVW-2-M

Total ^{210}Pb activity in LVW-2-M reaches an approximate equilibrium with supported ^{210}Pb activity *ca.* 24cm (Figure 5.7**).** Unsupported ^{210}Pb increases from the core surface to a peak *ca.* 9cm, suggesting a rapid PAR near the surface. Unsupported ^{210}Pb activity declines below 9cm in an irregular manner and does not decline uniformly until below *ca.* 14cm. There is a clear peak in the ^{137}Cs record at 12cm (Figure 5.7). ^{241}Am activity is less pronounced and does not match the ^{137}Cs record. SCP occurrences below 26cm were excluded with A assigned to 24cm. The non-monotonic variation in unsupported ^{210}Pb necessitates the use of the CRS model. The initial CRS model (Figure 5.8) did not agree with the ^{137}Cs independent radionuclide record for AD 1986 or both the ^{137}Cs and ^{241}Am records for AD 1963. On advice, the CRS model was adjusted to reflect the AD 1986 marker (Yang, H. pers. comm. 2014). The AD 1850 SCP marker is beyond the limits of the CRS model which culminates AD 1879. Extrapolating the model to a depth of 25cm, assuming a constant PAR (‘Extrap. CRS’; Figure 5.8), shows the SCP date for AD 1850 may be reliable. SCP AD 1955 and 1976 agree with both CRS models.

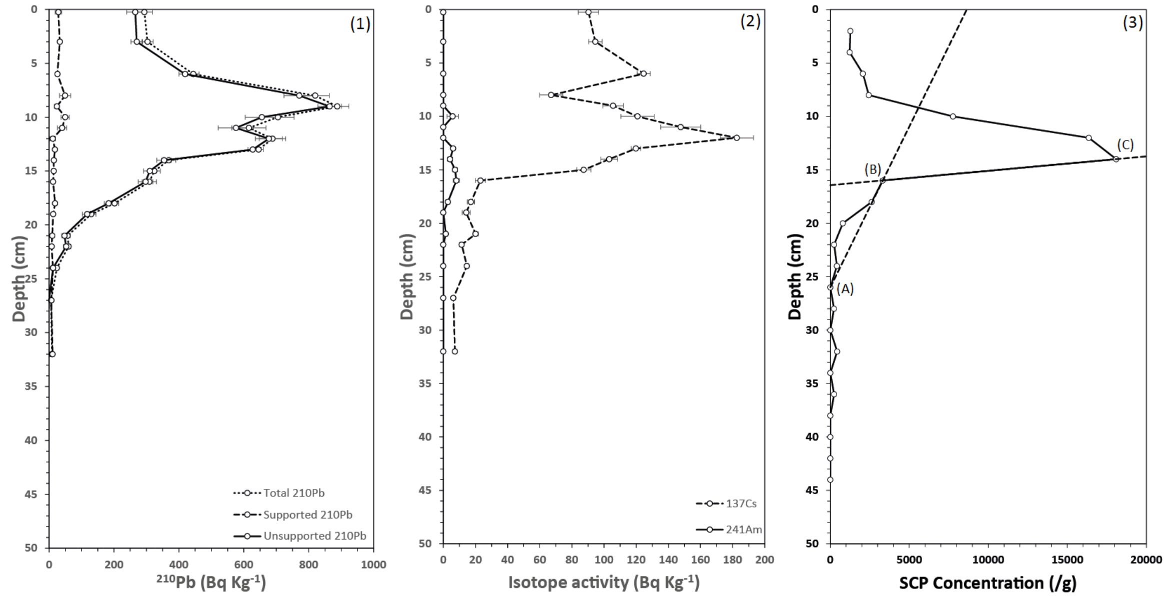


Figure 5.7: Dating results for core LVW-2-M.

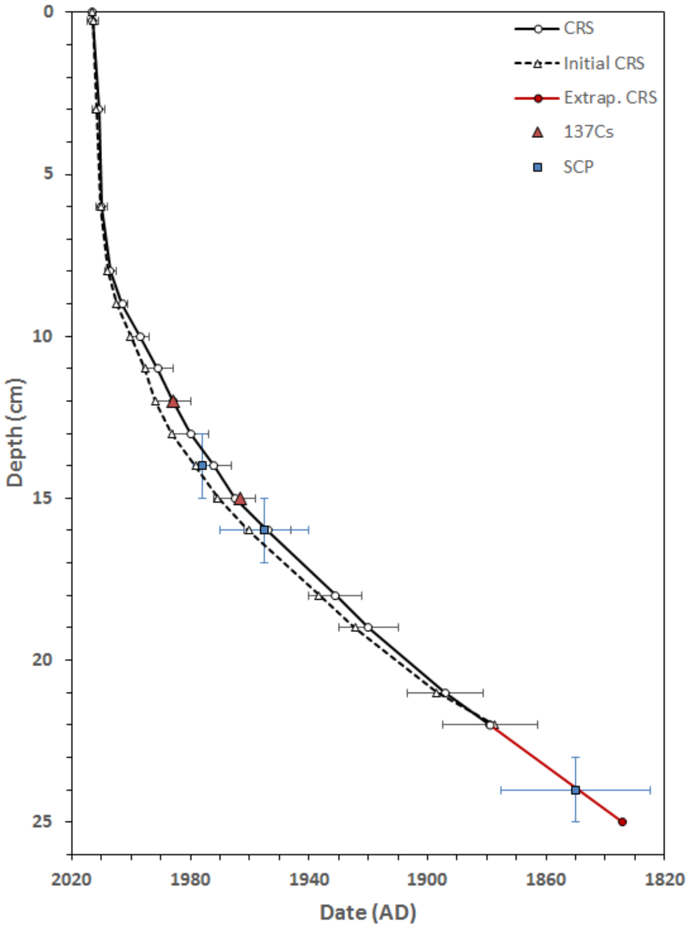


Figure 5.8: Comparison of CRS age/depth model with independent age markers for core LVW-2-M.

5.2.5 LVW-3-M

Total ^{210}Pb activity in LVW-3-M reaches an approximate equilibrium with supported ^{210}Pb activity *ca*. 40cm (Figure 5.9). Unsupported ^{210}Pb increases from the core surface to a peak *ca.* 9cm, suggesting a rapid PAR in the upper part of the core. Activity thereafter declines in a relatively uniform fashion until 19cm, suggesting a uniform PAR. There is a strong peak in the ^{137}Cs record at 11cm (Figure 5.9). The SCP record begins at 40cm (A) and exhibits a broadly 'typical' form. The non-monotonic variation in unsupported ^{210}Pb necessitates the use of the CRS model. The model (Figure 5.10) determines AD 1986 at 11cm, just above the peak in ^{137}Cs activity at 11cm. Whilst the CRS model again fails to reach the AD 1850 SCP marker, the extrapolated 40cm CRS age is beyond the error for this SCP horizon. The AD 1955 SCP date shows excellent agreement with the CRS model whilst the AD 1976 marker does not agree with the model.

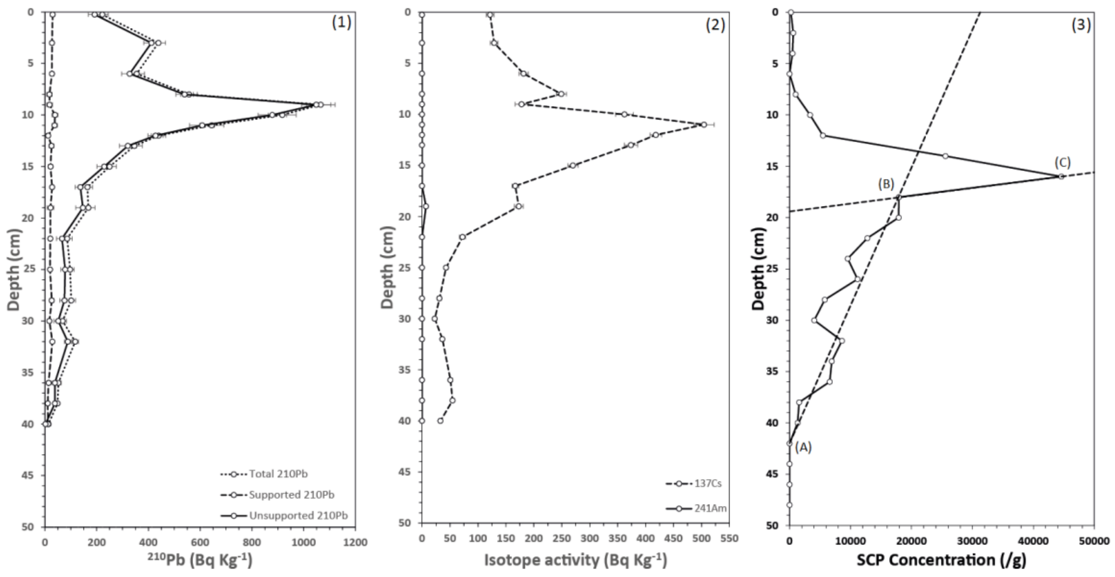


Figure 5.9: Dating results for core LVW-3-M.

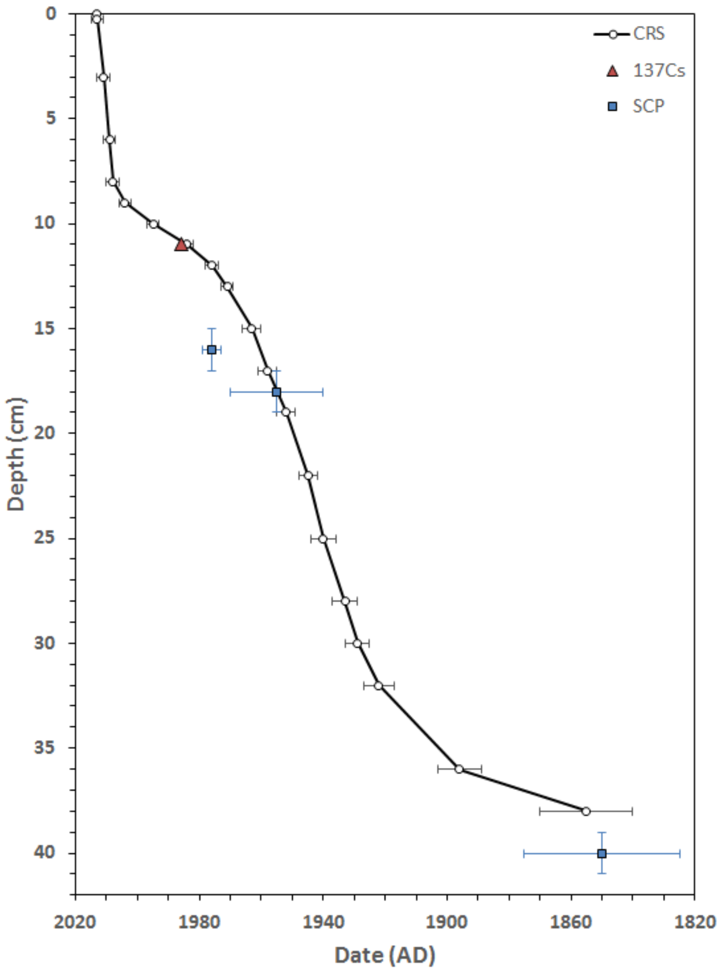


Figure 5.10: Comparison of CRS age/depth model with independent age markers for core LVW-3-M.

5.2.6 MIG-1-M

Total ^{210}Pb activity in MIG-1-M does not reach equilibrium with supported ^{210}Pb activity (Figure 5.11). Unsupported ^{210}Pb activity declines rapidly from 5cm to 30cm, but below this there is little change in activity suggesting a rapid PAR through much of the core. There are two prominent peaks in the ^{137}Cs record at 18cm and 30cm with no activity registered in the ^{241}Am profile (Figure 5.11). The AD 1850 SCP marker cannot be assigned because SCPs were recorded in all samples. The pre-B extrapolated gradient hence follows the trajectory of the broad increase in SCP concentration recorded in the lower portions of the core. The non-monotonic variation in unsupported ^{210}Pb necessitates the use of the CRS model. The CRS model (Figure 5.12) places AD 1986 proximal to the 18cm ^{137}Cs spike and AD 1963 just below the 30cm ^{137}Cs peak. These slight discrepancies are acceptable considering the rapid PAR in the core. The AD 1955 SCP marker agrees perfectly with the CRS model with the AD 1976 marker younger than the CRS model predicts for that depth.

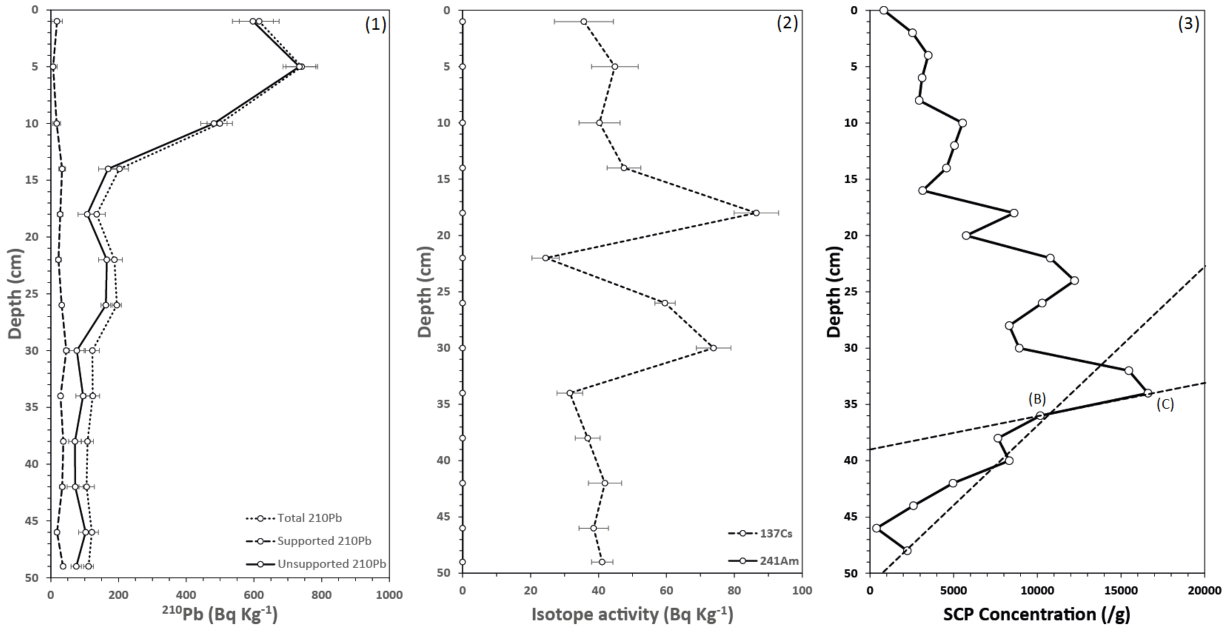


Figure 5.11: Dating results for core MIG-1-M.

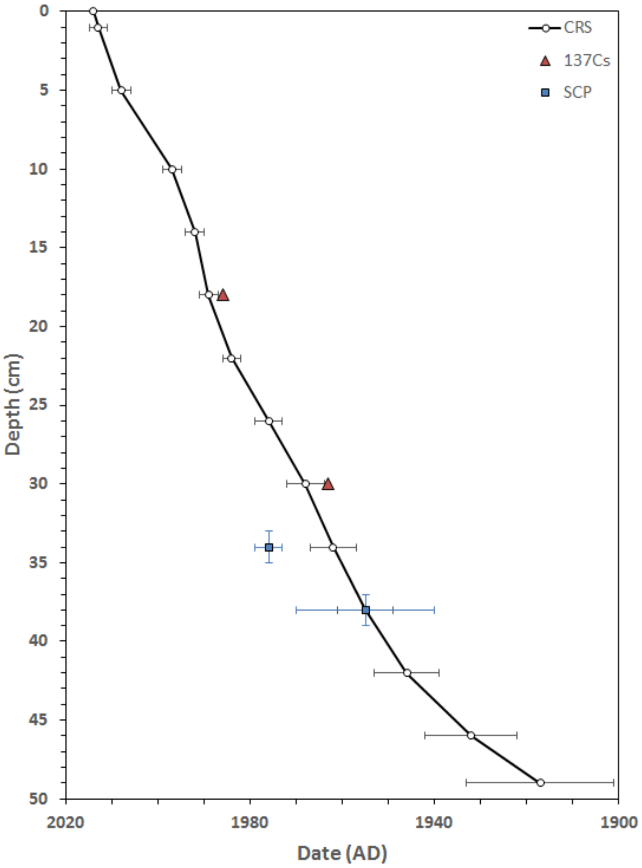


Figure 5.12: Comparison of CRS age/depth model with independent age markers for core MIG-1-M.

5.2.7 MIG-2-M

Total ^{210}Pb activity in MIG-2-M reaches approximate equilibrium with supported ^{210}Pb activity *ca*. 27cm (Figure 5.13). Unsupported ^{210}Pb increases from the core surface to a peak *ca*. 10cm suggesting an increase in PAR. Unsupported ^{210}Pb activity declines relatively uniformly with depth until equilibrium is reached *ca.* 27cm. There is a single usable peak in the ^{137}Cs record at 12cm and there are no registered readings in the ^{241}Am profile (Figure 5.13). Isolated recordings of SCPs at 46 and 38cm were not considered to represent the AD 1850 marker. AD 1850 was assigned at 28cm depth, with the SCP profile above A broadly typical. The non-monotonic variation in unsupported ^{210}Pb necessitates the use of the CRS model. The determined model (Figure 5.14) agrees reasonably well with the ^{137}Cs record, placing AD 1986 just below (*ca*. 13cm) the ^{137}Cs at 12cm. Both SCP AD 1850 and 1955 markers suggest peat is younger at a given depth than the CRS model, though the AD 1976 SCP peak shows good agreement with the model.

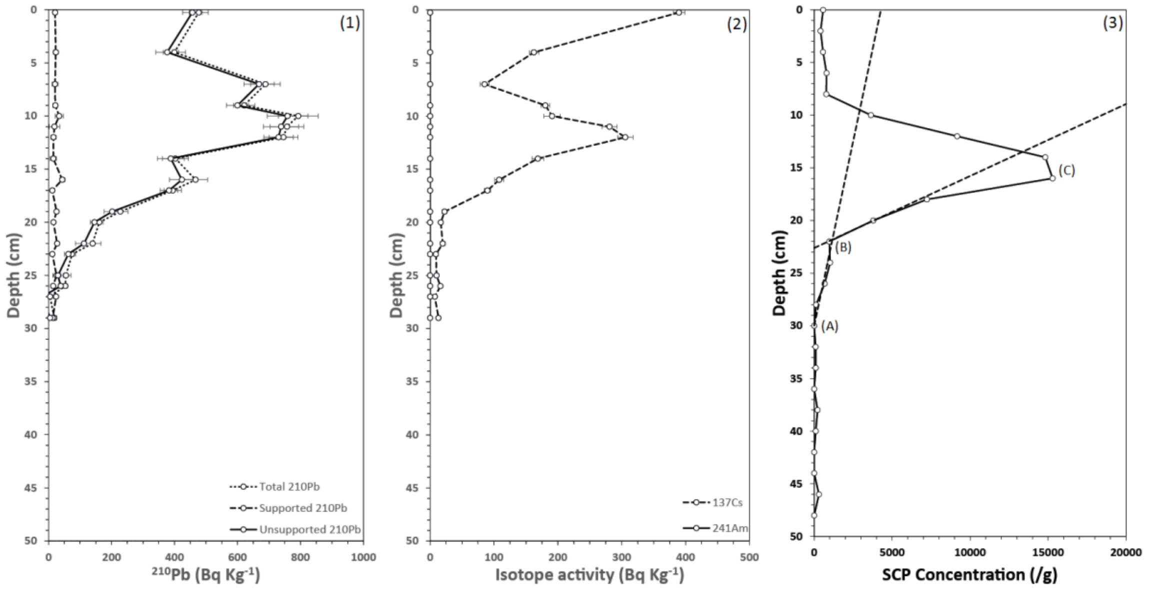


Figure 5.13: Dating results for core MIG-2-M.

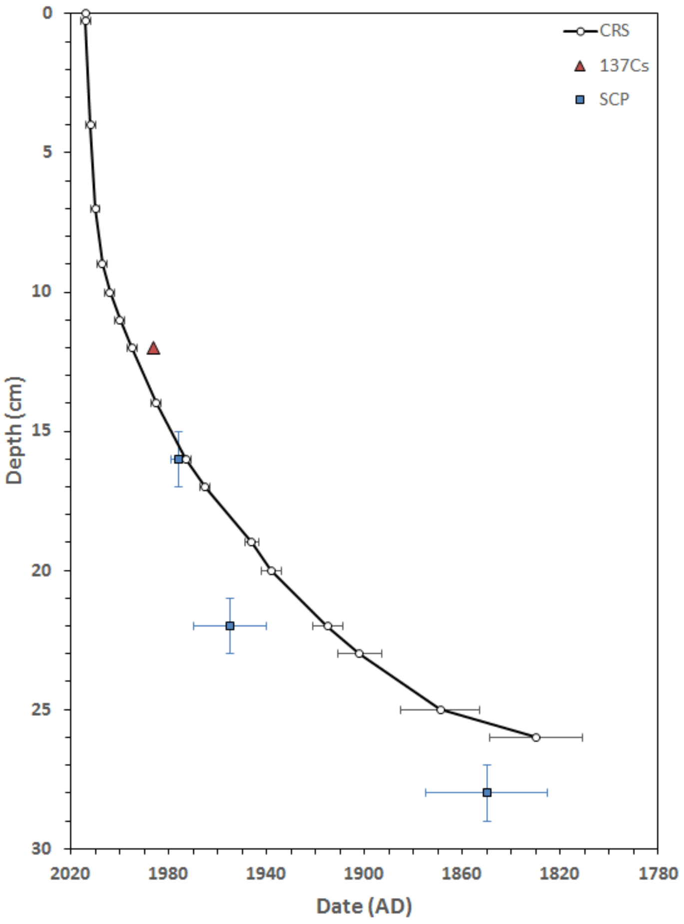


Figure 5.14: Comparison of CRS age/depth model with independent age markers for core MIG-2-M.

5.2.8 MIG-3-M

Total ^{210}Pb activity in MIG-3-M reaches an approximate equilibrium with supported ^{210}Pb activity *ca.* 40cm depth (Figure 5.15). Unsupported ^{210}Pb increases from the core surface to a peak *ca.* 14cm suggesting an increased PAR. Unsupported ^{210}Pb activity declines irregularly below 14cm suggesting a variable PAR with a relatively uniform PAR below *ca.* 29cm. There are two peaks in the ^{137}Cs profile at 18cm and 27cm, with two slight peaks in ^{241}Am offset slightly below these readings at 21cm and 30cm (Figure 5.15). The SCP profile is relatively typical with A located at 38cm. The non-monotonic variation in unsupported ^{210}Pb necessitates the use of the CRS model. The determined CRS model (Figure 5.16) agrees with the upper ^{137}Cs record, placing AD 1986 just below 18cm. AD 1963 is placed *ca.* 25cm, just above the 27cm peak in ^{137}Cs. The AD 1850 SCP marker is beyond the limits of the CRS model which culminates AD 1857. Extrapolating the model to a depth of 38cm shows that A is only just beyond the limits of the CRS model and would likely overlap within error of the CRS model. SCP AD 1955 and 1976 markers both suggest peat is younger at a given depth than the CRS model.

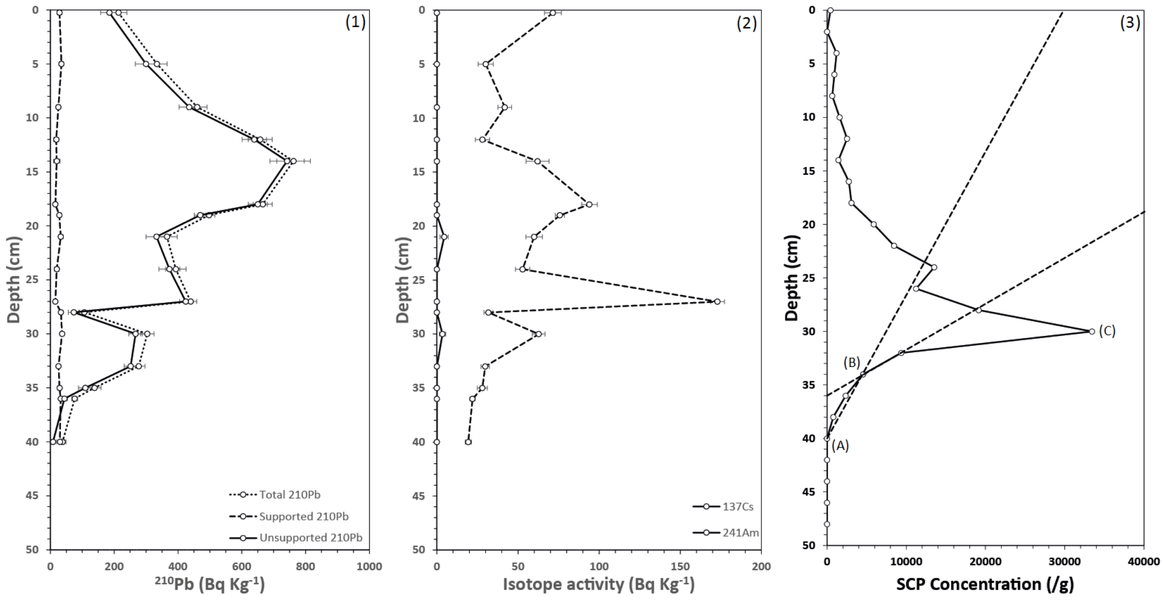


Figure 5.15: Dating results for core MIG-3-M.

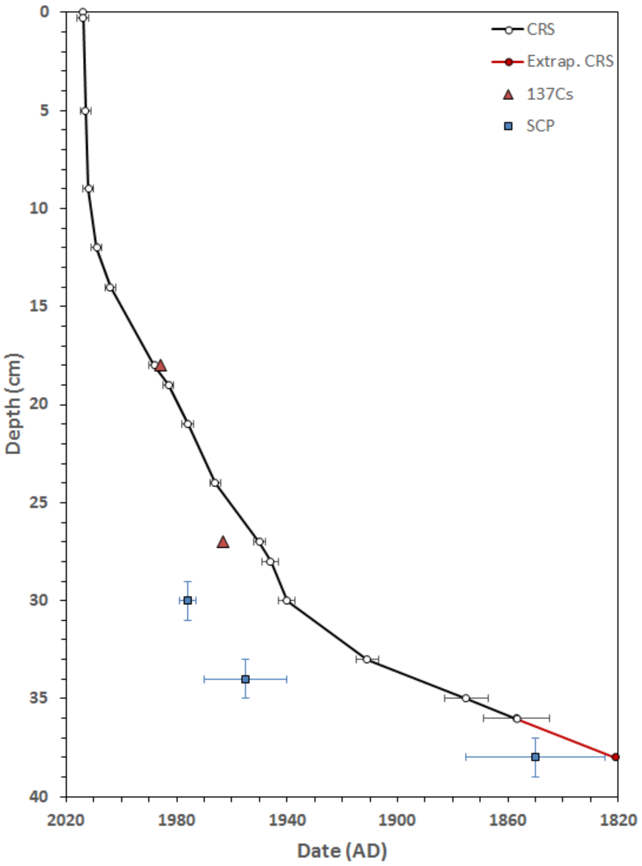


Figure 5.16: Comparison of CRS age/depth model with independent age markers for core MIG-3-M.

5.2.9 PLY-1-M

Total ^{210}Pb activity in PLY-1-M reaches an approximate equilibrium with supported ^{210}Pb activity *ca*. 39cm depth (Figure 5.17). Unsupported ^{210}Pb increases from the core surface to a peak *ca.* 7cm suggesting an increased PAR. Below this the uniform decline in unsupported ^{210}Pb activity suggests a constant PAR. There is a well resolved ^{137}Cs peak at 18cm and a lesser peak at 14cm (Figure 5.17). The isolated SCP recording at 36cm was excluded when assigning dates to the SCP curve (following Section 3.4.4). Feature A (AD 1850) was designated at 32cm. Non-monotonic variation in unsupported ^{210}Pb necessitates the use of the CRS model. The initial model (Figure 5.18) did not agree with radionuclide markers, placing AD 1986 at 16cm and AD 1963 at 23cm. These are offset from the respective ^{137}Cs peaks at 14cm and 18cm. On advice, the CRS model was adjusted to reflect the radionuclide AD 1986 and 1963 markers (Yang, H. pers. comm. 2014). Whilst the depth of the AD 1850 SCP marker is older than the CRS model both AD 1955 and 1976 horizons were within error of the initial CRS model.

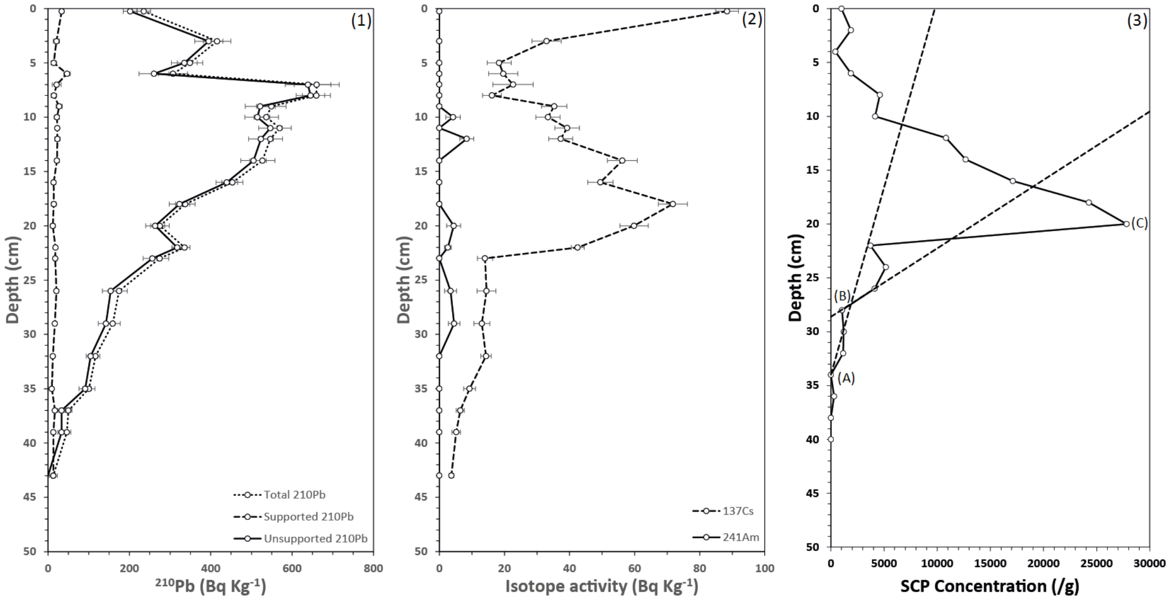


Figure 5.17: Dating results for core PLY-1-M.

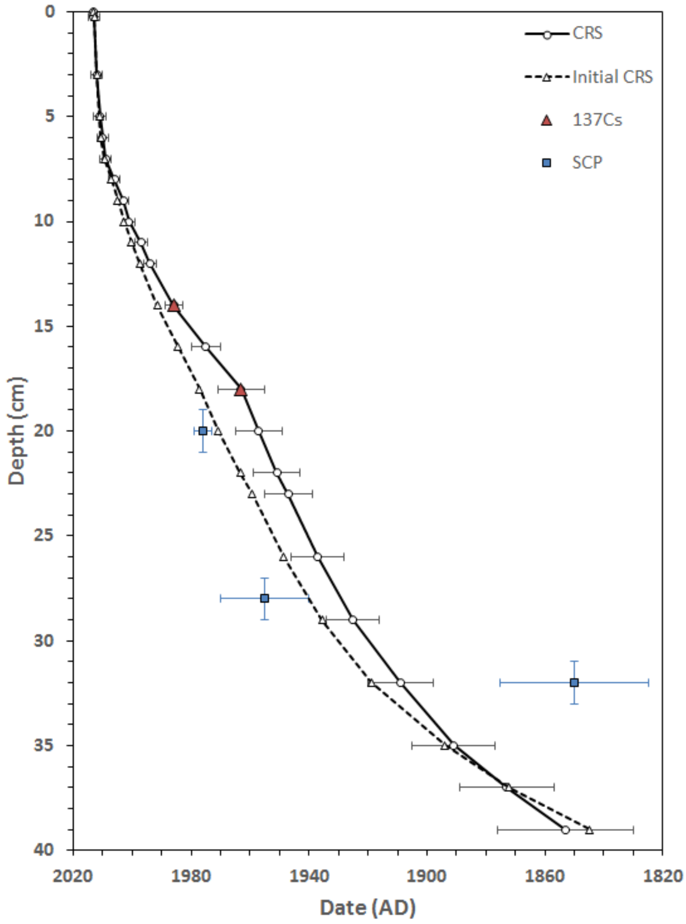


Figure 5.18: Comparison of CRS age/depth model with independent age markers for core PLY-1-M.

5.2.10 PLY-2-M

Total ^{210}Pb activity in PLY-2-M reaches an approximate equilibrium with supported ^{210}Pb activity *ca*. 26cm depth (Figure 5.19). Unsupported ^{210}Pb increases from the core surface to a peak at 16cm suggesting an increased PAR relative to a reasonably constant PAR below this point. There is a single prominent peak in ^{137}Cs activity at 19cm (Figure 5.19). ^{241}Am activity is recorded around this point (23-16cm depth) though no substantial peaks were detected. The isolated SCP recording at 27cm was excluded, with A located at 23cm. The non-monotonic variation in unsupported ^{210}Pb necessitates the use of the CRS model. The CRS model (Figure 5.20) calculates AD 1963 as being between 19 and 18cm. This is in good agreement with the ^{137}Cs record. All SCP age marker errors are within the error limits of the CRS model with AD 1976 showing the greatest agreement.

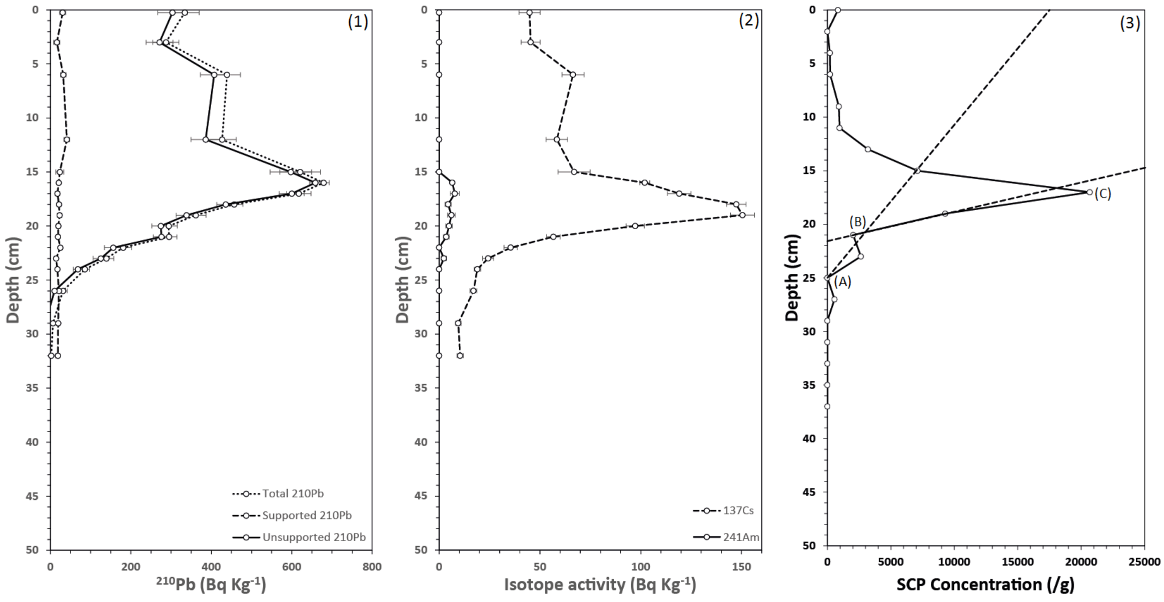


Figure 5.19: Dating results for core PLY-2-M.

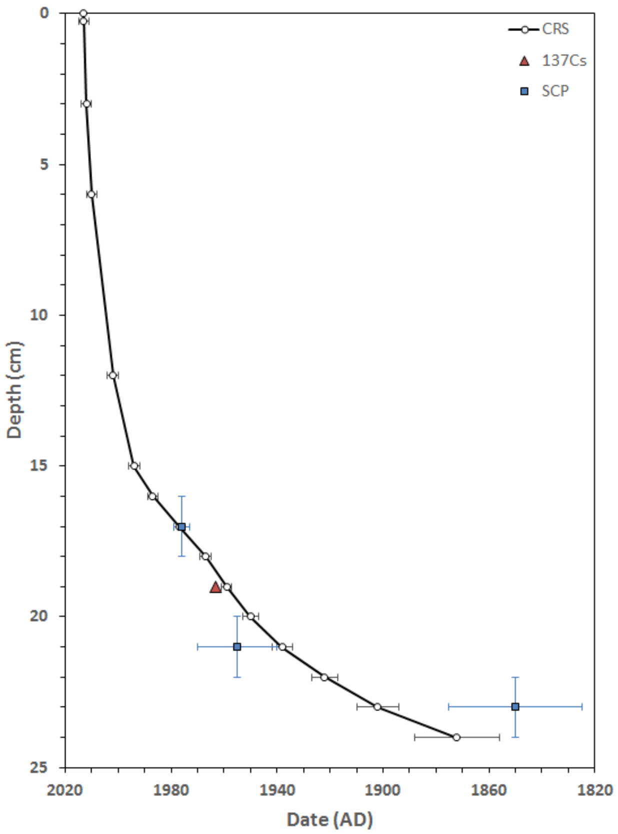


Figure 5.20: Comparison of CRS age/depth model with independent age markers for core PLY-2-M.

5.3 Discussion of master core dating results

The dating of recent peats (*ca*. last 200 years) is a challenge given concerns regarding the accuracy of radiocarbon dating across this timeframe (Charman, 2002). The radioisotope ^{210}Pb is considered one of the most valuable tools for constructing continuous chronologies for this period (Turetsky *et al*. 2004), hence its use in this thesis. The potential for ^{210}Pb mobility (Oldfield *et al*. 1979, 1995; Parry *et al*. 2013; Urban *et al*. 1990) undermines the use of the CRS model as a key assumption is that ^{210}Pb remains immobile following deposition (Appleby, 2001). As such, potential ^{210}Pb mobility is now explored to justify the use of the method and to assess its reliability for reconstructing peat accumulation rates (PARs). This thesis evaluates the potential for post-deposition ^{210}Pb mobility using three approaches employed by previous studies (Belyea and Warner, 1994; Parry *et al*. 2013); 1) comparison of fallout ^{210}Pb inventories (Section 5.3.1), 2) comparison of unsupported ^{210}Pb profiles (Section 5.3.2) and 3) comparisons of modelled dates with independent chrono-stratigraphic markers (Section 5.3.3).

5.3.1 Comparison of fallout ^{210}Pb inventories

Fallout ^{210}Pb inventories represent the cumulative unsupported ^{210}Pb activity in a core. ^{210}Pb inventories range from 3452-8688 Bq m^{-2} across the dataset (Table 5.1). Such variability across Mid- and North-Wales is not inherently surprising given Smith *et al.* (1997) showed variability in fallout ^{210}Pb inventories of 3810-7511 Bq m^{-2} within a single peatland. No statistical differences were observed in fallout inventories between the four sites (single factor ANOVA, P= 0.769) and the inventory means for Bryniau Pica (BRP) (5389 Bq m^{-2}), Lake Vyrnwy (LVW) (5863 Bq m^{-2}) and the Migneint (MIG) (4783 Bq m^{-2}) are all similar to the means of Parry *et al.* (2013) and Smith *et al.* (1997). The mean inventory for Plynlimmon (PLY) is greater (6376 Bq m^{-2}), though not significantly so (see above ANOVA).

Post-deposition mobilisation and leaching of ^{210}Pb may have occurred if the reconstructed annual ^{210}Pb flux of a core is lower than would be expected given a site's annual atmospheric ^{210}Pb fallout. Annual ^{210}Pb flux can be reconstructed for a core (Appleby *et al.* 1997; Parry *et al.* 2013) following Equation 5.1 (Smith *et al.* 1997) where F is the reconstructed annual flux, λ is the decay constant for ^{210}Pb (0.03114) and A is the total unsupported ^{210}Pb inventory:

$$F = \lambda A \qquad \text{Equation 5.1}$$

Direct measurements of atmospheric ^{210}Pb fallout over the period these cores have accumulated is not possible (i.e. there was no *in situ* monitoring of fallout over the last *ca.* 150 years). Atmospheric ^{210}Pb fallout can be estimated for a site if one assumes ^{210}Pb fallout has been constant as a function of rainfall. The value for mean annual atmospheric ^{210}Pb fallout of 77 ± 14 Bq m^{-2} yr^{-1} per 1,000mm of rainfall across the British Isles (Smith *et al.* 1997) was used to estimate annual regional ^{210}Pb fallout for each master core according to UKCP09 modelled annual precipitation (Section 4.3). Seven reconstructed annual ^{210}Pb fluxes are within error limits of the estimated annual regional ^{210}Pb fallout for that specific core (Table 5.1). The reconstructed annual ^{210}Pb flux of two master cores is notably greater than estimated annual ^{210}Pb regional fallout (BRP-2-M and LVW-1-M). Smith *et al.* (1997) found that the annual flux reconstruction method used here (Equation 5.1) was likely to overestimate annual ^{210}Pb flux relative to rainfall-measured annual ^{210}Pb fallout. Therefore, overestimations of annual ^{210}Pb flux (BRP-2-M and LVW-1-M) are not in themselves reasons to suspect inaccuracy in the chronologies. MIG-1-M estimates annual ^{210}Pb flux below that expected from annual regional fallout (Table 5.1). The discrepancy does not inherently mean that

^{210}Pb has been leached given the MIG-1-M AD 1955 SCP marker exhibits good agreement with the CRS model (Figure 5.12), implying the CRS chronology is accurate. Despite the coarseness of the method, seven of the ten reconstructed annual ^{210}Pb fluxes being within error limits of the estimated annual ^{210}Pb fallout suggests no substantial loss of ^{210}Pb has occurred.

Table 5.1: Master core fallout ^{210}Pb inventories, comparison of reconstructed core annual ^{210}Pb flux and regional ^{210}Pb fallout estimates. The 'Fallout difference' value is italicised where the difference exceeds estimated regional fallout error margins.

Core	**Fallout ^{210}Pb inventory (Bq m^{-2})**	**Reconstructed annual ^{210}Pb flux (Bq m^{-2} yr^{-1})**	**Estimated annual regional ^{210}Pb fallout (Bq m^{-2} yr^{-1})**	**Fallout difference (Bq m^{-2} yr^{-1})**
BRP-1-M	4512.8 ± 110.6	140.53 ± 3.44	137.93 ± 28	2.60
BRP-2-M	6264.8 ± 198.2	195.09 ± 6.17	137.93 ± 28	*+57.16*
LVW-1-M	8688.2 ± 167.1	270.55 ± 5.20	171.09 ± 28	*+99.46*
LVW-2-M	4749 ± 110.9	147.912 ± 3.54	171.09 ± 28	-23.17
LVW-3-M	4150 ± 124.9	129.23 ± 3.89	148.39 ± 28	-19.16
MIG-1-M	3452.6 ± 150.2	107.51 ± 4.68	180.74 ± 42	*-73.22*
MIG-2-M	4887.9 ± 149.4	152.21 ± 4.65	173.01 ± 42	-20.80
MIG-3-M	6008.5 ± 180.6	187.10 ± 5.62	173.01 ± 42	+14.09
PLY-1-M	7131 ± 172.8	222.06 ± 5.02	196.31 ± 42	+25.75
PLY-2-M	5621 ± 172.8	175.05 ± 5.38	196.31 ± 42	-21.26

5.3.2 Comparisons of unsupported ^{210}Pb profiles

Unsupported ^{210}Pb activity may be expected to decline exponentially with depth given ongoing decay of the radioisotope (Section 3.4.3). It has been noted that most master cores display a non-monotonic unsupported ^{210}Pb profile (Section 5.2) which may indicate evidence of 1) post-deposition isotope mobility (Damman, 1978; Vile *et al.* 1999) and/or 2) the unconsolidated upper parts of the profile are not appropriately transferring ^{210}Pb into the peatland (Belyea and Warner, 1994). Either phenomena violate the assumptions of the CRS model. This thesis now addresses these two concerns in turn to demonstrate that a non-monotonic unsupported ^{210}Pb profile does not inherently indicate post-deposition radioisotope mobility in these master cores.

A variable PAR would mean proximal portions of sediment receive differing initial concentrations of unsupported ^{210}Pb (Parry, 2011). Indeed, the CRS model explicitly accounts for the impact of variable sediment accumulation rates on unsupported ^{210}Pb activity (Appleby, 2001). The non-monotonic profiles of unsupported ^{210}Pb that most master cores exhibit likely occur due to an increase in apparent PAR in the upper core sections. Acrotelm peat accumulates at a faster rate relative to catotelm peat as the former has yet to undergo the same degree of decomposition as the latter (Section 2.2.2). Notable changes in bulk density (BD) can be used to infer the position of the acrotelm/catotelm transition (ACT) (Lindsay, 2010). Unsupported ^{210}Pb activity predominantly declines exponentially following a below-surface increase in BD (Figure 5.21). The relationship is particularly clear in cores LVW-2-M (Panel D), MIG-3-M (Panel H), PLY-1-M (Panel I) and PLY-2-M (Panel J). A sudden increase in apparent PAR in the acrotelm is therefore likely responsible for producing non-monotonic unsupported ^{210}Pb profiles. Levels of unsupported ^{210}Pb activity first increase with depth given peat is accumulating fast and receiving low initial concentrations of unsupported ^{210}Pb relative to the *measured* concentrations of catotelm peat. Unsupported ^{210}Pb then, predominantly, declines with depth in the catotelm, agreeing with Damman (1978) and Vile *et al.* (1999) (see above). The potential for PAR to change over time therefore negates the criticisms of Damman (1978) and Vile *et al.* (1999) in that the non-monotonic ^{210}Pb profiles in this thesis do not indicate post-deposition isotope mobility.

The second concern regarding the reliability of non-monotonic unsupported ^{210}Pb profiles was presented by Belyea and Warner (1994) who suggested such profiles indicate that unconsolidated surface peat is not transferring ^{210}Pb into the peatland. In this case, low BD substrates may yield a poor chronology. It could be argued that MIG-1-M produces the most monotonic unsupported ^{210}Pb profile as there is an exponential decline below 4cm (Panel F; Figure 5.21). MIG-1-M BD remains <0.06 g cm^{-3} for its duration, by far the lowest average BD of all master cores (Figure 5.21). Variations in near-surface BD do not therefore suggest ^{210}Pb is not being transferred effectively into the peatland. Changes in PAR at the ACT (see above) are responsible for producing the non-monotonic unsupported ^{210}Pb profiles, thereby justifying the use of the CRS over the CIC model (Appleby, 2008). Deviations from the exponential decline in unsupported ^{210}Pb are therefore not indicative of isotope mobility or an inability to transfer ^{210}Pb below the surface.

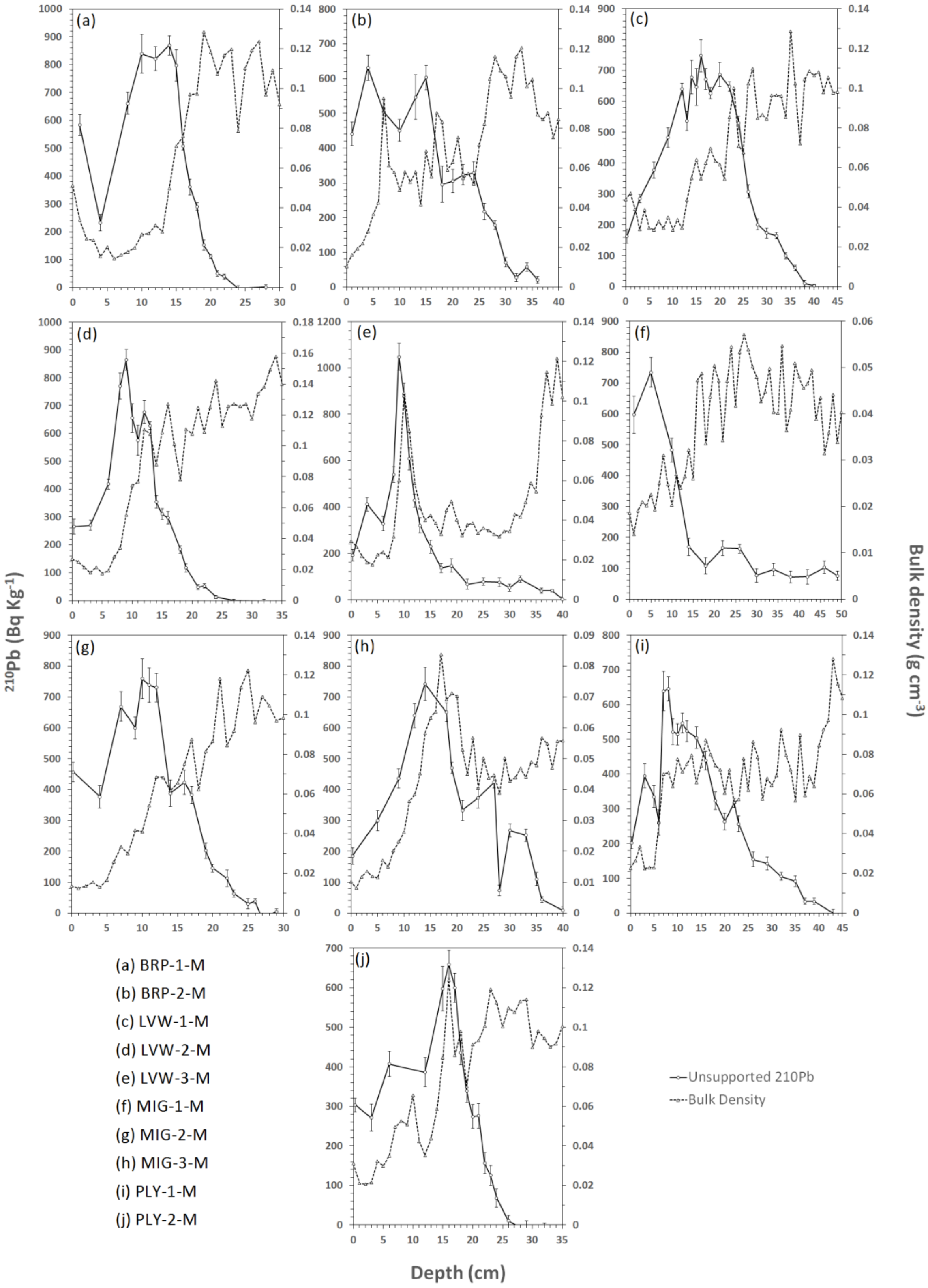


Figure 5.21: Comparison of unsupported ^{210}Pb activity and bulk density. Axes optimised by core. Note how unsupported ^{210}Pb activity declines in a, predominantly, uniform manner with depth following an increase in bulk density.

5.3.3 Comparisons of modelled dates with independent chrono-stratigraphic markers

The results of differing chronological approaches (i.e. radiometric and SCP as in this thesis) can be compared to offer an assessment of the relative efficacy of each approach and to determine whether one technique is systematically unreliable. One must consider that all techniques may be inaccurate meaning agreement between approaches does not inherently indicate a reliable chronology. As such, the performance of the independent radiometric (Section 5.3.3.1) and SCP chrono-stratigraphic markers (Section 5.3.3.2) are discussed in isolation before these results are compared to the CRS models (Section 5.3.3.3).

5.3.3.1 ^{137}Cs and ^{241}Am behaviour

^{137}Cs and ^{241}Am are indicators of nuclear activity (Section 3.4.3). The two primary age markers for 20th Century nuclear activity are the AD 1963 nuclear weapons testing fallout (though this can be temporally variable) and the AD 1986 Chernobyl nuclear accident (Appleby, 2008; Turetsky *et al.* 2004). Results for ^{137}Cs and ^{241}Am were variable across the suite of cores. Some profiles recorded small errors (e.g. LVW-1-M) whilst others recorded large errors (e.g. BRP-2-M). Long ‘tails’ in ^{137}Cs activity may indicate mobility if the activity is recorded in pre-AD 1940 substrate as determined by a CRS model (e.g. BRP-2-M; Figure 5.3).

^{241}Am decays *in situ* from ^{241}Pu released by nuclear weapons activity. ^{241}Pu is released and deposited in sedimentary systems very rapidly following such activity. Hence, Appleby *et al.* (1991) suggest ^{241}Am profiles, in their case from lakes, should consist of clear ^{241}Am peaks with limited background activity. Several studies assessing ^{241}Am in ombrotrophic peats broadly support this notion (Clymo *et al.* 1990; Oldfield *et al.* 1995; Sanders *et al.* 1995; Smith *et al.* 1997). BRP-1-M, BRP-2-M, LVW-3-M and MIG-3-M conform to such a pattern. LVW-1-M, LVW-2-M and PLY-1-M show less prominent peaks whilst PLY-2-M shows no prominent peak, merely general ^{241}Am activity between 24-15cm. Such responses are not common in ombrotrophic peats, albeit there is a general lack of studies that assess ^{241}Am. Parry *et al.* (2013) reported similar results in that ^{241}Am records from Dartmoor exhibited no clear peak, merely general background activity across a range of depths. Parry *et al.* attributed this to be a result of mobility of ^{241}Am in the peat column. Mitchell *et al.* (1992) identified post-deposition ^{241}Am mobility as they recorded ^{241}Am in peat which, according to other dating techniques, accumulated prior to nuclear testing. It may be that ^{241}Am is mobile in those cores (LVW-2-M and PLY-2-M) which show deviation in ^{241}Am activity from the assumed profile (*sensu* Appleby *et al.* 1991). No ^{241}Am activity was recorded in MIG-1-M and MIG-2-M. It is uncertain why this was the case. Expected (*sensu* Appleby *et al.* 1991) ^{241}Am activity was recorded in MIG-3-M which is located between the cores of MIG-1-M and MIG-2-M (Section 4.3.3). Lack of ^{241}Am activity in MIG-1-M and MIG-2-M is unlikely therefore to be a result of a phenomena that affects the whole of the Migneint.

5.3.3.2 SCP Concentration profiles

SCPs are indicators of industrial activity (Section 3.4.4) and were recorded in all master cores. Peak SCP concentrations ranged from 16,609 - 44,574 g DM^{-1}, with an average of 25,872 g DM^{-1}. Variability in peak concentration may be due to differences in SCP fallout which is linked to both regional rainfall and industrial activity (Rose *et al.* 1995), alongside peatland specific differences in microtopography (Parry *et al.* 2013). No UK wide or regional mean SCP fallout per unit rainfall has been published akin to the Smith *et al.* (1997) ^{210}Pb fallout values (Section 5.3.1). Researchers publish SCP results in a variety of manners so not all studies are directly comparable with the SCP concentration values produced by this thesis (g DM^{-1}). Values are published elsewhere as raw SCP numbers (Garnett *et al.* 2000; Mauquoy *et al.* 2002; McCarroll, 2014) or concentration per volume (Hendon and Charman, 2004; Schoning *et al.* 2005; Watson *et al.* 2015). Yang *et al.* (2001) recorded peak SCP concentration values from *ca.* 4,500 - 22,000 g DM^{-1} within the Lochnagar peatland catchment. Parry *et al.* (2011) reported peak concentrations for cores within a 10km area on Dartmoor of 9,692 - 73,825 g DM^{-1}. There is therefore no reason to suggest mass mobilisation and loss of SCPs from the cores recovered in this thesis as the recorded range of peak concentrations (16,609 - 44,574 g DM^{-1}) all exceed the lowest peak values of 4,500 and 9,692 gDM^{-1} recorded by Yang *et al.* (2001) and Parry *et al.* (2013) respectively. Peak SCP concentrations likely vary according to peatland specific characteristics (i.e. PAR, rainfall, sampled nanotope).

Divergence from the expected SCP concentration model (Rose *et al.* 1995) may indicate post-deposition SCP mobility (Rose and Appleby, 2005). Many of the master cores follow the typical SCP concentration model so mobility is not of concern for these cores. Divergence from the typical model is most clear in core BRP-2-M with three separate peaks identifiable at 26, 16 and 8cm (Figure 5.3). Multiple peak concentrations do not mean the BRP-2-M SCP profile is unreliable. Rose *et al.* (1995) suggest assigning the deepest peak as the AD 1976 peak. The deepest ‘peak’ at BRP-2-M (26cm) is the lowest in terms of concentration and it is hard to assign the AD 1850 and 1955 dating features below this point in the profile. The greatest concentration of SCPs was recorded at 16cm depth, hence as the deepest of the two shallower peaks this feature was designated the ‘true’ AD 1976 peak (Rose *et al*. (1995) which produced better agreement with the CRS model than if 26 or 8cm were designated as the AD 1976 horizon (Figure 5.4). The peak SCP concentration peak at 8cm in BRP-2-M may be related to the increased BD at this depth (Panel B; Figure 5.21). The effect of BD on SCP concentrations is discussed below.

An assumption of SCP dating is that PAR is reasonably constant because sudden changes may affect SCP concentration (Garnett *et al*. 2000). The decline in SCP concentrations at the surface of all cores may therefore not be occurring due to the reduction in SCP emissions and atmospheric fallout, but instead be due to an increase in apparent PAR (and so a decline in BD) ‘diluting’ SCP concentration (Clement, 2005). It seems unlikely that the declines in SCP concentration are a result of changes in PAR affecting concentration profiles. The relationship between SCP concentration and BD suggests the former is largely independent of the latter given all R^2 values are relatively small (<0.3) and no relationships are significant (Figure 5.22). The logic of this test is slightly flawed. One may expect a slight positive relationship between SCP concentration and BD as the upper levels of the acrotelm (i.e. less consolidated peat) are likely to have fewer SCPs relative to the lower levels of the acrotelm/catotelm (i.e. peat of greater BD) because UK atmospheric deposition of SCPs has been falling for approximately forty years (Rose and Appleby, 2005). Nevertheless, that none of the relationships between SCP concentration and BD are significant suggests SCP concentration is independent of substrate characteristics and the decline in near-surface SCP concentrations is likely to reflect the decline in UK-wide SCP deposition.

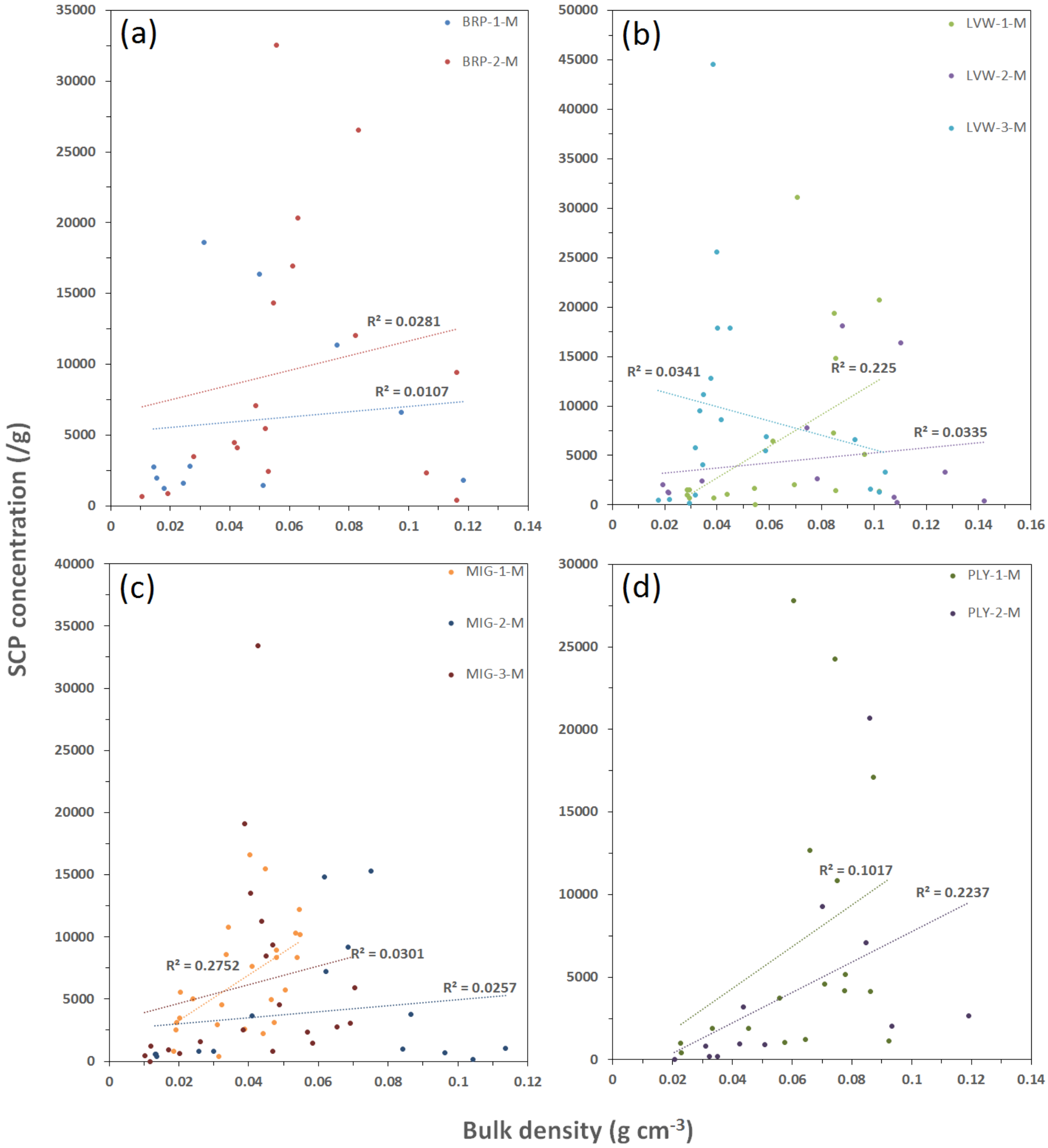


Figure 5.22: Relationship between SCP concentration and bulk density for master core samples. Panel A: BRP, Panel B: LVW, Panel C: MIG and Panel D: PLY. Only those samples deemed to represent peat that had accumulated since the start of the SCP record (AD 1850-present) were included in this analysis. Axes optimised by site.

5.3.3.3 Independent age markers as evidence for ^{210}Pb mobility

Comparing differing chronological methods can reveal systematic differences between approaches (e.g. *'SCPs produce consistently younger ages than ^{210}Pb modelled ages'*). A range of results were reported when independent chrono-stratigraphic markers (^{137}Cs, ^{241}Am and SCPs) were compared with CRS models (Figure 5.23). Nuclear activity markers (^{137}Cs and ^{241}Am) generally exhibited good agreement with CRS dates, aside from aforementioned 'adjusted' models (Sections 5.2.4 and 5.2.9). If one uses a potentially mobile chronological marker (^{137}Cs) to validate a potentially mobile dating method (^{210}Pb) then one approaches the realm of circular reasoning (*sensu* Parry, 2011). It may not be possible to validate a CRS model based solely on radiometric analyses (Section 3.4.3) despite good agreement occurring between differing methods in many cases (e.g. BRP-1-M, BRP-2-M, LVW-1-M, LVW-3-M, MIG-1-M, MIG-2-M and MIG-3-M). Many report the importance of water table depth (WTD) in determining the robustness of a ^{210}Pb chronology in that wetter sites produce poor chronologies by mobilizing unsupported ^{210}Pb (Belyea and Warner, 1994; Parry *et al*. 2013). Quantitative WTD is not presented in this thesis, merely observations from field notes. The two cores which show poor agreement between initial CRS models and independent markers (LVW-2-M and PLY-1-M) were both recovered from dry sites. In contrast, MIG-1-M and MIG-3-M were both recovered from wet sites and both produced reasonable agreement between CRS and ^{137}Cs age estimates. On a qualitative basis WTD does not appear to be affecting radiometrically determined chronologies, or it is affecting radiometric methods in the same way (i.e. circular reasoning).

There appears to be no systematic difference between ages expected by SCP dates and those expected by the CRS model (Figure 5.23). The AD 1850 SCP marker has been deemed the hardest to accurately identify given lower fallout around this time (Parry, 2011). Within error agreement is recorded for sites at LVW and the MIG whilst both BRP and PLY AD 1850 markers occur at depths where the CRS model predicts a more recent age. Better agreement is recorded for the AD 1955 SCP marker. It is worth noting that where the AD 1955 SCP marker does deviate from the CRS model, all deviations occur so that the CRS dates are older than the SCP horizon. Variable results were recorded for the AD 1976 SCP marker. At three of the four sites where the AD 1976 SCP marker deviates from the CRS dates (LVW, MIG and PLY) deviations again occur as CRS dates being older than SCP dates. It may that the SCPs have become mobile and have migrated downward meaning CRS dates are older than SCP horizons, though this is uncertain.

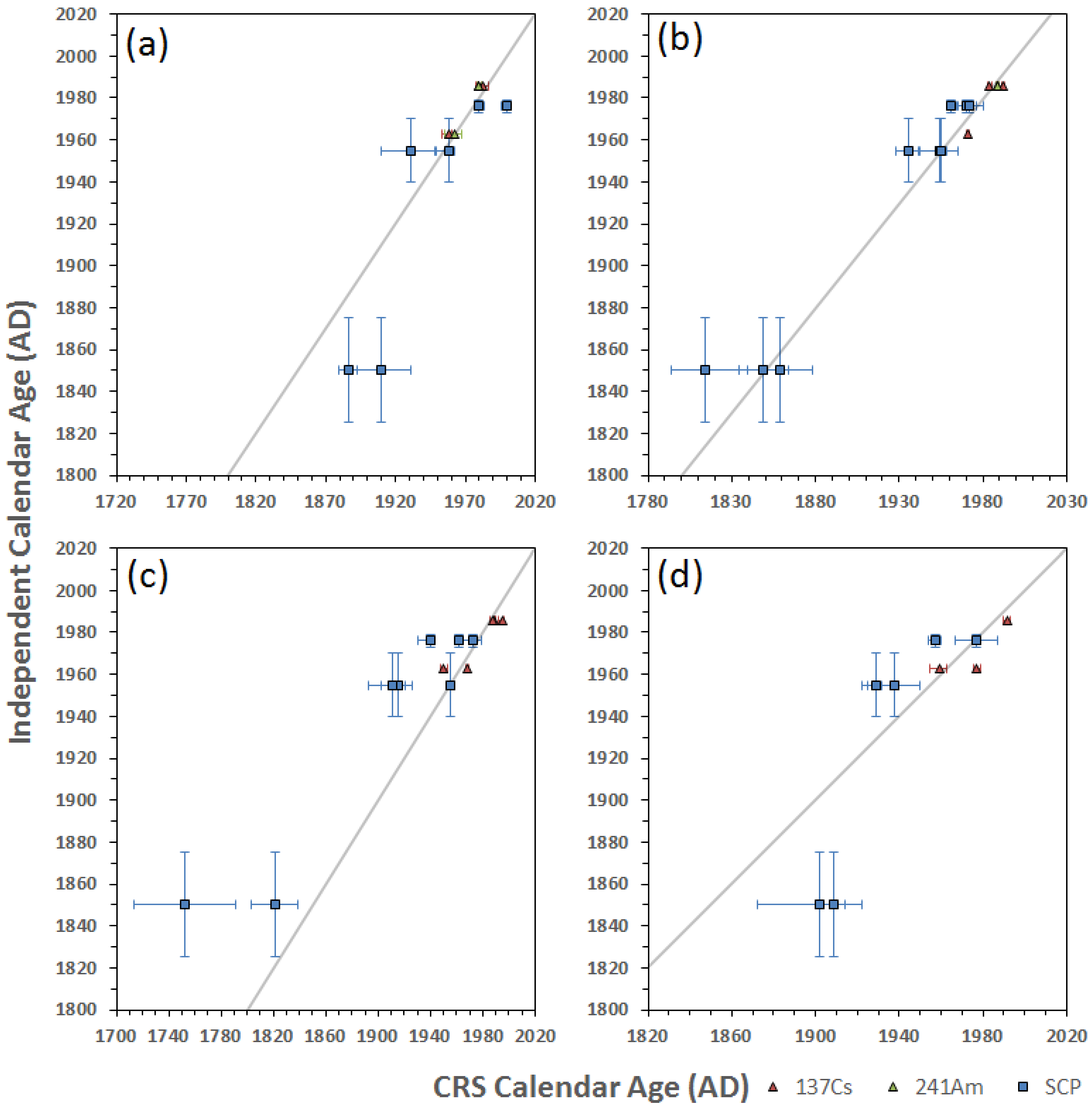


Figure 5.23: Comparison of independent age markers with the CRS modelled ages. Panel A: BRP, Panel B: LVW, Panel C: MIG and Panel D: PLY. 1:1 line represents agreement. X-axis errors relate to sampling error according to the CRS timescale and SCP y-axis errors relates to chronological error. Accumulation rates were considered constant should SCPs be located at a depth beyond the limits of the CRS model. CRS chronologies for LVW-2-M and PLY-1-M are the ‘initial’ CRS chronologies. Axes optimised by site.

Identifying dateable horizons in an SCP concentration profile can be problematic (Section 5.3.3.2) and will affect the comparison in Figure 5.23. The AD 1850 marker was assigned to the deepest SCP occurring sample above which SCPs were consistently recorded (Section 3.4.4) to identify the signal for mass industrialisation the horizon represents (Rose and Appleby, 2005). On balance, this approach to identifying AD 1850 produced less error across the dataset compared to assigning AD 1850 to the deepest sample where an SCP was recorded. ‘Isolated’ SCPs were recorded below the designated AD 1850 marker horizon in BRP-1-M, LVW-2-M, MIG-2-M, PLY-1-M and PLY-2-M. The designated AD 1850 SCP horizon agrees with the CRS model in PLY-2-M (Figure 5.20) and LVW-2-M if assuming a constant PAR beyond CRS model limits (Figure 5.8). The AD 1850 horizon is younger than the equivalent CRS date in MIG-2-M (Figure 5.14), though assigning it to 28cm produces greater agreement with the model than if AD 1850 were assigned to isolated SCPs at 46 or 38cm (Figure 5.13). The AD 1850 horizon is older than the CRS date in BRP-1-M (Figure 5.2) and PLY-1-M (Figure 5.17). Assigning AD 1850 to deeper SCP horizons (34 or 26cm) for BRP-1-M (Figure 5.1) produces worse agreement with the CRS model which places AD 1850 *ca.* 22cm (Figure 5.2). Assigning AD 1850 to 36cm in PLY-1-M rather than 32cm (Figure 5.17) does improve the agreement with the CRS model (Figure 5.18), but this is the only master core where using an alternative method to identifying AD 1850 improves agreement with the CRS model. ‘Isolated’ SCPs are likely indicators of small scale (pre-AD 1850) industry in the regions surrounding BRP-1-M, LVW-2-M, MIG-2-M, PLY-1-M and PLY-2-M. Low numbers of SCPs were released by pre-AD 1850 industrialisation, as evidenced by Central- and North-Wales lake records (Rose *et al.* 1995). Gedye (1998) also found evidence of SCPs in Cumbrian mires biostratigraphically dated to *ca.* AD 1550.

Rose and Appleby (2005) proposed the ‘cumulative %’ approach to limit subjectivity in dating SCP profiles. In this case, 0% is the start of the record (SoR) and 100% concentration is the peak, with dates assigned to each 10% increment between these features of the concentration curve. The cumulative % approach was modelled to examine the effect on the age/depth relationship for each master core relative to the ‘visual’ SCP method used thus far (‘SCP Cum’ and ‘SCP Vis.’ respectively; Figure 5.24). It was not possible to do this for MIG-1-M given the AD 1850 SCP horizon was not recorded (Section 5.2.6).

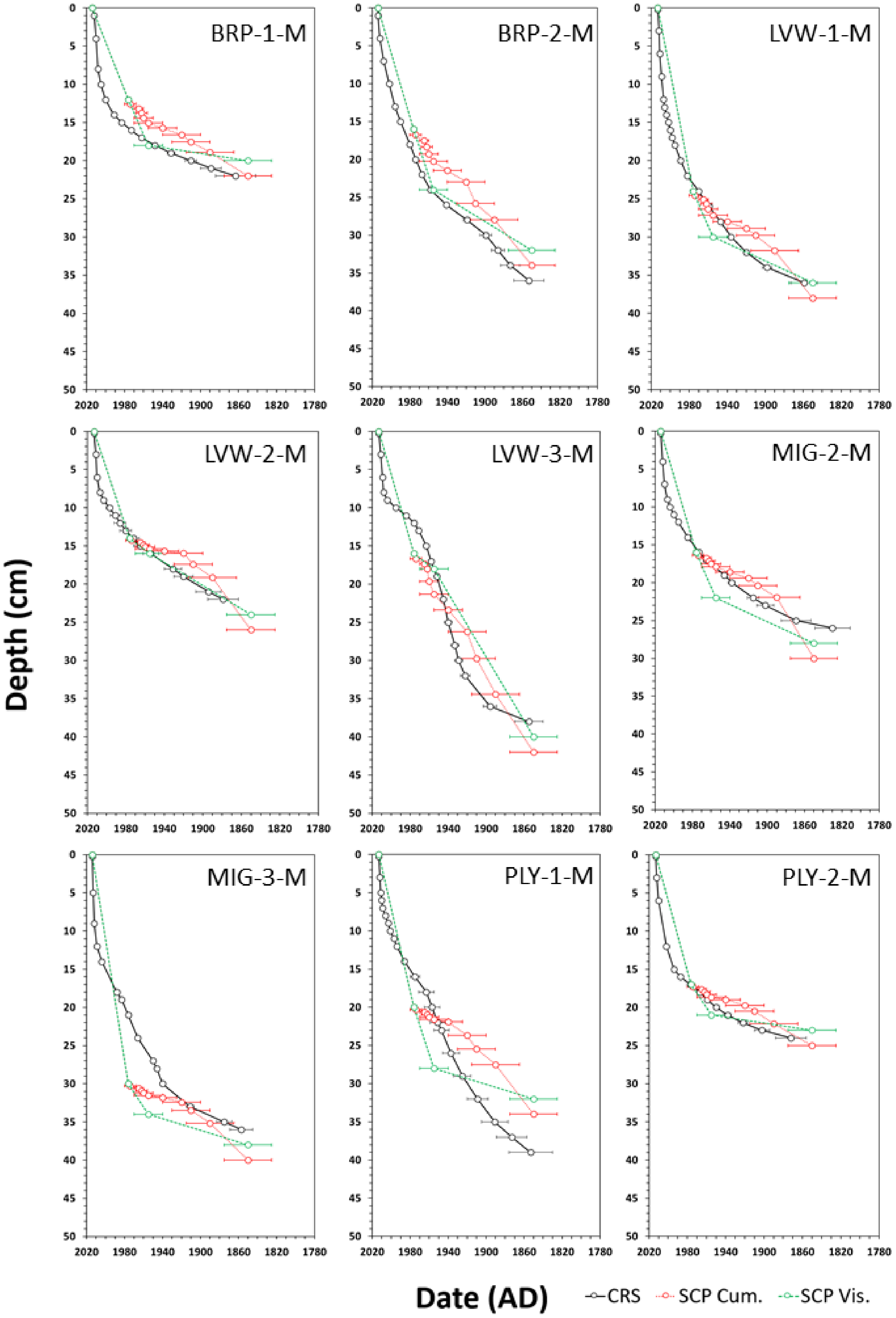


Figure 5.24: Comparison of CRS master core age/depth models with those constructed using dating approaches referred to in the text.

A key difference between the cumulative and visual SCP models is their approach to determining the AD 1850 SoR. Rose and Appleby's (2005) implementation of the cumulative approach considers the SoR as the measured increment below the deepest occurrence of SCPs. The cumulative % approach thereby predicts a deeper AD 1850 horizon relative to the visual SCP model. Following Rose and Appleby (2005) on balance reduces agreement between CRS models and the AD 1850 SCP horizon (Figure 5.25).

The measured increment below the deepest occurrence of SCPs may not truly represent 0% given SCP concentrations may 'stop' anywhere between these two measured points. One can in theory determine the 0% concentration horizon by assuming a constant decline in SCP concentration below the last measured point (i.e. between the visual SoR and the Rose and Appleby (2005) cumulative SoR). Such a method, hereafter termed 'adjusted 0%', assumes a constant PAR and minimal SCP fallout variability; a constant PAR is an assumption of the SCP technique (Section 5.3.3.2) and the assumption of only minor changes in SCP production *ca.* AD 1850 fits with industrial records (Rose *et al.* 1995; Swindles *et al.* 2015b). The adjusted 0% method in cases reduces the error between the Rose and Appleby (2005) cumulative method for identifying AD 1850 and the CRS model (e.g. LVW-1-M, LVW-3-M, MIG-2-M and MIG-3-M; Figure 5.25), though it can also increase the disparity between the horizons (e.g. BRP-1-M, BRP-2-M and PLY-1-M). The adjusted 0% SoR occurs deeper than the Rose and Appleby (2005) SoR where cumulative SCP increases are 'slow' (e.g. LVW-2-M and PLY-2-M; Figure 5.25). The Rose and Appleby SoR is certainly more appropriate in these cases given the SoR has been confirmed by a zero SCP concentration count (e.g. 26 cm in LVW-2-M; Figure 5.7).

It is inconclusive whether the adjusted 0% method offers a robust alternative to the Rose and Appleby (2005) approach to identifying the cumulative SoR. The adjusted 0% method may be a preferable alternative where SCP sampling resolution is relatively coarse (≥2cm). Ultimately, no SCP method (visual, Rose and Appleby cumulative or adjusted 0% cumulative) was consistently better at replicating the CRS model. It seems prudent therefore to retain all SCP models to assess their ability to reconstruct carbon accumulation in subsequent chapters.

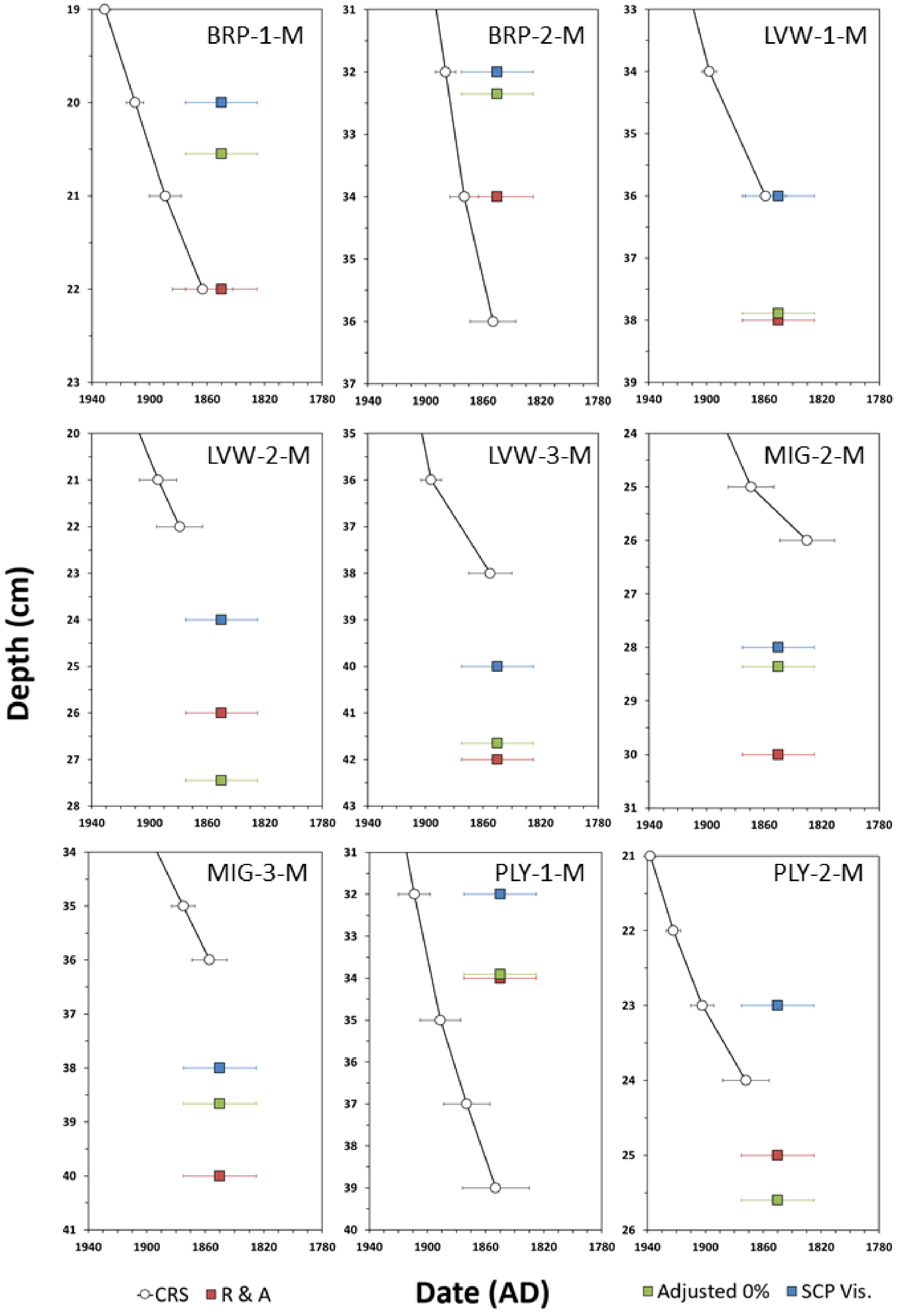


Figure 5.25: Comparison of CRS age/depth model with SCP SoR AD 1850 markers according to methods discussed in the text. SCP marker is assigned according to the cumulative Rose and Appleby (2005) method (‘R & A’), the newly proposed cumulative method (‘adjusted 0%’) and the visual method (‘SCP Vis.’). Y-axis optimised by core.

The potential for regional scale differences in SCP fallout histories has not been addressed as SCPs were used in this thesis to 1) validate CRS models and 2) to assess their ability to reconstruct PARs across the suite of replicate cores (Section 5.4). UK SCP fallout is spatially and temporally variable due to regional-level industrial variability (Rose and Harlock, 1998). The average SCP horizon ages used in this thesis mask variation in SCP fallout (Rose and Appleby, 2005) as the AD 1850 and 1955 markers have been recorded across periods of AD 1830-1860 and AD 1940-1965 respectively (Rose *et al.* 1995; Swindles *et al.* 2015b). Poor regional calibration of SCP chronologies may therefore explain observed differences between SCP and CRS dates (Figure 5.23). No post-deposition mobility of either dating method may have occurred and the CRS models may be providing more accurate dates for temporal changes in SCP fallout than the average dates of Rose and Appleby (2005). It seems prudent to use SCP markers to validate CRS models rather than assuming all chronological methods are accurate, hence regional-scale SCP fallout is not elaborated on here.

5.3.4 Summary of master core dating results

Master core CRS models are deemed appropriate to reconstruct carbon accumulation in subsequent chapters because most models display reasonable agreement with independent age markers. Some discrepancies were found between CRS and SCP dates, though not in a manner which suggests a systematic issue with SCP records. As such, SCPs are appropriate for constructing age/depth models for the replicate cores (see below).

5.4 Replicate core dating results

Replicate cores were in many cases specifically selected for their management history/degraded condition (Section 4.3). SCPs were the only method used to reconstruct chronologies for the replicate cores with dating features assigned to each concentration profile using both visual and cumulative methods (Section 5.3.3.3). The AD 1850 cumulative SoR is determined using the adjusted 0% method unless the 0% concentration horizon occurs deeper than the next measured increment. In this case, the Rose and Appleby (2005) SoR is used. The results for each replicate core are discussed below, before patterns across the replicate dataset are highlighted in Section 5.4.5.

5.4.1 Bryniau Pica replicate core SCP results

SCPs were recorded in all seven BRP replicate cores. All cores followed the typical model of SCP accumulation, save for BRP-2-REP-B which exhibits peaks at 24 and 16cm (Panel 4; Figure 5.26). The 24cm BRP-2-REP-B peak was designated as AD 1976 following Rose *et al*. (1995). The range of peak values (11,076 - 20,974 g DM^{-1}) was within acceptable limits of previous studies (Section 5.3.3.2). Whilst SCP accumulation can be affected by several factors (Section 5.3.3.2) a basic comparison can be performed between replicates and their relevant master core. The SCP records for BRP-1-REP-A and B begin at 28 and 32cm respectively. This is lower than the BRP-1-M SoR at 22cm, though given isolated SCPs were recorded below this depth in BRP-1-M (Figure 5.1), the lower replicate sampling resolution may not be detecting the 'true' AD 1850 SCP horizon. BRP-2-REP-A and B SCP records both begin at 40cm, deeper than the start at 32cm for BRP-2-M (Figure 5.3). The resultant age/depth models are presented in Figure 5.27.

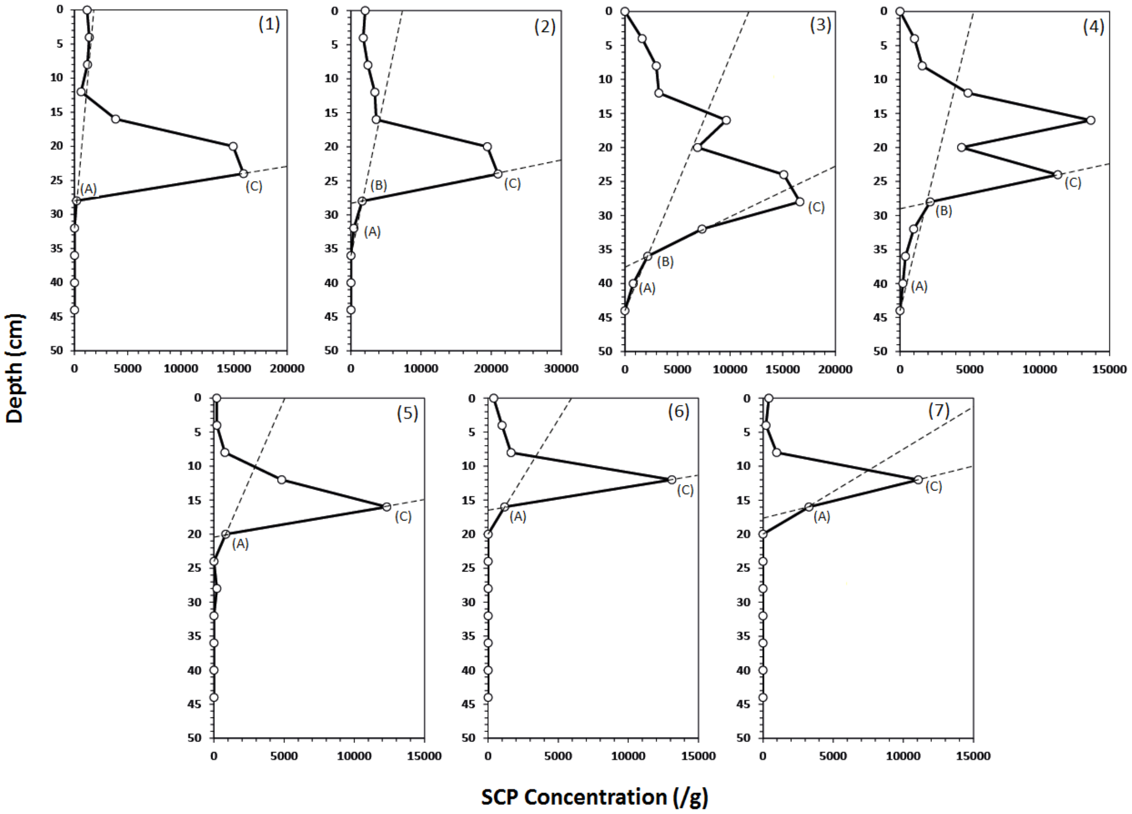


Figure 5.26: SCP concentration profiles for all BRP replicate cores. Panel 1: BRP-1-REP-A, Panel 2: BRP-1-REP-B, Panel 3: BRP-2-REP-A, Panel 4: BRP-2-REP-B, Panel 5: BRP-3-REP-A, Panel 6: BRP-4-REP-A and Panel 7: BRP-4-REP-B. X-axes optimised by core.

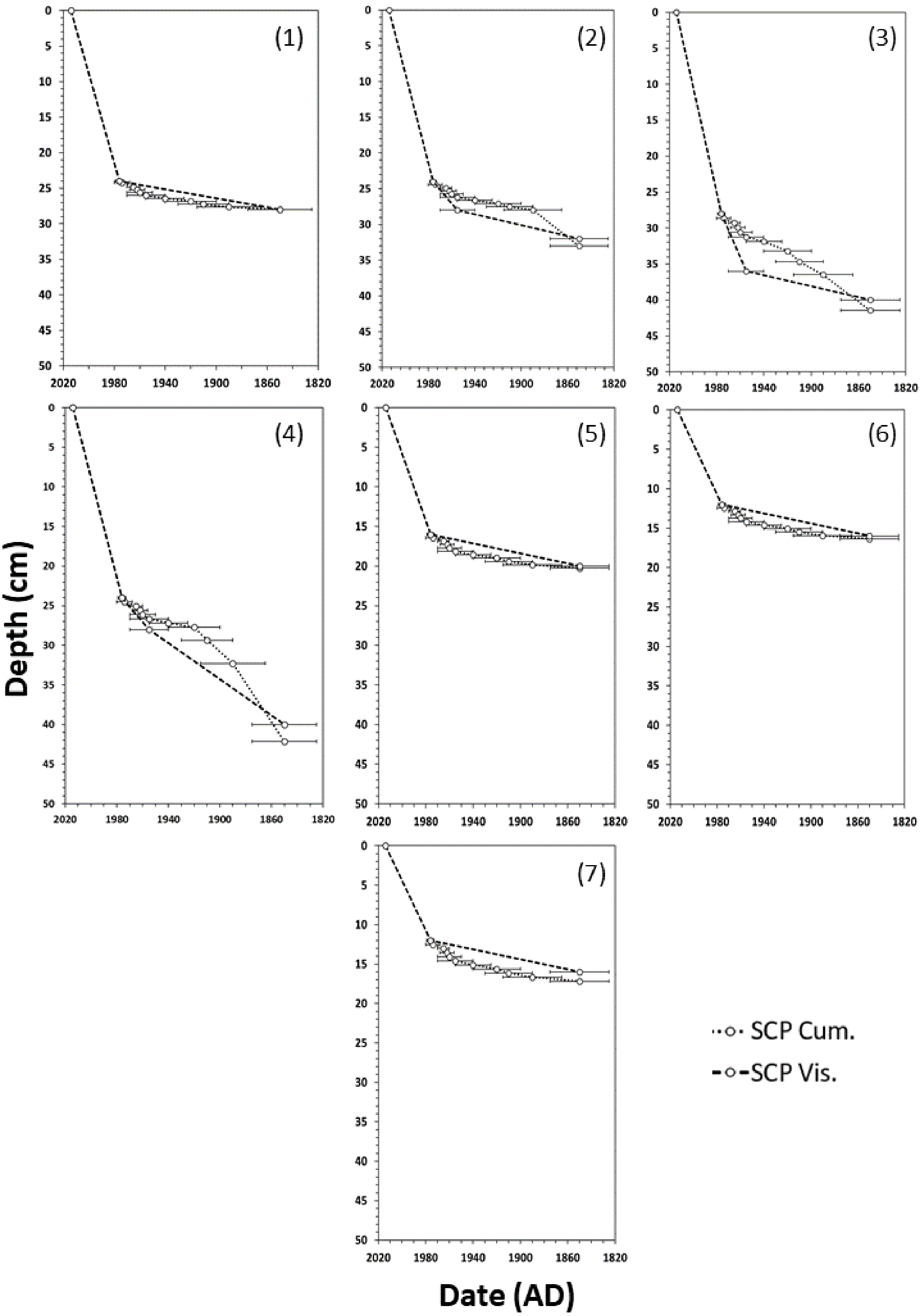


Figure 5.27: Age/depth models for all BRP replicate cores. Panel lettering same as Figure 5.26.

5.4.2 Lake Vyrwny replicate core SCP results

SCPs were recorded in all four LVW replicate cores. Most cores followed the typical model of SCP accumulation though LVW-1-REP-C exhibits an early peak at 36cm (Panel 3: Figure 5.28). This does not seem to be a true double peak, but rather just variability in SCP concentration that is accentuated by the 4cm sampling resolution. Peak concentrations ranged from 10,765 - 25,076 g DM^{-1} with an average of 15,835 g DM^{-1}. This range is similar to other studies (Section 5.3.3.2). The LVW-1-REP-A SCP record commences at 24cm and at 20cm in LVW-1-REP-B, both shallower than LVW-1-M (36cm) (Figure 5.5). The peak for both LVW-1-REP-A and B is also nearer to the surface than the 24cm depth concentration peak of LVW-1-M. The SCP record for LVW-1-REP-C begins much deeper at 44cm, though it was recovered from a saddle at LVW rather than the watershed site of LVW-1-M (Section 4.3.2) and it may be accumulating at a faster rate as the core was characterised by abundant *Sphagnum* (Section 6.2.2). The resultant age/depth models are presented in Figure 5.29.

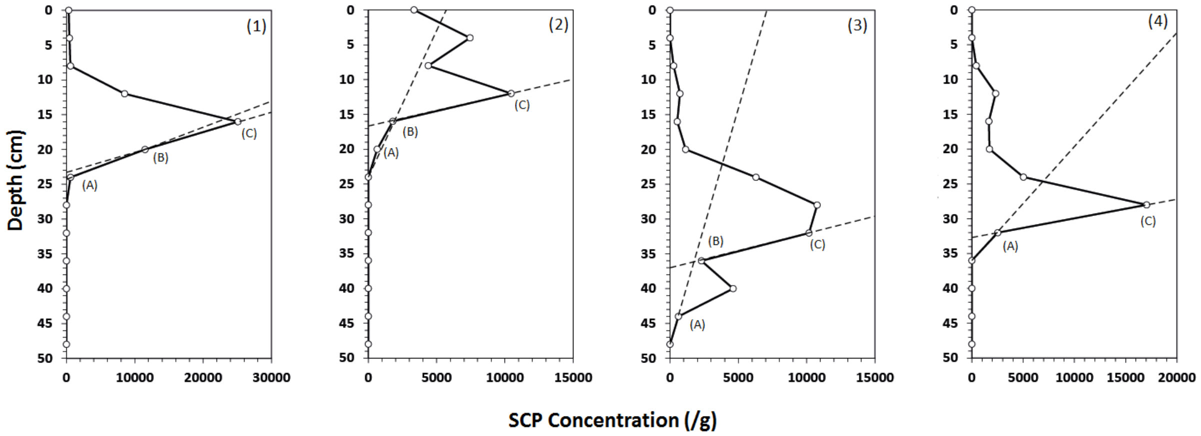


Figure 5.28: SCP concentration profiles for all LVW replicate cores. Panel 1: LVW-1-REP-A, Panel 2: LVW-1-REP-B, Panel 3: LVW-1-REP-C and Panel 4: LVW-1-REP-D. X-axes optimised by core.

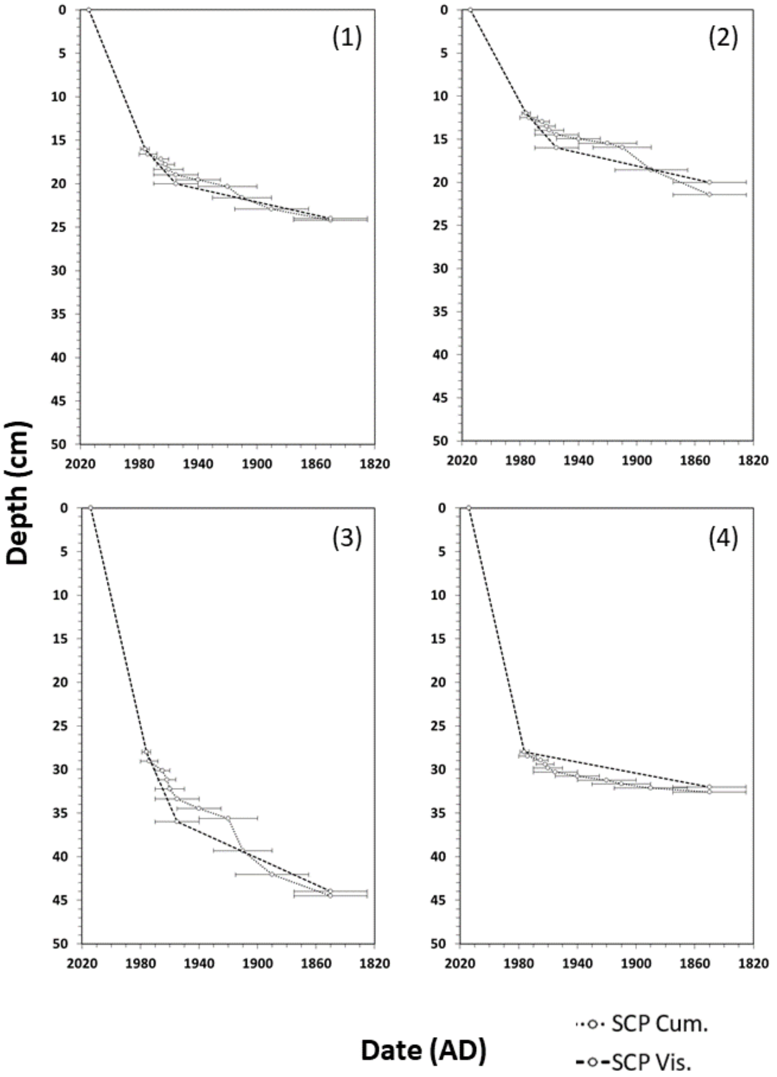


Figure 5.29: Age/depth models for all LVW replicate cores. Panel lettering same as Figure 5.28.

5.4.3 Migneint replicate core SCP results

SCPs were recorded in all twenty-four MIG replicate cores. Peak concentrations ranged from 5,214 - 33,746 g DM^{-1} with an average of 17,976 g DM^{-1}. Most peak values are within expected ranges (Section 5.3.3.2). Many cores followed the typical model of SCP accumulation. There were unexpected features in the profiles with cores exhibiting double peaks (e.g. Panel 5; Figure 5.30) or long tails (e.g. Panel 12; Figure 5.30). Regarding similarities between master and replicate cores, MIG-1-REP A and C (Panels 1 and 3 respectively; Figure 5.30) both exhibit a similarity with MIG-1-M (Figure 5.11) in that the start of the SCP profile (AD 1850) is not recorded in either replicate. The start of the profile is recorded in MIG-1-REP-B (Panel 2; Figure 5.30) though this was recovered from a higher density substrate near to MIG-1-M. Peat at MIG-1-REP-B may therefore not be accumulating as fast as at MIG-1-M, MIG-1-REP-A or B. This variability in PARs between proximal cores demonstrates the heterogeneity of peat accumulation. The SCP record for MIG-2-REP-A (Panel 4; Figure 5.30) matches that MIG-2-M (Figure 5.13), as both records begin at 28cm and peak at 16cm. MIG-3-REP-B and C were recovered from a drained area and so are not directly comparable with MIG-3-M. The SCP record for MIG-3-REP-A (Panel 11; Figure 5.30) is similar to MIG-3-M (Figure 5.15), these two beginning at 40 and 38cm respectively. The resultant age/depth models are presented in Figure 5.32, Figure 5.33 and Figure 5.34.

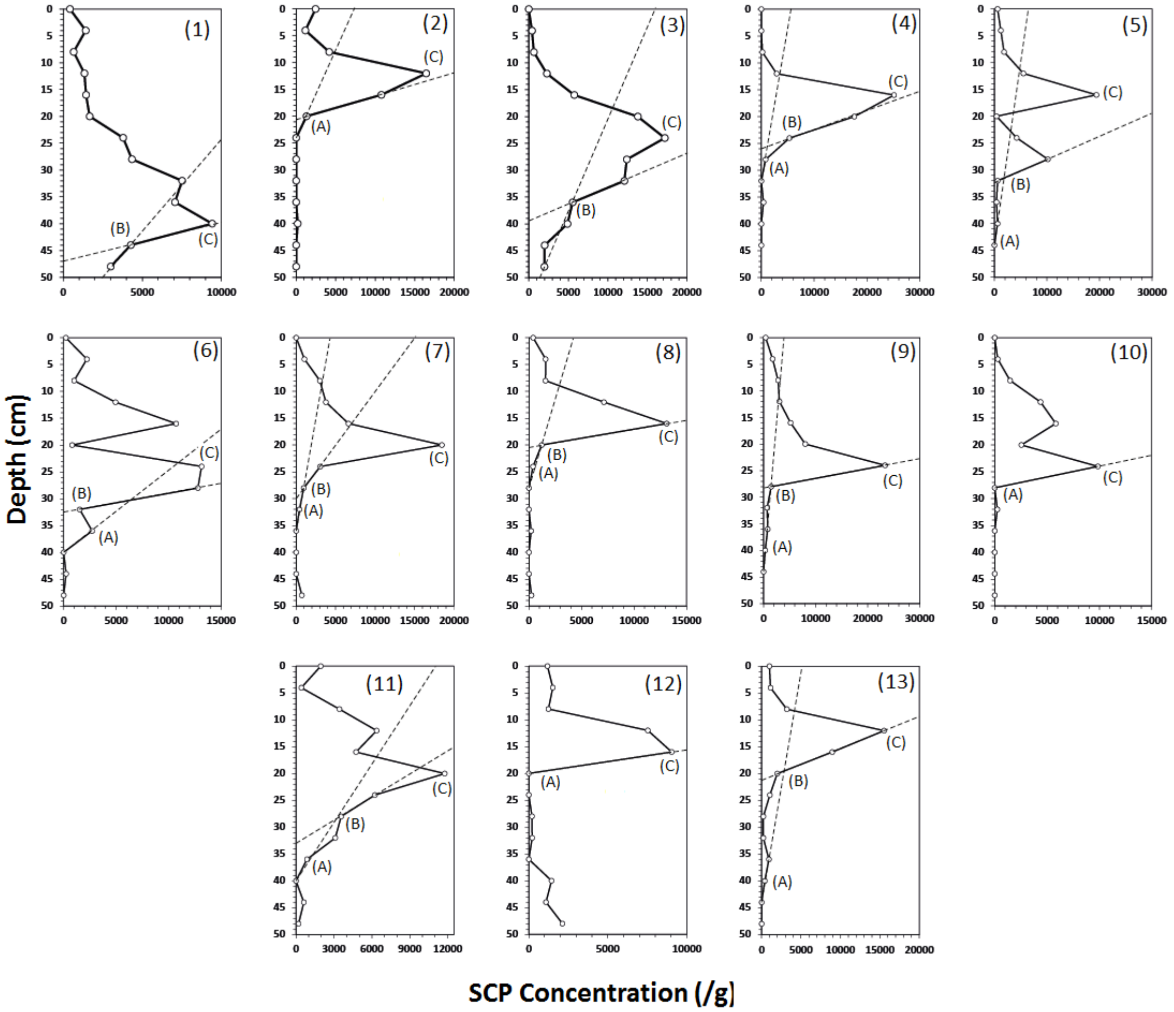


Figure 5.30: SCP concentration profiles for all MIG replicate cores. Panel 1: MIG-1-REP-A, Panel 2: MIG-1-REP-B, Panel 3: MIG-1-REP-C, Panel 4: MIG-2-REP-A, Panel 5; MIG-2-REP-B, Panel 6; MIG-2-REP-C, Panel 7: MIG-2B-REP-A, Panel 8: MIG-2B-REP-B, Panel 9: MIG-2B-REP-C, Panel 10: MIG-2B-REP-D, Panel 11: MIG-3-REP-A, Panel 12: MIG-3-REP-B and Panel 13: MIG-3-REP-C. X-axes optimised by core.

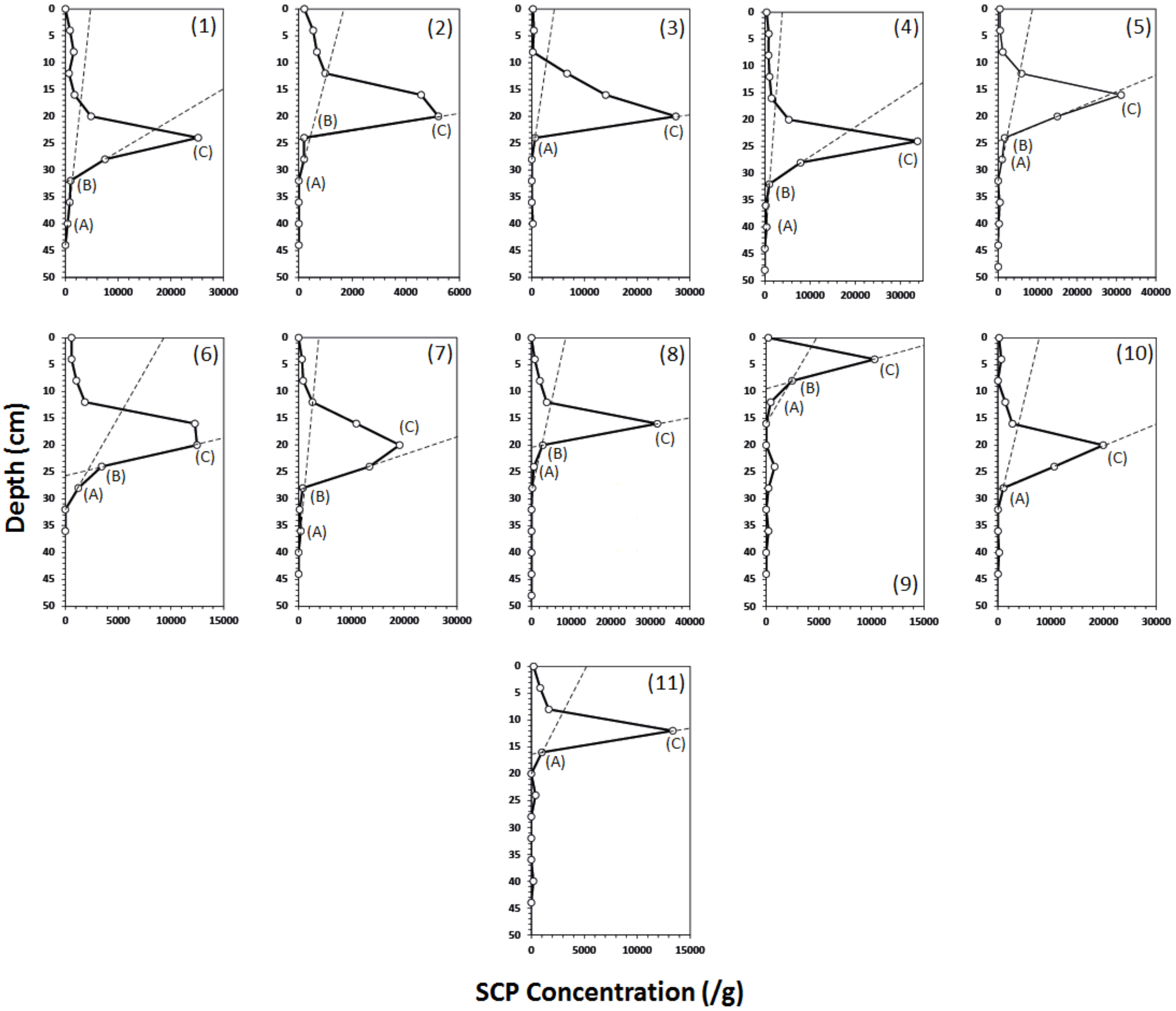


Figure 5.31: SCP concentration profiles for all MIG replicate cores. Panel 1: MIG-4-REP-A, Panel 2: MIG-4-REP-B, Panel 3: MIG-4-REP-C, Panel 4: MIG-4-REP-D, Panel 5; MIG-5-REP-A, Panel 6; MIG-5-REP-B, Panel 7: MIG-5-REP-C, Panel 8: MIG-5-REP-D, Panel 9: MIG-6-REP-A, Panel 10: MIG-6-REP-B and Panel 11: MIG-6-REP-C. X-axes optimised by core.

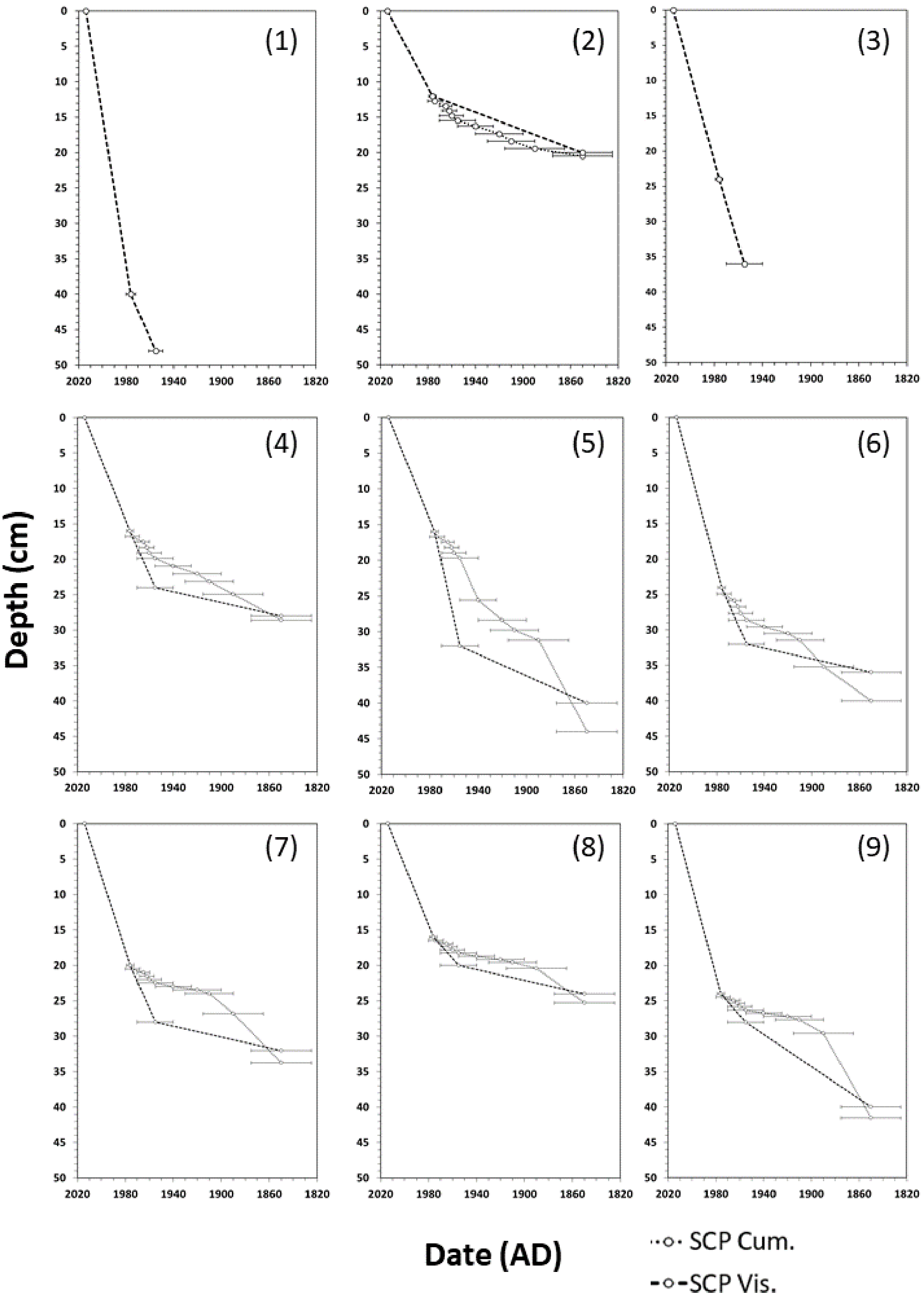


Figure 5.32: Age/depth models for all MIG replicate cores. Panel 1: MIG-1-REP-A, Panel 2: MIG-1-REP-B, Panel 3: MIG-1-REP-C, Panel 4: MIG-2-REP-A, Panel 5; MIG-2-REP-B, Panel 6; MIG-2-REP-C, Panel 7: MIG-2B-REP-A, Panel 8: MIG-2B-REP-B and Panel 9: MIG-2B-REP-C.

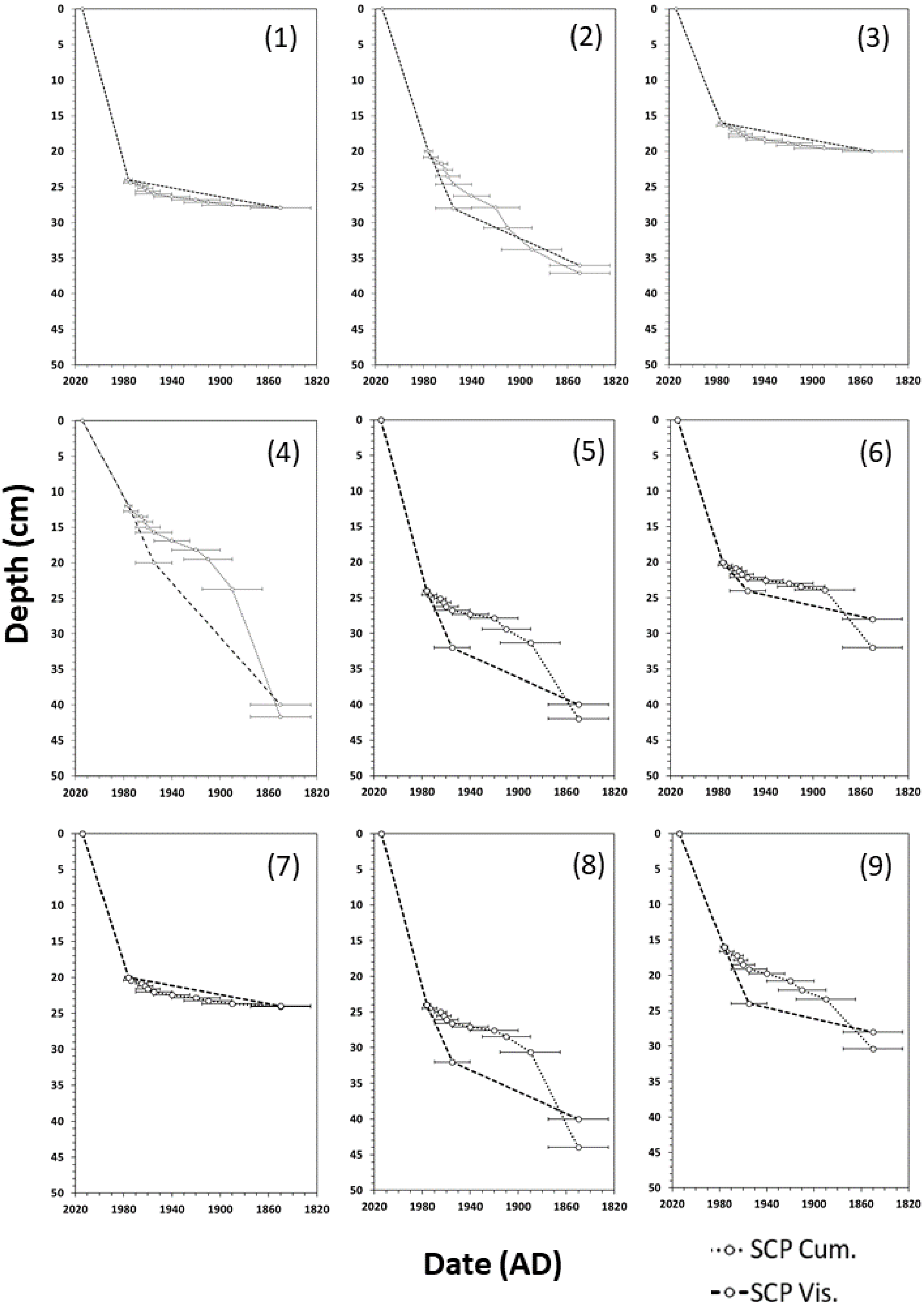


Figure 5.33: Age/depth models for all MIG replicate cores. Panel 1: MIG-2B-REP-D, Panel 2: MIG-3-REP-A, Panel 3: MIG-3-REP-B, Panel 4: MIG-3-REP-C, Panel 5: MIG-4-REP-A, Panel 6: MIG-4-REP-B, Panel 7: MIG-4-REP-C, Panel 8: MIG-4-REP-D and Panel 9; MIG-5-REP-A.

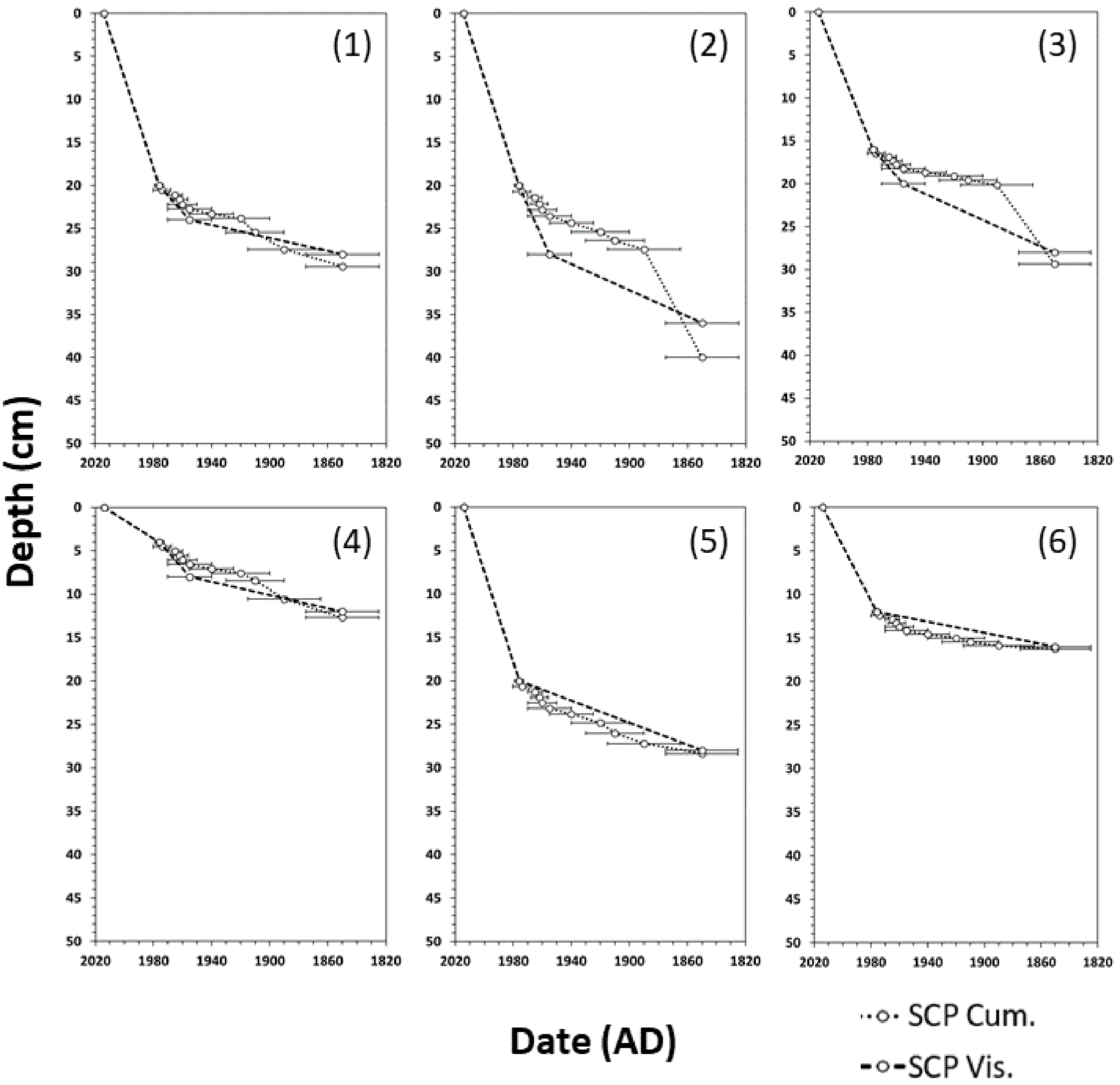


Figure 5.34: Age/depth models for all MIG replicate cores. Panel 1; MIG-5-REP-B, Panel 2: MIG-5-REP-C, Panel 3: MIG-5-REP-D, Panel 4: MIG-6-REP-A, Panel 5: MIG-6-REP-B and Panel 6: MIG-6-REP-C.

5.4.4 Plynlimmon replicate core SCP results

SCPs were recorded in all five PLY replicate cores. Peak concentrations ranged from 6,962 - 14,944 g DM^{-1} with an average of 10,995 g DM^{-1}. The range is within acceptable limits (Section 5.3.3.2). Most cores followed the ‘typical’ model of SCP accumulation though some isolated SCPs occurred below the designated AD 1850 SCP horizon. Regarding similarities between master and replicate cores, PLY-2-REP-B, C and D (Panels 2, 3 and 4 respectively; Figure 5.35) all agree with PLY-2-M (Figure 5.19) in placing the beginning of the SCP record *ca*. 24cm. PLY-2-REP-A and E (Panels 1 and 5 respectively; Figure 5.35) both have SCP records beginning nearer to the surface at 16cm. The resultant age/depth models are presented in Figure 5.36.

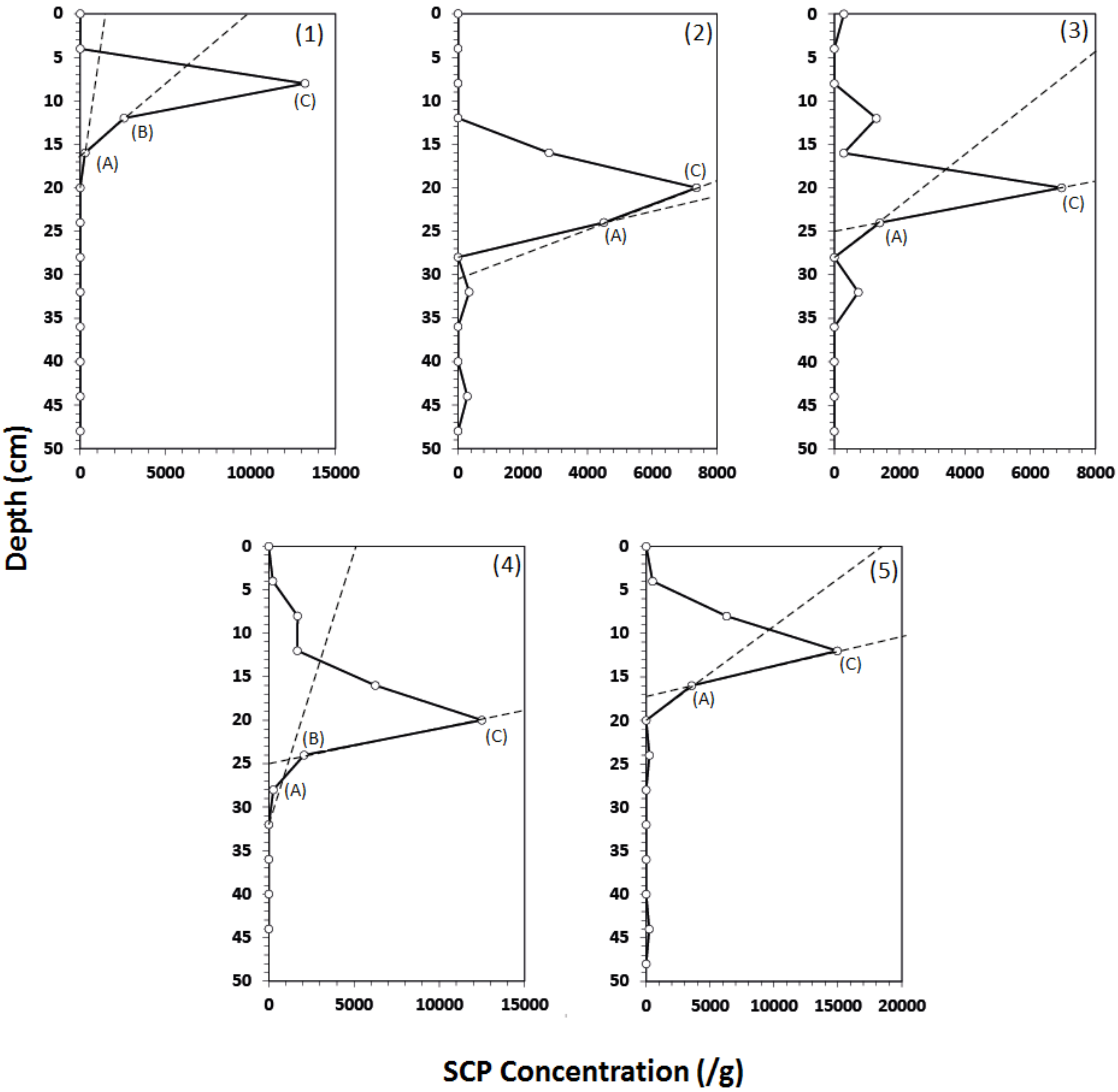


Figure 5.35: SCP concentration profiles for all PLY replicate cores. Panel 1: PLY-2-REP-A, Panel 2: PLY-2-REP-B, Panel 3: PLY-2-REP-C, Panel 4: PLY-2-REP-D and Panel 5; PLY-2-REP-E. X-axes optimised by core.

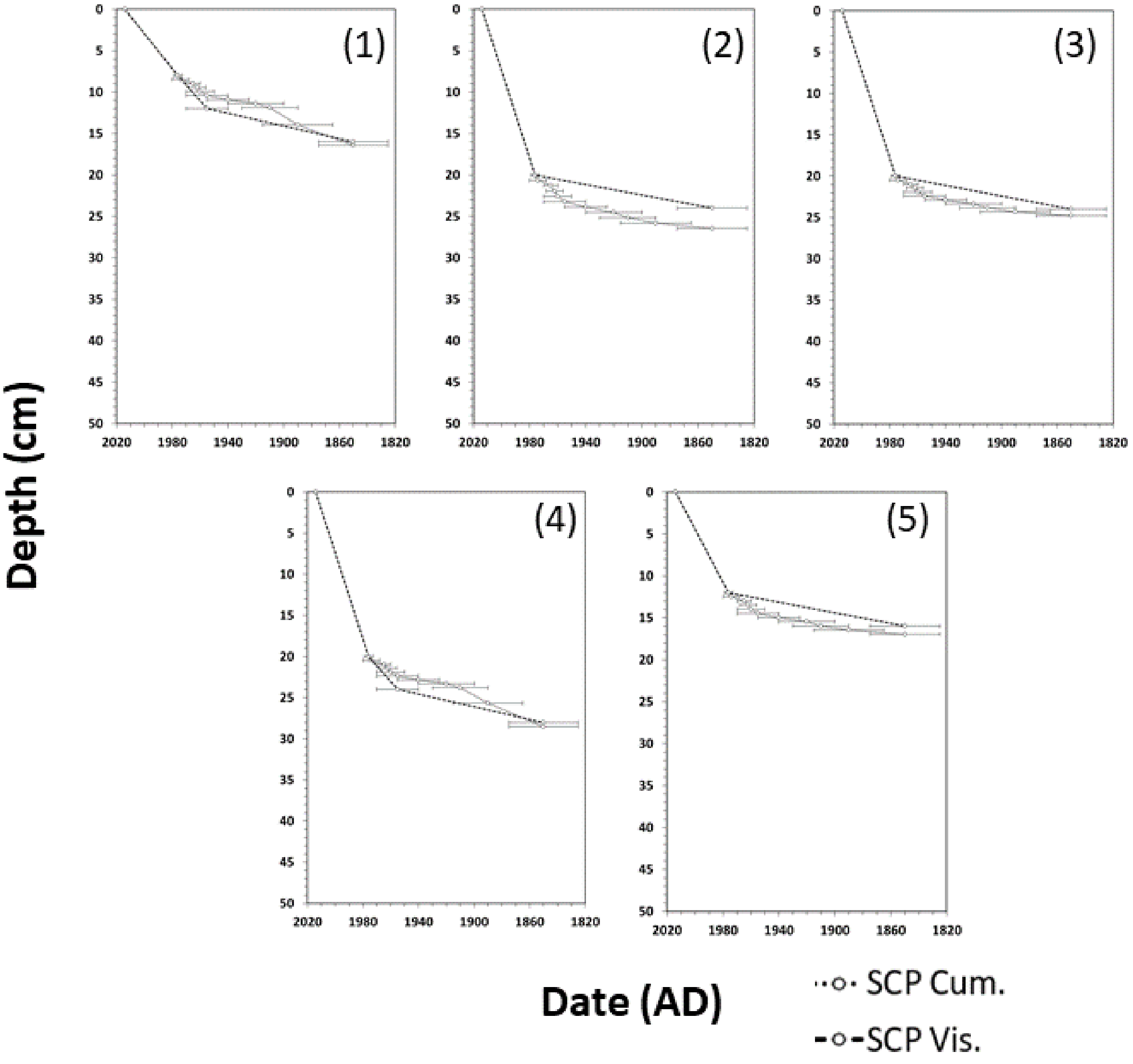


Figure 5.36: Age/depth models for all PLY replicate cores. Panel lettering same as Figure 5.35.

5.4.5 Discussion of replicate core chronologies

The reliability of the replicate chronologies cannot be assessed in the same manner as the master core chronologies (Section 5.3) because SCP analysis was the sole tool used to date replicate cores. Comparisons between master and replicate core SCP profiles do not inherently provide a reasonable assessment of the SCP chronologies given PARs can vary over fine spatial scales (Section 2.2.3). Dissimilar SCP profiles from cores recovered within the same sub-site may therefore be providing accurate chronologies for their respective sampling point. Such dissimilarity likely demonstrates that nearby peat cores have been affected by differing environmental drivers which have caused PARs to vary over fine spatial scales. Plotting replicate core SCP concentrations against BD suggests the former is largely independent of the latter given no R^2 exceeds 0.19 and no relationship is significant (Figure 5.37). The reliability of this test is weakened when applied to replicates because SCPs were sampled at 4cm non-contiguous increments whilst BD was sampled as 4cm contiguous blocks (Section 3.4.4). There is therefore an inconsistency in that the SCP concentration for 8cm is being compared against homogenized BD for peat 8-12cm depth. These results nevertheless suggest that variations in BD are not substantially affecting replicate SCP chronologies.

Dating recent peat accumulation in the replicate cores using a single method is not ideal (Turetsky *et al*. 2004), though in this thesis it was necessitated by financial and temporal restraints. There seems to be no systematic issue affecting SCP profiles across the replicate cores and in some cases they produce good agreement with proximal master core SCP profiles. Divergence from master core SCP profiles is not in itself a reason to consider the replicate chronologies to be inaccurate. These replicate cores were often sampled from areas that had been managed in a specific way (e.g. drained) whilst master cores were purposefully recovered from supposedly intact sites. Differing PARs between master and replicate cores may therefore be expected due to differing management histories. Peat accumulation can also vary substantially over small areas of intact peatlands (Ireland *et al*. 2013; Watson *et al*. 2015; Yang *et al*. 2001) so differing chronologies should not be discounted as inaccurate purely for demonstrating SCP profile variability. Whilst the precision of the age/depth models can be critiqued, the variety of replicate core SCP concentration profiles is likely an excellent illustration of the heterogeneous nature of peat accumulation in blanket peatlands. Therefore, provided uncertainty is appropriately acknowledged, the replicate chronologies are considered suitable to reconstruct apparent rates of carbon accumulation in subsequent chapters. The appropriateness of the SCP Vis. and SCP Cum. models are also discussed in subsequent chapters.

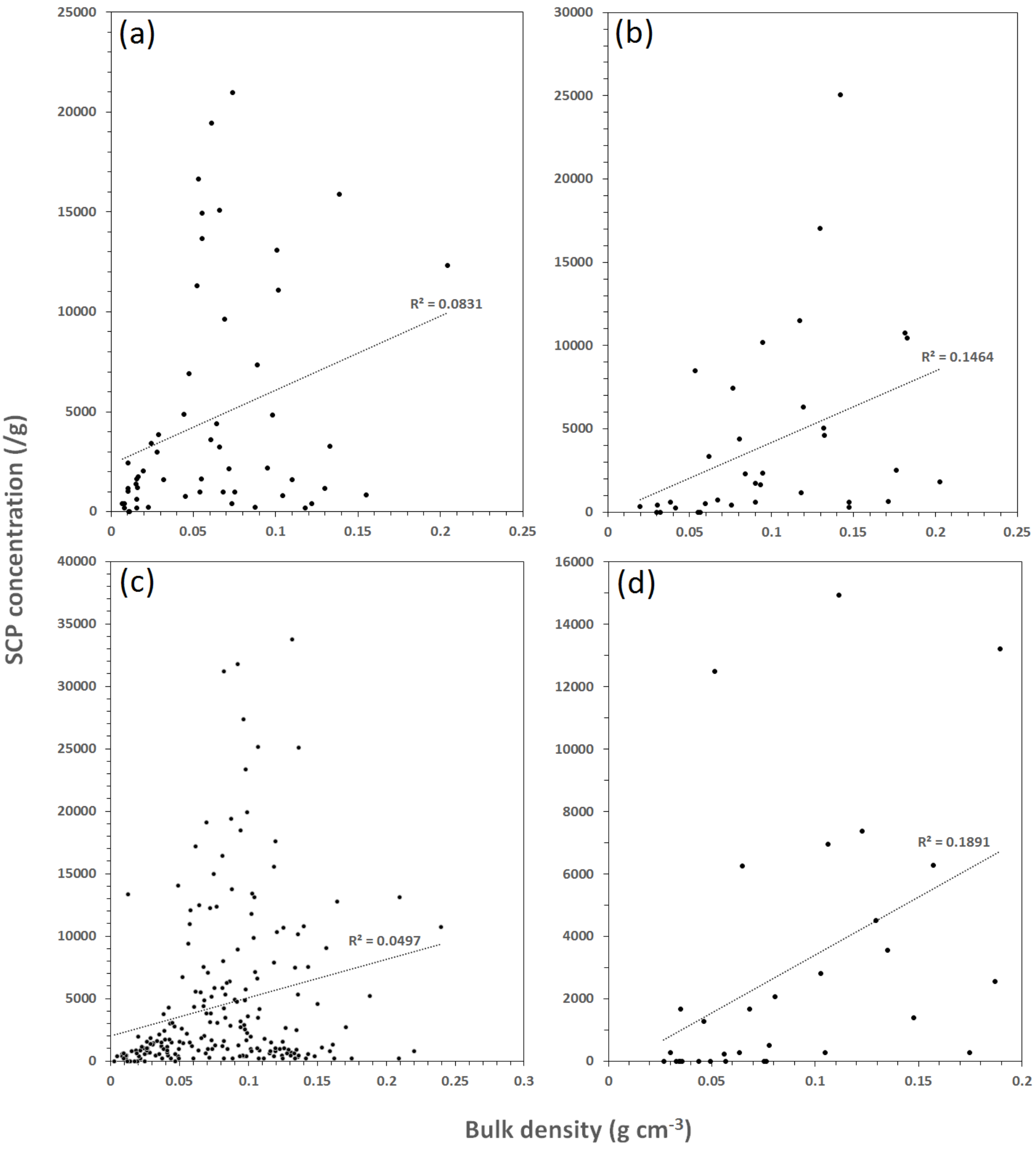


Figure 5.37: Relationship between SCP concentration and bulk density for samples from each replicate core. Panel A: BRP, Panel B: LVW, Panel C: MIG and Panel D: PLY. Only those samples deemed to represent peat that had accumulated since the start of the SCP record (AD 1850-present) were included in this analysis. Axes optimised by site.

5.5 Conclusions

The analysis conducted in this chapter has produced dating frameworks which are used to present data in subsequent chapters. In addition to providing these frameworks, the main conclusions from this chapter are:

- Results suggest that all cores recovered in this study are from actively accumulating blanket peat though variable peat accumulation rates (PAR) were recorded.
- Constant Rate of Supply (CRS) was the most appropriate ^{210}Pb model for reconstructing PAR in this thesis as the non-monotonic unsupported ^{210}Pb profiles all master cores exhibited precluded the use of the Constant Initial Concentration model.
- ^{210}Pb, ^{137}Cs, ^{241}Am and Spheroidal Carbonaceous Particle (SCP) analyses all displayed variable levels of agreement. Agreement between CRS and SCP age estimates could often be improved by excluding isolated SCP occurrences from the profile and considering the start of the SCP record (AD 1850) as that SCP horizon above which SCPs were consistently recorded.
- Multiple cores should be recovered if one is trying to understand spatial heterogeneity in PAR given the notable differences in SCP profiles between proximal replicate cores.

Chapter 6 Peat stratigraphy as evidence of environmental change

6.1 Introduction

The vegetation that grows on or near the surface of a peatland responds to both internal and external drivers of peatland development. Peat therefore acts as a "*scientific archive*" of change (Barber, 1993, page 474) if one analyses changes in vegetation over time. As vegetation is known to respond to the impacts of land use (Section 2.2.4.1), it makes plant macrofossil analysis (Section 3.4.5) an appropriate technique for examining the effect of land use at the sites examined within this thesis.

This chapter presents data acquired through plant macrofossil analysis of peat cores and discusses the resultant trends in species turnover. The first section presents the plant macrofossil data according to site (Bryniau Pica (Section 6.2.1), Lake Vyrnwy (Section 6.2.2), Migneint (Section 6.2.3) and Plynlimmon (Section 6.2.4)). Each of these sub-sections details the approaches used to create ordinations, before the final ordinations are interpreted using the ecological requirements of recorded taxa. The second section considers the assembled macrofossil dataset as a whole and discusses broad similarities and differences exhibited in the vegetation development of the selected blanket peatlands (Section 6.3) before chapter conclusions are presented (Section 6.4).

6.2 Stratigraphy and macrofossil analysis

Plant macrofossil analysis followed the protocol outlined in Section 3.4.5 where macrofossils were quantified using an ordinal five-point abundance scale. These five terms (i.e. 'absent', 'rare') are used exclusively hereafter to describe taxa according to their position on the scale. Macrofossil assemblage diagrams were created in TILIA (Grimm, 2015) with an example diagram provided for each site. Taxa are reported according to their position on the five-point abundance scale with values exaggerated (x10) to aid visual interpretation. A green horizontal line is indicated on each diagrm to denote the approximate position of the acrotelm/catotelm transition (see Chapter 7 for a full discussion on the transition). Remaining macrofossil assemblage diagrams are available in

Appendix A. Simplified stratigraphic diagrams are also presented to aid interpretation of vegetation change and to allow rapid comparisons between cores. A key for interpreting these diagrams is provided in Table 6.1. Each 'unit symbol' represents either a single taxon or an assemblage, with a rationale provided for the categorisation of each assemblage. Up to five 'units' are used in the diagrams at any specific depth to show relative peat composition. Charcoal results are expressed as hollow arrows (small, local fires; 'limited charcoal') or solid arrows (more severe fires; 'plentiful charcoal') according to their position on the five-point scale (1-2 and 3-5 respectively) (Langdon, 1999). Summary diagrams show data plotted against depth (cm) from the peat surface at time of core recovery. Some profiles are shorter than 50cm because of recovery issues or sampling requirements for other analyses.

Statistical analysis followed the protocol outlined in Section 3.5.4 with data presented against date (AD) using chronologies detailed in Chapter 5. Taxa of less relevance to the discussion are presented as opaque in ordination bi-plots to aid interpretation.

Table 6.1: Key for vegetation types indicated in figures presented through Section 6.2.

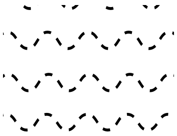
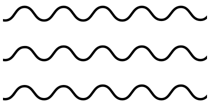
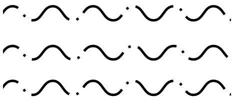
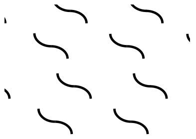
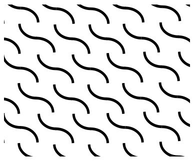
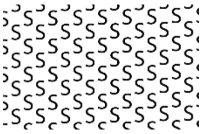
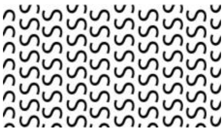
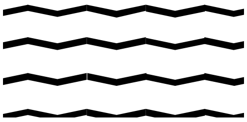
Unit Symbol	Unit Name	Unit Symbol	Unit Name
	*Sphagnum* sect. *Acutifolia*		*Sphagnum* sect. *Sphagnum*
	*Sphagnum* sect. *Cuspidata*		Limited mixed bryophytica: mixed assemblage of two or more brown mosses recorded in sparse numbers (1-2 on abundance scale)
	Abundant mixed bryophytica: mixed assemblage of two or more brown mosses recorded in greater numbers (3-5 on abundance scale)		*Plagiothecium* sp.
	*Pleurozium schreberi*		*Polytrichum* sp.
	*Rhytidiadelphus loreus*		*Dicranum scoparium*

Table 6.1 (cont.)

〰	*Racomitrium lanuginosum*	≶	*Eriophorum vaginatum*
≷	*Eriophorum angustifolium*	‖	*Scirpus cespitosus*
|	*Rhynchospora alba*	¦	*Molinia caerulea*
E	Assorted *Ericaceae*: Indicates either *Ericaceae* undiff., *Empetrum nigrum*, *Vaccinium myrtillus* or *Vaccinium oxycoccos*	e	*Ericaceae* undiff. rootlets
C	*Calluna vulgaris*	T	*Erica tetralix*
Ψ	Forestry litterfall: *Picea sitchensis* bark and needles	⋏	Forestry root growth: *Picea sitchensis* roots

Table 6.1 (cont.)

▤	Unidentifiable organic matter (UOM)	▦	Mixed UOM and monocot roots: dominated by UOM and monocot roots, with little other material identifiable
◁	Limited charcoal (1-2 on abundance scale)	◀	Plentiful charcoal (3-5 on abundance scale)

6.2.1 Bryniau Pica stratigraphy

Bryniau Pica (BRP) was selected for investigation given *M. caerulea* is abundant across most of the site (Section 4.3.1). *M. caerulea* is recorded in all cores and appears to have become abundant in the AD 1980s. Cores were recovered from 1) the main blanket bog complex (BRP-1 and 2) and 2) from shallower peat on the fringes of the complex (BRP-3 and 4). These two areas differ as cores from the main complex tended to be characterised by *S.* s. *Sphagnum* whilst those from the fringes tend be characterised by *S.* s. *Acutifolia* and *S. cespitosus*. There is also plentiful charcoal recorded in cores recovered from the fringes. These findings are expanded upon below. An example assemblage diagram is presented in Figure 6.1 whilst summary stratigraphic diagrams are detailed in Figure 6.2. Details of stratigraphic changes within the cores are presented in Table 6.2.

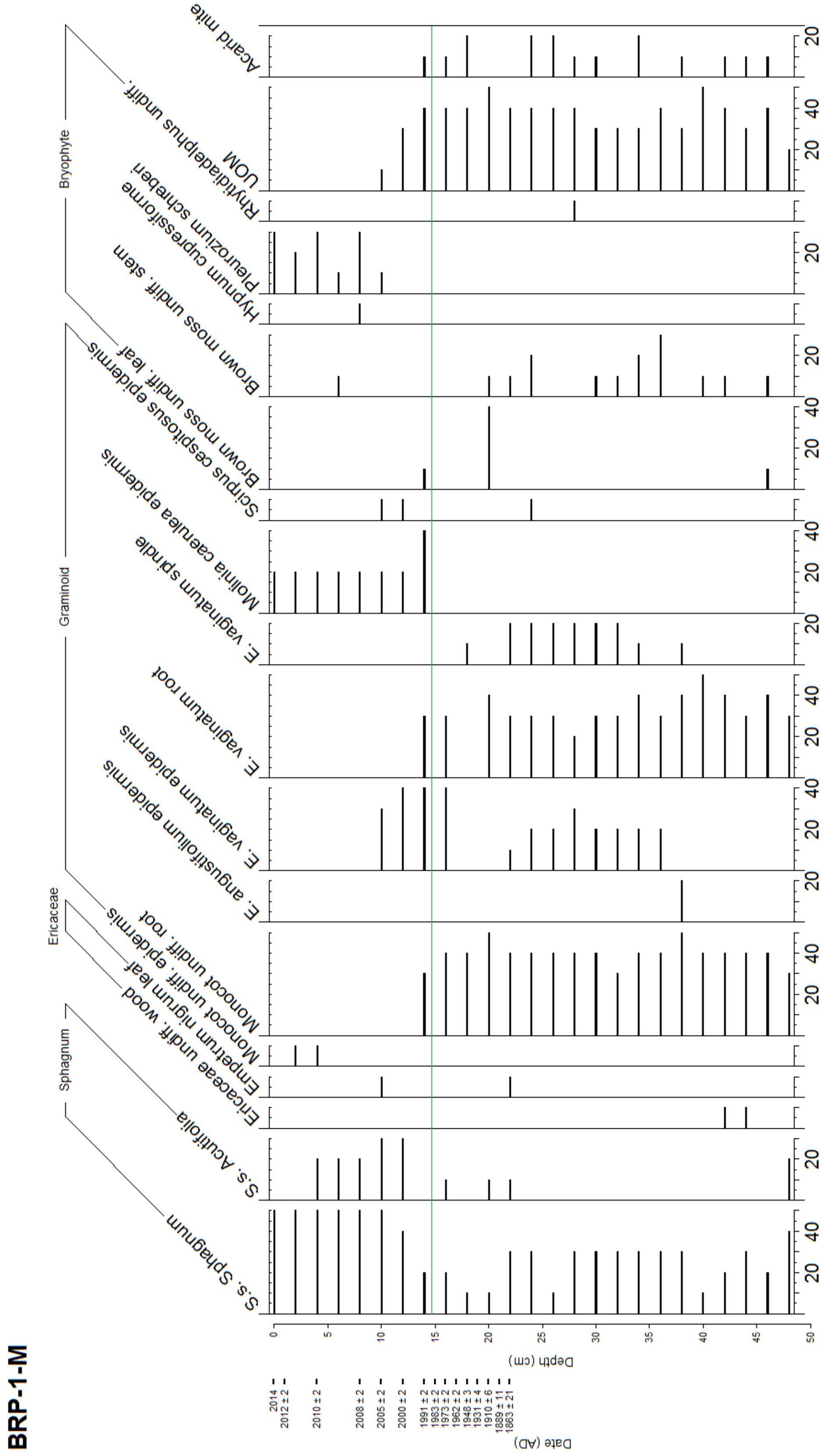


Figure 6.1: Plant macrofossil assemblage diagram for core BRP-1-M plotted against depth (cm) and age estimates (AD). Green line indicates the possible position of the acrotelm/catotelm transition.

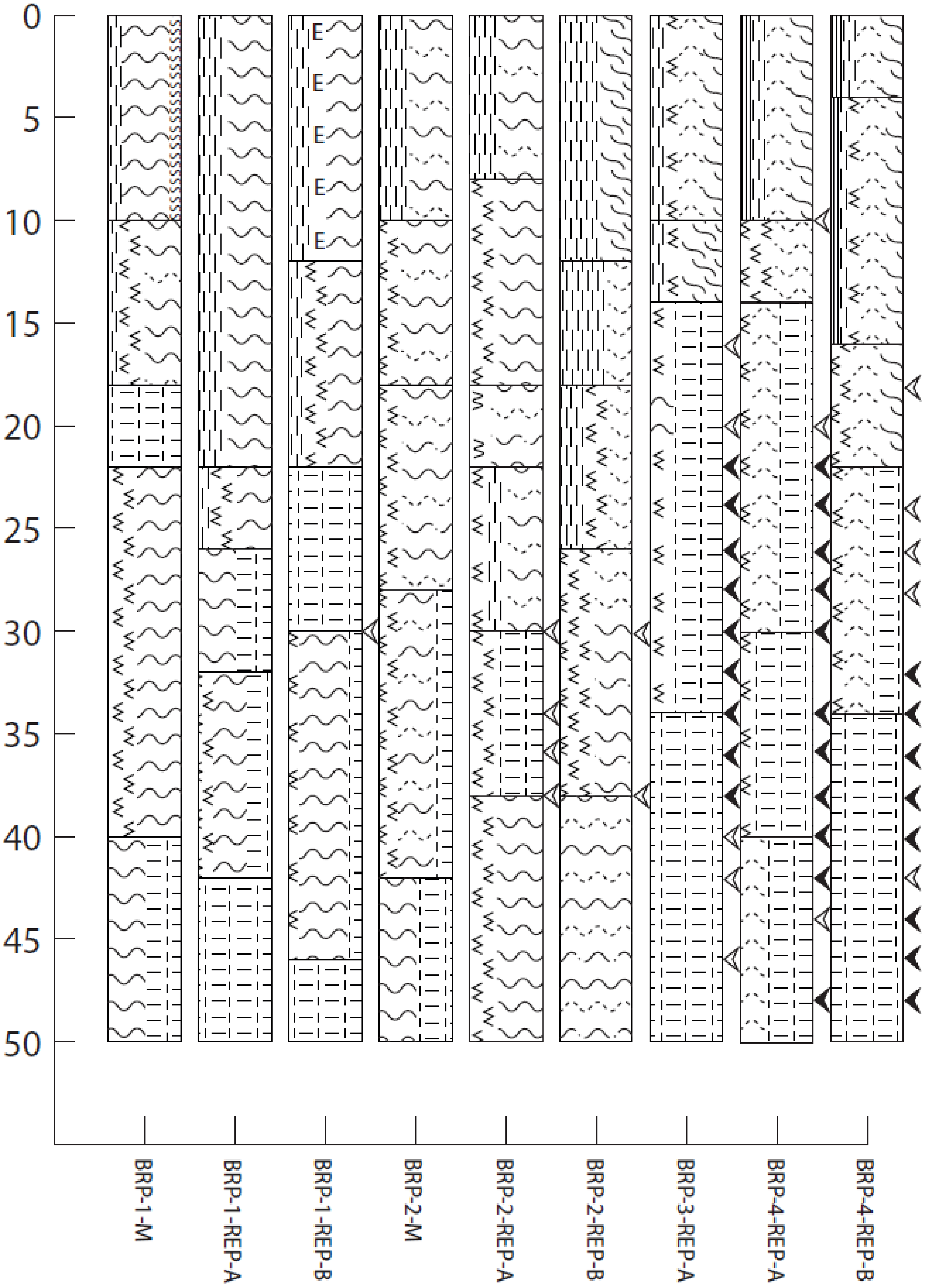


Figure 6.2: Stratigraphic diagrams for cores recovered from BRP.

Table 6.2: BRP plant macrofossil assemblage descriptions.

Core	Vegetation assemblage description
BRP-1-M	*S.* s. *Sphagnum* is persistent throughout and has likely been the primary peat former. The major change exhibited is the shift from *E. vaginatum* dominance as the primary monocot to *M. caerulea ca.* 10cm. *M. caerulea* is only occasional at the surface which suggests it has declined slightly in abundance from an earlier more common phase.
BRP-1-REP-A	There is a trend of increasing *S.* s. *Sphagnum* through the core. *M. caerulea* again replaces *E. vaginatum* as the primary monocot. *M. caerulea* is recorded initially at 34cm then is not present again until 24cm with it recorded as frequent or common from 16cm to the surface.
BRP-1-REP-B	*S.* s. *Sphagnum* is the primary peat former for the majority of the record and is only absent from 30-24cm. *M. caerulea* replaces *E. vaginatum* as the primary monocot, both co-dominating from 20-12cm, after which *M. caerulea* is the sole monocot recorded.
BPR-2-M	*S.* s. *Sphagnum* is present through much of the sequence, though declines to occasional or rare (36-22cm) soon after the arrival of *E. vaginatum* at 42cm and *M. caerulea* at 40cm. Dominant *E. vaginatum* co-exists with intermittent *M. caerulea* from 38-18cm. *M. caerulea* becomes re-established at 8cm and displaces *E. vaginatum*.
BRP-2-REP-A	*S.* s. *Sphagnum* is initially common (50-46cm) though is absent from 40-26cm apart from a rare occurrence at 32cm. This absence occurs alongside rare charcoal particles at 38-30cm. *M. caerulea* has not completely displaced *E. vaginatum*, though *Molinia* is the dominant monocot at the surface following previous growth at 48 and 26-24cm.

Table 6.2 (cont.)

BRP-2-REP-B	A drying trend is exhibited within the core. Lower sections (50-40cm) imply an environment with a high water table given *S.* s. *Sphagnum* and *S.* s. *Cuspidata* are key constituents and *R. alba* is also recorded. A drying trend is suggested above this by an *E. vaginatum* then *M. caerulea* expansion.
BRP-3-REP-A	Charcoal is recorded throughout the majority of the record (46-16cm) reaching peak abundance 28-22cm. Peat is highly humified across these depths but *E. vaginatum* and *S.* s. *Sphagnum* remains are intermittently recorded. A diverse brown moss community is established following the cessation of burning and their presence alongside *E. vaginatum* and *M. caerulea* suggest a predominantly dry environment.
BRP-4-REP-A	Charcoal is present in most samples ≤18cm depth. These depths are characterised by common or abundant UOM with intermittent *S.* s. *Acutifolia* also present. The surface appears dry given the co-dominance of *M. caerulea* and *S. cespitosus* alongside a range of brown mosses.
BRP-4-REP-B	Charcoal is present in most samples <18cm depth. A decline in burning intensity occurs 30-18cm. UOM dominates the lower portions of the record though a rare occurrence of both *S.* s. *Sphagnum* and *S.* s. *Cuspidata* (44cm) suggests a peat forming community. The system appears to have dried and *M. caerulea* and *S. cespitosus* co-dominate surface samples.

Datasets generated through plant macrofossil analysis represent complex changes in biotic assemblages through space and time. The ordination techniques of Detrended Correspondence Analysis (DCA), Principal Components Analysis (PCA) and non-Metric Multidimensional Scaling (nMDS) (Section 3.4.5) were used in this thesis to numerically represent the effect of environmental drivers (e.g. land use) on peatland plant communities. Macrofossil assemblages from each BRP core were combined so they could be analysed in the same ordination space to detect key differences/similarities in how sampling points had developed over time. Analysis was first conducted on the full dataset with

UOM included, where Axis one of the DCA had an eigenvalue of 0.3021 and a gradient length of 3.092. This eigenvalue is not especially high as eigenvalues between 0.2-0.3 may mean either a linear or unimodal response model can be used (Section 3.5.1). The gradient length of 3.092 does exceed the 3 standard deviation (SD) units suggested for using a unimodal response model (Ter Braak and Prentice, 1988; van Bellen *et al*. 2016). An nMDS model was run to compare the outputs with the results of the DCA. The nMDS had a stress score of 0.16 indicating the model is suitable for interpretation (Section 3.5.2). Normalised Axis one sample scores for both the DCA and a PCA, were compared with their equivalent Axis one scores in the nMDS model. The DCA was significantly positively correlated with the nMDS, whilst the PCA was significantly negatively correlated (Table 6.3). Procrustes rotation to compare the nMDS and DCA ordinations produced a sum of squares value of 0.17 and a significant positive correlation ($r = 0.91$, $p = 0.001$) between the ordinations which suggests the ordinations are displaying similar directions of change (Section 3.5.3).

Table 6.3: Comparison of ordination techniques applied to the BRP macrofossil dataset Axis one sample scores where UOM was included.

	nMDS vs. DCA		**nMDS vs. PCA**	
	***R*-value**	***P*-value**	***R*-value**	***P*-value**
BRP-1-M	0.97	<0.001	-0.63	<0.001
BRP-1-REP-A	0.96	<0.001	-0.79	<0.001
BRP-1-REP-B	0.88	<0.001	-0.62	<0.001
BRP-2-M	0.67	<0.001	-0.62	<0.001
BRP-2-REP-A	0.90	<0.001	-0.88	<0.001
BRP-2-REP-B	0.81	<0.001	-0.68	<0.001
BRP-3-REP-A	0.94	<0.001	-0.99	<0.001
BRP-4-REP-A	0.98	<0.001	-0.99	<0.001
BRP-4-REP-B	0.96	<0.001	-0.98	<0.001

A second set of ordinations were performed where UOM was removed to explore the effect of this pseudo-taxon on the ordination results. The removal of UOM improved DCA performance as it increased the Axis one eigenvalue to 0.430, the gradient length to 4.342 and also reduced the nMDS stress score to 0.13. A comparison of normalised Axis one sample scores (Table 6.4) shows agreement between the DCA and nMDS has improved for some cores (BRP-2-M, BRP-2-REP-A, BRP-2-REP-B) but declined for others (BRP-1-M, BRP-1-REP-A, BRP-1-REP-B). Removal of UOM strengthened the relationship between the DCA and nMDS in symmetric procrustes rotation ($r = 0.95$, $p = 0.001$, sum of squares = 0.09), suggesting improved overall agreement between the ordinations regarding taxa response to the primary environmental gradient. It appears therefore that UOM may be reducing the explanatory power of the DCA as it is behaving as a mid-gradient taxon where only presence/absence values are used because UOM can originate from any peatland plant (Section 3.5.4). Given these findings, the DCA output where UOM was excluded was adopted for analysis.

Table 6.4: Comparison of ordination techniques applied to the BRP macrofossil dataset Axis one sample scores where UOM was excluded.

	nMDS vs. DCA		**nMDS vs. PCA**	
	***R*-value**	***P*-value**	***R*-value**	***P*-value**
BRP-1-M	0.91	<0.001	0.45	<0.05
BRP-1-REP-A	0.94	<0.001	0.29	0.07
BRP-1-REP-B	0.85	<0.001	0.39	<0.05
BRP-2-M	0.75	<0.001	0.32	0.06
BRP-2-REP-A	0.95	<0.001	0.88	<0.001
BRP-2-REP-B	0.91	<0.001	0.32	0.07
BRP-3-REP-A	0.99	<0.001	0.83	<0.001
BRP-4-REP-A	0.98	<0.001	0.95	<0.001
BRP-4-REP-B	0.98	<0.001	0.88	<0.001

Visual interpretation of taxa distribution within the ordination space was used to analyse the DCA. The environmental gradient that drives vegetation change may be determined by examining the separation of taxa along Axes one and two and by applying ecological knowledge to interpret what environmental gradient may drive species response (Section 3.4.5).

The arrangement of taxa along Axis one of the DCA is indicative of a burning gradient (Figure 6.3). Charcoal is the pseudo-taxon ordinated in the most positive location on Axis one. All *Sphagna* sections plot mid-gradient as, with a few exceptions, most *Sphagna* are recorded at depths in the cores across BRP where there is no evidence of burning. This point is especially true for occurrences of *S.* s*. Sphagnum* at sites BRP-1 and BRP-2. These cores were recovered from the main blanket bog complex where minimum burning has apparently occurred. Figure 6.4 shows that most BRP-1 and BRP-2 samples plot on the negative side of Axis one. Fire has never observably dominated these sites, though limited charcoal is recorded at BRP-1-REP-B, BRP-2-REP-A and B (Figure 6.2).

BRP-3 and BRP-4 were recovered from the fringes of the bog on shallower peat. These fringe cores display little evidence of prior *S.* s. *Sphagnum* communities and exhibit the most frequent occurrence of charcoal, hence their plotting predominantly on the positive side of Axis one in Figure 6.4. Some sample scores from the BRP-3 and BRP-4 cores do plot on the negative side of Axis one (Figure 6.4) as these sites have transitioned away from an environment characterised by fire. The importance of the charcoal decline in the upper *ca.* 20cm (Figure 6.2) of most cores is evidenced by Figure 6.5. Samples where charcoal is recorded plot near exclusively on the positive side of Axis one whilst samples where no charcoal is recorded plot predominantly on either the negative side, or between 0-1 SD units on the positive side of the Axis one gradient. The appearance of *M. caerulea* does not appear to relate to firing as it plots on the negative end of Axis one (Figure 6.3) and its sustained rise occurs at sites where burning has not observably occurred (e.g. BRP-1-M, BRP-1-REP-A) or at sites after burning has stopped for some time (e.g. BRP-1-REP-B, BRP-2-REP-B) (Figure 6.6).

Taxa dispersion on Axis two of the DCA show taxa indicative of blanket bog (e.g. *S.* s. *Cuspidata*, *S.* s. *Sphagnum* and *E. angustifolium*) arranged at the positive end with xeric bryophytes affiliated to humid heath conditions (e.g. *A. palustre*, *P. schreberi*, *Polytrichum alpestre*) plotting on the negative end (Figure 6.3). The gradient therefore represents a shift from blanket bog (positive) to heath communities (negative) in that hydrological conditions vary from wet to dry. A comparison of sample scores supports this interpretation of the DCA (Figure 6.4) as samples from cores recovered from the main bog complex characterised by *S.* s. *Sphagnum* (BRP-1 and 2) plot predominantly on the positive side of Axis two. Those cores from the fringes of the blanket bog on shallower peat where *S.* s. *Acutifolia* and brown mosses are prominent (BRP-3 and 4) plot primarily on the negative side (Figure 6.4). Taxa grouped on the negative end of Axis two (Figure 6.3) are largely indicative of the M15d *V. myrtillus* sub-community of *S. cespitosus* – *E. tetralix* wet heath (Elkington *et al.* 2001). *H. cupressiforme*, *P. schreberi*, *Rhytidiadelphus* sp. and *S. cespitosus* would be expected in an M15d assemblage (Elkington *et al.* 2001) and all plot <-1 SD units on Axis two (Figure 6.3).

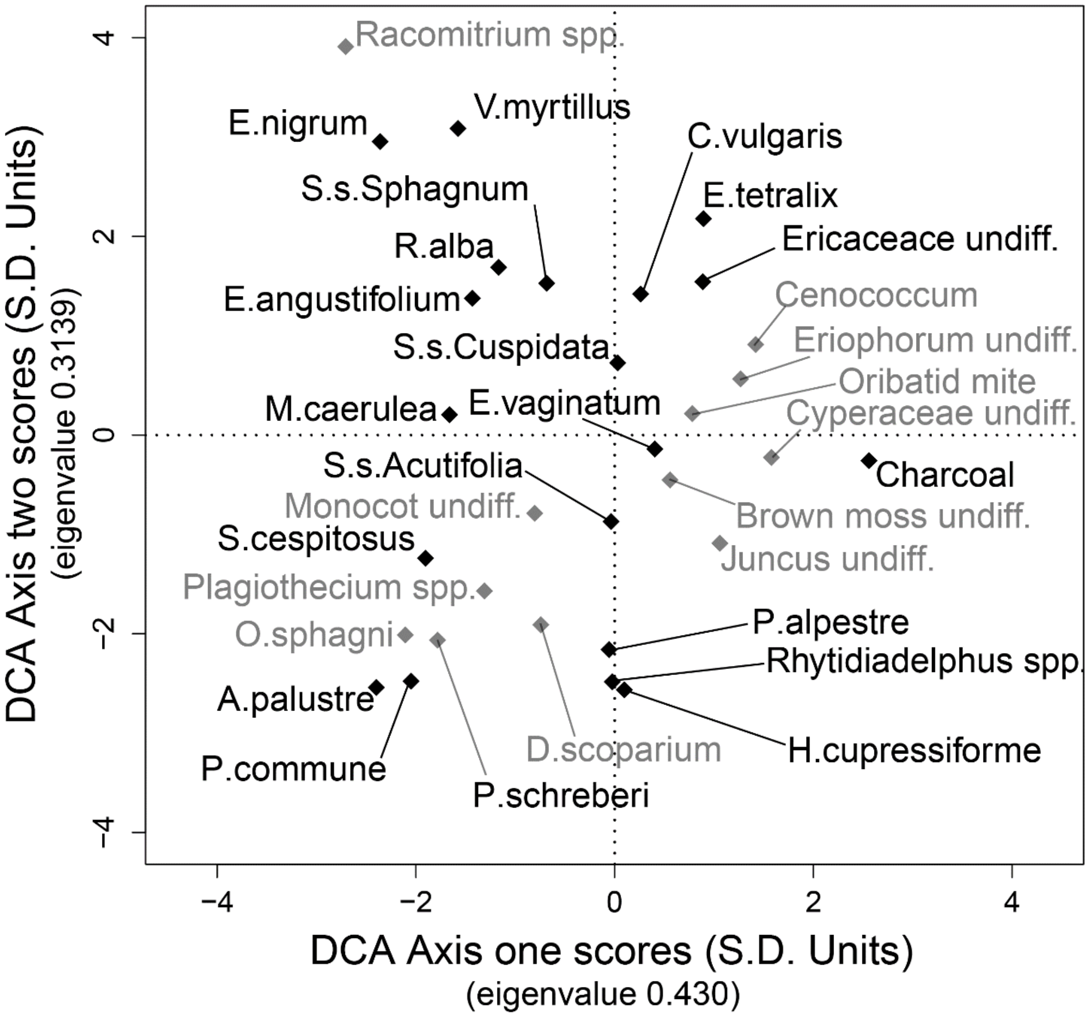


Figure 6.3: DCA bi-plot of the BRP plant macrofossil data showing taxa distribution according to their position on Axis one and two.

One may expect heath assemblages at BRP-3 and 4 because peat at these sites is shallower than on the main blanket bog. There is evidence that these heath-like sites once exhibited assemblages more characteristic of blanket bog. *S.* s. *Sphagnum* is recorded between 32-12cm at BRP-3-REP-A and at 44cm in BRP-4-REP-B (Figure 6.2). *S.* s. *Acutifolia* is recorded through the majority of BRP-4-REP-A and above 32cm depth in BRP-4-REP-B. *Sphagnum* is clearly present at these sites prior to the commencement of the SCP record and often alongside *E. vaginatum*. Charcoal is near continuously recorded as 3-5 on the abundance scale during these periods at BRP-3 and 4 (Figure 6.2). Fire (pre-AD 1850) may therefore be responsible for forcing BRP-3 and 4 towards their heath-like state.

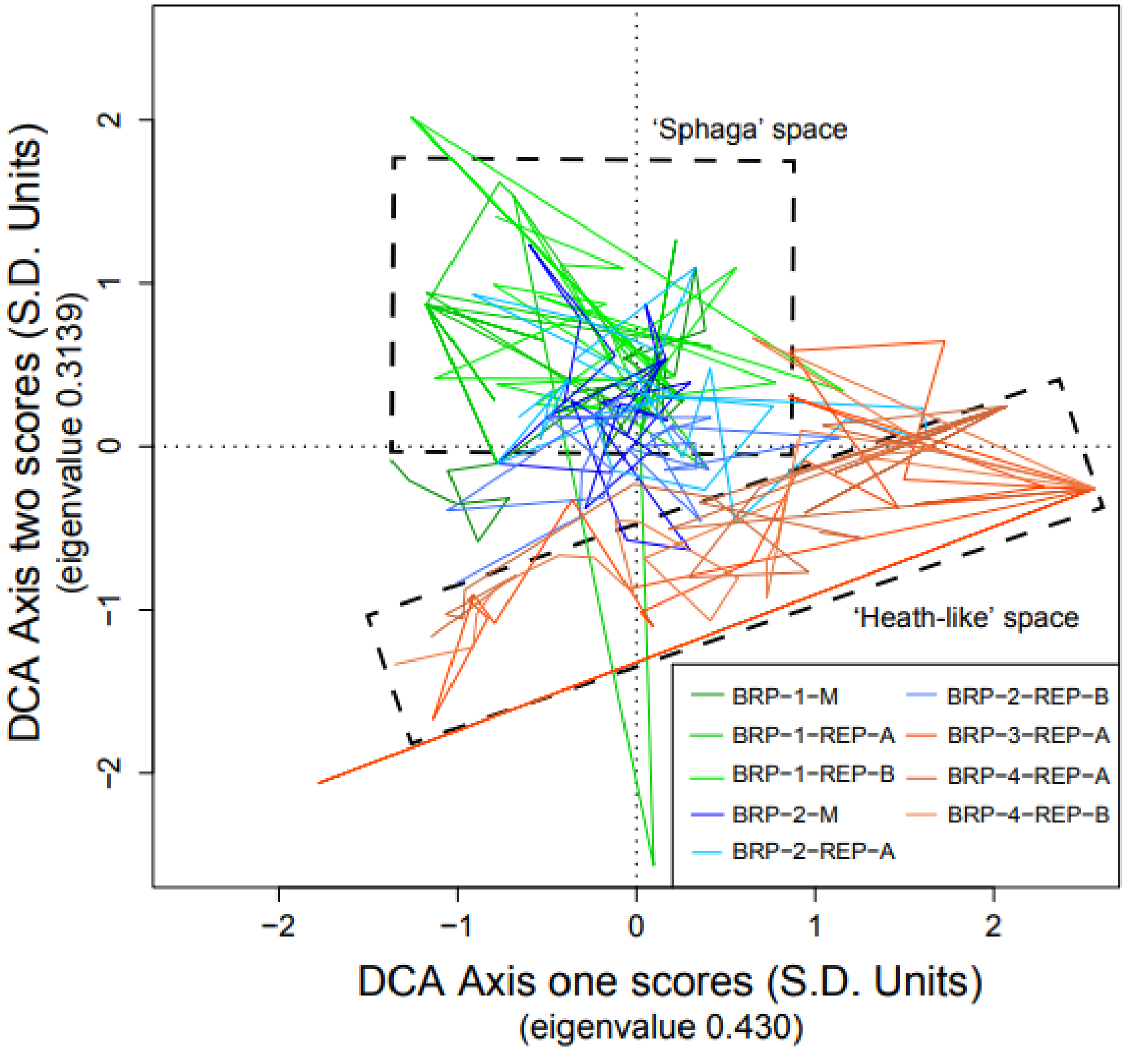


Figure 6.4: DCA bi-plot of the BRP macrofossil data showing sample distribution according to their position on Axis one and two.

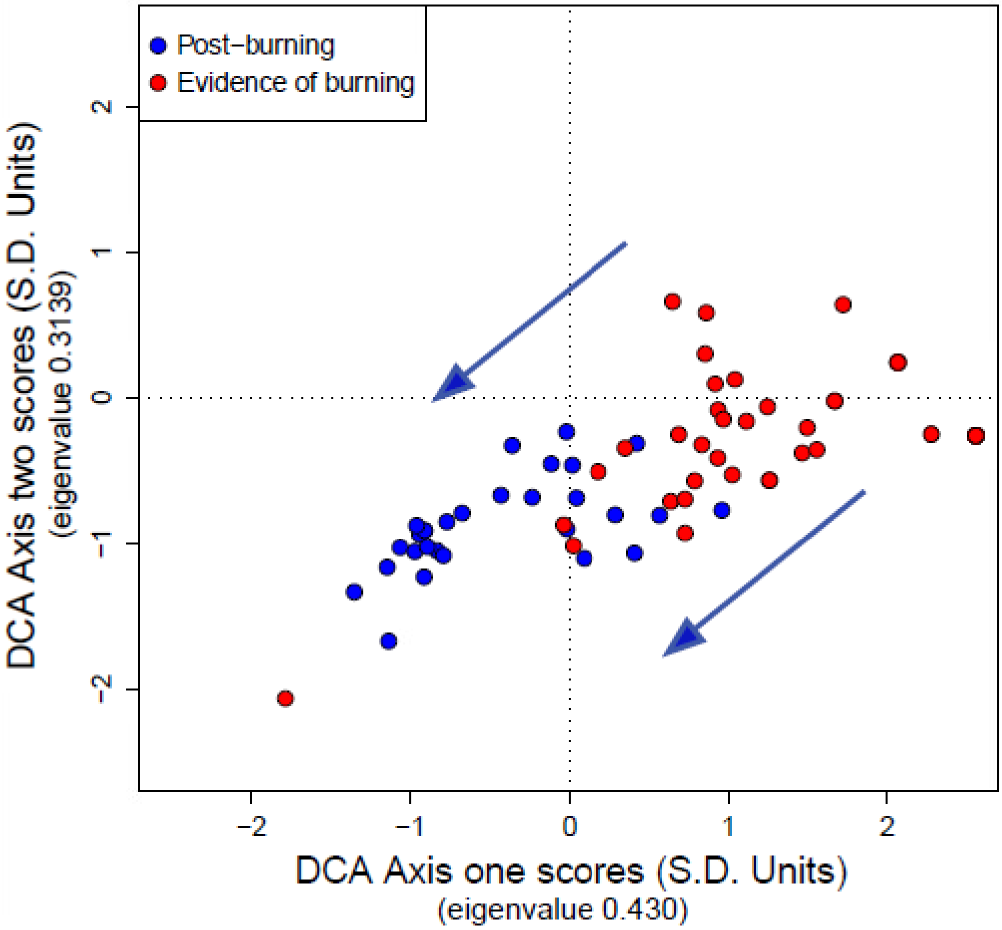


Figure 6.5: DCA sample score bi-plot of 'heath-like' BRP sites (BRP-3-REP-A, BRP-4-REP-A and B). Arrows indicate direction of change away from environment characterised by fire. Sample scores are colour coordinated according to their position pre- and post the last recorded charcoal horizon.

The development of blanket peat at BRP can be broadly separated into two differing types. Cores from the main blanket bog complex are predominantly characterised by *S. s. Sphagnum* whilst cores from the fringes of the main deposit are characterised by drier affiliated taxa. The dominant land use at BRP over the previous hundred and fifty years has been grazing, primarily using sheep. Fire has clearly been used as a form of management at these sites, with burning apparently more intense prior to the AD 1850 SCP horizon. The use of fire may have played a role in forcing sub-sites BRP-3 and 4 toward heath like conditions, either directly by encouraging graminoids or by altering hydrological conditons and making these areas less ecologically suitable for *S.* s. *Sphagum*. Fire has likely not caused the dominance of *M. caerulea* at BRP given it is present in all contemporary assemblages regardless of each core's burning history.

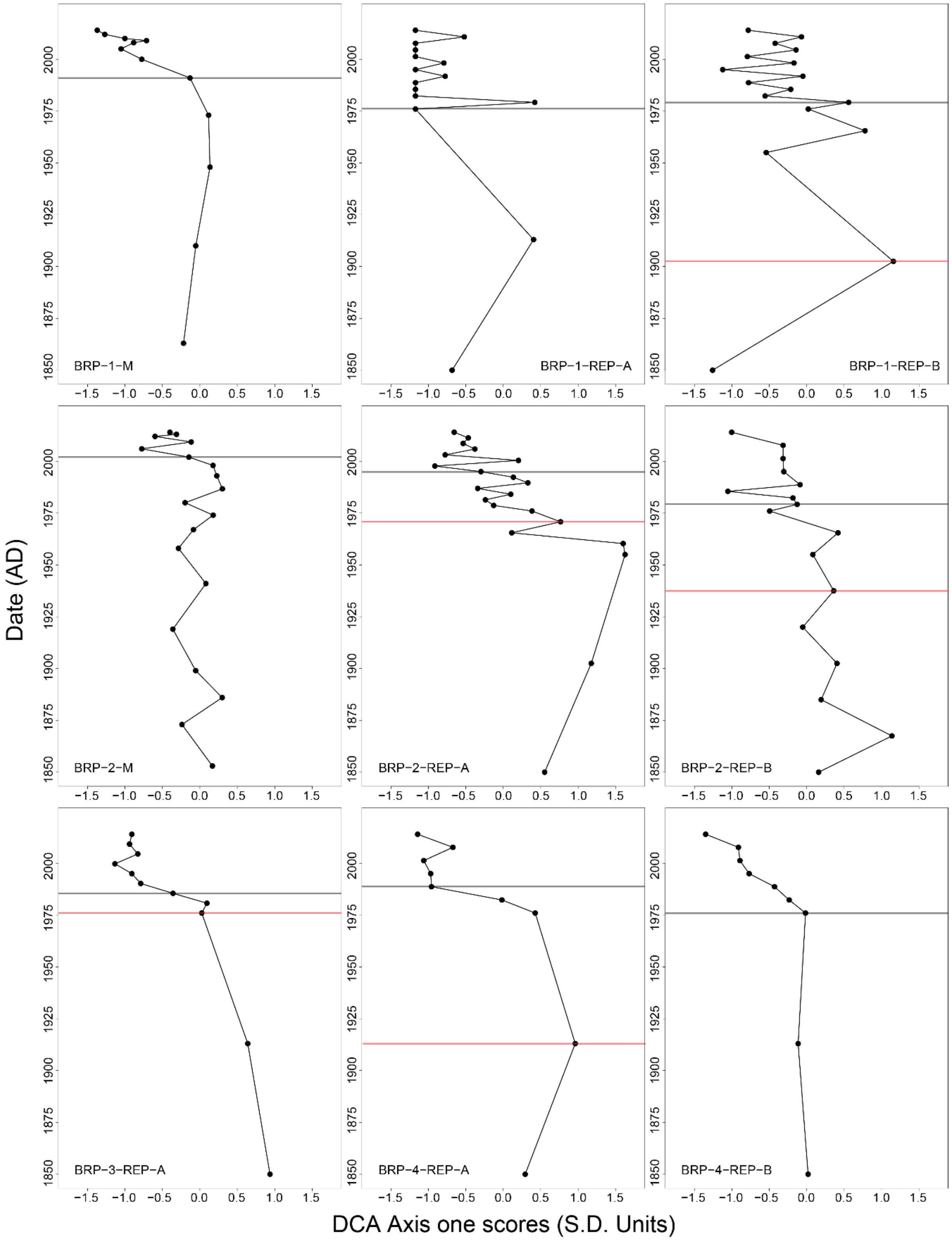


Figure 6.6: Plots showing BRP DCA Axis one sample scores plotted against calendar age. The grey line indicates where *M. caerulea* becomes established in the record (excluding intermittent appearances). The red line indicates the most recent evidence of fire in the profile.

6.2.2 Lake Vyrnwy stratigraphy

Lake Vyrnwy (LVW) was selected for investigation because of its historic use as a grouse shooting estate (Section 4.3.2). Charcoal is present in all but one of the seven cores recovered from this site. There are a variety of vegetation records reported from these cores with a broad hydrological gradient being present amongst the results. Cores dominated by *Calluna* were typically recovered from the water-shedding mesotopes and exhibit vegetation affiliated to drier conditions. Two cores recovered from topographical saddles (LVW-1-REP-C and D) exhibit assemblages more characteristic of mesic to waterlogged conditions. These findings are expanded upon below. An example assemblage diagram is presented in Figure 6.7 whilst summary stratigraphic diagrams for cores recovered at LVW are detailed in Figure 6.8. Details of stratigraphic changes within the cores are presented in Table 6.5.

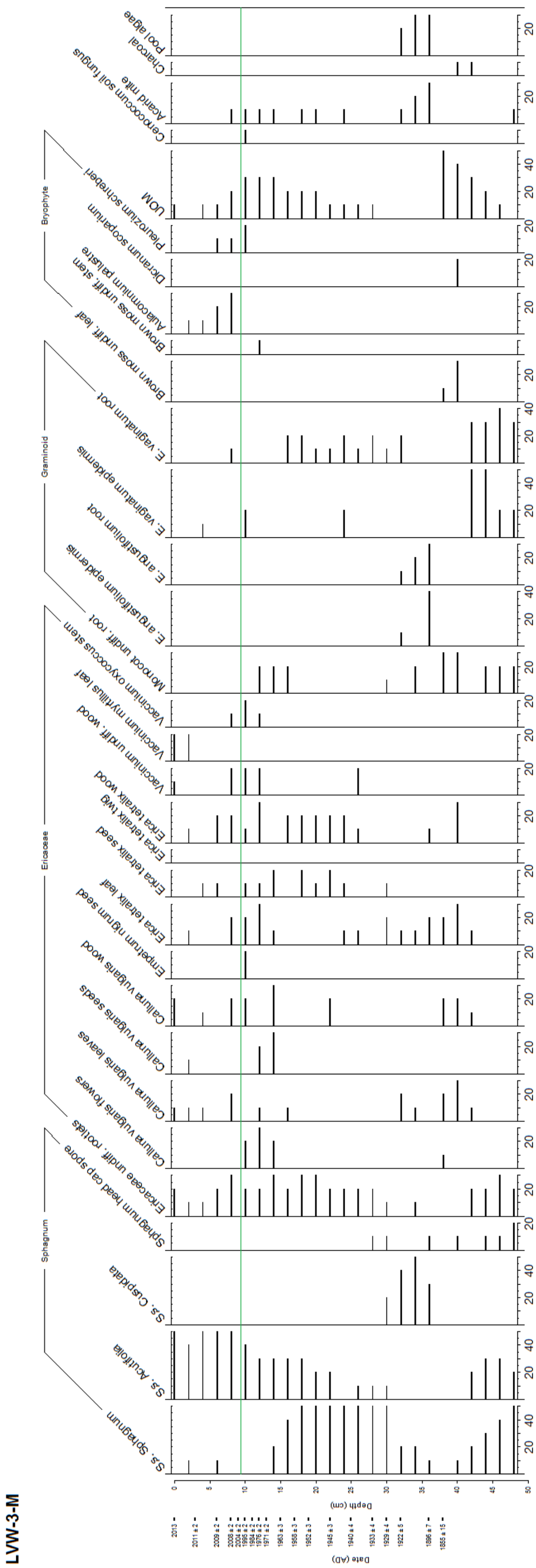


Figure 6.7: Plant macrofossil assemblage diagram for core LVW-3-M plotted against depth (cm) and age estimates (AD). Green line indicates the possible position of the acrotelm/catotelm transition.

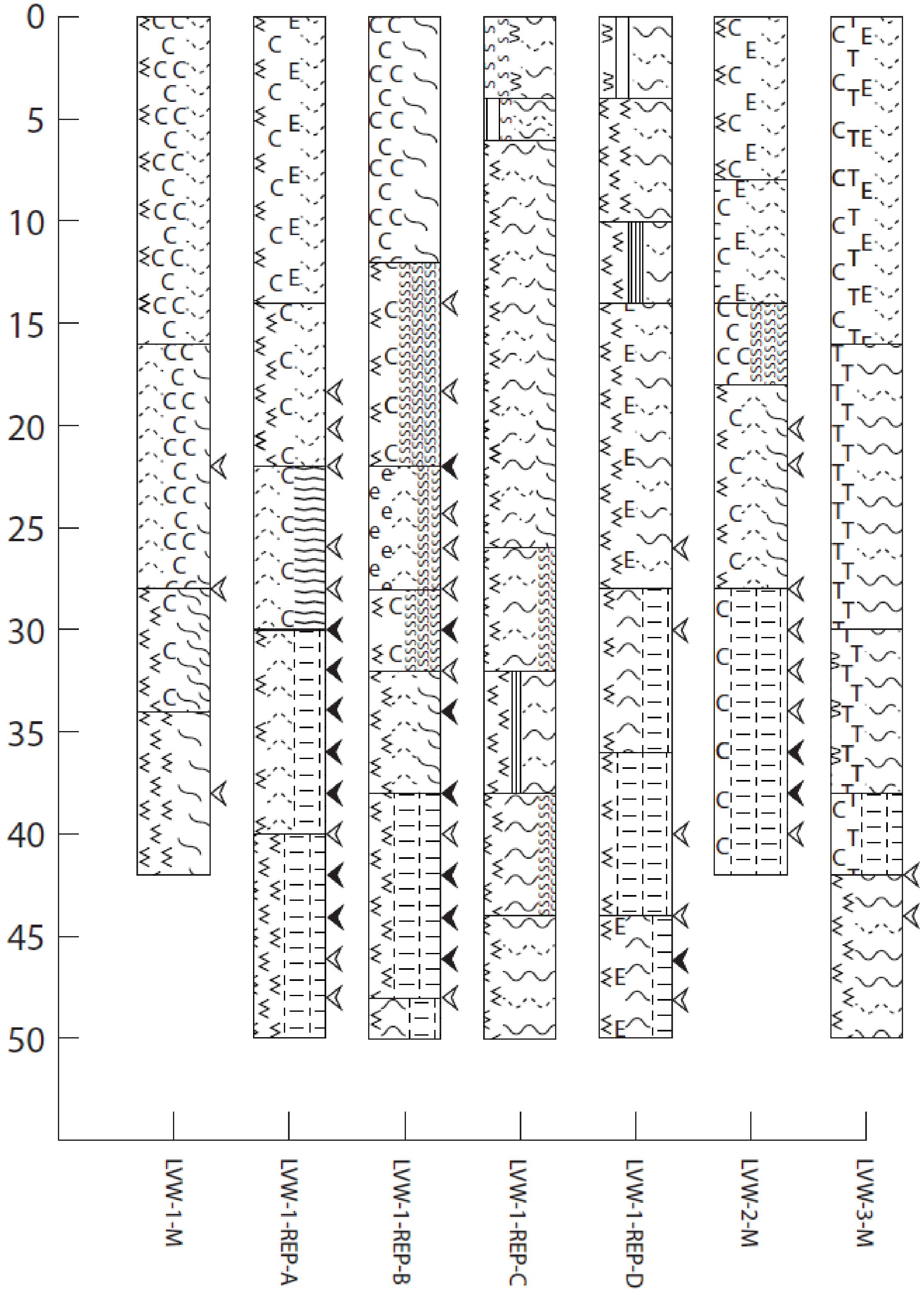


Figure 6.8: Stratigraphic diagrams for cores recovered from LVW.

Table 6.5: LVW plant macrofossil assemblage descriptions.

Core	Vegetation assemblage description
LVW-1-M	Predominantly dry record. *E. vaginatum* dominates 40-28cm, recorded first alongside an assortment of rare or occasional brown mosses (*A. palustre*, *D. scoparium* and *P. schreberi*) before *C. vulgaris* becomes established at 32cm. *S.* s. *Acutifolia* appears at 28cm and increases rapidly to become common and abundant (as *S. rubellum*) at the mire surface.
LVW-1-REP-A	Predominantly dry record. UOM common or abundant from 48-28cm with *E. vaginatum* likely the primary peat former alongside intermittent *S.* s. *Acutifolia*. Charcoal often frequent or common across these depths. *C. vulgaris* established at 28cm alongside an expansion of *R. lanuginosum*. *S.* s. *Acutifolia* (*S. rubellum*) and *E. vaginatum* increase above 20cm.
LVW-1-REP-B	Predominantly dry record. UOM is the major constituent from 48-40cm. Isolated *E. vaginatum* epidermis fragments and spindles and a single *S.* s. *Sphagnum* leaf suggest a peat forming community. *Calluna* remains dominate above 20cm. Charcoal near consistently recorded from 48-22cm.
LVW-1-REP-C	Predominantly wet record. *S.* s. *Sphagnum* dominant throughout and never recorded less than frequent. Slight dry phase occurs 36-32cm (*P. schreberi*, *S. cespitosus* and *Calluna*). System likely shifting toward wetter conditions as *S.* s. *Cuspidata* is recorded above 22cm and *E. angustifolium*, *P. commune* and *R. alba* become established in upper horizons.
LVW-1-REP-D	Likely switching between wet and dry conditions, though neither being extreme as *E. vaginatum* persists through the majority of the record. Inferred wet phases occur 48-42cm (*E. angustifolium*, *E. tetralix* and *S.* s. *Sphagnum*), 26-14cm (*E. tetralix*, *S.* s. *Sphagnum* and *S.* s. *Cuspidata*) and 4-0cm (*E. angustifolium*, *R. alba* and *S.* s. *Cuspidata*). Inferred dry phases occur 42-28cm (*E. vaginatum*, minimal *Sphagnum*) and 14-10cm (*S. cespitosus* and *E. nigrum*).

Table 6.5 (cont.)

LVW-2-M	Predominantly dry record as *C. vulgaris* is present through the bulk of the profile. Peat from 40-24cm is highly humified and charcoal is recorded as occasional or frequent through this zone. Mixed *D. scoparium* and *P. schreberi* assemblage occurs 24-14cm before *S. rubellum* increases to common or abundant above 10cm.
LVW-3-M	Likely switching between wet and dry conditions. Inferred wet phases occur 48-44cm (*S.* s*. Sphagnum*, low UOM), 36-30cm (*E. angustifolium*, *S.* s*. Cuspidata*, absent UOM, pool algae). Inferred dry phases occur 42-38cm (*C. vulgaris*, *E. vaginatum*, low *Sphagnum* and high UOM) and 12-0cm (*Calluna*, *V. myrtillus*, *S. rubellum*, mixed *A. palustre* and *P. schreberi* phases) with an intermediate phase occurring 28-14cm (*E. tetralix*, declining *S.* s*. Sphagnum*, increasing UOM).

LVW macrofossil datasets were combined so that vegetation assemblages could be assessed in the same ordination space for key differences/similarities in their development over time. Analysis was first conducted on the full dataset with UOM included. Where UOM was included the DCA had an eigenvalue of 0.2813 and a gradient length of 2.5109. This eigenvalue is located within the 0.2-0.3 range where either a linear of unimodal response model may be used to explore the data. Normalised Axis one sample scores for both the DCA and a PCA were compared to their equivalent Axis one scores in an nMDS model. The PCA was significantly positively correlated with the nMDS whilst the DCA broadly showed no relationship to the nMDS ordination (Table 6.6).

Table 6.6: Comparison of ordination techniques applied to the LVW macrofossil dataset Axis one sample scores where UOM was included.

	nMDS vs. DCA		nMDS vs. PCA	
	***R*-value**	***P*-value**	***R*-value**	***P*-value**
LVW-1-M	0.03	0.42	0.80	<0.001
LVW-1-REP-A	0.02	0.44	0.81	<0.001
LVW-1-REP-B	-0.03	0.41	0.65	<0.05
LVW-1-REP-C	0.36	<0.01	0.87	<0.001
LVW-1-REP-D	0.13	0.18	0.88	<0.001
LVW-2-M	0.06	0.36	0.63	<0.05
LVW-3-M	0.05	0.37	0.78	<0.001

A secondary set of ordinations were performed where UOM was removed from the combined LVW datasets to explore the effect of this pseudo-taxon on the ordination output. Where UOM was excluded, Axis one of the DCA had an eigenvalue of 0.3494 and a gradient length of 3.7963. The LVW nMDS had a lower stress score where UOM was excluded (0.15) relative to the same model where UOM was included (0.16) suggesting the removal of UOM is improving the reliability of the nMDS. The DCA and nMDS were though significantly positively correlated in symmetric procrustes rotation ($r = 0.87$, $p = 0.001$) despite a relatively high sum of squares (0.24). The procrustes rotation therefore suggests that both ordinations are returning broadly similar directions of environmental change where UOM is excluded. Their plotting on different orientations explains why they do not produce positively correlated Axis one sample scores (Table 6.7) but are significantly positively correlated under procrustes. Hence, both the LVW DCA and nMDS were deemed appropriate to interpret environmental change where UOM was excluded given the greater ordination eigenvalue and lower stress score respectively where the pseudo-taxon was removed.

Table 6.7: Comparison of ordination techniques applied to the LVW macrofossil dataset Axis one sample scores where UOM was excluded.

	nMDS vs. DCA		**nMDS vs. PCA**	
	***R*-value**	***P*-value**	***R*-value**	***P*-value**
LVW-1-M	-0.75	<0.001	0.32	0.08
LVW-1-REP-A	-0.93	<0.001	0.36	<0.05
LVW-1-REP-B	-0.97	<0.001	-0.26	0.11
LVW-1-REP-C	-0.91	<0.001	0.83	<0.001
LVW-1-REP-D	-0.94	<0.001	0.97	<0.001
LVW-2-M	-0.97	<0.001	0.13	0.29
LVW-3-M	-0.78	<0.001	0.72	<0.001

Visual interpretation of taxa distribution within the ordination space was again used to analyse the LVW DCA and nMDS. Water table depth (WTD) is likely the primary determinant of vegetation changes at LVW according to Axis one of both the DCA (Figure 6.9) and nMDS (Figure 6.10). Drier affiliated *A. palustre*, *C. vulgaris*, *D. scoparium*, *H. cupressiforme*, *O. sphagni*, *P. schreberi* and *Rhytidiadelphus* sp. all plot <0 SD units on Axis one of the DCA (Figure 6.9). Taxa on the positive end of the gradient are broadly indicative of wetter conditions, *E. angustifolium*, pool algae, *P. commune*, *S. s. Cuspidata*, and *R. alba* all plotting >2 SD units. *S. cespitosus* can be used to infer drier mire conditions though here plots on the positive, wetter, end of Axis one (Figure 6.9). *S. cespitosus* can exhibit a wide distribution along the moisture gradient (Rodwell, 1991) hence in this ordination it may represent intermediate conditions (i.e. lawns or low hummock) rather than a nanotope with a large WTD (i.e. high hummock). The nMDS has ordinated taxa on a different orientation to the DCA, hence why the negative end of Axis one represents wetter and the positive end drier conditions (Figure 6.10). Similar groupings of taxa along Axis one, albeit on differing oriented axes, explains why these two ordinations are positively correlated in procrustes rotations ($r = 0.87$, $p = 0.001$).

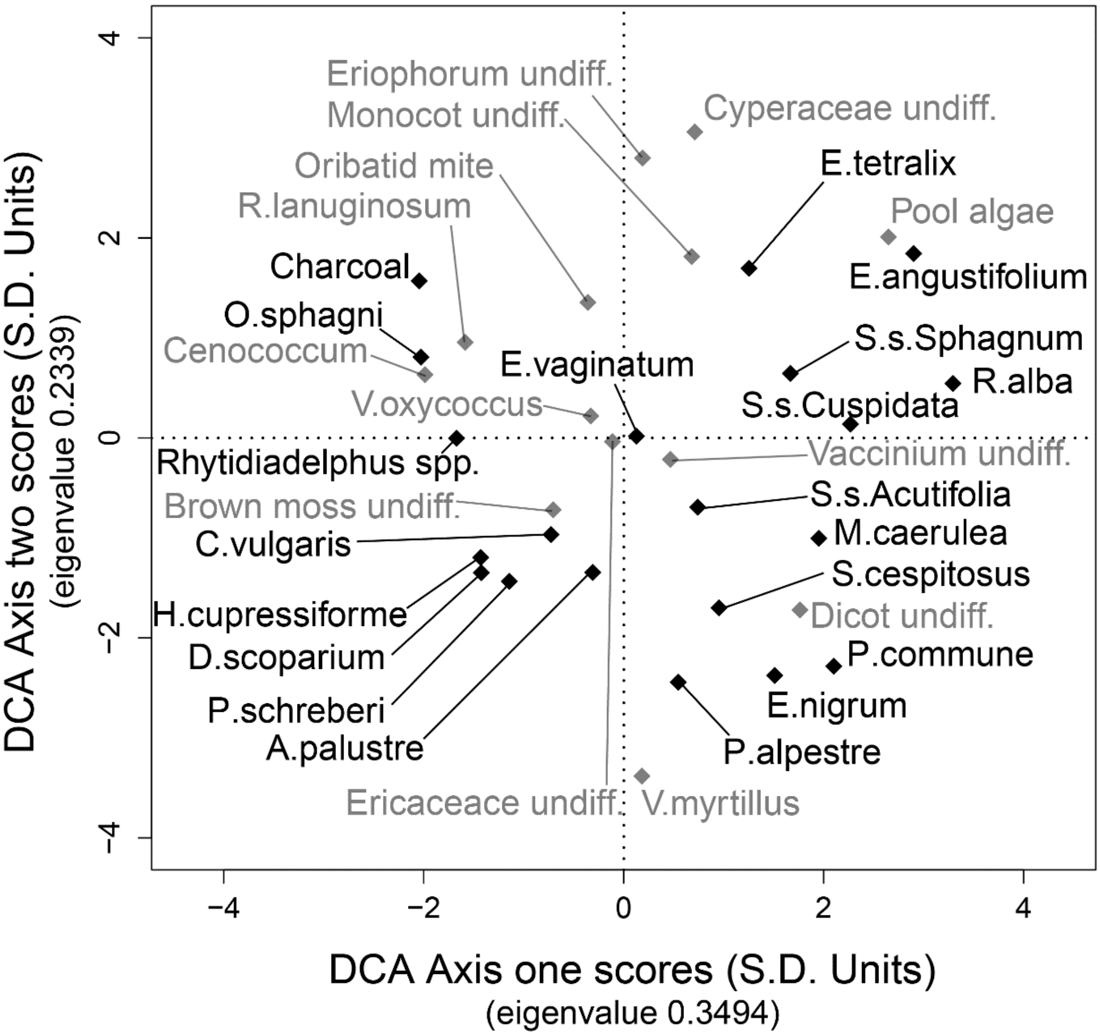


Figure 6.9: DCA bi-plot of LVW plant macrofossil data (UOM excluded) showing taxa distribution according to their position on Axis one and two.

DCA sample scores indicate broad hydrological differences between the cores recovered from LVW (Figure 6.11). Records which are *Sphagnum* rich plot mostly on the positive (wetter) side of Axis one (e.g. LVW-1-REP-C and LVW-3-M), whilst those samples where *C. vulgaris* and brown mosses dominate plot predominantly on the negative (drier) side (e.g. LVW-1-M, LVW-1-REP-B and LVW-2-M) (Figure 6.11). Sites where charcoal is either never recorded, or recorded only rarely/occasionally, plot predominantly on the positive side of Axis one (LVW-1-REP-C and LVW-3-M). Sites which are *Sphagnum* rich, primarily *S.* s. *Sphagnum* and *S.* s. *Cuspidata*, at LVW therefore appear to have been less affected by fire than sites where these taxa are less prevalent. Sites at LVW where fire is often recorded (e.g. LVW-1-REP-A and B; Figure 6.11) have not been completely devoid of *Sphagnum* (Figure 6.8). *Sphagnum* simply appears to have not been as prevalent at these sites relative to those sites less affected by fire.

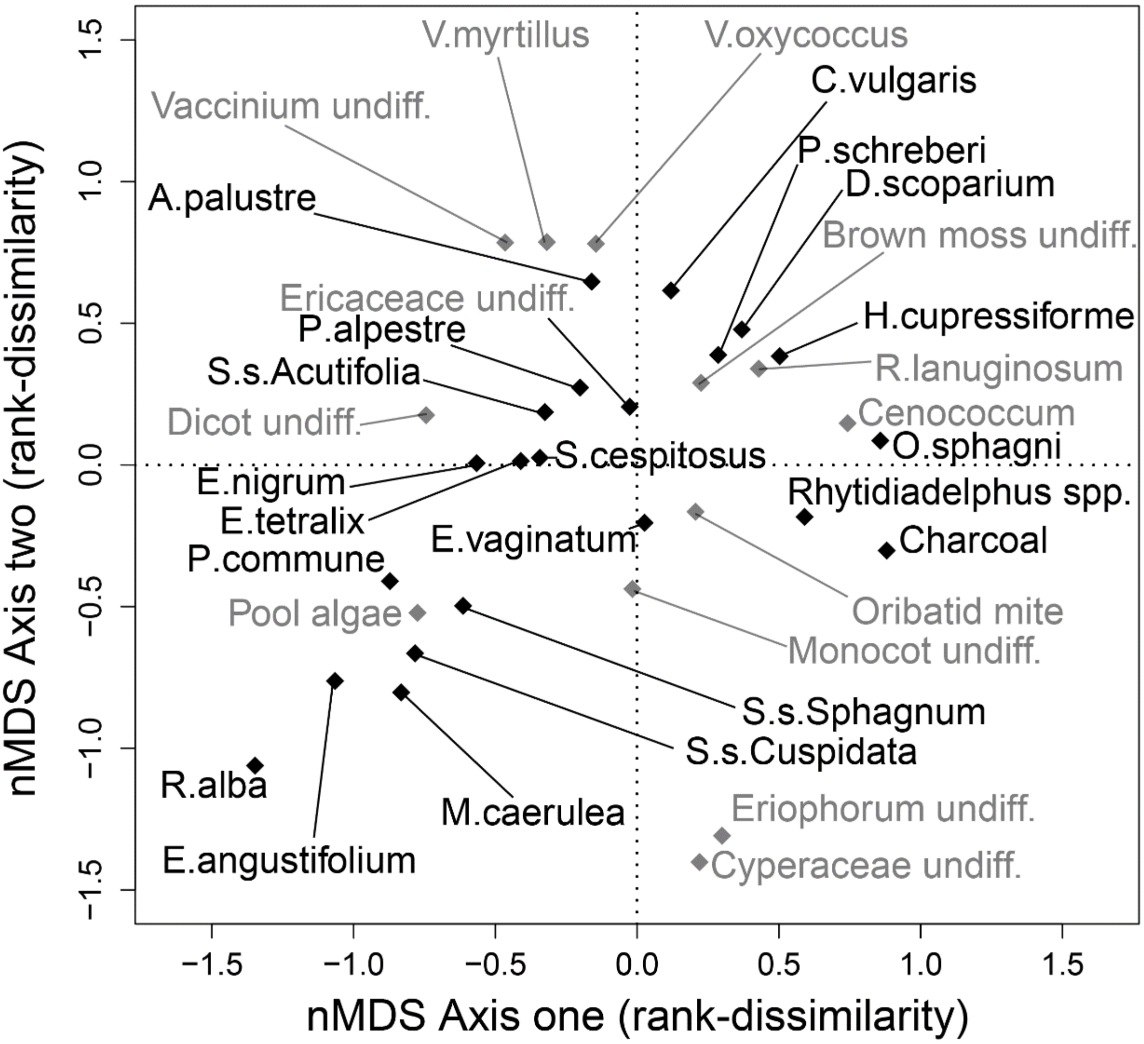


Figure 6.10: nMDS bi-plot of LVW plant macrofossil data (UOM excluded) showing taxa distribution according to their position on Axis one and two.

It is unclear what variable Axis two of the DCA represents. The low Axis two eigenvalue (0.2339) may explain why there is more visible difference in Axis two taxon groupings between the DCA (Figure 6.9) and nMDS (Figure 6.10) than for Axis one. *E. tetralix* and *S. cespitosus* plot closely together on Axis two of the nMDS but not the DCA and *R. alba* is a mid-gradient taxon on Axis two of the DCA (Figure 6.9) whilst it is in a negative position in the nMDS ((Figure 6.10). The ordinated responses of certain taxa may be confused by the relative complacency of the LVW-1-REP-B and C macrofossil records, as both display distributions of only 1-1.2 SD units on Axis two of the DCA. Whilst Axis two of both the DCA and nMDS may represent a true gradient of environmental change, that gradient is not robustly determinable from data presented here.

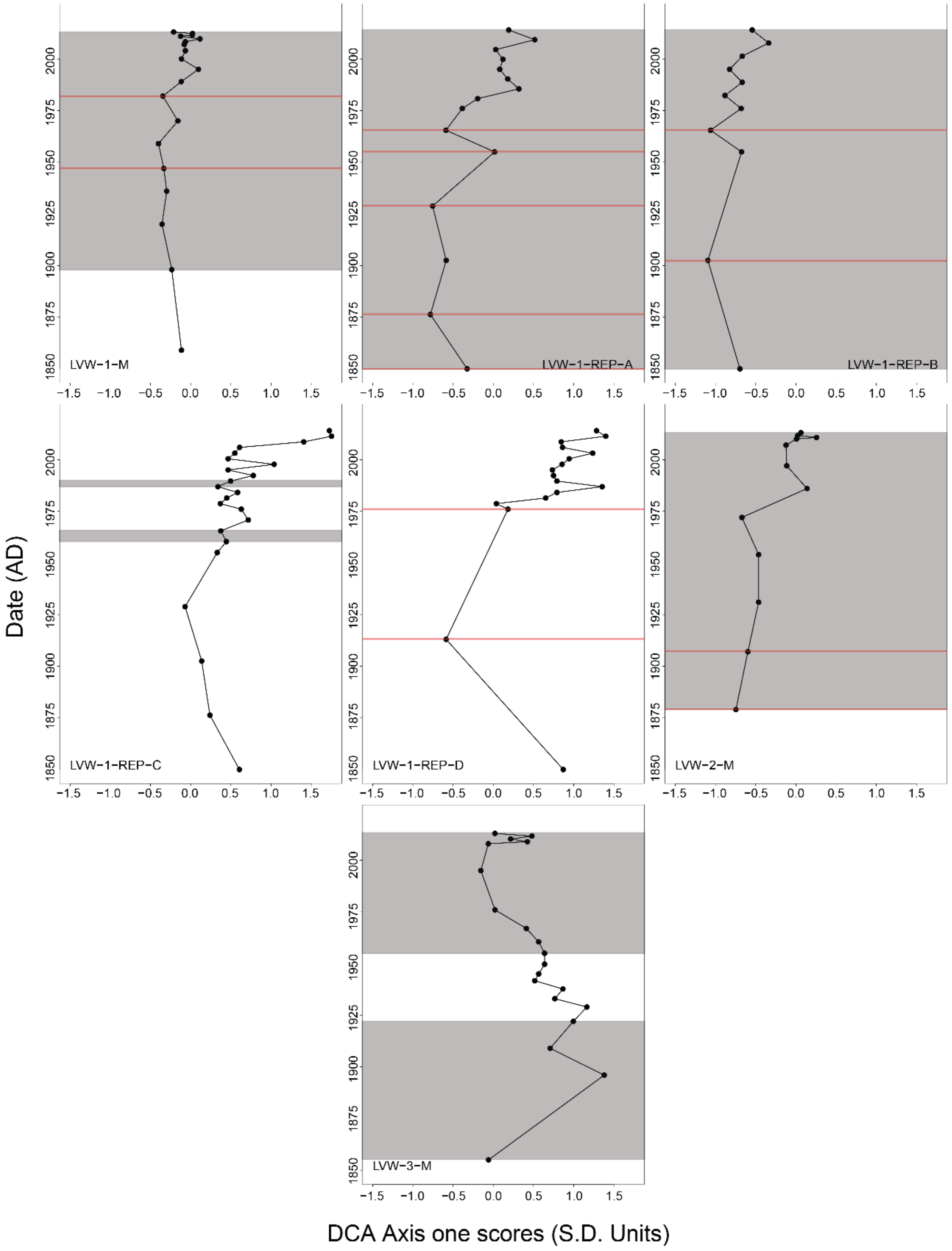


Figure 6.11: Plots showing LVW DCA (UOM excluded) Axis one sample scores plotted against calendar age. The red line indicates evidence of firing in the profile, and the grey shaded area indicates where *C. vulgaris* is present in the cores.

A grouse estate was established at LVW in AD 1800 (Section 4.3.2). Burning across the estate was more frequent and intensive from *ca.* AD 1800-1940 where small-scale rotational patch burns were implemented (Walker, M. pers. comm. 2015). Burning intensity declined following World War II (WW II) as the estate's popularity declined. During this period (*ca.* AD 1950-1975) firing occurred on a less intensive scale than the pre-WW II small-scale, intensive rotational burning. Only isolated, light burns occurred post-AD 1985 at several locations within the estate until burning stopped *ca.* AD 2003 (Walker, M. pers. comm. 2015).

Data here broadly reflect these changing firing patterns. Where charcoal is recorded post-AD 1850, firing in most cases appears to have ceased AD 1970-1982 (Figure 6.11). There is no evidence of burning at LVW-1-REP-C (Figure 6.8) and only minor evidence of burning at LVW-3-M which occurs at depths (44-42 cm) beyond the limit of the ^{210}Pb record (AD 1853 ± 23 at 39cm). LVW-1-REP-A suggests that burning was most prevalent prior to AD 1850, as this chronological marker occurs proximal to the most recent occurrence of plentiful charcoal at 30cm. The same is true at LVW-1-REP-B where the most recent occurrence of plentiful charcoal at 22cm occurs *ca.* AD 1850. The other cores where charcoal is recorded as plentiful are 38-36cm in LVW-2-M and 46cm in LVW-1-REP-D, both depths beyond the limits of their chronologies. It seems likely that firing was prevalent in the LVW region prior to the establishment of the grouse estate *ca.* AD 1800 and firing declined in intensity in the second half of the 19th Century.

Evidence of spatial and temporal heterogeneity in burning at LVW is clear, but it is unclear whether rotational burning for grouse has altered vegetation. One may expect an increase in *C. vulgaris* if firing was used for grouse management (Section 2.3.1). *Calluna* is present in the surface samples of five of the sampling points; LVW-1-M, LVW-1-REP-A and B, LVW-2-M and LVW-3-M (Figure 6.8). *C. vulgaris* is only present through the entire record of LVW-2-M. The timing of the *C. vulgaris* arrival in other LVW cores is variable and there appears to be no obvious relationship between firing and *Calluna* growth (Figure 6.11). Relatively intense burning has occurred at LVW-1-REP-A and B for some time (*ca.* 20cm of peat accumulation) before *Calluna* expansion occurs prior to AD 1850. The main expansion of *C. vulgaris* at these sites appears to occur following a decline in burning intensity. Other work shows *Calluna* may come to dominate an area *ca.* 20-25 years after burning halts (Alday *et al.* 2015; Hobbs, 1984), often alongside brown mosses (Lee *et al.* 2013) as at LVW-1-REP-B (Figure 6.8).

WTD is the key regulator of vegetation community structure at LVW (i.e. Axis one of the DCA (Figure 6.9) and nMDS (Figure 6.10)). Management has likely played a role in regulating WTD. Sites where charcoal is absent or occurs in limited occurrences (LVW-1-REP-C and LVW-3-M) are ordinated on the ‘wetter’ (positive) end of Axis one in the DCA (Figure 6.11). LVW-1-REP-D also shifts towards the ‘wetter’ end of Axis one once firing halts (Figure 6.11). Both *Calluna* and charcoal plot on the supposed drier end of Axis one in the DCA and nMDS. *C. vulgaris* and firing both promote peat desiccation, the former through promoting evapotranspiration (Armstrong *et al.* 2012; Dixon *et al.* 2015) and the latter by increasing peat temperatures (Brown *et al.* 2015a; Thompson and Waddington, 2013). Fire likely played a role in promoting peat desiccation at LVW, thereby altering vegetation communities to those favouring drier conditions.

Fire has played a role in the development of the LVW blanket peatland. Firing occurred prior to the establishment of the grouse estate *ca.* AD 1800, with burning for grouse representing a decline in burning intensity relative to earlier periods. A hydrological gradient appears to be the primary control of vegetation structure at LVW. Hydrological conditions have likely been affected by historic land use, with the use of fire which pre-dates the grouse estate causing drier affiliated plant communities to dominate much of the LVW catchment.

6.2.3 Migneint stratigraphy

The Migneint (MIG) was selected for investigation given the variety of land uses which have occurred across the blanket peatland (Section 4.3.3). The most notable variation in the macrofossil records occurs between those cores recovered within the Blaenycwm plantation and those recovered from the ‘open’ peatland. There is a hydrological gradient across the suite of cores from ‘open’ sub-sites. In most cases those sampling points characterised by drier plant communities have been affected by burning and drainage. The effects of burning at many sub-sites likely pre-date those of drainage given drains were installed exclusively during the 20th Century. These findings are expanded upon below. An example assemblage diagram is presented in Figure 6.7 whilst summary stratigraphic diagrams are detailed in Figure 6.13, Figure 6.14, Figure 6.15 and Figure 6.16. Details of stratigraphic changes are presented in Table 6.8, Table 6.9, Table 6.10 and Table 6.11.

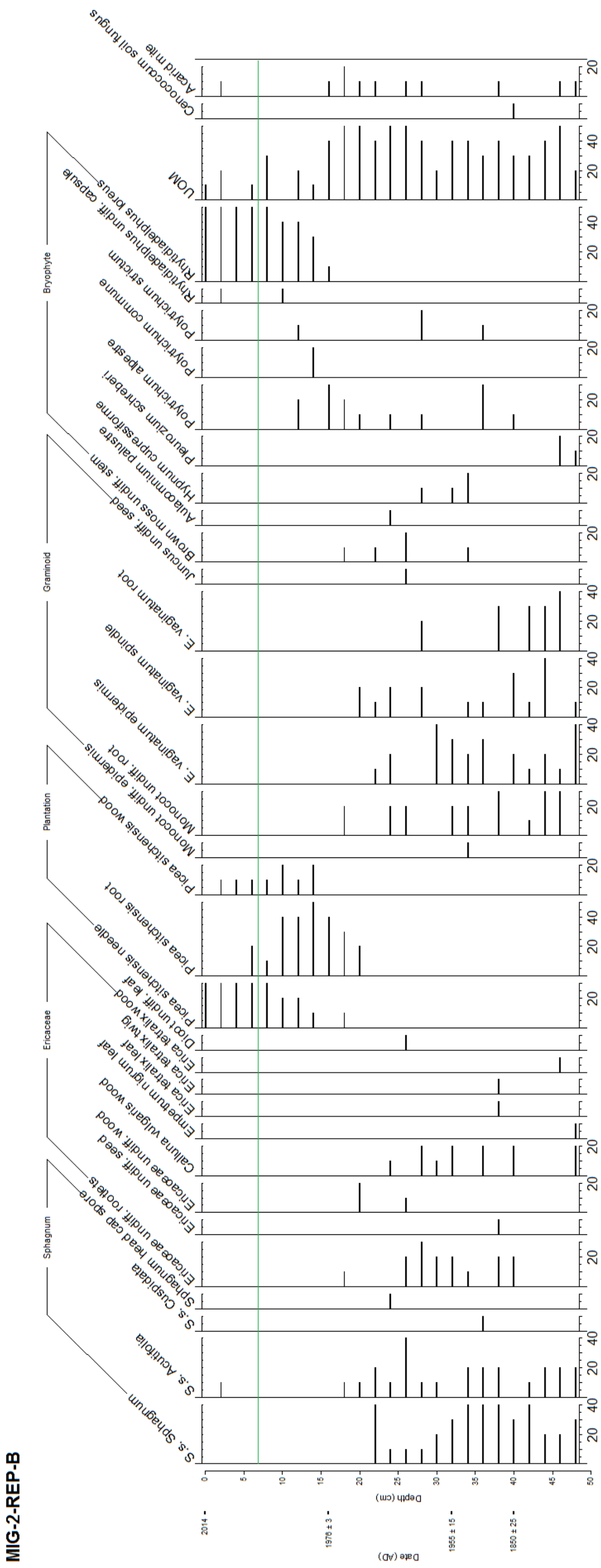


Figure 6.12: Plant macrofossil assemblage diagram for core MIG-2-REP-B plotted against depth (cm) and age estimates (AD). Green line indicates the possible position of the acrotelm/catotelm transition.

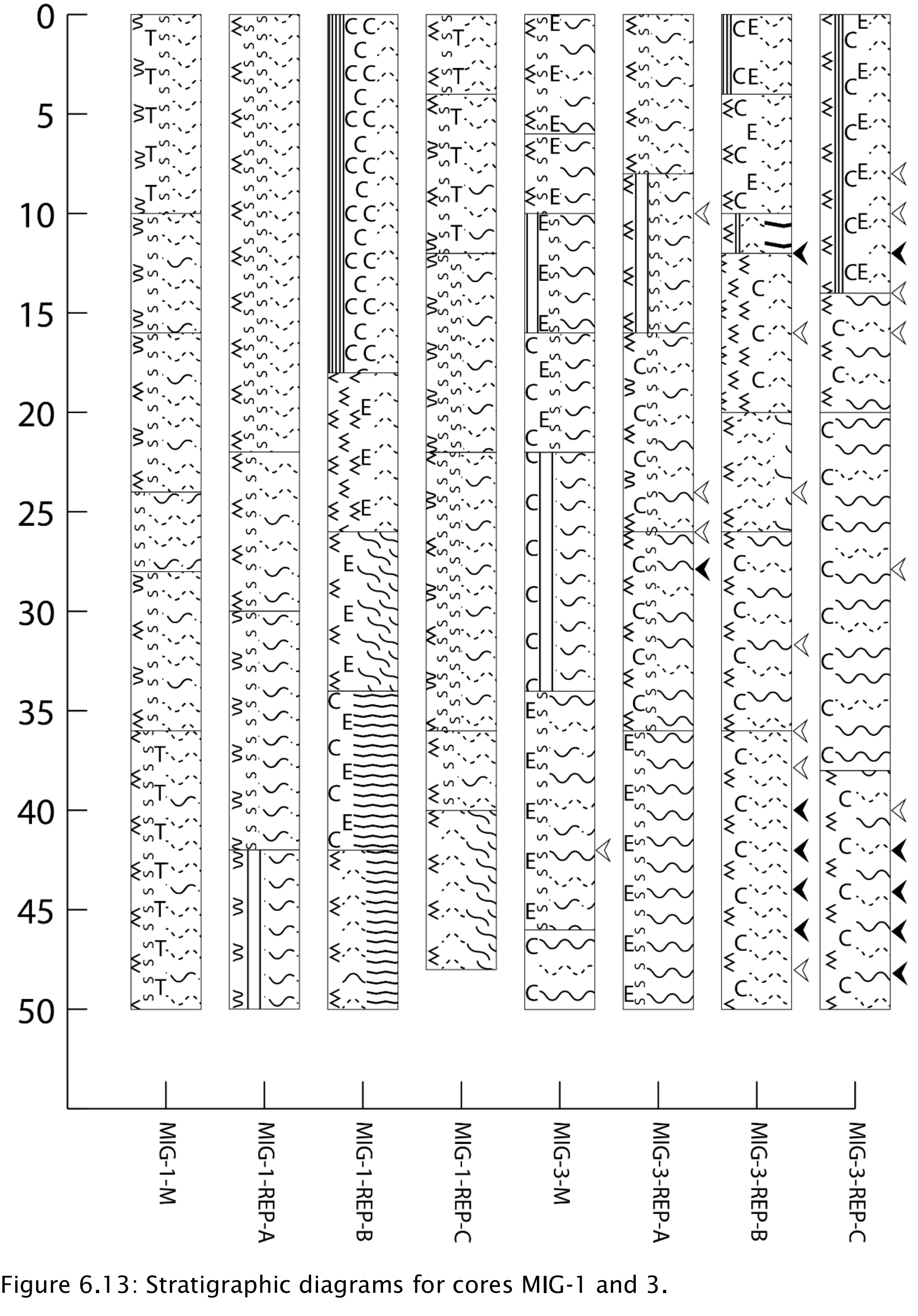


Figure 6.13: Stratigraphic diagrams for cores MIG-1 and 3.

Table 6.8: MIG-1 and 3 plant macrofossil assemblage descriptions.

Core	Vegetation assemblage description
MIG-1-M	Predominantly wet record. *S.* s. *Acutifolia* and *P. commune* persistent with *S.* s. *Cuspidata* present through most of the record. Switch from *E. vaginatum* to *E. angustifolium* as the dominant monocot at 12cm suggests increasing site wetness.
MIG-1-REP-A	Wet to dry shift occurs *ca.* 28-22cm. *S. rubellum* replaces *S.* s. *Cuspidata*, *P. alpestre* replaces *P. commune* and *E. vaginatum* replaces *E. angustifolium*.
MIG-1-REP-B	Predominately dry record. *S.* s. *Acutifolia* present 48-38cm before displaced by mixed brown mosses and *E. vaginatum* assemblage. *S.* s. *Acutifolia* abundance increases alongside *Calluna* and *S. cespitosus* above 20cm.
MIG-1-REP-C	Intermediate record. *S.* s. *Acutifolia* present throughout whilst *S.* s. *Cuspidata* absent above 8cm (*S. Cuspidatum* intermittently recorded below this point). *E. angustifolium* and *E. vaginatum* fluctuate in abundance.
MIG-3-M	Predominantly wet record. *S.* s. *Cuspidatum* near constant (*S. recurvum* at surface) with *S. papillosum* present 48-32cm and 16-2cm. *S. papillosum* decline associated with increased *Calluna*, *R. alba* and *P. commune*.
MIG-3-REP-A	Predominantly intermediate record. *S.* s. *Sphagnum* abundant from 48-22cm, but displaced by *S. recurvum* and *S. s. Acutifolia*. *E. nigrum*, *E. tetralix* and *P. commune* expand above 16cm following displacement of *S.* s. *Sphagnum*.
MIG-3-REP-B	Predominantly dry record. *E. vaginatum* and *C. vulgaris* intermittent throughout. *S.* s. *Acutifolia* near constant with *S. papillosum* present 34-26cm. *E. nigrum* and *S. cespitosus* present at surface. Evidence of plentiful charcoal.
MIG-3-REP-C	Intermediate record. *S. papillosum* displaced at 14cm by *S. rubellum* as *E. nigrum*, *E. vaginatum* and *S. cespitosus* abundance increases. Burning phases 48-40 and 16-10cm.

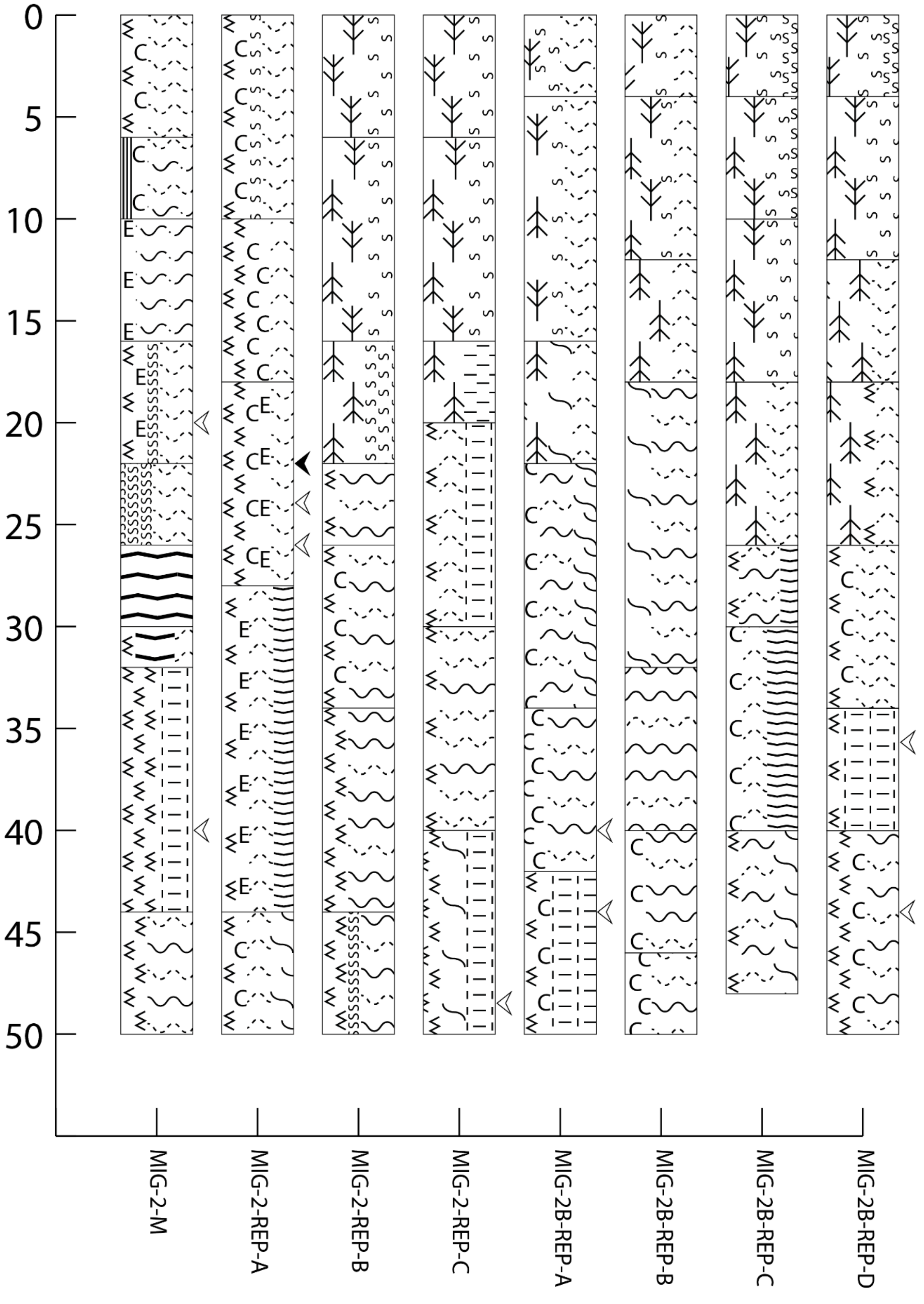


Figure 6.14: Stratigraphic diagrams for cores MIG-2 and 2B.

Table 6.9: MIG-2 and 2B plant macrofossil assemblage descriptions.

Core	Vegetation assemblage description
MIG-2-M	Likely switching between intermediate and dry conditions. Inferred intermediate phases occur 48-44cm (*S. papillosum*, UOM frequent or less abundant) and 16-6cm (*E. tetralix*, *S.* s. *Cuspidata*). Inferred dry phases 42-18cm (*E. vaginatum*, minimal *Sphagnum*, mixed brown mosses) and 4-0cm (*C. vulgaris*, *E. vaginatum*, *S. cespitosus*).
MIG-2-REP-A	Predominantly dry record. *C. vulgaris*, *E. vaginatum* and *S.* s. *Acutifolia* (*S. rubellum* at surface) present throughout.
MIG-2-REP-B	Switching from intermediate to dry afforested. Mixed *S. papillosum*, *S.* s. *Acutifolia* and *E. vaginatum* assemblage replaced by *P. sitchensis* and *R. loreus*.
MIG-2-REP-C	Switching from dry to dry afforested. *E. vaginatum* and *D. scoparium* peat (48-38cm) precedes a mixed *S. papillosum*, *S.* s. *Acutifolia* and *E. vaginatum* assemblage which is then replaced by *P. sitchensis* and *R. loreus*.
MIG-2B-REP-A	Switching from dry to intermediate afforested. *S.* s. *Acutifolia* persists across the transition. Mixed *E. vaginatum*, *C. vulgaris*, *S.* s. *Acutifolia* and *S.* s. *Sphagnum* community replaced by *P. sitchensis*, *S.* s. *Acutifolia*, *S. recurvum* and R. *loreus*.
MIG-2B-REP-B	Switching from intermediate to intermediate afforested. *S.* s. *Acutifolia* persists across the transition. Mixed *C. vulgaris*, *S.* s. *Acutifolia* and *S.* s. *Sphagnum* community with intermittent *S. cuspidatum* replaced by *P. sitchensis*, *S. s. Acutifolia*, and R. *loreus*.
MIG-2B-REP-C	Switching from dry to dry afforested. *E. vaginatum*, *C. vulgaris*, *R. lanuginosum* and *S.* s. *Acutifolia* replaced by *P. sitchensis*, *P. laetum* and *R. loreus*.
MIG-2B-REP-D	Switching from dry to dry afforested. *E. vaginatum*, *C. vulgaris* and *S.* s. *Acutifolia* replaced by *P. sitchensis* and *R. loreus*.

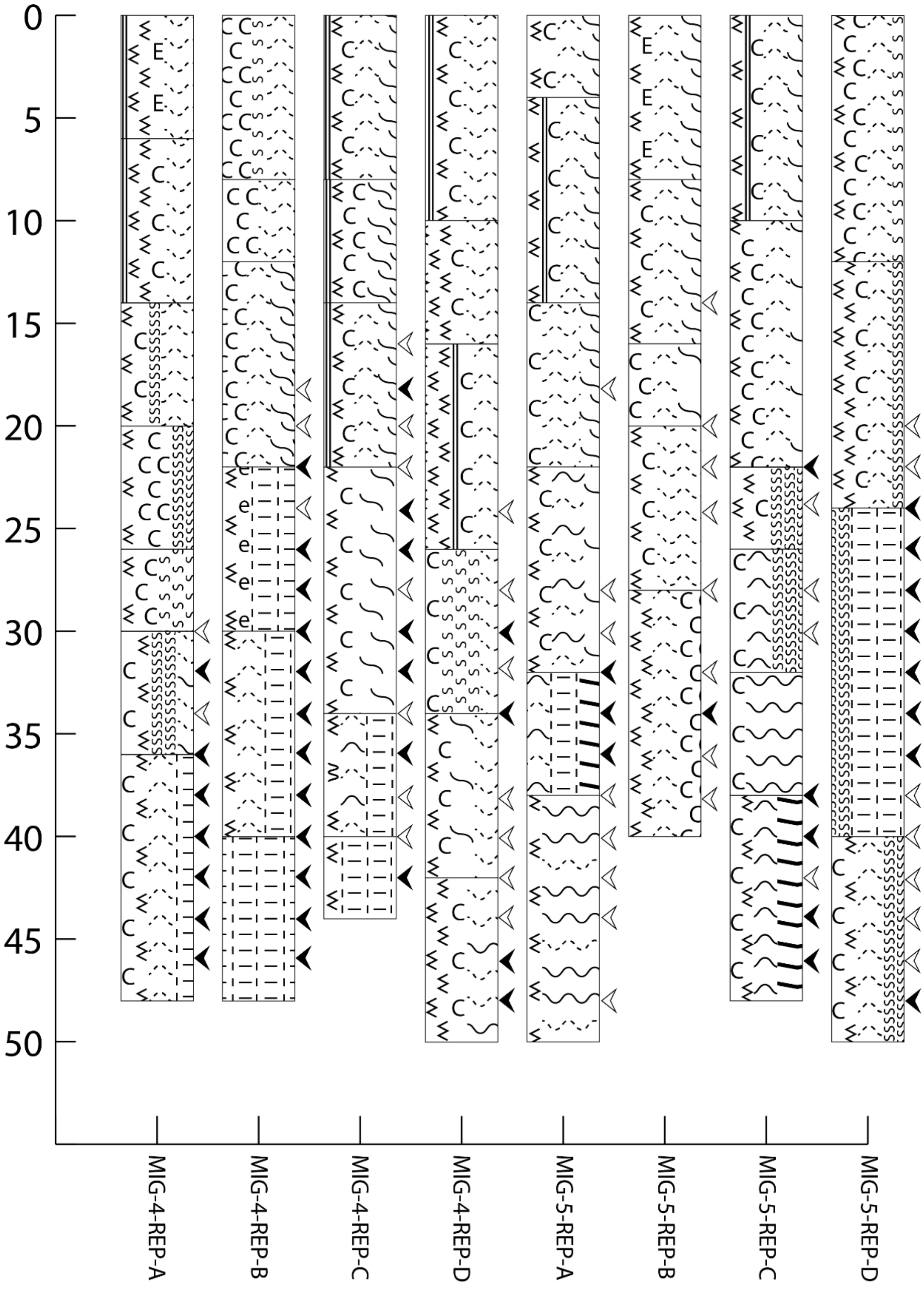


Figure 6.15: Stratigraphic diagrams for cores MIG-4 and 5.

Table 6.10: MIG-4 and 5 plant macrofossil assemblage descriptions.

Core	Vegetation assemblage description
MIG-4-REP-A	Predominantly dry record. Plentiful charcoal alongside intermittent *E. vaginatum* and *S.* s. *Acutifolia* (46-32cm). *P. alpestre* and *P. schreberi* dominate following cessation of burning. Increasingly dry at surface with *C. vulgaris*, *E. nigrum*, *S. capillifolium* subsp. *capillifolium* and *S. cespitosus*.
MIG-4-REP-B	Predominantly dry record. Plentiful charcoal alongside intermittent *E. vaginatum* and *S.* s. *Acutifolia* (46-22cm). Dry at surface with expansion of *C. vulgaris* and *S. capillifolium* subsp. c*apillifolium*.
MIG-4-REP-C	Predominantly dry record. Plentiful charcoal alongside intermittent *E. vaginatum* and *S.* s. *Acutifolia*. *C. vulgaris*, *E. nigrum*, *H. cupressiforme* and *P. schreberi* indicate mire relatively dry.
MIG-4-REP-D	Predominantly dry record. *C. vulgaris*, *E. vaginatum* and *S.* s. *Acutifolia* present through majority. *Calluna*, *E. nigrum*, *S. rubellum* and *V. myrtillus* indicate mire surface relatively dry.
MIG-5-REP-A	Shift from intermediate to dry conditions. Mixed *E. vaginatum* and *S.* s. *Sphagnum* (48-38cm) displaced by plentiful charcoal and *S.* s. *Acutifolia* and *D. scoparium* (36-32cm). *S. papillosum* increases following firing but is displaced above 20cm by *A. palustre*, *C. vulgaris*, *P. schreberi* and *S. rubellum*.
MIG-5-REP-B	Predominantly dry record. *S.* s. *Acutifolia* (*S. rubellum* at surface) and *E. vaginatum* largely present throughout. *C. vulgaris* remains through much of the core, and *P. schreberi* and *P. alpestre* frequent or common in upper 14cm.
MIG-5-REP-C	Intermediate to dry shift. *S. papillosum* expands (36-32cm) following plentiful charcoal. *S. papillosum* replaced by *S.* s. *Acutifolia* above 22cm as *E. vaginatum* expands. *A. palustre* and *S. rubellum* prevalent at surface.
MIG-5-REP-D	Predominantly dry record. Mixed *E. vaginatum* and *S.* s. *Acutifolia* (48-40cm) followed by UOM with plentiful charcoal (38-24cm). *S. rubellum* recovers above 22cm alongside *Calluna*/*E. vaginatum*.

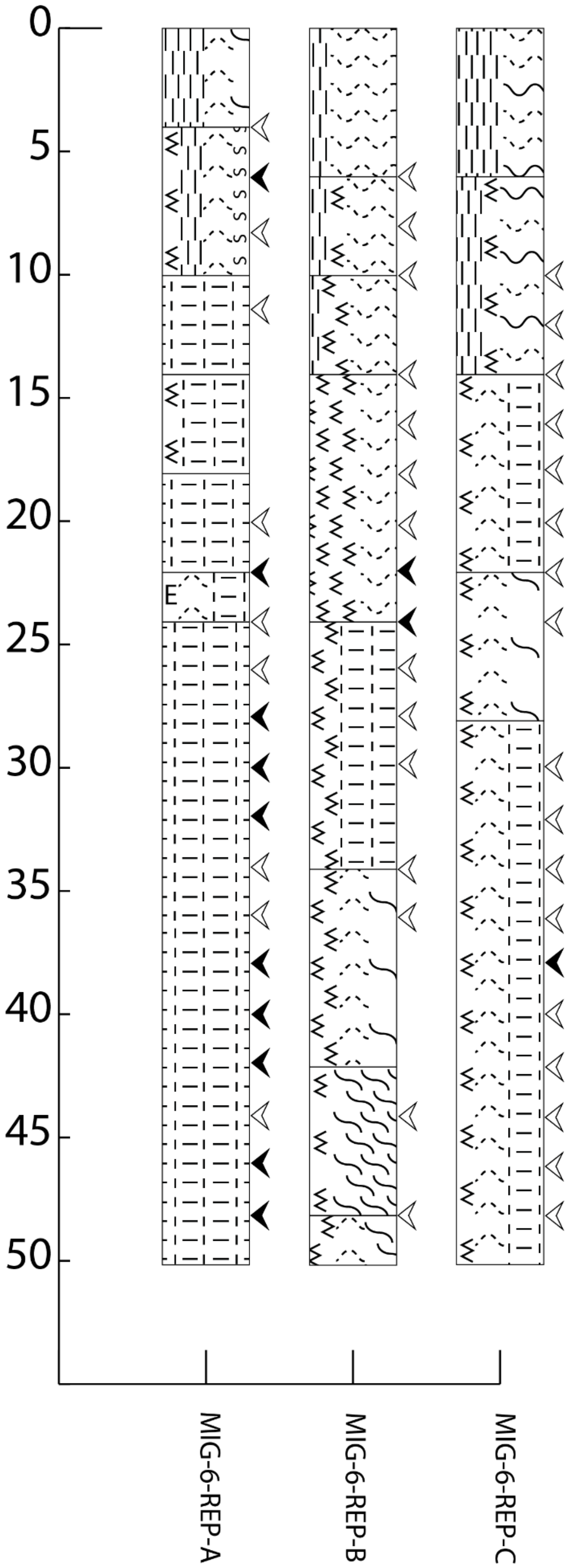


Figure 6.16: Stratigraphic diagrams for MIG-6 cores.

Table 6.11: MIG-6 plant macrofossil assemblage descriptions.

Core	Vegetation assemblage description
MIG-6-REP-A	Predominantly dry record with little identifiable material for much of the profile. Peat highly humified with plentiful charcoal from 48-24cm. *S.* s. *Acutifolia* and *Ericaceae* undiff. identifiable at 22cm, with *E. vaginatum* and *M. caerulea* recorded in the upper 16cm as another charcoal phase occurs. Rare *S.* s. *Acutifolia* at surface alongside *P. alpestre* and *P. schreberi*.
MIG-6-REP-B	Predominantly dry record. *E. vaginatum* present for much of the profile (48-34 and 30-6cm). *S.* s. *Acutifolia* intermittent during these phases, but it increases in abundance above 22cm. *M. caerulea* displaces *E. vaginatum* above 6cm, though *S.* s. *Acutifolia* (*S. rubellum*) remains abundant.
MIG-6-REP-C	Predominantly dry record. 48-16cm characterised by high UOM and constant charcoal alongside intermittent *E. vaginatum* and *S.* s. *Acutifolia*. *S. papillosum* occurs above 14cm, though is only rare at the surface as *M. caerulea* displaces *E. vaginatum*. *S.* s. *Acutifolia* frequent despite abundant *M. caerulea*.

MIG macrofossil datasets were combined so that vegetation assemblages could be assessed in the same ordination space for key differences/similarities in their development over time. Analysis was first conducted on the full dataset with UOM included where the DCA had an Axis one eigenvalue of 0.2493 and a gradient length of 4.9887. This eigenvalue is not especially high. An nMDS model was run (stress score = 0.14) for a comparison with the full MIG DCA. Procrustes rotation between the two ordinations produced a high sum of squares value of 0.57, though there was a significant positive correlation of moderate strength ($r = 0.65$, $p = 0.001$). These findings suggest that including all recorded plant macrofossil data in the MIG ordinations does not produce a reliable indicator of environmental gradients which affect blanket peatland development.

Several further ordinations were performed on the MIG dataset to explore the effect of removing specific taxa from the ordinations (Table 6.12). UOM was removed from the first ordination whilst both UOM and *Ericaceae* undiff. rootlets (UOMEUR) were removed from the second. Removing UOMEUR was the most effective alteration to the full MIG dataset as the Axis one DCA eigenvalue increased to 0.3446. Sample scores of cores recovered from Blaenycwm notably shift on Axis one to values ≥1.5 SD units following conversion to forestry (Figure 6.17) as macrofossil assemblages become characterised by *P. sitchensis* and *R. loreus* (Figure 6.18). As such, the dominant gradient of change within the full MIG dataset is the creation of forestry plantations on previously 'open' blanket bog.

It is not surprising that taxa associated with afforested sites plot away from taxa found on open bog (Figure 6.18). *P. laetum* and *R. loreus* plot ≥3 SD units on Axis one of the DCA as they occur primarily at plantation sites (Figure 6.14), though both can be found on open bogs. The shift in DCA Axis one scores and/or visible changes in peat stratigraphy occurs *ca.* AD 1969-1980 as pre-plantation vegetation declines/is lost, before forestry affiliated taxa dominate assemblages post-AD 1976-1990 (Figure 6.19). Forestry establishment at Blaenycwm occurred AD 1967-1969 (Jones, J. pers. comm. 2015). The shift in DCA Axis one scores and/or visible changes in peat stratigraphy at most sites broadly correspond to these dates for forestry plantation, or the years following establishment. Bio-stratigraphic evidence here therefore broadly agrees with SCP chronologies (Section 5.4.3). The 4cm SCP sampling resolution may have affected the age/depth relationship in MIG-2-REP-C and MIG-2B-REP-D where the transition period does not agree as well with anecdotal evidence.

Table 6.12: MIG macrofossil dataset ordination statistics. All *r* values for procrustes rotations were $p = <0.001$.

MIG Dataset	**Axis one eigenvalue**	**Axis one gradient length (SD units)**	**nMDS stress score**	**DCA vs. nMDS procrustes (*r*)**
Full	0.2493	4.9887	0.14	0.65
UOM excluded	0.3231	4.6222	0.15	0.92
UOM and *Ericaceae* undiff. rootlets excluded	0.3725	4.4858	0.15	0.91
UOM and afforested cores excluded	0.3765	4.4332	0.15	0.91
UOM, *Ericaceae* undiff. rootlets and afforested cores excluded	0.4003	4.4289	0.15	0.88

Afforestation is such a strong gradient of change that it may mask other, more subtle, environmental gradients within the MIG dataset. As such, two additional ordinations were run, the first where UOM and afforested cores (MIG-2-REP-B and C, MIG-2B-REP-A, B, C and D) were removed and the second where UOMEUR and the afforested cores were removed. The latter of these two ordinations was the most effective as the Axis one DCA eigenvalue increased to 0.4003 (Table 6.12). The DCA and an nMDS were significantly positively correlated in a procrustes rotation ($r = 0.88$, $p = 0.001$) with a lower sum of squares (0.22) than that of the full MIG dataset (0.57). These comparisons suggest the two ordinations are displaying similar results where UOMEUR and afforested cores were excluded. As such, these DCA and nMDS outputs were adopted for identification of environmental drivers of change at non-forested sites on the MIG.

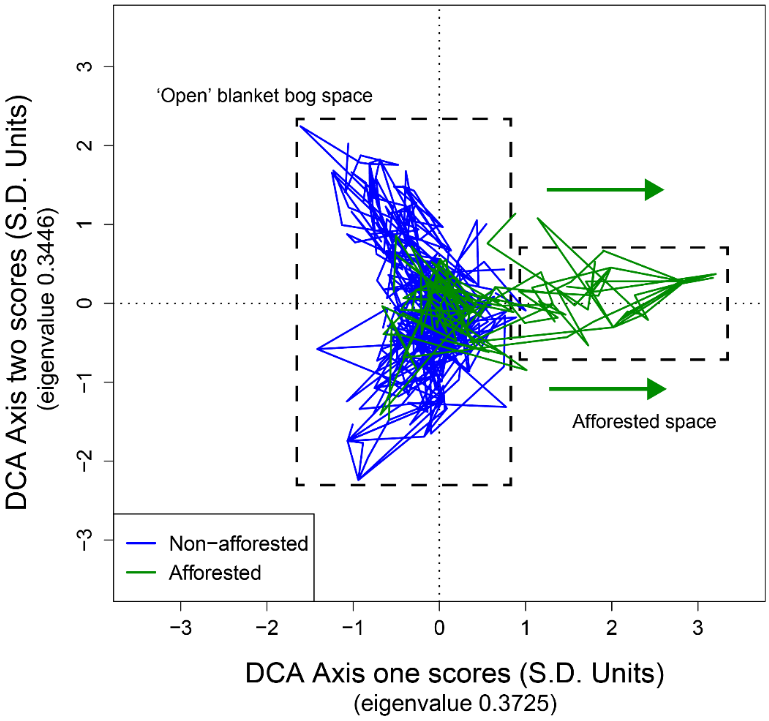


Figure 6.17: DCA bi-plot of MIG plant macrofossil data (UOM and *Ericaceae* undiff. rootlets excluded) showing sample score trajectory for each core. Note how 'afforested' cores move from a position characterised by 'open' blanket bog taxa toward a position characterised by forestry taxa.

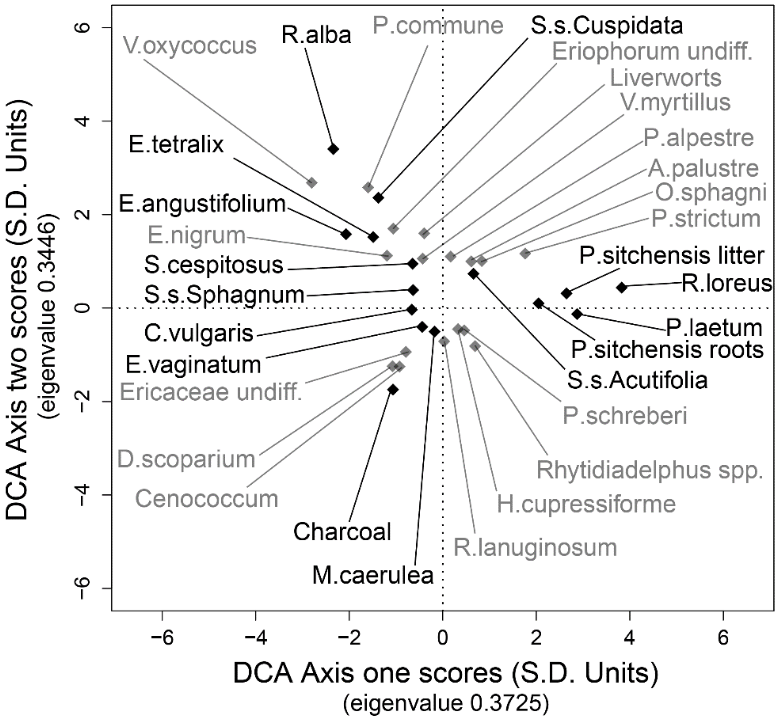


Figure 6.18: DCA bi-plot of selected taxa in the MIG plant macrofossil dataset (UOM and *Ericaceae* undiff. rootlets excluded) showing taxa distribution according to their position on Axis one and two.

Taxa distribution along Axis one (Figure 6.20) are broadly indicative of a hydrological gradient. Taxa affiliated to drier conditions are ordinated on the negative end (*D. scoparium*, *H. cupressiforme*, *P. schreberi*, *R. lanuginosum*) whilst taxa affiliated to wetter environments are predominantly ordinated >1.5 SD units (*E. angustifolium*, *E. tetralix*, *P. commune*, *R. alba* and *S.* s. *Cuspidata*) (Figure 6.20). The distribution of species from wetter to drier affiliation along the primary nMDS gradient (Figure 6.21) supports the notion that hydrological conditions are the main driver of vegetation community structure on the MIG. The nMDS orientation is reversed, hence why wetter affiliated taxa plot on the negative end of Axis one and xeric taxa on the positive end.

There is a reasonable distribution of MIG DCA Axis one sample scores across the supposed hydrological gradient (Figure 6.22). Samples which plot predominately on the negative (drier) side of Axis one are characterised by the presence of charcoal, assorted *Ericaceae* and/or *M. caerulea*. The *M. caerulea* rise at MIG-6 appears unrelated to burning in isolation given fire has clearly been used at the site prior to the arrival of *Molinia* (Figure 6.16). The ‘dry’ status of certain sites is likely related to the evidence of burning and that many ‘dry’ profiles were recovered from drained sub-sites. Not all ‘dry’ sites have a clear history of management, e.g. MIG-2-M and MIG-2-REP-A were not recovered from drained sites and have limited charcoal in their profiles. Such findings complicate attempts to determine whether firing and drainage have affected the MIG in systematic manners if ‘dry’ vegetation assemblages can develop in apparent isolation from these land uses. Sites which plot consistently on the positive (wetter) side of Axis one (Figure 6.22) are characterised primarily by mixed *Sphagnum* assemblages with little evidence of *Ericaceae* and/or charcoal (MIG-1-M, MIG-1-REP-A and C, Figure 6.13). There therefore appears to be a hydrological gradient exhibited across the MIG dataset where afforestation has not occurred and some currently ‘dry’ sites have exhibited DCA samples scores similar to ‘wet’ vegetation assemblages in the past (e.g. MIG-3-REP-C; Figure 6.22).

Taxa located on the positive end of Axis two of the DCA (Figure 6.20) may signify sites transitioning toward heath-like conditions. Most brown mosses plot >0 SD units on Axis two, as do all drier affiliated *Ericaceae*, and none of the taxa >0 SD units on Axis two of the DCA (save for *Eriophorum* undiff.) plot >1.5 SD units on Axis one, the supposed wetter end of the gradient. The Axis two eigenvalue is only 0.2534 so this interpretation should be treated with caution.

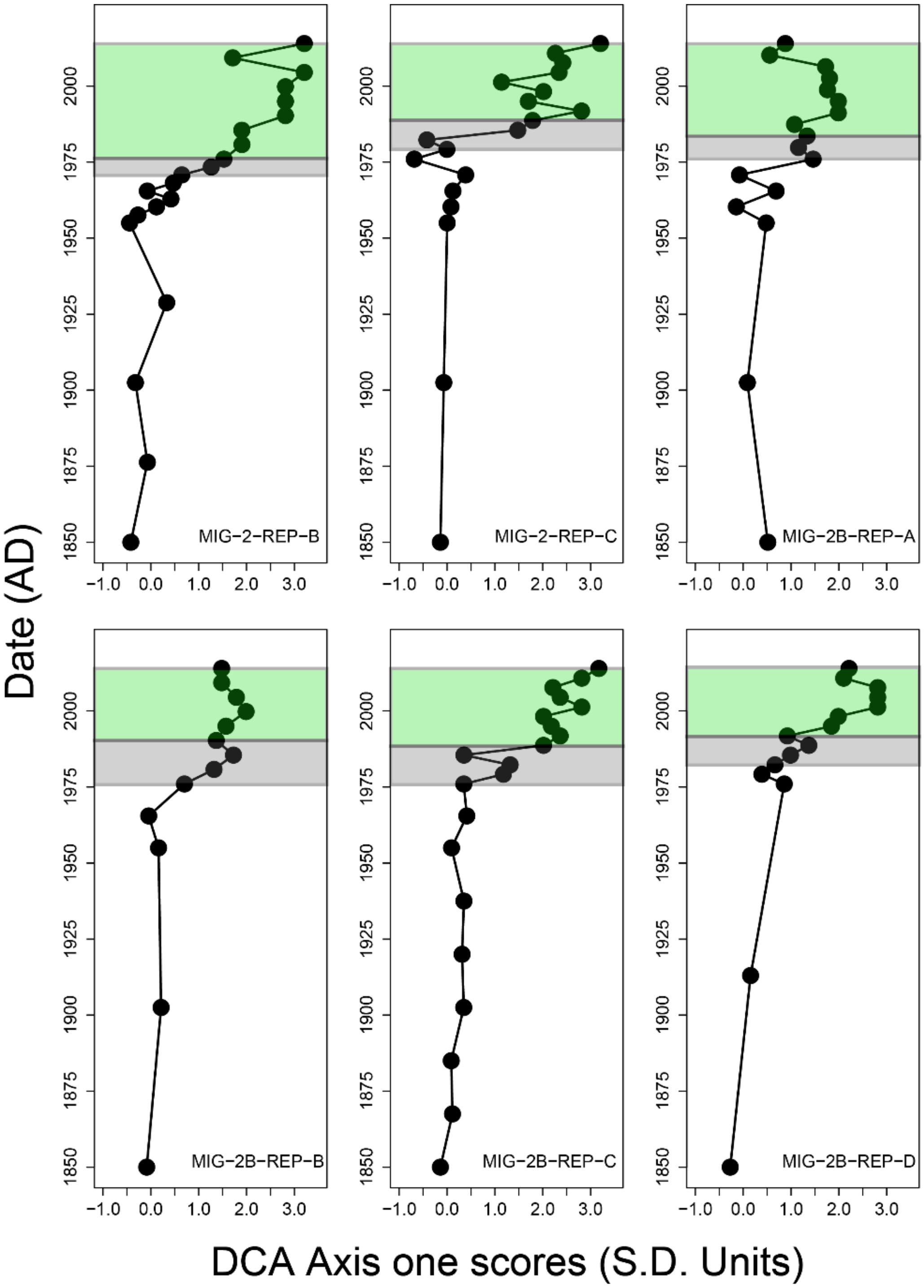


Figure 6.19: Plots showing MIG DCA (UOMEUR) Axis one sample scores for afforested cores plotted against calendar age. Grey shaded areas indicate a transition period where previous bog vegetation declines/is lost, whilst green shaded areas indicate where plantation affiliated taxa dominate as discussed in the text.

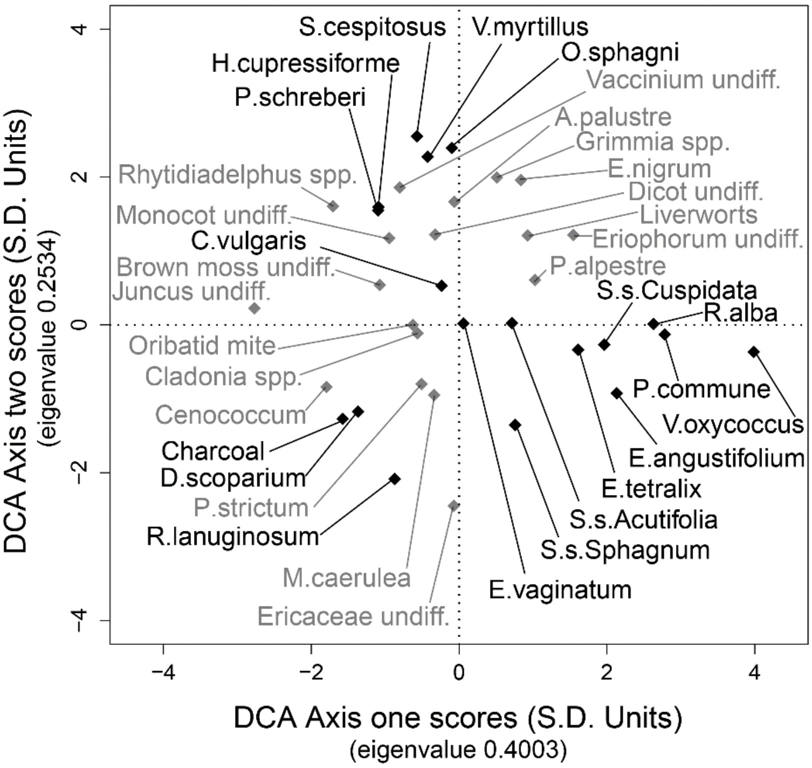


Figure 6.20: DCA bi-plot of the MIG plant macrofossil data (UOMEUR and afforested cores excluded) showing taxa distribution according to their position on Axis one and two.

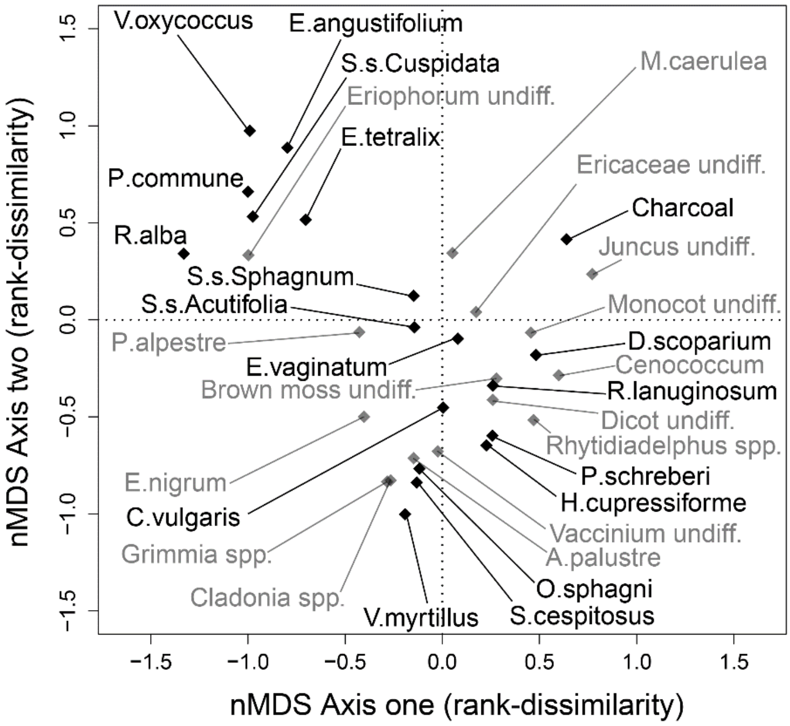


Figure 6.21: nMDS bi-plot of the MIG plant macrofossil data (UOMEUR and afforested cores excluded) showing taxa distribution according to their position on Axis one and two.

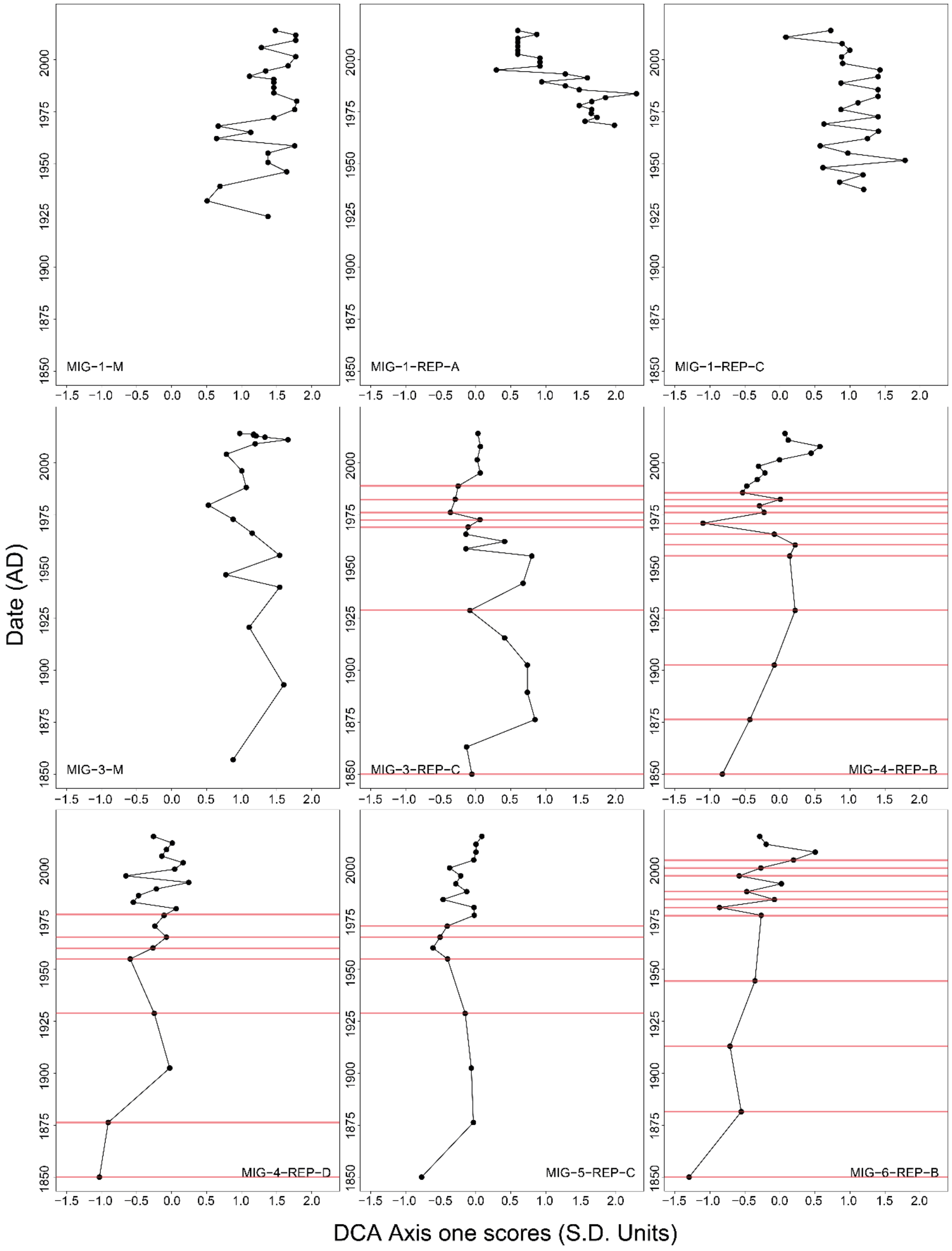


Figure 6.22: Plots showing MIG DCA (UOMEUR and afforested cores excluded) Axis one sample scores for selected cores against calendar age. Red line indicates evidence of firing.

There is much variability recorded in vegetation records recovered from the MIG. Forestry has clearly caused the most notable change in peatland vegetation. It appears that burning and drainage have caused some sites to transition to drier conditions, as inferred from the expansion of xeric taxa. Firing and/or drainage have not been responsible for all expansions of *Calluna* and other xeric taxa across the MIG and firing is unlikely to have played a role in the expansion of *M. caerulea* at MIG-6. As such, macrofossil results from the MIG demonstrate the difficulty in identifying a consistent impact of a given land use when using palaeoenvironmental records.

6.2.4 Plynlimmon stratigraphy

Plynlimmon (PLY) was selected for investigation because of its history of being used for grazing (Section 4.3.4). There appears to be a hydrological gradient which dictates vegetation community structure at PLY. Cores which display taxa favouring drier conditions were typically recovered from amongst peat haggs. PLY-1-M was also recovered from a hagged site, but it exhibits an assemblage characterised by more mesic taxa for the majority of the 50cm core window. Several cores also exhibit periods of peat accumulation with large amounts of UOM and charcoal, after which *S.* s. *Sphagnum* recovers and is dominant at present. These cores were all recovered from *Sphagnum* lawns and appeared relatively undisturbed when cores were recovered. There is evidence of burning in all records recovered from PLY, so it is uncertain whether burning alone played a key role in altering hydrological conditions. These findings are expanded upon below. An example assemblage diagram is presented in Figure 6.23 whilst summary stratigraphic diagrams are presented in Figure 6.24. Details of stratigraphic changes are presented in Table 6.13.

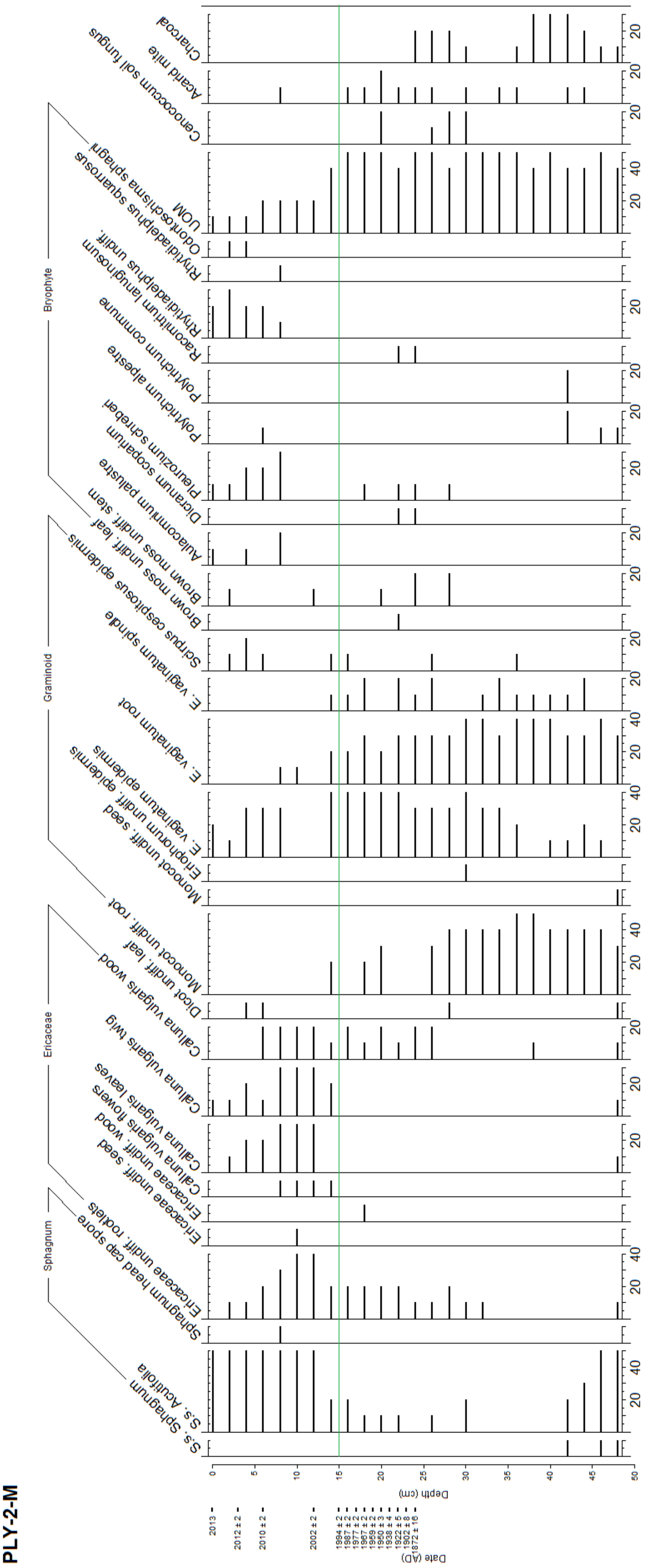


Figure 6.23: Plant macrofossil assemblage diagram for core PLY-2-M plotted against depth (cm) and age estimates (AD). Green line indicates the possible position of the acrotelm/catotelm transition.

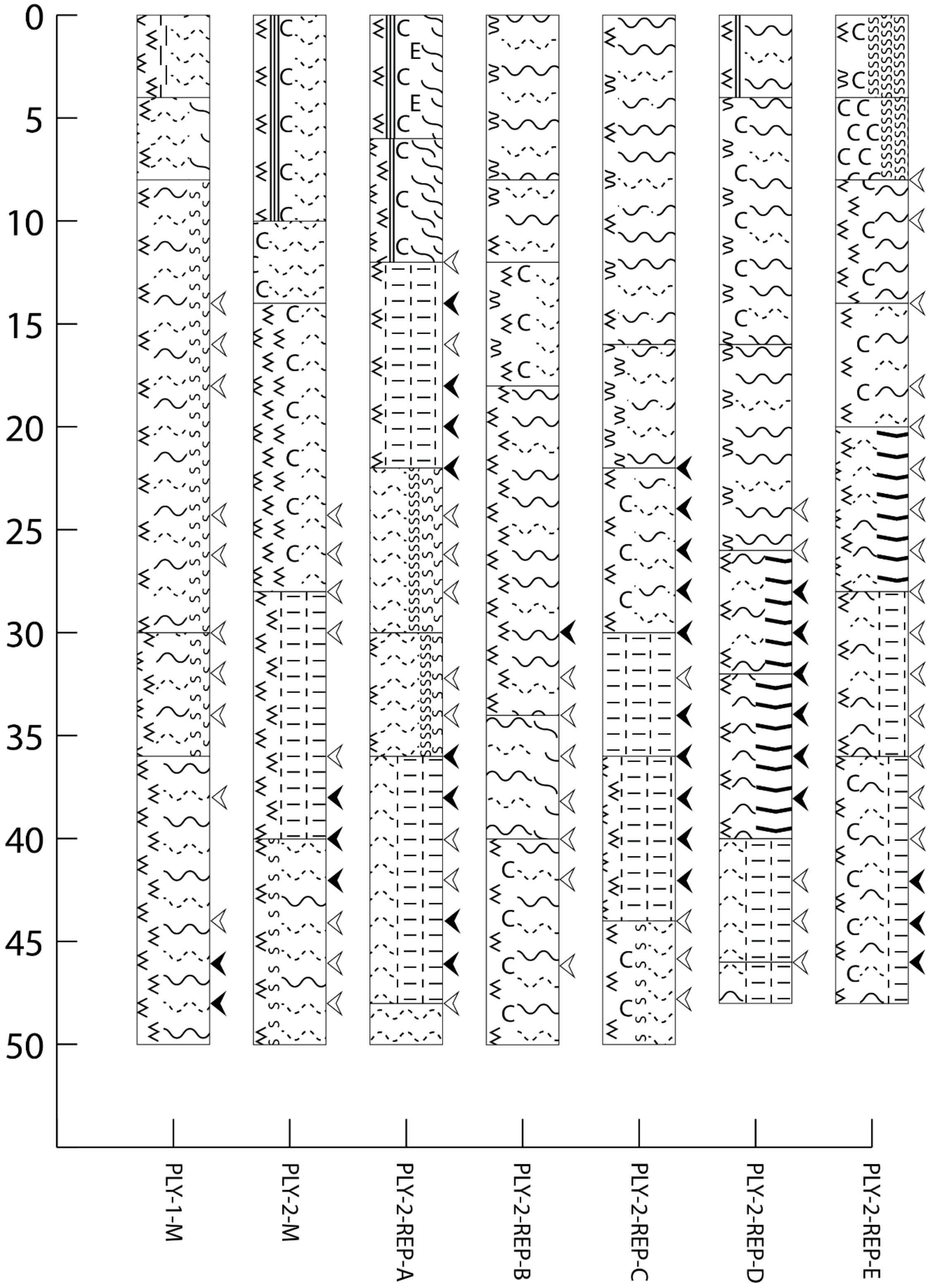


Figure 6.24: Stratigraphic diagrams for cores recovered from PLY.

Table 6.13: PLY plant macrofossil assemblage description.

Core	Vegetation assemblage description
PLY-1-M	Slight drying trend in the record. *E. vaginatum* present throughout whilst *S.* s. *Sphagnum* declines and *S.* s. *Acutifolia* increases alongside xeric *P. alpestre* and *P. schreberi*. Intermittent burning in the record.
PLY-2-M	Predominantly dry record. Mixed *E. vaginatum* and *S.* s. *Acutifolia* assemblage with rare *S.* s. *Sphagnum* 48-42cm replaced by *E. vaginatum* dominated community following burning 42-38cm. *C. vulgaris* established at 26cm. A relatively dry mire surface is inferred from the presence of *Calluna* alongside *S. rubellum*, *E. vaginatum* and *P. schreberi* above 10cm.
PLY-2-REP-A	Predominantly dry record. *S.* s. *Acutifolia* was previously abundant but is absent at the surface, and only recorded rarely at two increments in the upper 20cm. *Calluna* dominates at present alongside *E. vaginatum*, *S. cespitosus* and a range of xeric bryophytes. Abundant evidence of burning through the record.
PLY-2-REP-B	Dry to wet shift in the record. *S.* s. *Sphagnum* and *S.* s. *Acutifolia* co-dominate through much of the profile, with *E. vaginatum* the primary monocot and only intermittent, rare *Calluna* wood. From 18-12cm, *S.* s. *Sphagnum* is largely absent, though it recovers (*S. papillosum*) above this as *E. angustifolium* and *V. oxycoccos* appear, suggesting a recent wetting trend.
PLY-2-REP-C	Dry to wet shift in the record. 48-28cm characterised by high UOM alongside rare *E. vaginatum* remains, predominately rare *P. alpestre*, and largely devoid of *Sphagnum*. Plentiful charcoal recorded during this period. *S.* s. *Sphagnum* presence increases above 22cm, alongside *E. angustifolium*.

Table 6.13 (cont.)

PLY-2-REP-D	Change from dry to intermediate conditions through the record. 46-26cm characterised by high UOM, limited *Sphagnum* and intermittent *E. vaginatum* remains. Expansion of *D. scoparium* occurs during phase of intense burning 38-26cm. *S.* s. *Sphagnum* and *S.* s. *Acutifolia* expand above this alongside *E. angustifolium*. Intermittent *Calluna* remains 14-4cm suggest the system is not waterlogged.
PLY-2-REP-E	Predominantly dry record. Macrofossil remains, apart from mixed monocot roots, are sparse from 48-26cm. During this period, *S.* s. *Sphagnum*, *S.* s. *Acutifolia*, *E. vaginatum*, *P. schreberi* and *Calluna* are identifiable, though all are only ever recorded as rare or occasional. *D. scoparium* expands following a brief increase in charcoal abundance, before *E. vaginatum*, *S.* s. *Sphagnum* and *S.* s. *Acutifolia* expand from 18-8cm. *Calluna* and *P. schreberi* displace *Sphagnum* from the record above 8cm, apart from a single rare *S.* s. *Sphagnum* occurrence at 0cm.

PLY macrofossil datasets were combined so that vegetation assemblages could be assessed in the same ordination space for key differences/similarities in their development over time. Ordinations were initially conducted on the full dataset with UOM included where the DCA had an eigenvalue of 0.2302 and a gradient of 2.8057. Both DCA and PCA normalised Axis one sample scores were positively correlated with their equivalent sample scores in an nMDS (stress score =0.167) (Table 6.14). Both unimodal (DCA) and linear (PCA) models are likely exhibiting significant positive correlations with the nMDS because the DCA Axis one eigenvalue (0.2309) is within the range (0.2-0.3) where either model may be used.

Table 6.14: Comparison of ordination techniques applied to the PLY macrofossil dataset Axis one sample scores where UOM was included.

	nMDS vs. DCA		**nMDS vs. PCA**	
	***R*-value**	***P*-value**	***R*-value**	***P*-value**
PLY-1-M	0.81	<0.001	0.89	<0.001
PLY-2-M	0.69	<0.001	0.71	<0.001
PLY-2-REP-A	0.45	<0.001	0.75	<0.001
PLY-2-REP-B	0.87	<0.001	0.94	<0.001
PLY-2-REP-C	0.97	<0.001	0.96	<0.001
PLY-2-REP-D	0.93	<0.001	0.95	<0.001
PLY-2-REP-E	0.83	<0.001	0.80	<0.001

Two further ordinations were performed on the PLY dataset to explore the effect that removing specific taxa (UOM and *Ericaceae* undiff. rootlets) had on the ordination results. The most effective ordination was where both UOM and *Ericaceae* undiff. rootlets (UOMEUR) were removed as it exhibited the greatest Axis one eigenvalue (Table 6.15). UOM was likely reducing the explanatory power of the DCA given it can originate from any taxon and *Ericaceae* rootlets from overlying dry phases were likely affecting the responses of taxa within the sampling period of interest. DCA and nMDS axis one sample scores remain significantly positively correlated following the exclusion of UOMEUR (Table 6.16) and procrustes correlation between DCA and nMDS ordinations was strongest where UOMEUR were excluded (Table 6.15). These findings suggest the exclusion of UOMEUR produces the most reliable results for interpreting environmental change at PLY, hence this ordination is discussed below.

Table 6.15: PLY macrofossil dataset ordination statistics. All *r* values for procrustes rotations were $p = 0.001$.

PLY Dataset	**Axis one eigenvalue**	**Axis one gradient length (SD units)**	**nMDS stress score**	**DCA vs. nMDS procrustes (*r*)**
Full	0.2302	2.8057	0.167	0.83
UOM excluded	0.2785	3.0853	0.168	0.83
UOM and *Ericaceae* undiff. rootlets excluded	0.293	2.8148	0.168	0.88

Table 6.16: Comparison of ordination techniques applied to the PLY macrofossil dataset Axis one sample scores where UOMEUR were excluded.

	nMDS vs. DCA		**nMDS vs. PCA**	
	***R*-value**	***P*-value**	***R*-value**	***P*-value**
PLY-1-M	0.54	<0.005	-0.69	<0.001
PLY-2-M	0.54	<0.005	-0.72	<0.001
PLY-2-REP-A	0.40	<0.05	-0.56	<0.005
PLY-2-REP-B	0.74	<0.001	-0.85	<0.001
PLY-2-REP-C	0.98	<0.001	-0.90	<0.001
PLY-2-REP-D	0.94	<0.001	-0.90	<0.001
PLY-2-REP-E	0.72	<0.001	-0.68	<0.001

Taxa orientation within the accepted DCA ordination (Figure 6.25) suggests that Axis one represents a hydrological gradient. Taxa favouring wetter environments are ordinated on the positive end of Axis one (*E. angustifolium*, *P. commune S.* s. *Cuspidata* and *V. oxycoccos*). Taxa ordinated between -1 and 1SD units may indicate fluctuating water conditions as *C. vulgaris*, *E. tetralix* and *E. vaginatum* all tolerate such conditions. *S.* s. *Acutifolia* plots within this supposed fluctuating water table range which is logical because this taxon is only reported here at the sub-genus level and there are several species within the group with wet and dry requirements (Daniels and Eddy, 1985). The negative end of Axis one of the DCA indicates drier hydrological conditions given taxa located ≤-1 SD units often favour xeric environments (*E. nigrum*, *H. cupressiforme*, *O. sphagni* and *P. schreberi*) (Figure 6.25). The nMDS supports the conclusion that a hydrological gradient regulates vegetation assemblages at PLY (Figure 6.26). *M. caerulea* has shifted toward intermediate water tables within the nMDS (Figure 6.26) relative to its strong dry affiliation in the DCA (Figure 6.25). *M. caerulea* does inhabit wetter areas of peatlands as it is only waterlogged conditions that it cannot tolerate (Mauquoy and van Geel, 2007). The assemblages it is recorded in at PLY are characterised also by *S.* s. *Acutifolia* and *E. vaginatum*, both taxa which favour intermediate moisture conditions. Results cannot therefore be used to identify what environmental driver has caused *M. caerulea* to appear at PLY because its ordinated location differs between the two techniques used (DCA and nMDS).

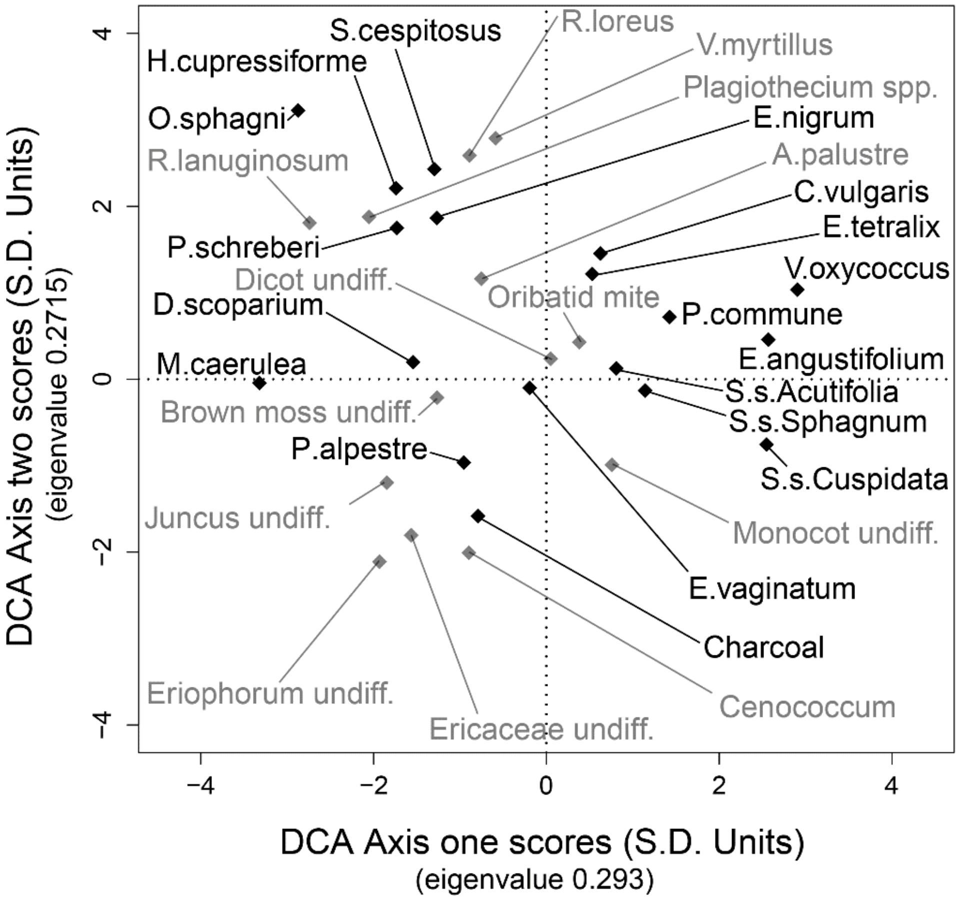


Figure 6.25: DCA bi-plot of PLY plant macrofossil data (UOMEUR excluded) showing taxa distribution according to Axis one and two position.

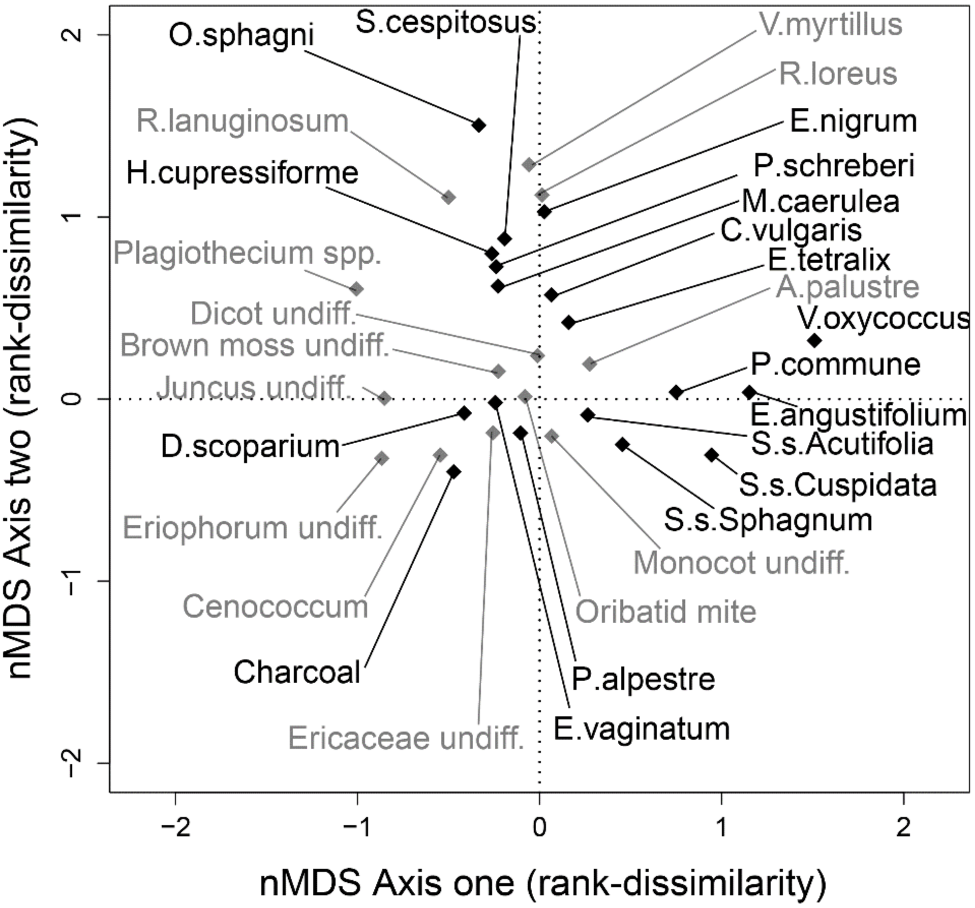


Figure 6.26: nMDS bi-plot of PLY plant macrofossil data (UOMEUR excluded) showing taxa distribution according to Axis one and two position.

Axis two of the DCA displays a gradient of taxa response to fire. The negative values on Axis two of both the DCA (Figure 6.25) and nMDS (Figure 6.26) indicate taxa inhabiting environments where fire had a role in dictating vegetation community structure, whilst positive values indicate samples where fire is not prevalent. *D. scoparium* and *E. vaginatum* are ordinated close to charcoal on the negative end of Axis two in the DCA (Figure 6.25) and nMDS (Figure 6.26). These are reasonable ordinations given known taxon affinity to burning (e.g. nutrient release). *S.* s. *Sphagnum* and *S.* s. *Cuspidata* plot on the negative side of Axis two. Such plotting may be surprising given the prevailing, albeit often untested, view that firing is detrimental to *Sphagnum* growth (Bain *et al.* 2011). *S.* s. *Sphagnum* and *S.* s. *Acutifolia* manage to persist through some burning phases across the PLY datasets, though they are often not very abundant (≤2 on the five-point abundance scale) during these phases (Figure 6.24). As such, the ordinated results show that both charcoal and *Sphagnum* occupy a similar environmental niche given they are both present in these samples. Evidence here may therefore suggest that *Sphagnum* can persist alongside burning, though its abundance does seem to increase following periods of burning (e.g. PLY-2-REP-B, C and D).

A hydrological gradient was detected in ordinations performed on the PLY macrofossil dataset. The primary land use at PLY has been grazing, though fire has clearly been used in conjunction with grazing. As such, it is uncertain which of these two land uses has most affected vegetation development by affecting the hydrological conditions at these sites. Cores characterised by ‘drier’ taxa were typically recovered from actively hagging areas. As such, fire and/or grazing may have caused the initiation of these erosion complexes. Cores which are currently characterised by *S. papillosum* (PLY-2-REP-B, C and D) present an interesting ecological history. The cessation of burning at these sites seems to have allowed the expansion of *Sphagnum*, hence these systems may be exhibiting a recovery from previously intense phases of land use.

6.3 Assessment of the full palaeoecological assemblage

This chapter has so far highlighted controls on vegetation change at individual study sites as well as commenting on the difficulties in accurately reconstructing the effects of a given land use (e.g. drainage, grazing) on peatland development. It seems premature to assume the macrofossil records presented here do not reflect the effects of land use, but rather that identifying the discrete impacts of each land use type may not be possible. This section explores the environmental controls on blanket peatland development across the whole macrofossil dataset (Section 6.3.1) and assesses whether grouping sites according to those which exhibit similar patterns of vegetation change over time is an appropriate way to identify land use impacts (Section 6.3.2).

6.3.1 Identifying environmental controls on vegetation development across the study sites

BRP, LVW, MIG and PLY macrofossil datasets were combined into a single ‘ALL CORE’ dataset to identify forcings which may have driven key differences/similarities in sampling point development over time. Grouping cores in this way allows broad scale changes across the dataset to be identified, which is a key point given this thesis seeks to determine whether land use types may cause ‘systematic’ changes in peatland development, somewhat regardless of site specific conditions.

Several ordinations were performed on the ALL CORE dataset. The most effective ordination (DCA Axis one eigenvalue 0.3561) occurred where UOM, *Ericaceae* undiff. rootlets, afforested cores and rare taxa were excluded (UEAR) (Table 6.17; Figure 6.27). Rare taxa were *Cladonia*, *Cyperaceae* undiff., *Juncus* sp., Liverworts undiff., Monocot undiff., *Polytrichum* undiff. and *Polytrichum strictum*. These taxa were selected for removal as 1) they were not recorded at many sampling points in this thesis and 2) it increased the Axis one eigenvalue relative to the next best performing DCA ('UOM and afforested cores excluded'; Table 6.17).

Ordinations conducted on the suite of cores recovered in this study, save for afforested cores (Section 6.2.3), suggest that Welsh blanket peats have developed under a variety of hydrological conditions. Xeric taxa are ordinated ≤-1 SD units on Axis one of the DCA (*Cenococcum*, *D. scoparium*, *H. cupressiforme* and *P. schreberi*) alongside charcoal (Figure 6.27). Taxa ordinated between -0.5 to 1 SD units on Axis one appear to represent fluctuating/intermediate water tables (*E. vaginatum*, *S.* s. *Acutifolia*) whilst taxa ordinated ≥1 indicate wetter conditions (*P. commune*, *R. alba*, *S.* s. *Cuspidata* and *S.* s. *Sphagnum*). Axis one of the nMDS supports this interpretation (Figure 6.28) with the orientation reversed so xeric taxa plot on the positive end of Axis one. Axis two of the DCA (Figure 6.27) represents a gradient from communities more indicative of blanket bog on the negative end of the gradient (*E. angustifolium*, *E. tetralix*, *S.* s. *Cuspidata*, *S.* s. *Sphagnum*) and taxa representing humid heath conditions on the positive end (*E. nigrum*, *H. cupressiforme*, *P. schreberi*, *Rhytidiadelphus* sp., *S. cespitosus*). All taxa which plot >1 SD units on Axis two of the DCA are ordinated <1SD units on Axis one of the DCA and hence represent intermediate/drier conditions (Figure 6.27). Axis two of the nMDS supports this interpretation (Figure 6.28), with the gradient reversed so heath taxa plot on the negative end.

The identification of Axis one as a hydrological gradient for both the DCA and nMDS ALL CORE ordinations matches the interpretation of Axis one in three of the four site-specific ordinations presented in Section 6.2. Clearly therefore, hydrological controls play an important role in driving vegetation community composition across Welsh blanket peats. Internal or climatic forcings are not necessarily the drivers of such hydrological changes. As commented on in Section 6.2, land use has the capacity to alter peatland hydrological conditions and so may dictate vegetation community structure. It seems likely this is the case with many records in this thesis. Cores chacterised by xeric vegetation communities were often either recovered from areas of visible disturbance (e.g. amongst drains or from peat haggs) or display evidence of disturbance in their record (e.g. abundant charcoal). As such, whilst the expression of a discrete land use type (e.g. burning, drainage) may not be identifiable in a given record, it does seem possible to identify the general expression of management/disturbance using plant macrofossil analyses. The applicability of this finding is explored below.

Table 6.17: ALL CORE dataset ordination statistics. All procrustes rotation r values were $p = 0.001$.

ALL CORE Dataset	**Axis one eigenvalue**	**Axis one gradient length (SD units)**	**nMDS stress score**	**DCA vs. nMDS procrustes (*r*)**
Full	0.1876	5.4575	0.15	0.53
UOM excluded	0.2393	4.4251	0.15	0.74
UOM and *Ericaceae* undiff. rootlets excluded	0.2518	4.4946	0.16	0.87
UOM and afforested cores excluded	0.3382	4.6730	0.15	0.90
UOM, *Ericaceae* undiff. rootlets, afforested cores and rare taxa excluded	0.3561	4.5004	0.15	0.90

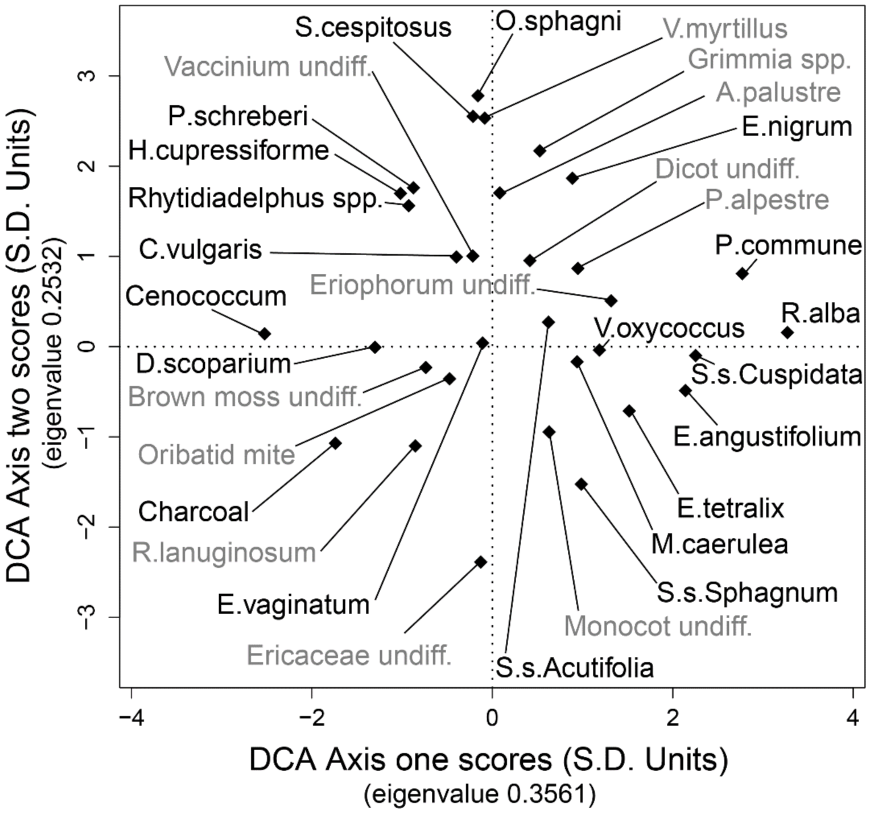


Figure 6.27: DCA bi-plot of the ALL CORE plant macrofossil data (UEAR excluded) showing taxa distribution according to Axis one and two position.

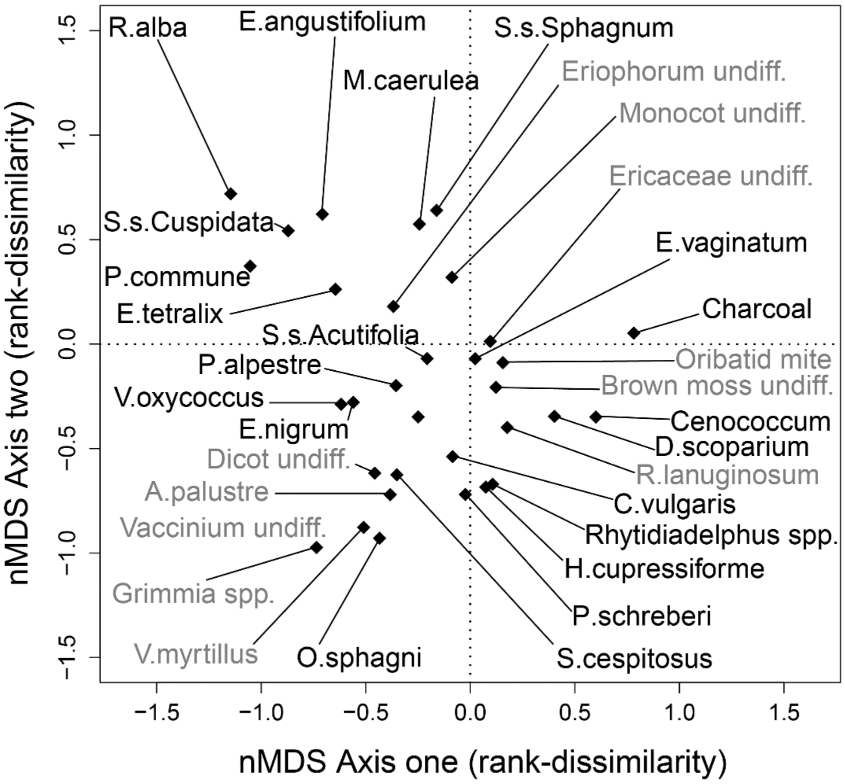


Figure 6.28: nMDS bi-plot of the ALL CORE macrofossil data (UEAR excluded) showing taxa distribution according to Axis one and two position.

6.3.2 Developing ‘Peat Accumulation Types’

Section 6.3.1 showed that the blanket peatlands studied in this thesis have developed under a variety of hydrological conditions. Some sampling points appear predominantly wet, others predominantly dry, whilst others switched between either state (discussed below). The ALL CORE dataset is now used to determine whether records recovered within this study display similar pathways of vegetation development. Grouping cores into discrete developmental classes of ‘Peat Accumulation Types’ (PATs) is a useful method for assessing systematic impacts of land management on blanket peatland vegetation communities because vegetation cover is already used as an indicator of peatland condition. As such, identifying similarities in vegetation development over time would better contextualise what drivers may have led certain vegetation assemblages to occur on Welsh blanket peatlands. PATs could also be used to examine the relationship between carbon accumulation and vegetation change given vegetation is a key control on peatland carbon cycling (Ward *et al*. 2007, 2012).

Cores were grouped into a PAT according to records which displayed broadly similar vegetation assemblages at similar points in time. DCA sample scores for each core were allocated to one of four time-slices; 1) pre-AD 1850, 2) AD 1850-1955, 3) AD 1955-1976 and 4) post-AD 1976, as devised using the three dateable features of the Spheroidal Carbonaceous Particle profiles. For consistency, master core DCA sample scores were allocated to one of these time-slices as determined by the relevant age/depth model. Cores were allocated to a PAT according to 1) an initial assessment of relative abundance of key vegetation types (i.e. *Ericaceae*, monocots, *Sphagnum*) and 2) examining the groupings of DCA sample scores in the ALL CORE ordinations when samples were divided into time-slice categories. Four broad PATs were determined using this method (Table 6.18): A) sites where *Sphagnum* is consistently dominant (Figure 6.29), B) sites where *C. vulgaris* is persistently present or has become established (Figure 6.30), C) sites where *Sphagnum* cover was previously poor but are now *Sphagnum* rich (Figure 6.31) and D) sites where *M. caerulea* has become established (Figure 6.32). Afforested sites (Figure 6.14) were classified into a fifth PAT (‘E’; Table 6.18) given forestry alters vegetation to a greater degree than changes recorded in other PATs, as evidenced by an ALL CORE ordination (‘UOM and *Ericaceae* undiff. rootlets excluded’; Table 6.17) where afforested cores are included (Figure 6.33).

Table 6.18: PAT macrofossil assemblage descriptions.

Code	PAT	Assemblage description
A	Sites where *Sphagnum* is consistently dominant	Samples scores across all time slices are predominantly ordinated on the positive side of Axis one of the DCA (Figure 6.29). Assemblages are often characterised by *E. angustifolium*, *E. tetralix*, *R. alba*, *S.* s. *Sphagnum* and/or *S.* s. *Cuspidata*.
B	Sites where *C. vulgaris* is persistently present or has become established at some point	Samples scores plot predominately from -1 to 1 on Axis one of the DCA (Figure 6.30). Macrofossil assemblages from these sites are often characterised by *C. vulgaris*, *E. vaginatum*, *S.* s. *Acutifolia* and assorted brown mosses.
C	Sites where *Sphagnum* cover was previously poor but which are now *Sphagnum* rich	Samples scores plot predominately from -1.5 to 1.5 on Axis one of the DCA (Figure 6.31). Macrofossil profiles characterised by shifts from *Sphagnum* poor, often *Eriophorum* dominated states, to contemporary *Sphagnum* rich assemblages.
D	Sites where *M. caerulea* has become established	Samples scores plot predominately between -1.5 to 1 on Axis one of the DCA (Figure 6.32). There is variability within this group but all cores are characterised by the establishment of *M. caerulea* at some point in the profile.
E	Sites where conversion to *P. sitchensis* plantation has occurred	Samples plot predominantly between -0.5 to 3.5 on Axis one of the 'UOM and *Ericaceae* undiff. rootlets excluded' DCA (see text for justification) (Figure 6.33). Macrofossil profiles are characterised by a shift from mixed blanket bog vegetation (e.g. *E. vaginatum*, *S.* s. *Sphagnum*) to plantation conditions with *P. sitchensis* remains and *R. loreus*.

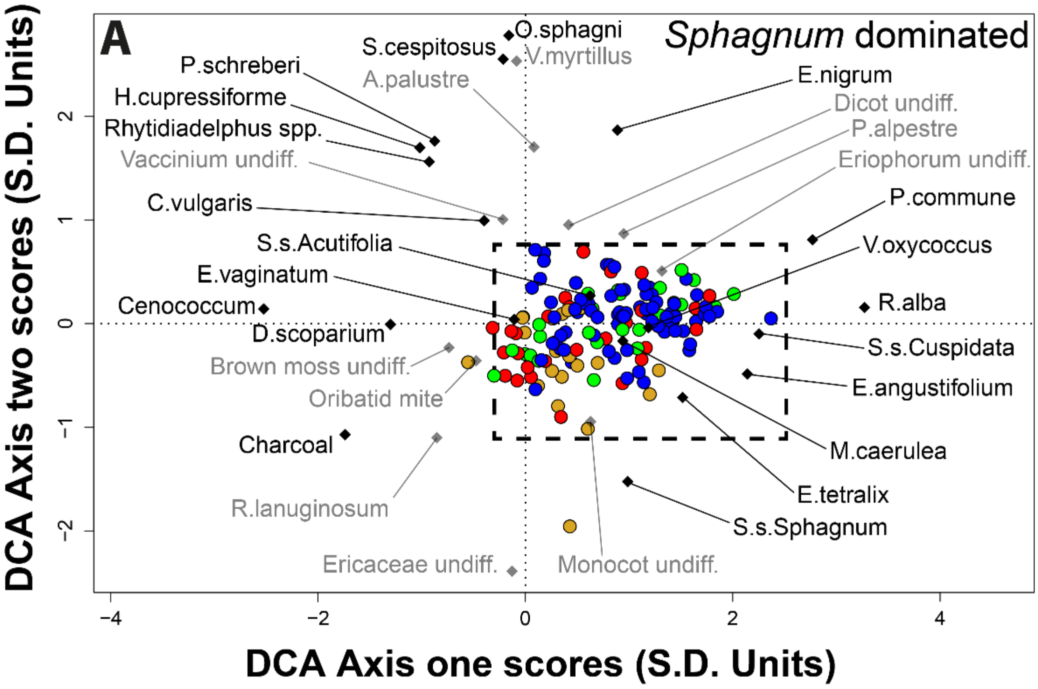


Figure 6.29: DCA ordination output detailing sample scores by time-slice for the PAT A as defined in Table 6.18. Orange represents pre-AD 1850, red indicates AD 1850-1955, green indicates AD 1955-1976 and blue indicates post-AD 1976.

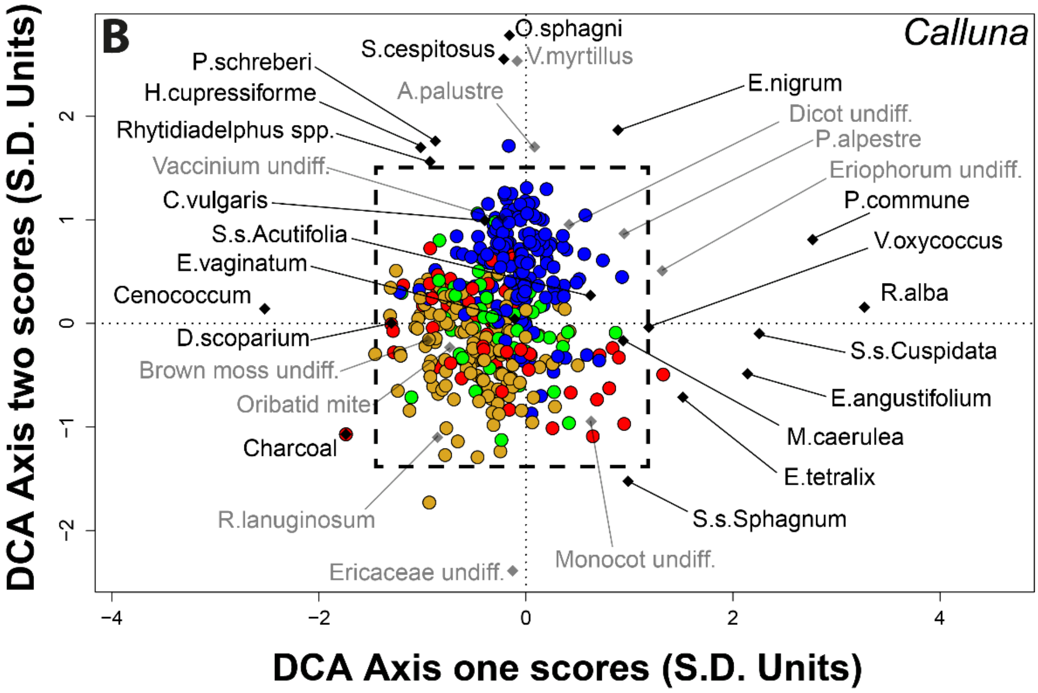


Figure 6.30: DCA ordination output detailing sample scores by time-slice for the PAT B as defined in Table 6.18. Orange represents pre-AD 1850, red indicates AD 1850-1955, green indicates AD 1955-1976 and blue indicates post-AD 1976.

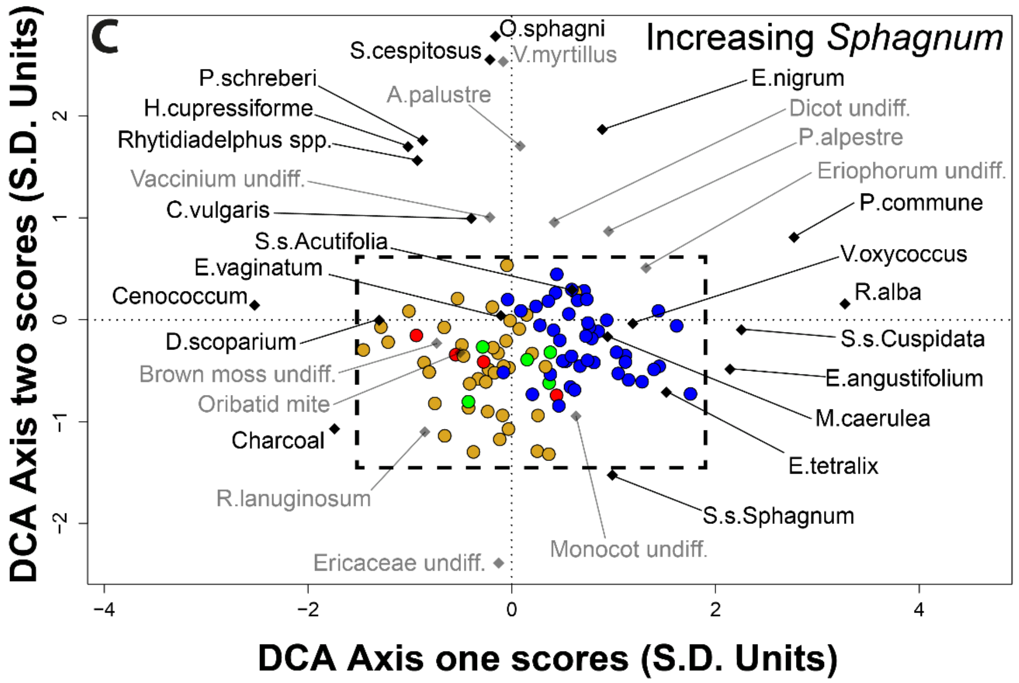


Figure 6.31: DCA ordination output detailing sample scores by time-slice for the PAT C as defined in Table 6.19. Orange represents pre-AD 1850, red indicates AD 1850-1955, green indicates AD 1955-1976 and blue indicates post-AD 1976.

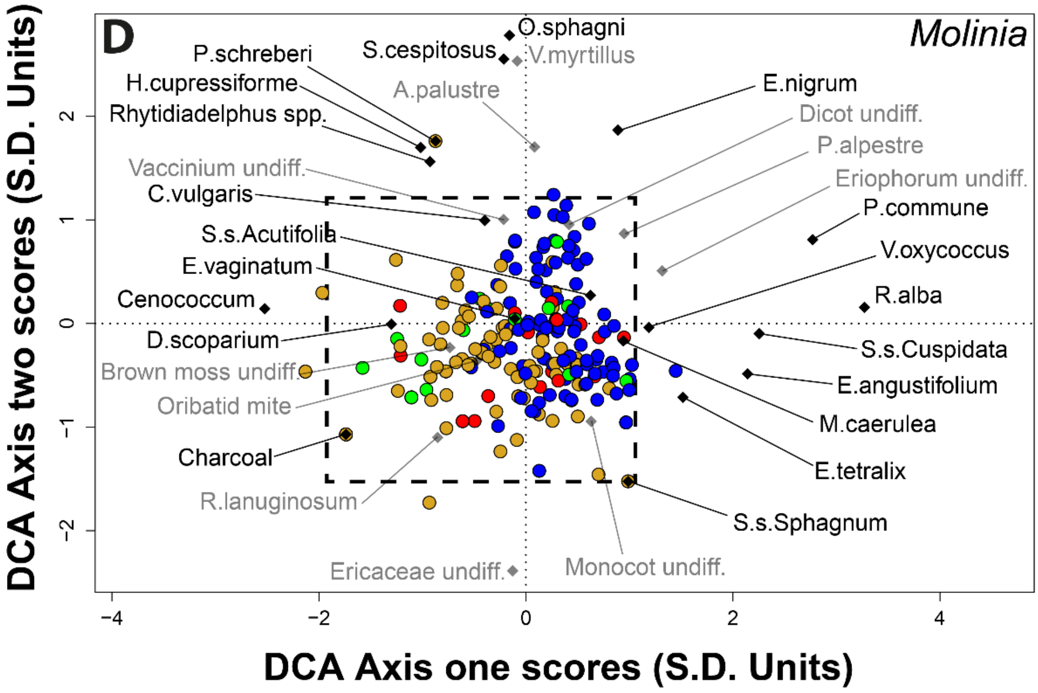


Figure 6.32: DCA ordination output detailing sample scores by time-slice for the PAT D as defined in Table 6.18. Orange represents pre-AD 1850, red indicates AD 1850-1955, green indicates AD 1955-1976 and blue indicates post-AD 1976.

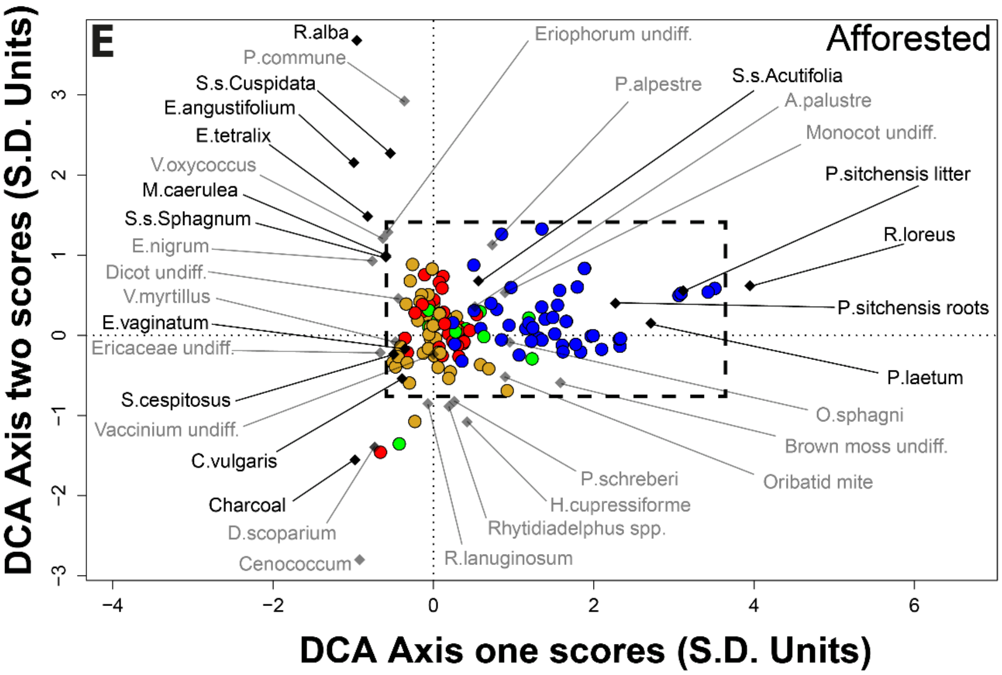


Figure 6.33: DCA ordination output detailing sample scores by time-slice for the PAT E as defined in Table 6.18. Orange represents pre-AD 1850, red indicates AD 1850-1955, green indicates AD 1955-1976 and blue indicates post-AD 1976. Ordinated outputs represent the 'UOM and *Ericaceae* undiff. rootlets excluded' ordination in Table 6.17.

Significant differences in DCA sample scores between PATs suggest there are significant differences in vegetation development over time. A Kruskal-Wallis (KW) Test was used to determine whether significant differences existed in the distributions of PATs A-D DCA sample scores for each time-slice. The KW test was used as it can accommodate non-normal distributions of independent samples (Chapman McGrew Jr and Monroe, 2000). Post-hoc analysis was conducted using the Dunn Test (DT) to examine which PAT DCA samples score were displaying significant differences in their distribution (i.e. which PATs exhibited significantly different vegetation community composition). The DT is appropriate in this case as it is a post-hoc KW test capable of analysing groups (i.e. PATs) with differing numbers of observations (i.e. number of sample scores) (Zar, 2010). These analyses were not conducted on PAT E given conversion to a forestry plantation was deemed an obvious change in peatland development.

The KW test showed there are significant differences in Axis one sample score distribution between PATs across the time-slices (Table 6.19), i.e. differing PATs exhibit differing vegetation communities over time. PATs have therefore developed under differing hydrological conditions as Axis one of the DCA was determined in Section 6.3.1 to represent a hydrological gradient. PAT A displays significantly different Axis one DCA sample scores compared to all other PATs at all time-slice increments (Table 6.20). The distribution of PAT A Axis one sample scores predominantly on the positive side of Axis one (Figure 6.34) shows that these sites have been primarily composed of *Sphagnum* and other taxa affiliated to wetter environments. Sample score distributions for PATs where *Sphagnum* is not consistently dominant (B-D) are not significantly different for time slices AD 1850-1955 and AD 1955-1976 where they plot predominantly between -0.5 to 0.5 SD units (Figure 6.34). These sample score distributions suggest that PATs B-D have been characterised by relatively intermediate taxa during the AD 1850-1976 periods relative to the *Sphagnum* dominated PAT A. Sample score distribution for PAT B displays relatively little change over time. These scores are distributed on the negative side of the Axis one gradient, reflecting the drier conditions of these sites as they are characterised by *Calluna* (Figure 6.30). PAT C sample scores shift to the positive end of the Axis one gradient for the post-AD 1976 time-slice (Figure 6.34), reflecting the expansion of *Sphagnum* at these sites relative to pre-AD 1976 conditions which were characterised by *E. vaginatum* and charcoal (Figure 6.31). Therefore, whilst PAT A sites have been characterised by *Sphagnum* and other wetter favouring taxa through the dateable period, the other PATs have shifted between wet (positive values) and dry (negative values) states, or have remained predominantly dry.

There are also significant differences in Axis two PAT DCA values when analysed by time-slice (Table 6.21). Axis two was determined in Section 6.3.1 to represent some forcing which may dictate a gradient from blanket bog to heath-like conditions. As such, one may interpret shifts in PAT sample score distributions along Axis two as indicating shifts in habitat state. This is an interesting finding given the often cited, but scarcely evidenced, potential for land use to cause shifts in habitat condition from blanket bog to heath (Thompson *et al*. 1995). The DT shows all PATs display significantly different Axis two sample scores at the most recent time-slice (Table 6.22). Type B exhibits the greatest median Axis two sample score for the post-AD 1976 time-slice (Figure 6.35), indicative of *C. vulgaris* alongside mixed *Ericaceae* and brown mosses which often characterise recent samples of cores from this PAT (Figure 6.27). Vegetation assemblages tend to be more similar in older time-slices as the number of statistically significant differences declines in older time slices (Table 6.22). These findings may therefore suggest that some sites are moving toward a heath-like status if their average Axis two sample scores increase over time (e.g. PAT B).

Table 6.19: Kruskal-Wallis test results for comparison between time-slice allocated DCA Axis one sample scores of the PATs A-D.

Time-slice	**Kruskal-Wallis Chi-squared**	***P*- value**
post-AD 1976	257.41	<0.001
AD 1955-1976	50.85	<0.001
AD 1850-1955	36.31	<0.001
pre-AD 1850	40.10	<0.001

Table 6.20: Dunn test results for pairwise comparisons between time-slice allocated DCA Axis one sample scores of the PATs A-D. Values above the diagonal indicate the *z*-test-statistic for each comparison and values below the diagonal indicate significance of the result. Results significant at <0.05 in a two-tailed test indicated in italics.

	A	**B**	**C**	**D**
post-AD 1976				
A	/	14.8	2.2	7.1
B	*<0.001*	/	-9.5	-7.7
C	*<0.001*	*<0.001*	/	3.6
D	*<0.001*	*<0.001*	*<0.001*	/
AD 1955-1976				
A	/	7.0	2.0	5.0
B	*<0.001*	/	-1.4	-6.0
C	*<0.001*	0.16	/	1.0
D	*<0.001*	0.56	0.32	/
AD 1855-1955				
A	/	6.0	2.6	4.0
B	*<0.001*	/	-0.3	-1.3
C	*<0.01*	0.74	/	-3.0
D	*<0.001*	0.18	0.76	/
pre-AD 1850				
A	/	6.0	4.0	5.6
B	*<0.001*	/	-2.5	-0.7
C	*<0.001*	*<0.05*	/	2.0
D	*<0.001*	0.50	*<0.05*	/

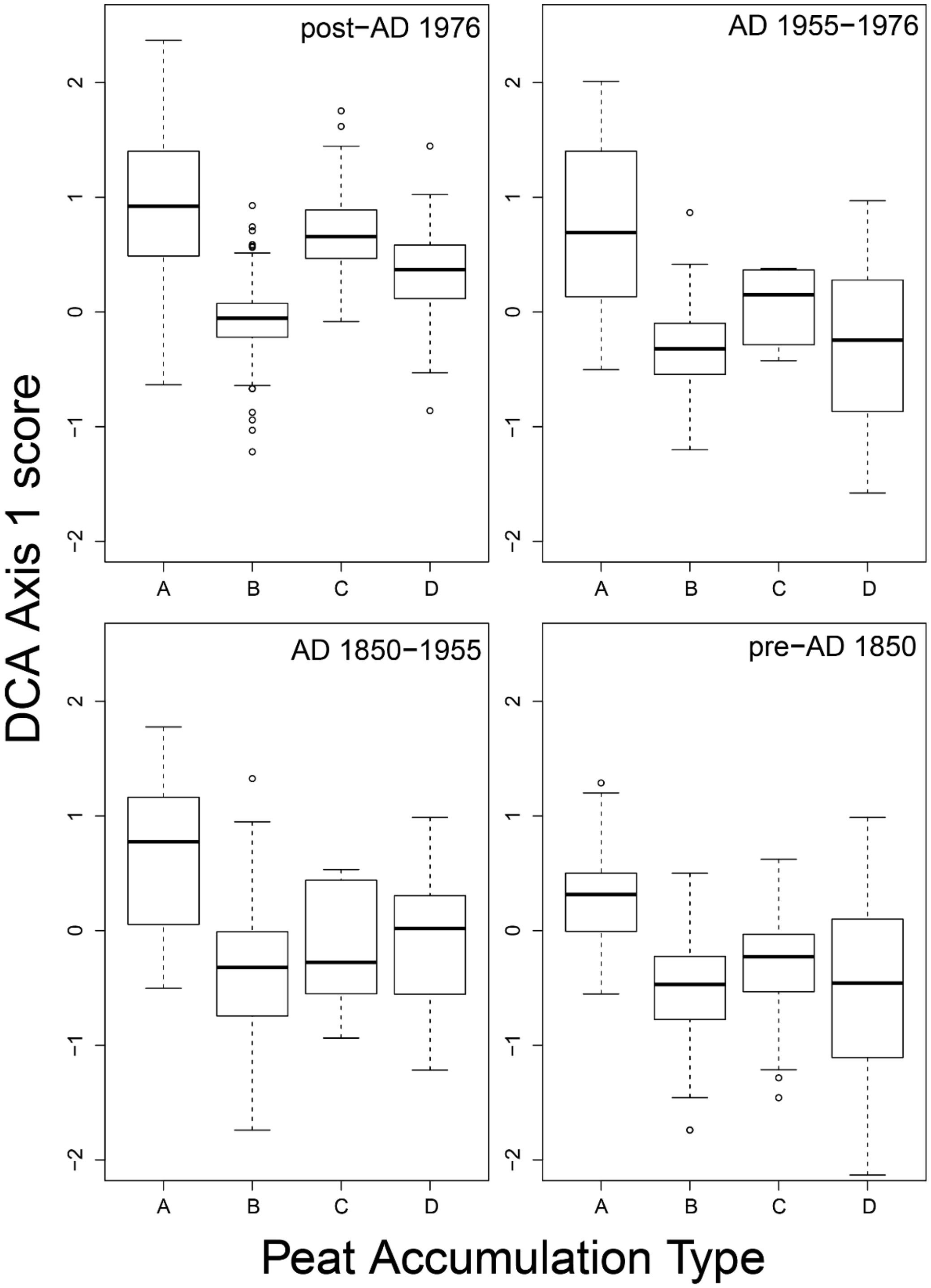


Figure 6.34: DCA Axis one sample scores by time-slice for PATs A-D. Letters correspond to designations in Table 6.18. Limits of the box represent the first and third quartiles and the horizontal line indicates the median. Whiskers indicate minimum and maximum values within 1.5 times the interquartile range whilst circles indicate values >1.5 times the interquartile range. Y-axis consistent for comparative purposes.

Table 6.21: Kruskal-Wallis test results for comparison between time-slice allocated DCA Axis two sample scores of the PATs A-D.

Time-slice	**Kruskal-Wallis Chi-squared**	***P*- value**
post-AD 1976	166.60	<0.001
AD 1955-1976	14.00	<0.001
AD 1850-1955	11.06	<0.05
pre-AD 1850	23.58	<0.001

Table 6.22: Dunn test results for pairwise comparisons between time-slice allocated DCA Axis two sample scores of PATs A-D. Values above the diagonal indicate the *z*-test-statistic for each comparison and values below the diagonal indicate significance of the result. Results significant at <0.05 in a two-tailed test indicated in italics.

	A	**B**	**C**	**D**
post-AD 1976				
A	/	-7.2	3.5	2.0
B	*<0.001*	/	9.7	10.4
C	*<0.001*	*<0.001*	/	-2.0
D	*<0.05*	*<0.001*	*<0.05*	/
AD 1955-1976				
A	/	0.3	3.0	2.3
B	0.74	/	2.9	2.3
C	*<0.005*	*<0.005*	/	-1.5
D	*<0.05*	*<0.05*	0.12	/
AD 1855-1955				
A	/	1.3	2.2	2.9
B	0.18	/	1.6	2.2
C	*<0.05*	0.10	/	-0.5
D	*<0.005*	*<0.05*	0.60	/
pre-AD 1850				
A	/	-0.6	0.9	1.3
B	0.52	/	2.5	4.7
C	0.38	*<0.05*	/	0.5
D	0.20	*<0.001*	0.62	/

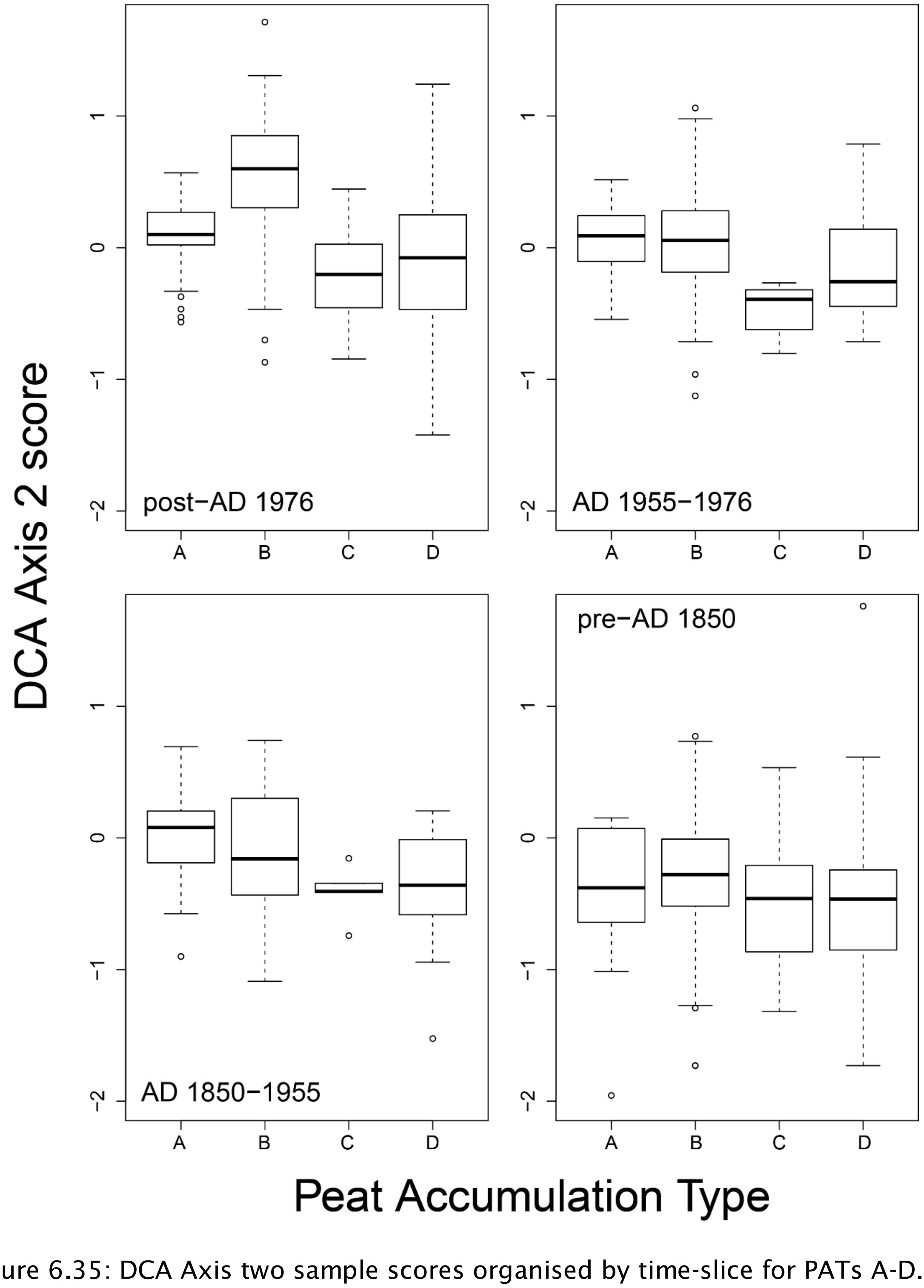


Figure 6.35: DCA Axis two sample scores organised by time-slice for PATs A-D. Letters correspond to designations in Table 6.18. Box limits represent the first and third quartiles and the horizontal line indicates the median. Whiskers indicate minimum and maximum values within 1.5 times the interquartile range whilst circles indicate values >1.5 times the interquartile range. Y-axis consistent for comparative purposes.

Five broad PAT categories exist within the records recovered in this investigation. There is variation in DCA sample score distribution within each PAT, meaning that assigning cores to each of these categories has to some extent masked details of sampling point specific vegetation development. Policy level changes in land management are devised according to large scale processes given policy instruments are not flexible enough in most cases to respond on a case-by-case basis (Gordon *et al*. 2004). As such, the identification of 'broad-themes' in peatland development is the most applicable use of data in this thesis, as opposed to a less applied use of data by reporting all possible pathways of peatland development. Hence, focussing on broad differences in peatland development (i.e. the PATs presented here) is appropriate when attempting to detect the impact of land management across the suite of cores in this study.

The main determinant of PAT development appears to be hydrological change given the dispersion of taxa within the ALL CORE ordinations (Table 6.20; Figure 6.34). It seems likely that land use is the driver of this hydrological change as opposed to an external and/or some other allogenic (e.g. climatic) driver. There is limited/absent charcoal recorded in cores within PAT A, hence this 'wetter' PAT may have developed in this way due to the lack of firing. Cores within PAT B on the other hand tend to have evidence of abundant charcoal and were often recovered from drained sites. Hence, PAT B cores likely represent the effect of disturbance by human activity. The drivers which may cause blanket peats to develop along certain PAT trajectories are discussed in more detail in Chapter 8.

6.4 Conclusions

Data presented within this chapter has demonstrated the variable ways which Welsh blanket peatlands have developed over time. Vegetation change at many of these sites has likely been affected by the actions of humans, with charcoal records often suggesting frequent firing of the landscape prior to AD 1850. Therefore, the contemporary state of many of these blanket peatlands cannot be considered solely a product of 19th and 20th Century land use. The exact effect of a given land use may not be reliably determined using these records given many sites have been subject to multiple land use types over the previous centuries to millennia. An exception to this point is that the impact of afforestation can be reliably identified given 1) all forestry cores were recovered from within plantations and 2) the creation of a plantation creates a clear gradient of change in the reported ordinations.

Results from plant macrofossil analysis have been used to identify the primary environmental gradients which have dictated vegetation community structure across the studied peatlands. In many cases this gradient was found to be a hydrological gradient. Analysis within this chapter has also presented peatland development frameworks (Peat Accumulation Types (PATs)) which are used to analyse data later in this thesis. The main conclusions from this chapter are:

- The dominant gradient of environmental change in the macrofossil datasets was related to hydrological conditions (four of the five ordinated datasets). Secondary gradients in three of the five ordinations were determined as some forcing capable of driving a blanket bog toward heath-like conditions.
- Vegetation presence/absence data was sufficient for detecting gradients of environmental change within all datasets, both at a site level and where all cores were included in the analysis.
- Variable fire records are presented across the recovered cores. Some records exhibit evidence of plentiful charcoal whilst others display no charcoal at all. The cause of the firing (e.g. burning to structure vegetation for grouse or sheep) may not be determined solely from data here.
- Vegetation change within afforested peatlands is markedly different from that of ‘open’ blanket peatlands. This is due to the introduction of taxa which are unlikely to be present on Welsh blanket bogs unless introduced by plantations (e.g. *Picea sitchensis*) or unlikely to dominate blanket bogs as they do under forest canopies (e.g. *Rhytidiadelphus loreus*). Removing these samples from ordinations increased the explanatory power (eigenvalues) of the ordinations of the Migneint and ALL CORE datasets.
- Cores were grouped into five PATs where sampling points had exhibited similar patterns in vegetation community composition over time. Differing sampling points have likely developed according to these PAT pathways due to similar environmental forcings being applied (e.g. land use pressures such as firing). PATs present useful frameworks for exploring what drivers have affected Welsh blanket peatland development and these PATs are used in subsequent chapters to present and analyse data.

Chapter 7 Peat chemical and physical properties

7.1 Introduction

Peatlands provide a valuable ecosystem service by sequestering carbon from the atmosphere. The effect of land use on this service is uncertain; this thesis aims to address this uncertainty (Section 2.8). Contemporary flux monitoring is a costly and time-consuming approach to understanding carbon dynamics (Section 2.2.3). An alternative approach is to analyse peat cores using physical and chemical techniques to reconstruct carbon accumulation over longer timescales. This approach forms the focus of this chapter. An issue that complicates the study of peatland carbon dynamics and the calculation of reliable estimates of carbon accumulation is the effect of the acrotelm/catotelm transition (ACT). This chapter will address this issue.

This chapter presents data acquired through chemical and physical analyses of cores recovered in this study. These findings are used in combination with peat accumulation data in Chapter 5 to produce carbon accumulation histories. Bulk density (BD), carbon (C) and nitrogen (N) content data are presented against depth (cm) whilst reconstructed carbon accumulation rates are presented against date (AD). The first section of this chapter (Section 7.2) presents peat BD, C content and N content for Bryniau Pica (BRP), Lake Vyrnwy (LVW), the Migneint (MIG) and Plynlimmon (PLY). Some example BD, C and N content diagrams are presented which detail broad trends across the dataset, with the remaining diagrams located in Appendix B. C/N content is not standardized against BD given the relevance of raw C/N content values in identifying the ACT (see below). A second section comments on the ACT (Section 7.3) before a third section presents the reconstructed carbon accumulation histories for each core (Section 7.4). A final section (Section 7.5) considers the carbon dynamics of each Peat Accumulation Type established in Section 6.3.2. Chapter conclusions are presented in Section 7.6.

7.2 Chemical and physical analytical results

As noted in Section 7.1, examination of the physical and chemical properties of peat is a pre-requisite for accurate reconstructions of carbon dynamics. Peat physical and chemical properties vary across the fifty-core dataset (Appendix B). Most records exhibit an increase in BD (Figure 7.1; Figure 7.2) and C and N content (Figure 7.3; Figure 7.4) *ca*. 10-20 cm below the surface. The predominant increase in BD, C and N with depth likely represents the transfer of less consolidated acrotelm peat into the catotelm, where peat density is higher. To examine the effect of peat density on C and N content, these two variables were plotted against BD for each sample. Samples with a BD, C or N content outside the 1.5x interquartile range for each metric were removed and the regression repeated (Figure 7.5; Figure 7.6; Figure 7.7; Figure 7.8). Excluding outliers tended to improve the correlation between BD and both C and N content when all sites were analysed individually. The relationship between BD and C content and between BD and N content is typically weak linear. LOESS smoothing suggests the linear component to the relationship between BD and C and N primarily occurs where BD is *ca*. ≤0.15 g cm^{-3} (Figure 7.5; Figure 7.6; Figure 7.7; Figure 7.8), likely as a result of low BD acrotelm material meaning unstandardized C and N content is also low. Once material is transferred to the catotelm (i.e. samples with greater BD) there is less of a linear relationship between BD and C and N content.

A trend noted across all sites was that BD, C and N content tended to increase with depth. To examine the effect of peat depth, these three variables were plotted against depth for each sample when all sites were combined into a single dataset and BD, C and N content values outside the 1.5x interquartile range for each metric were removed. The combined dataset was used in this analysis to determine if a consistent effect of depth on peat physical and chemical properties was present across the sampled blanket peatlands. To this end, the maximum number of data points available were included (save for outliers). The relationship between depth and BD, C and N content is weak linear (Figure 7.9) with LOESS smoothing suggesting the linear component to the relationship between depth and BD and N content occurs primarily at depths ≤25cm. The weak linear relationship between depth and C content occurs across the 50cm coring envelope. Variability in peat property values is therefore related to the depth in each core which a sample was recovered from, even where data are relatively noisy. This demonstrated variability in peat properties with depth may be used to infer the location of the acrotelm/catotelm transition (ACT). Identifying the ACT is important when attempting to produce reliable estimates of long-term carbon accumulation rates (see below).

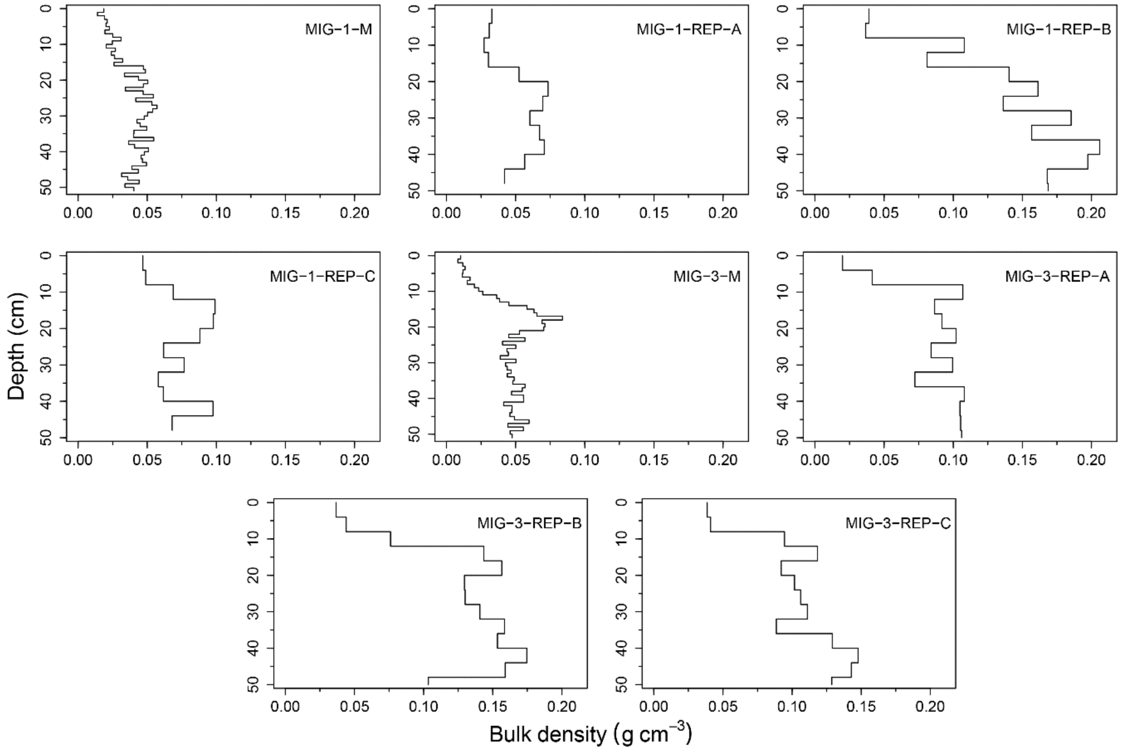


Figure 7.1: BD profiles for cores from the Migneint.

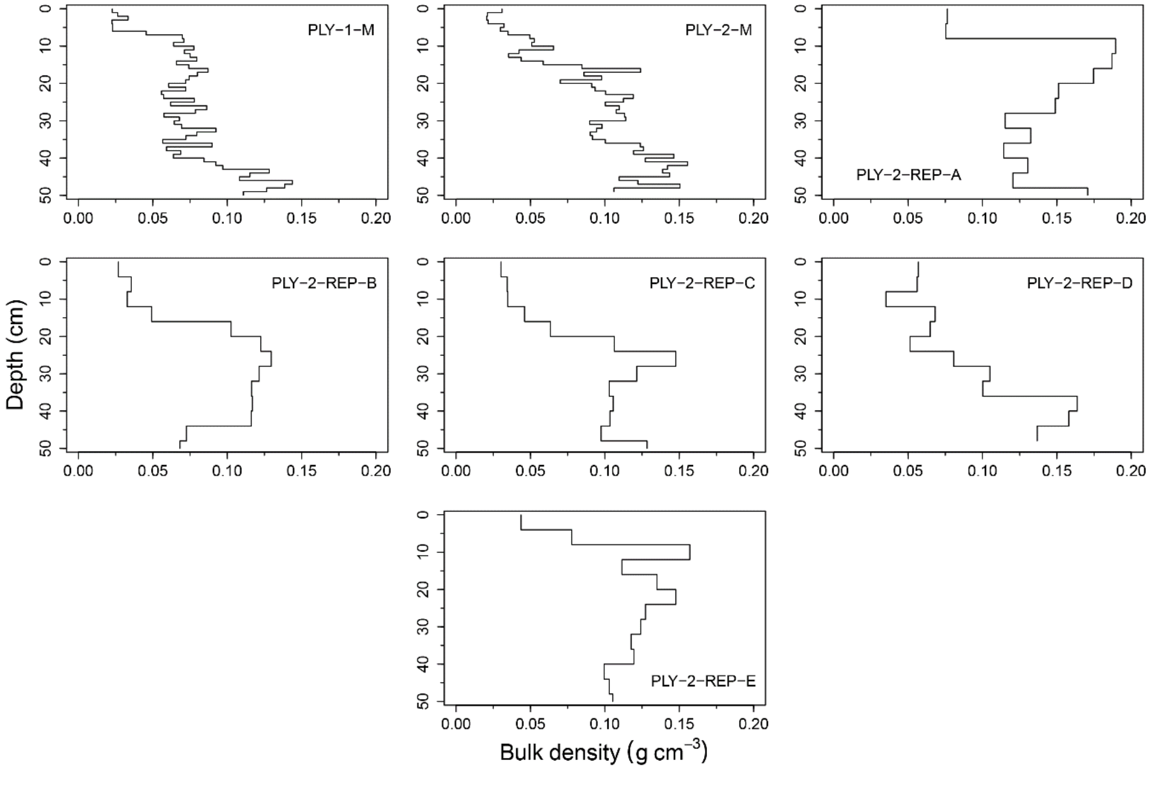


Figure 7.2: BD profiles for cores recovered from Plynlimmon.

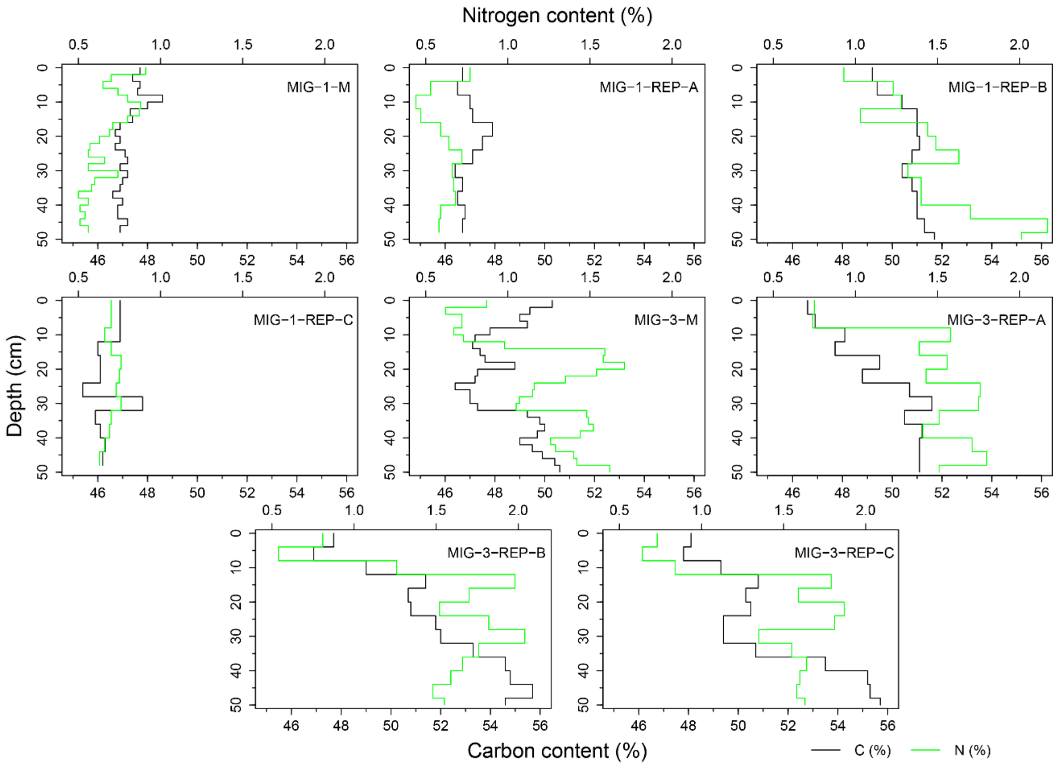


Figure 7.3: C and N content profiles for cores the Migneint.

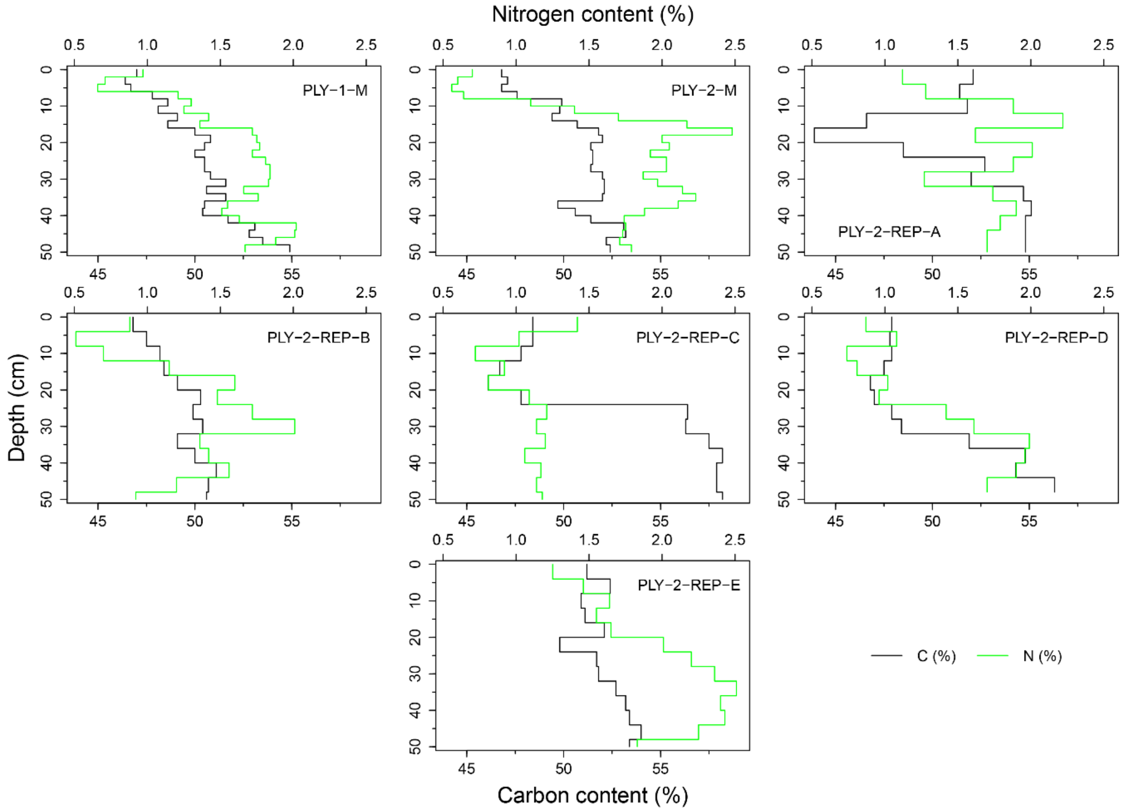


Figure 7.4: C and N content profiles for cores recovered from Plynlimmon.

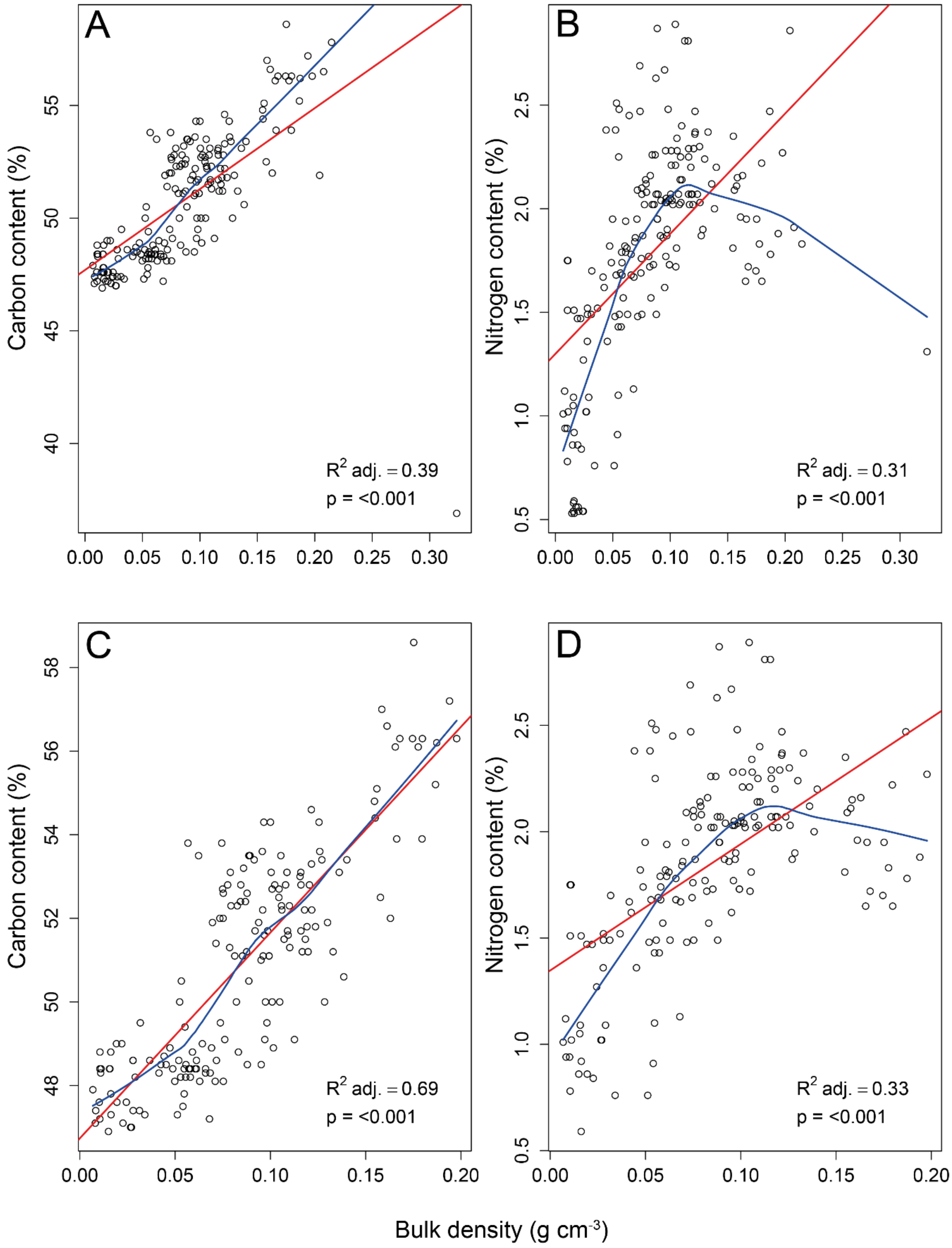


Figure 7.5: Relationship between BD and C content and N content for cores recovered from BRP, before (A and B) and after (C and D) outliers were removed as discussed in the text. Red lines and statistics indicate linear models, blue lines indicate LOESS smoothing. X and Y-axes optimised by dataset.

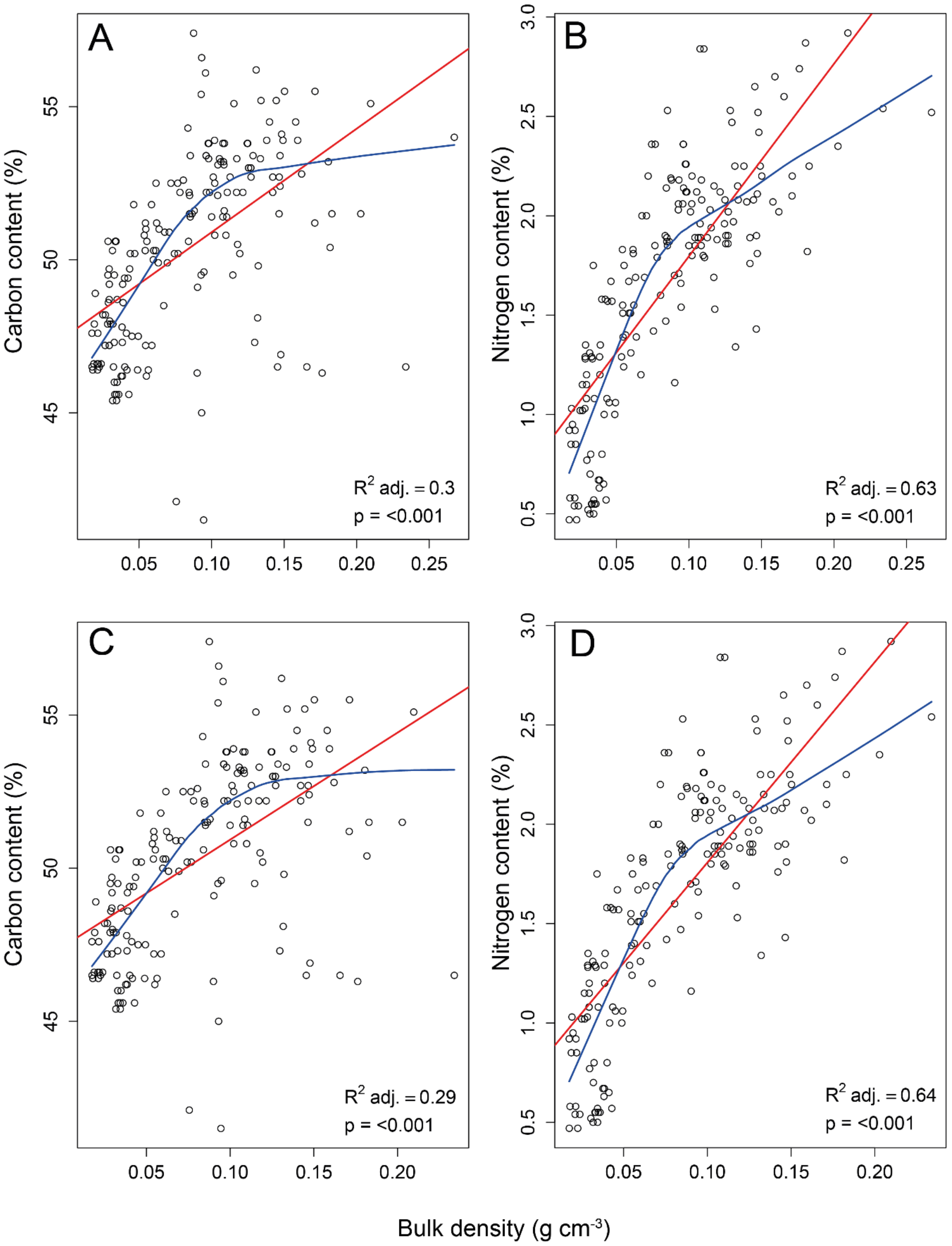


Figure 7.6: Relationship between BD and C content and N content for cores recovered from LVW, before (A and B) and after (C and D) outliers were removed as discussed in the text. Red lines and statistics indicate linear models, blue lines indicate LOESS smoothing. X and Y-axes optimised by dataset.

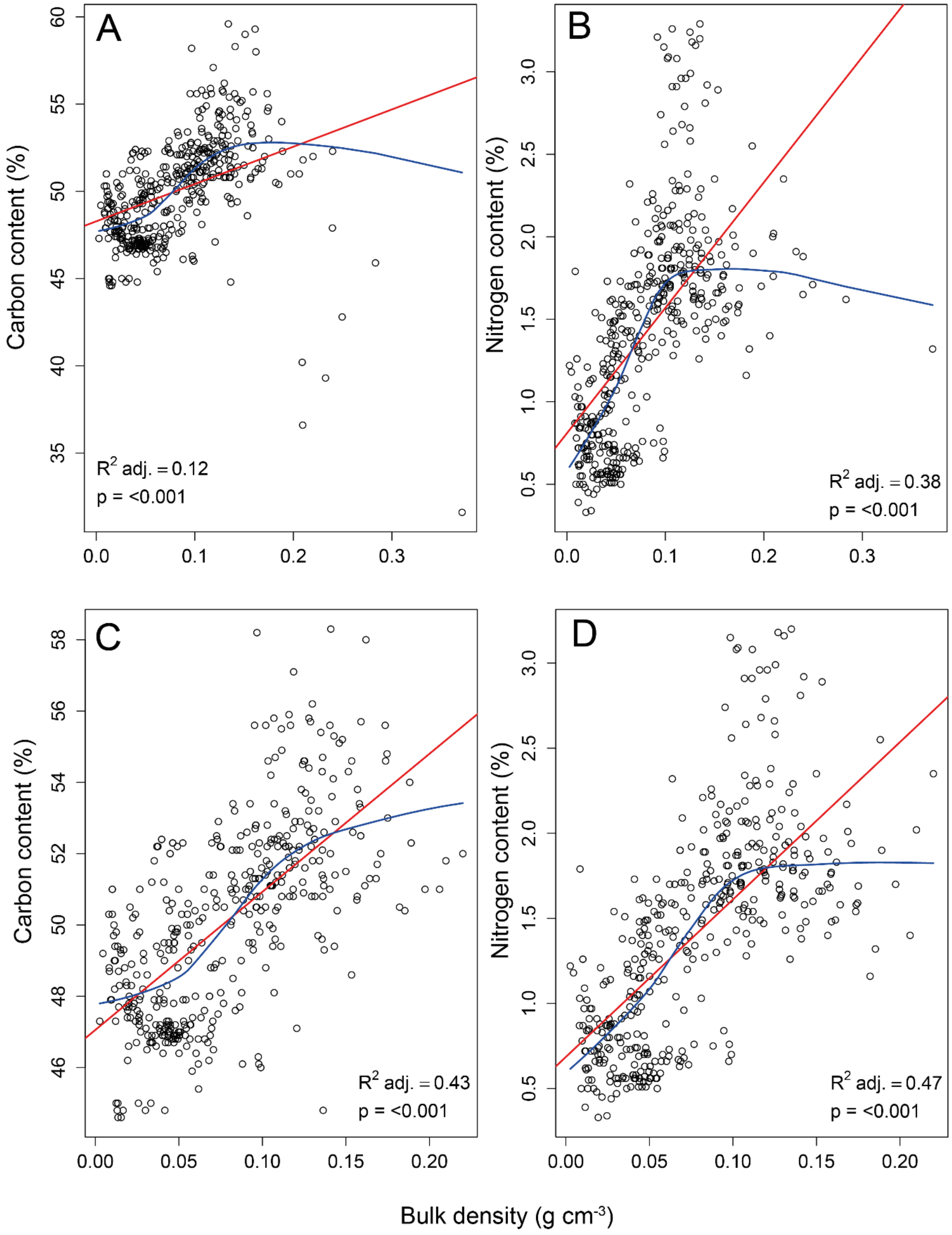


Figure 7.7: Relationship between BD and C content and N content for cores recovered from the MIG, before (A and B) and after (C and D) outliers were removed as discussed in the text. Red lines and statistics indicate linear models, blue lines indicate LOESS smoothing. X and Y-axes optimised by dataset.

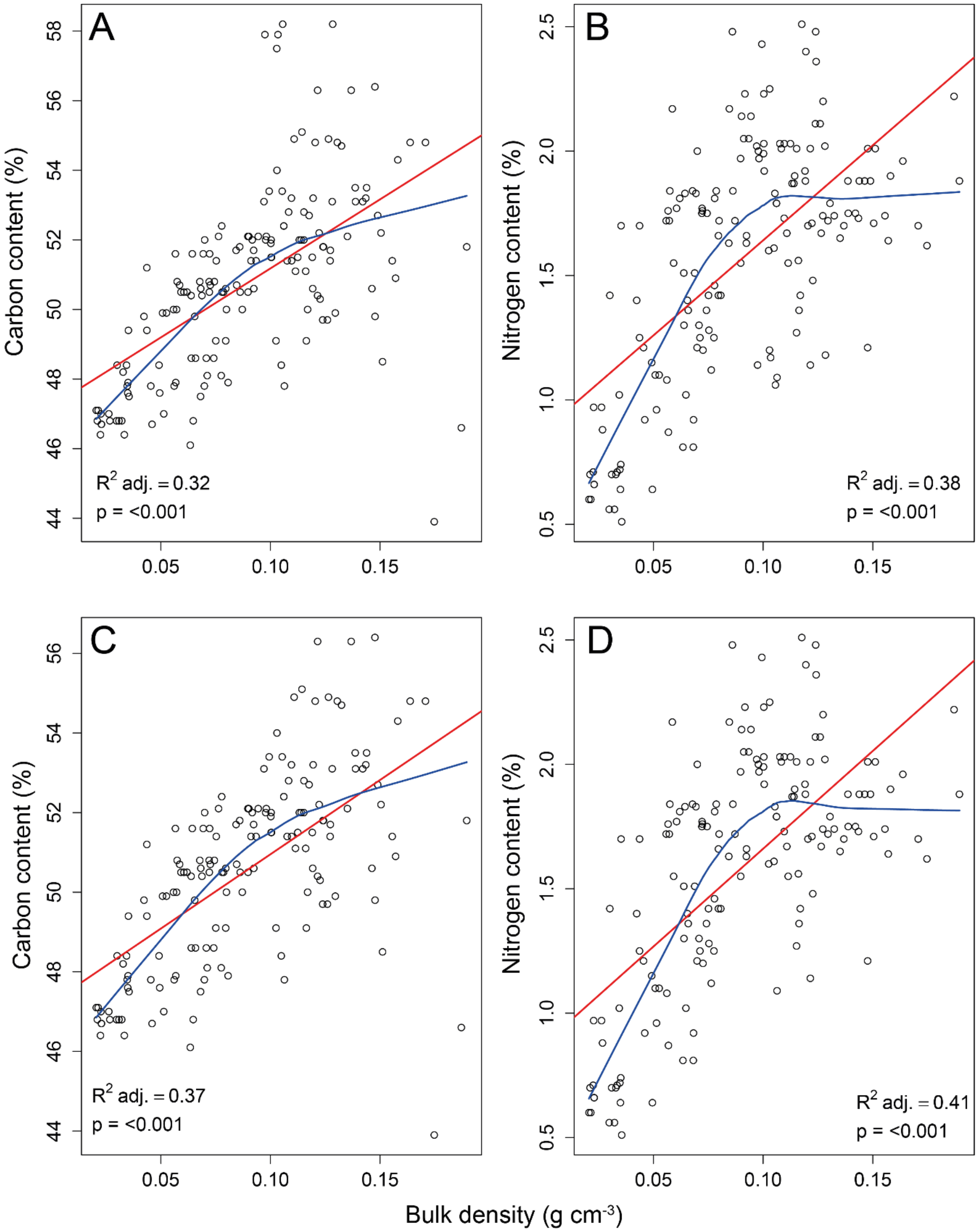


Figure 7.8: Relationships between BD and C content and N content for cores recovered from PLY, before (A and B) and after (C and D) outliers were removed as discussed in the text. Red lines and statistics indicate linear models, blue lines indicate LOESS smoothing. X and Y-axes optimised by dataset.

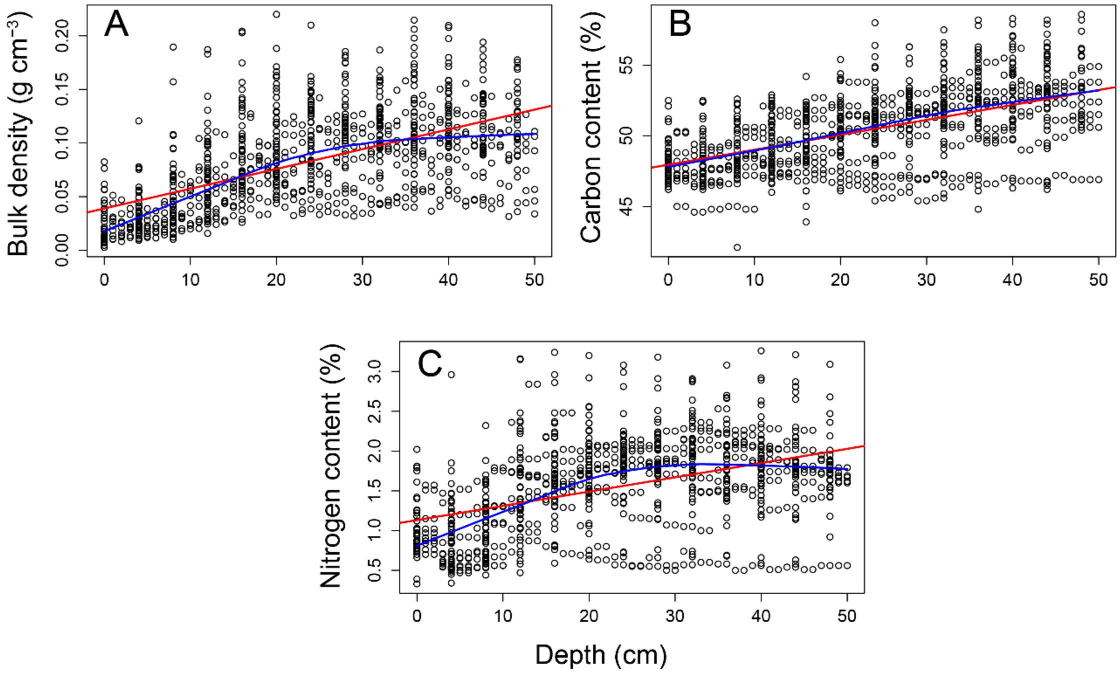


Figure 7.9: Relationship between depth and BD (A), C (B) and N (C) content after outliers were removed as discussed in text. Red lines indicate linear models, blue lines indicate LOESS smoothing. Statistics for linear models reported in Table 7.1

Table 7.1: Linear model statistics for relationship between peat properties and depth for the full dataset where outliers were removed. Statistics relate to linear models (red lines) in Figure 7.9.

Peat property	**Adjusted R^2**	***P*- value**
Bulk density	0.44	<0.001
Carbon content	0.29	<0.001
Nitrogen content	0.18	<0.001

7.3 Detecting the acrotelm/catotelm transition

Recovered cores represent dynamic peatland surface layers as they record both live vegetation and decaying plant matter, and they may include the ACT. Detecting the boundary between these two functional components is useful for estimating long-term rates of carbon accumulation. Hereafter, ‘below’ the ACT is used to signify the catotelm and ‘above’ the ACT is used to signify the acrotelm.

Approaches to reconstructing carbon accumulation rate follow definitions in Section 2.2.3.

Carbon accumulation in a peat profile is not a linear phenomenon, even when rates of productivity and decay are constant (Clymo, 1984). Apparent carbon accumulation for a given portion of peat decreases as the depth of that peat increases due to ongoing decomposition in the catotelm. This ongoing decomposition produces a concave long-term apparent rate of carbon accumulation (LORCA) profile (Clymo *et al*. 1992; Figure 7.10). Recent apparent rate of carbon accumulation (RERCA) overestimates carbon accumulation relative to LORCA as acrotelm material has yet to undergo the same degree of decomposition as catotelm peat. Carbon accumulation rate (CAR) may also overestimate carbon accumulation if acrotelm material is included in the calculation (Figure 7.10).

Parry (2011) suggests that RERCA represents average rates of carbon accumulation for the acrotelm only. However, RERCA may include peat recovered from the catotelm if the dated point above which RERCA is calculated is located below the ACT. RERCA has been converted to LORCA where 1) RERCA is believed to solely represent acrotelm carbon accumulation and 2) rates of decay in the catotelm are known (Parry, 2011). Such approaches have been criticised. Turunen (2002) noted RERCA and LORCA were not systematically related across a wide survey of boreal mires. Estimates of catotelm decay require the total depth and basal age of a peat deposit to be known (Clymo *et al*. 1998). As such, LORCA could not be calculated in this study as peat depths or basal ages were not known for these sites. An alternative approach to understanding long-term carbon accumulation is to identify the ACT using data presented thus far.

A trend of BD, C and N content increasing with depth was highlighted in Section 7.2. A notable change in peat property with depth may be a semi-quantitative/theoretical indication of the depth of the ACT (*sensu* Clymo, 1992). To this end, breakpoint regressions (Muggeo, 2017) were conducted on the entire peat property dataset (i.e. Figure 7.9) save for outliers (Section 7.2) to test for a change in the relationship between depth and BD, C and N content. Davies' tests were also conducted to test the significance of the change in the slope of the response variable (Davies, 2002). The estimated breakpoint in the Davies' Test is somewhat dependent on the number of evaluation points used, hence it should be used alongside the maximum likelihood breakpoint estimates as produced by the standard segmented regression (Muggeo, 2017).

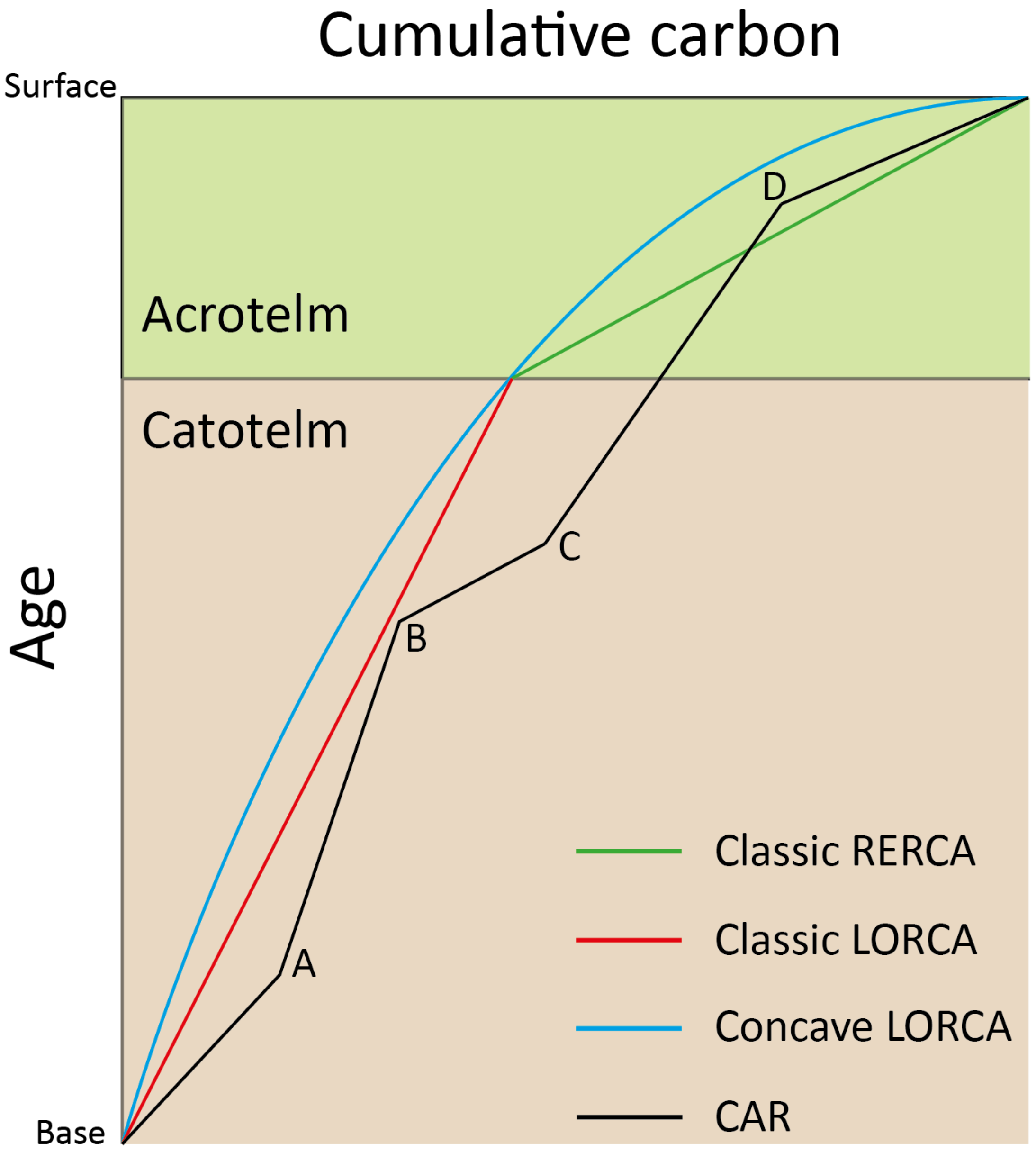


Figure 7.10: Conceptual diagram detailing carbon accumulation in a peat profile. Types of carbon accumulation defined in Section 2.2.3 save for 'Concave LORCA' (*sensu* Clymo *et al*. (1998). CAR profile gradient is arbitrary to illustrate the impact of variable peat accumulation between dated points A, B, C and D. CAR profile also demonstrates limitations of the technique if the ACT is not identified (adapted from Parry (2011)).

It is uncertain whether a universal depth for the ACT would be detectable because patterns of nanotopes across a site likely mean the depth of the ACT varies across a peatland (Lindsay, 2010). The whole dataset was used in this case to 1) enable the maximum number of data points to be included to detect whether a consistent change in peat properties could be detected with depth (see Section 7.2) and 2) to attempt to determine a universal depth which may be used to refine estimates of long-term carbon accumulation across the entire dataset.

Breakpoints were detected in all datasets where peat properties were compared to depth (Table 7.2; ‘Segmented regression’). These values broadly reflect the visible change in the slope of the LOESS smoothing for BD (*ca*. 22-24cm) and N content (*ca*. 18-22cm) (Figure 7.9). It is harder to identify a change in the LOESS slope where C content is plotted against depth which likely explains 1) why standard error on the breakpoint estimate for C is higher than for the other peat properties and 2) why the Davies’ Test could find no significant changes in the slope of the response variable (Table 7.2). Qualitatively therefore, one may deem that many cores within the study display a significant change in peat properties *ca*. 19-25cm depth and this change primarily affects BD and N content.

Table 7.2: Statistics comparing peat properties and depth for the entire dataset where outliers were removed as explained in the text. Maximum likelihood estimated breakpoint in a segmented regression and best estimated breakpoint in a Davies’ Test using 50 evaluation points. Davies’ Test *P*-values represent two-tailed testing.

Peat property	**Segmented regression**		**Davies’ Test**	
	Breakpoint (cm)	**Standard Error**	**Breakpoint (cm)**	***P*-value**
Bulk density	23.99	1.38	23.47	<0.001
Carbon content	40.16	4.56	39.80	0.138
Nitrogen content	19.46	1.47	19.39	<0.001

Findings discussed in this section have thus far focussed on identifying the ACT as a ‘depth’ within a peatland. Some authors do identify the ACT as a given depth (Clymo, 1992; Evans *et al*. 1999), but if carbon accumulation rates are calculated it is arguably more relevant to determine the ‘time’ taken for material to transfer to the catotelm (i.e. the y-axis in Figure 7.10). Lindsay (2010) suggested that BD is a key property for identifying the ACT. As such, BD values were tested as a function of age across several datasets. ‘Older’ peat typically exhibits a greater BD than ‘younger’ peat across all datasets (Figure 7.11). This finding is not surprising given BD increases with depth (i.e. analogous to age) (Panel A; Figure 7.9). Breakpoints were detected in all datasets where BD was compared to age (Table 7.3), with most breakpoints estimated *ca*. AD 1976-1986. Grouping of data in this way inevitably masks variability in the age of the ACT. The use of a large dataset, and the findings of significant breakpoints in the relationship between peat age and BD, does produce an arguably representative estimated age of peat at the ACT (*ca*. AD 1976-1986). Findings in this section can therefore be used to better understand how rates of carbon accumulation in the acrotelm relate to long term rates of carbon accumulation at sites in this thesis (see below).

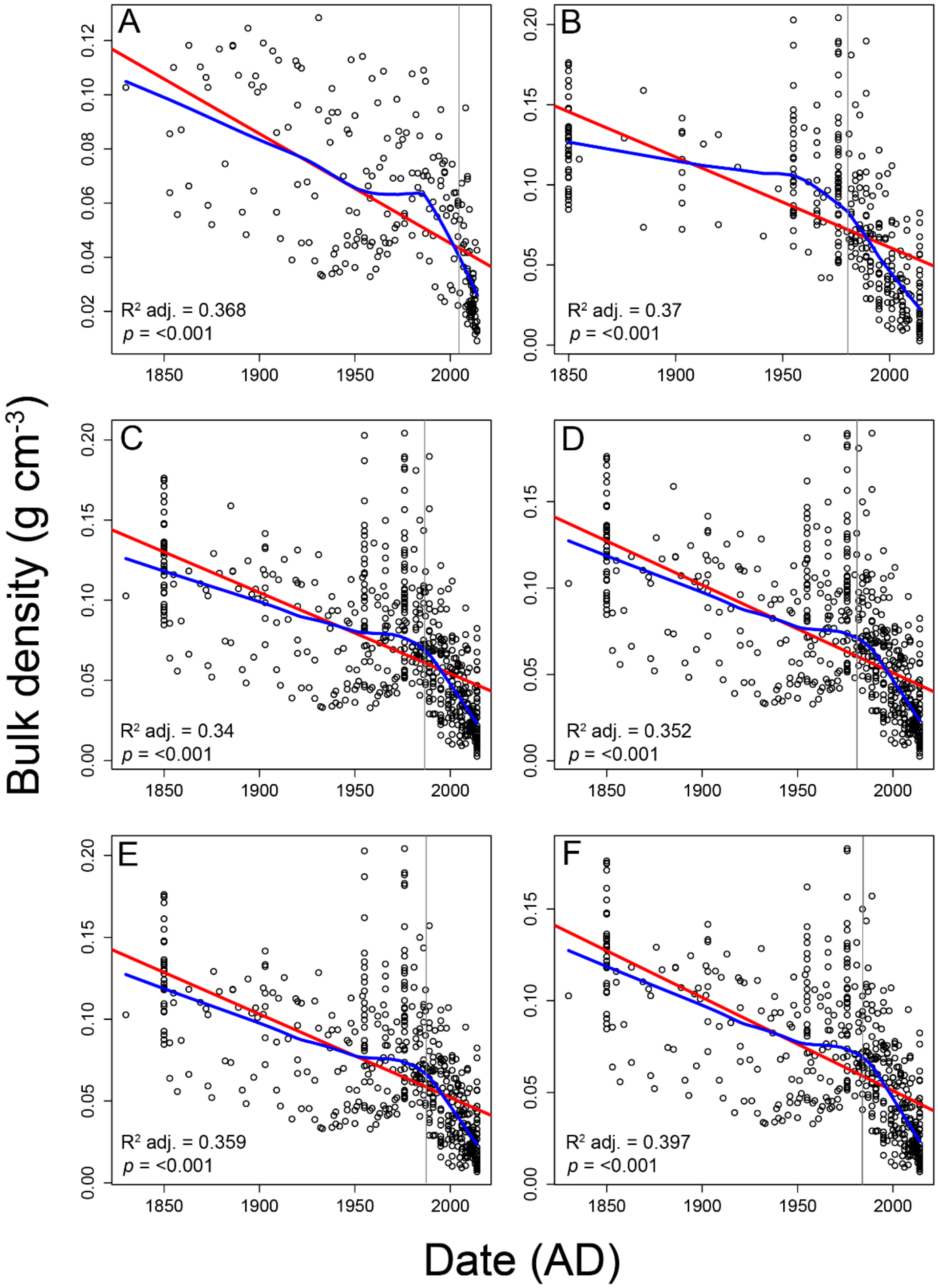


Figure 7.11: Relationships between peat age and BD across several datasets. Letters relate to Table 7.3. Grey lines indicate Davies' Test breakpoints in Table 7.3. Red lines indicate linear models, blue lines indicate LOESS smoothing. Statistics relate to linear models. X and Y-axes optimised according to dataset.

Table 7.3: Statistics comparing peat BD and age across several datasets. Maximum likelihood estimated breakpoint in a segmented regression and best estimated breakpoint in a Davies' Test using 50 evaluation points. Davies' Test *P*-values represent two-tailed testing. 'Outliers' relate to those bulk density samples >1.5x the interquartile range for each dataset.

Dataset	**Segmented regression**		**Davies' Test**	
	Breakpoint (AD)	**Standard Error**	**Breakpoint (AD)**	***P*-value**
A- Master cores	2003	2.02	2004	<0.001
B- Replicates	1976	3.01	1977	<0.001
C- Full dataset	1982	2.75	1981	<0.001
D- Full dataset excl. outliers	1983	2.67	1984	<0.001
E- Full dataset excl. afforested cores	1981	3.08	1981	<0.001
F- Full dataset excl. afforested cores and outliers	1986	2.81	1984	<0.001

7.4 Carbon accumulation histories

The main aim of this thesis is to characterise the impact of land management on carbon accumulation. Carbon accumulation must therefore be reconstructed for each recovered core. The carbon accumulation histories reported within this section were determined using the age/depth models in Chapter 5 and the peat properties presented in Section 7.2. The effect of the ACT (Section 7.3) is not explicitly acknowledged in this section, with the effect of the ACT on these carbon accumulation histories discussed in Chapter 8.

7.4.1 Comparisons of carbon accumulation histories according to selected chronological method

Differing chronological techniques may produce differing age estimates for a given depth in a peatland, and so produce differing estimates of peat accumulation rates (PAR) (Turetsky *et al*. 2004). Differing chronological methods (e.g. ^{210}Pb, Spheroidal Carbonaceous Particles (SCPs)) may therefore produce differing reconstructed carbon accumulation rates for the same peat profile as PAR is an important metric for calculating rates of carbon accumulation (Section 3.4.2). Section 5.3.3.3 presented findings on how Constant Rate of Supply (CRS) age/depth models compared with SCP age/depth models. This sub-section now uses these data to explore the differences in carbon accumulation histories as reconstructed using these methods.

Figure 7.12 and Figure 7.13 detail carbon accumulation rate (CAR) for all master cores. CAR is used here following its definition in Section 2.2.3 in that PARs change between dated horizons according to the difference in age and depth between those two horizons. Uncertainties in CAR values were calculated as the minimum and maximum possible CAR according to chronological uncertainty for that horizon. For example, the AD 1955 ‘SCP Vis.’ horizon has a ± error of five years (Rose and Appleby, 2005). As such, to produce minimum and maximum estimates for CAR at the AD 1955 horizon, PARs were altered to represent that this dated point in the core may be either AD 1950 or 1960.

Many CRS and SCP CAR profiles overlap for a given date within the envelope of uncertainty (Figure 7.12). There are some notable points where SCP reconstructed CAR deviates from CRS reconstructed CAR (e.g. MIG-2-M ‘SCP Cum.’ AD 1870-1900, PLY-1-M all SCP methods AD 1930-1960). All master core CAR values appear to increase in recent years, often post-AD 1980. The increase in CAR is likely the result of an increase in apparent PAR as peat transfers to the acrotelm. The increase in SCP-reconstructed CAR across all SCP methods occurs earlier than the increase in CRS CAR in most cases (e.g. LVW-3-M *ca*. AD 1985, MIG-2-M *ca*. AD 1970). SCP-reconstructed CAR profiles were compared to CRS-reconstructed CAR to assess the ability of SCPs to replicate results determined using continuous radiometric chronologies (Table 7.4). Chronological uncertainty was not accounted for in these tests. Where the full CAR profiles were compared (Table 7.4; 'FULL') 'SCP Vis.' produced the best agreement for six records, whilst 'SCP Cum' and 'SCP Adj. 0%' each exhibited best agreement with CRS CAR once.

Given the increase in CAR in recent years (*ca*. post-AD 1980), breakpoint analyses were conducted on CRS CAR profiles to determine any notable increases in CAR. Then, CRS-reconstructed CAR and SCP methods were compared, with post-CRS CAR breakpoint data removed (Table 7.4; 'B. POINT'). Removing post-breakpoint data improved the agreement between 'SCP Vis.' CAR and CRS CAR in most cases. The cumulative approaches ('SCP Cum.' and 'SCP Adj. 0%') showed more variability with no clear agreement across the assembled cores. Therefore, 'SCP Vis.' was determined to be the best method for calculating carbon dynamics using SCP chronologies. All subsequent references to SCP carbon accumulation and/or carbon stock relate to values determined using 'SCP Vis.'.

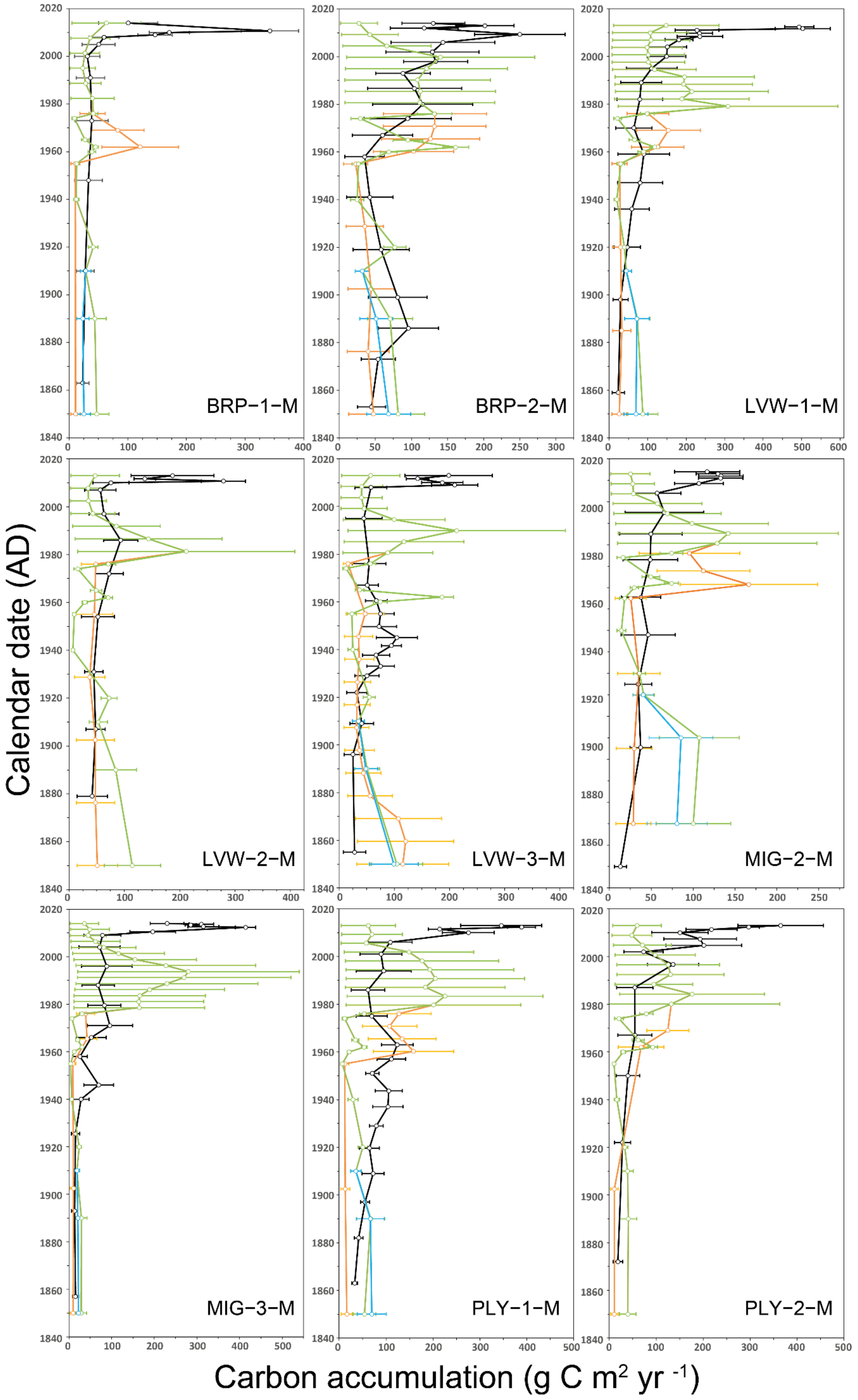


Figure 7.12: Reconstructed CAR for each master core according to differing age depth models. Black indicates ‘CRS model’, orange indicates ‘SCP Vis.’, green indicates ‘SCP Cum.’ and blue indicates ‘SCP Adj. 0%’. ‘SCP Vis’, ‘SCP Cum.’ and ‘SCP Cum. Adjusted 0%’ follow their definitions in Section 5.3.3.3. Y-axes consistent, x-axes optimised by core.

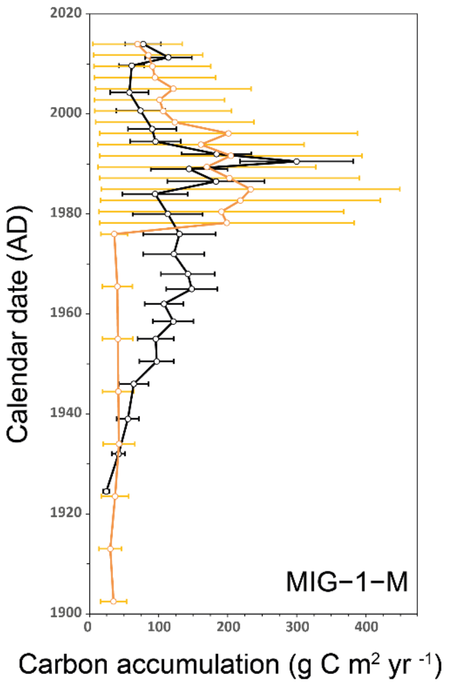


Figure 7.13: Reconstructed CAR for MIG-1-M according to differing age depth models. Black indicates ‘CRS model’ and orange indicates ‘SCP Vis.’. It was not possible to calculate ‘SCP Cum.’ and ‘SCP Cum. Adjusted 0%’ given the start of the SCP record (i.e. AD 1850) was not recorded (Section 5.2.6).

Table 7.4: Pearson correlation coefficients for CRS modelled CAR and CAR reconstructed using differing SCP age/depth models. ‘Full’ represents entire dataset, ‘B. POINT’ indicates where post-breakpoint data were excluded as discussed in the text.

Core		**SCP Vis.**		**SCP Cum.**		**SCP Adj. 0%**	
		***R*-value**	***P*-value**	***R*-value**	***P*-value**	***R*-value**	***P*-value**
BRP-1-M	FULL	0.263	<0.001	0.176	<0.05	0.442	<0.001
	B. POINT	0.560	<0.001	-0.648	<0.001	0.033	0.652
BRP-2-M	FULL	0.275	<0.001	0.189	0.008	0.217	<0.005
	B. POINT	0.408	<0.001	0.345	<0.001	0.167	<0.05
LVW-1-M	FULL	0.373	<0.001	0.269	<0.001	0.288	<0.001
	B. POINT	0.459	<0.001	0.271	<0.001	0.300	<0.001

Table 7.4 (cont.)

LVW-2-M	FULL	0.263	<0.005	0.18	<0.05	N/A	N/A
	B. POINT	0.635	<0.001	0.478	<0.001	N/A	N/A
LVW-3-M	FULL	-0.212	<0.005	-0.206	<0.005	-0.187	0.009
	B. POINT	-0.261	<0.001	-0.243	<0.005	-0.221	<0.005
MIG-1-M	FULL	0.458	<0.001	N/A	N/A	N/A	N/A
	B. POINT	0.426	<0.001	N/A	N/A	N/A	N/A
MIG-2-M	FULL	0.140	<0.05	-0.420	<0.001	-0.319	<0.001
	B. POINT	0.406	<0.001	-0.408	<0.001	-0.252	<0.001
MIG-3-M	FULL	0.510	<0.001	0.428	<0.001	0.445	<0.001
	B. POINT	-0.967	<0.001	0.502	<0.001	0.701	<0.001
PLY-1-M	FULL	0.306	<0.001	0.001	0.495	-0.029	0.358
	B. POINT	0.433	<0.001	0.002	0.488	-0.036	0.329
PLY-2-M	FULL	0.391	<0.001	0.312	<0.001	N/A	N/A
	B. POINT	0.841	<0.001	0.527	<0.001	N/A	N/A

In addition to providing continuous carbon accumulation histories, dated profiles can be used to determine carbon stock above a given horizon. Carbon stock can be used to calculate a constant annual rate of carbon accumulation for time elapsed since that dated horizon (i.e. classic RERCA). The three SCP horizons of AD 1850, 1955 and 1976 were used as markers to calculate carbon stock and RERCA using CRS and SCP age/depth relationships (Table 7.5). It was only possible to calculate CRS carbon stock above the AD 1850 horizon for MIG-2-M, as this was the only core where the CRS chronology extended to this horizon. Better agreement tends to be exhibited between the two methods for post-AD 1955 RERCA where values are often within 5 g C m^{2} yr^{-1} of each other (e.g. BRP-1-M). Similar agreement for the AD 1976 horizon is only exhibited at MIG-2-M and

PLY-2-M. In the one instance where post-AD 1850 RERCA is calculable for CRS chronologies (MIG-2-M) the two methods agree to within 7 g C m^{2} yr^{-1}.

Table 7.5: Comparisons of CRS and ‘SCP Vis.’ modelled carbon stocks and RERCA. A positive value in the ‘Diff.’ column indicates CRS estimates a greater value than SCP and a negative value indicates SCPs estimate a greater value than CRS.

Core	**Horizon (AD)**	**Carbon stock above horizon (g C m^{2})**			**RERCA (g C m^{2} yr^{-1})**		
		CRS	**SCP**	**Diff.**	**CRS**	**SCP**	**Diff.**
BRP-1-M	1850	N/A	4221	N/A	N/A	26	N/A
	1955	2922	3092	-170	50	52	-2
	1976	2153	1380	773	57	36	21
BRP-2-M	1850	N/A	9822	N/A	N/A	60	N/A
	1955	6380	6069	311	108	103	5
	1976	4836	6669	-1833	127	176	-49
LVW-1-M	1850	N/A	11589	N/A	N/A	71	N/A
	1955	6946	8475	-1530	118	144	-26
	1976	5406	5812	-406	142	153	-11
LVW-2-M	1850	N/A	8890	N/A	N/A	55	N/A
	1955	4252	4193	59	73	72	1
	1976	2952	3178	-226	80	86	-6
LVW-3-M	1850	N/A	9129	N/A	N/A	56	N/A
	1955	3552	3583	-31	61	62	-1
	1976	2314	3246	-932	63	88	-25

Table 7.5 (cont.)

MIG-1-M	1850	N/A	N/A	N/A	N/A	N/A	N/A
	1955	6568	6557	11	111	111	0
	1976	3885	5756	-1870	102	151	-49
MIG-2-M	1850	7207	8407	-1200	44	51	-7
	1955	3557	5241	-1684	60	89	-29
	1976	2621	2673	-52	69	70	-1
MIG-3-M	1850	N/A	7566	N/A	N/A	46	N/A
	1955	4920	6526	-1606	83	111	-28
	1976	3651	5669	-2018	96	149	-53
PLY-1-M	1850	N/A	9746	N/A	N/A	60	N/A
	1955	5876	8419	-2543	101	145	-44
	1976	4011	5645	-1634	108	153	-45
PLY-2-M	1850	N/A	6777	N/A	N/A	42	N/A
	1955	4962	5737	-775	86	99	-13
	1976	3963	3846	117	107	104	3

Where uncertainty at each horizon is included, agreement between the SCP and CRS data is again variable (Figure 7.14). CRS and SCP estimates for RERCA vary by 2-39 g C m^{2} yr^{-1} for the AD 1955 horizon and from 0-66 g C m^{2} yr^{-1} for the AD 1976 horizon. SCP chronologies tend to overestimate RERCA relative to CRS chronologies, as evidenced by more samples plotting above the 1:1 agreement line in Figure 7.14.

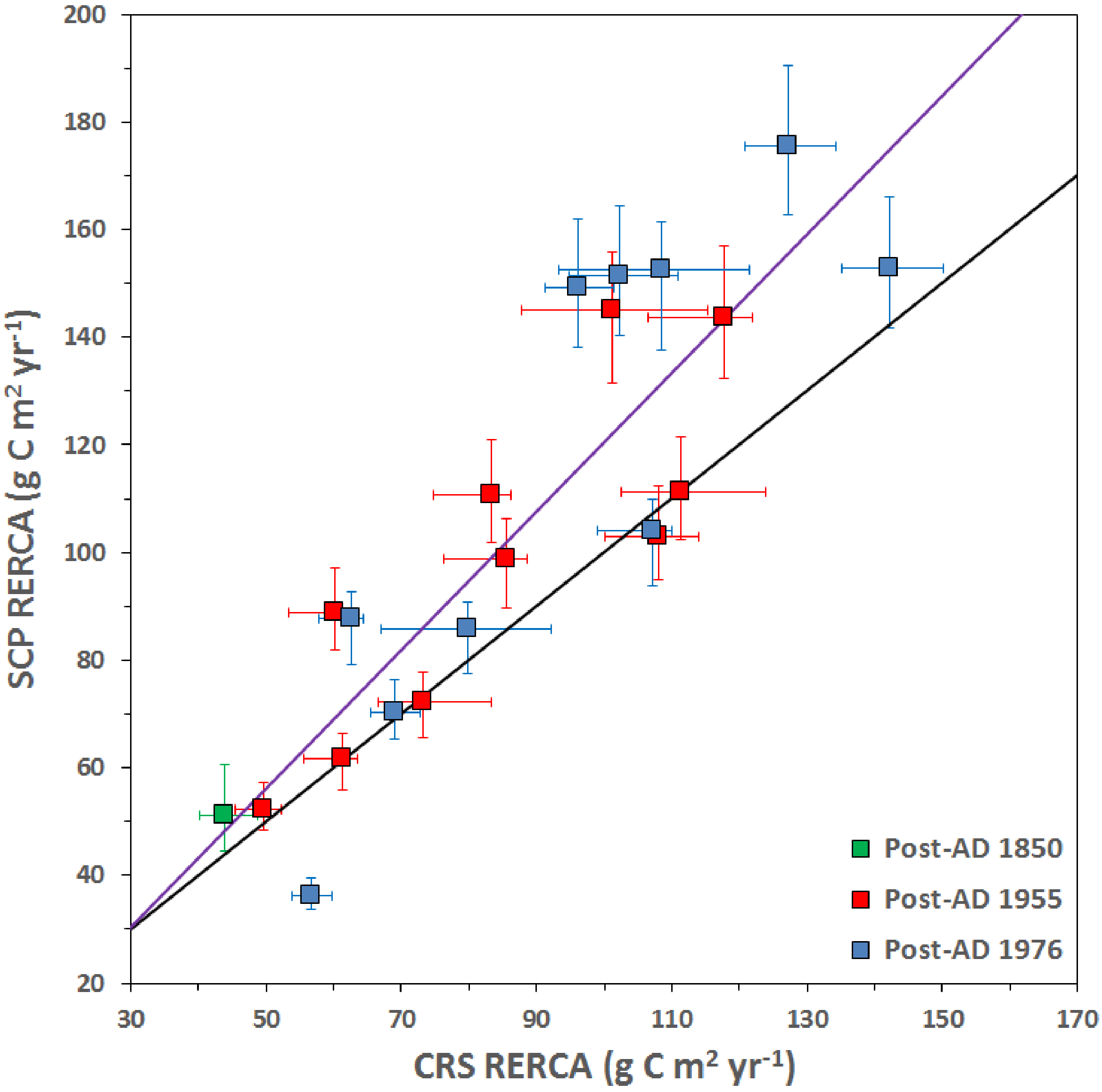


Figure 7.14: Comparison of CRS and SCP calculated RERCA for each master core. Black line represents 1:1 agreement, whilst the purple line indicates linear model for CRS vs SCP RERCA. Points above the 1:1 line indicate SCP age/depth models estimate greater RERCA than CRS models, whilst points below the 1:1 line indicate CRS models estimate greater RERCA than SCP models. X-axis errors relate to the impact of uncertainty in the CRS chronologies. Y-axis errors relate to the impact of uncertainty in the three dateable SCP horizons on SCP calculated RERCA.

7.4.2 Replicate core reconstructed carbon accumulation

It was only possible to reconstruct the carbon dynamics of replicate cores using the ‘SCP Vis.’ method given these cores were only dated using SCPs (Section 5.4). Some example CAR diagrams are presented here, with the remaining CAR and all RERCA results located in Appendix C.

Most replicate cores display similar patterns of SCP reconstructed CAR to those displayed in master cores in that CAR typically increases *ca*. AD 1955 relative to pre-AD 1955 CAR (Figure 7.15; Figure 7.16). If the AD 1955 horizon was not recorded, then CAR often does not rise until AD 1976 (i.e. the next dateable feature of the SCP curve) (e.g. LVW-1-REP-D; Figure 7.15). Pre-AD 1955 CAR is often *ca*. 30-50 g C m^2 yr^{-1}. Peak CAR of between 170-600 g C m^2 yr^{-1} is often recorded *ca*. AD 1980-1995. CAR in most cores appears to decline to present from this late 20th Century peak. It is uncertain whether this is a ‘true’ signal that carbon accumulation is slowing or whether it is more likely an effect of the lower BD of peat which has accumulated since *ca*. AD 1980 (Figure 7.11).

Replicate cores were assigned specifically to investigate the impact of a presumed land management regime (i.e. burning, drainage etc.). There is notable variability in CAR profiles between replicate cores recovered from the same sub-site, and hence under the same presumed management conditions. Some examples of this variability are provided here, but more are observable in Appendix C. Replicate cores from the MIG-1 site were all recovered within 50m of each other from a peatland where records suggest only grazing has been used to manage the land. The CAR profile for MIG-1-REP-B is markedly different from the profiles for MIG-1-REP-A and C because the AD 1850 SCP marker was not recorded in the latter two cores, hence CAR can only be reconstructed to *ca*. AD 1970 and 1940 respectively (Figure 7.16). All these MIG-1 cores have though been recovered from the same presumed management class (i.e. grazing). MIG-3-REP-B CAR remains *ca*. 25 g C m^2 yr^{-1} from AD 1850-1976 whilst MIG-3-REP-C fluctuates from *ca*. 75-350 g C m^2 yr^{-1} over the same period (Figure 7.16). These two cores therefore exhibit notably differing carbon accumulation regimes, but were both recovered from the same drained site <10m apart. As such, it is unlikely that management history alone can explain variations in carbon dynamics across the replicate cores. This finding supports observations made in the plant macrofossil results that cores apparently managed according to the same land use exhibited notably different vegetation records (Section 6.2.3).

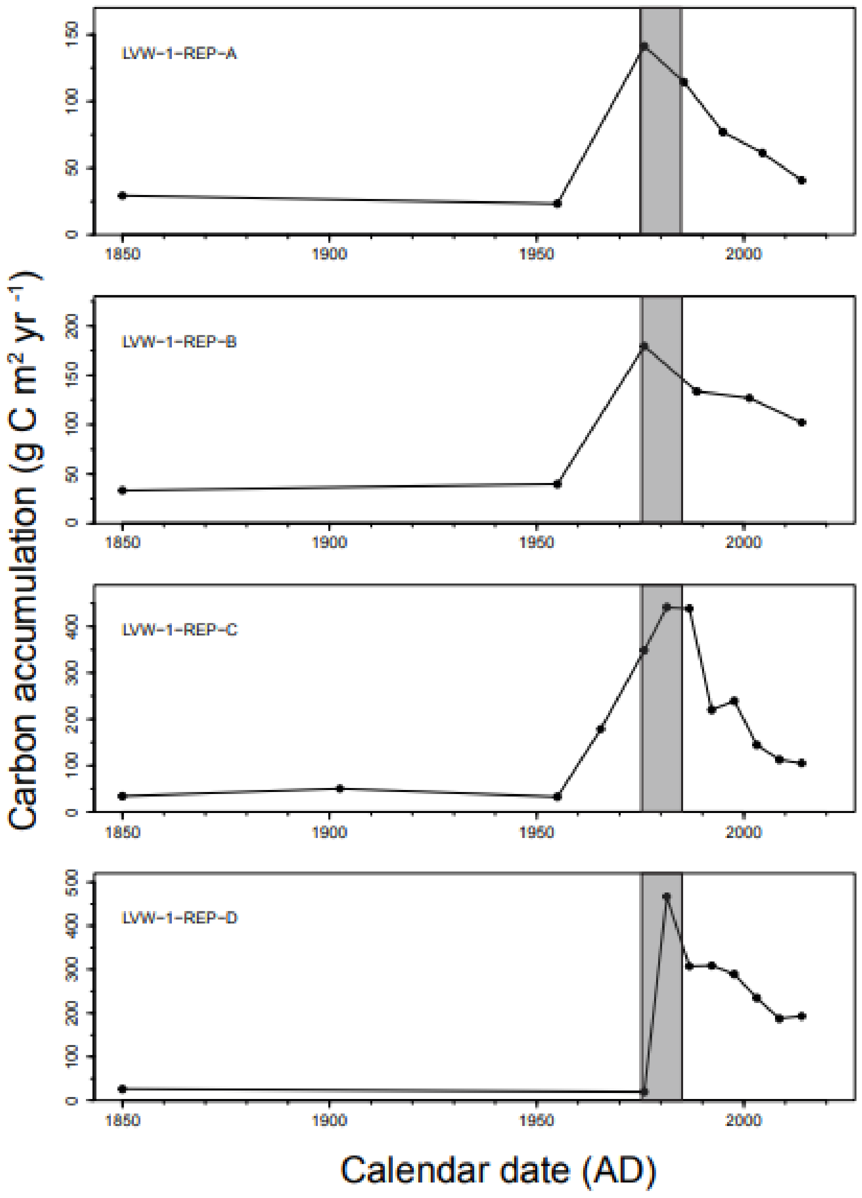


Figure 7.15: CAR for each replicate core recovered from LVW. Y-axes optimised by core. The grey box indicates AD 1976-1986, given the relevance of this time period to the effect of the acrotelm/catotelm transition, as discussed in Section 7.3.

The single outlier for both carbon stock and RERCA for each SCP horizon is from the MIG-2-REP-C core (Figure 7.17). MIG-2-REP-C exhibits the greatest rate of CAR across the replicate dataset, estimated at *ca*. 600 g C m^2 yr^{-1} post-AD 1955. Carbon dynamics at MIG-2-REP-C reported here may be accurate but the value is an outlier relative to the remainder of the replicate dataset and is therefore removed in subsequent analyses (see below).

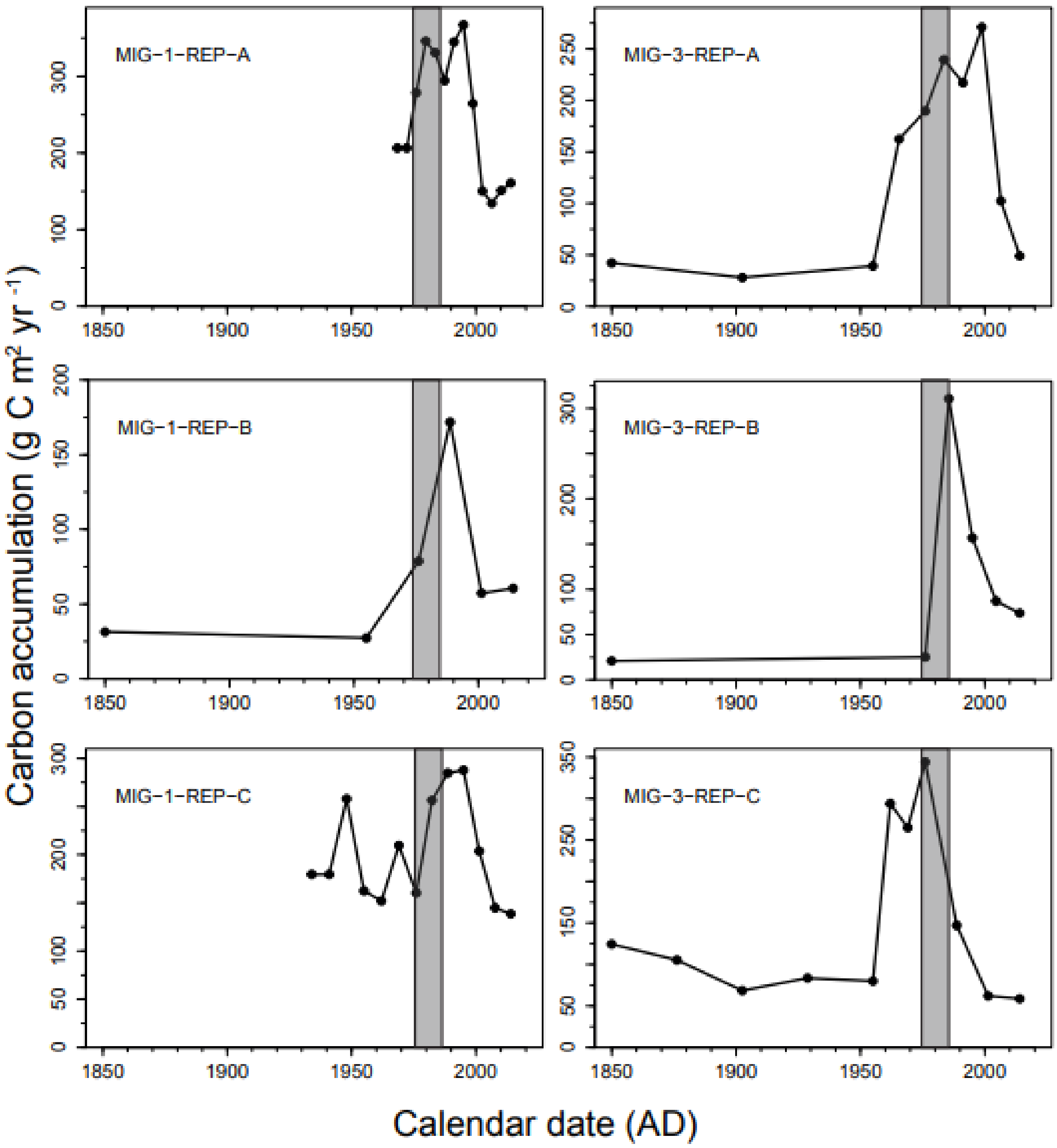


Figure 7.16: CAR for each replicate core recovered from MIG-1 and 3. Y-axes optimised by core. The grey box indicates AD 1976-1986, given the relevance of this time period to the effect of the acrotelm/catotelm transition, as discussed in Section 7.3.

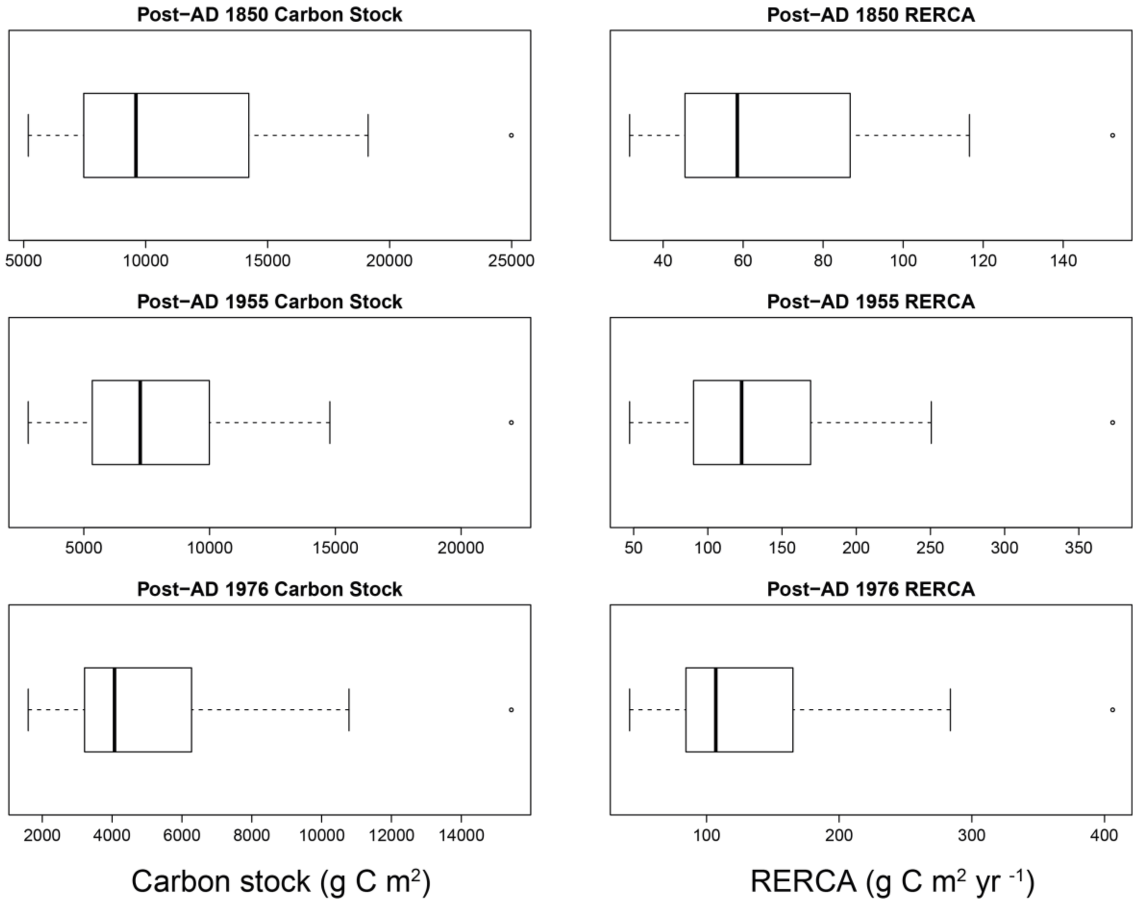


Figure 7.17: Distribution of carbon stock and RERCA values above each SCP horizon for all replicate cores recovered in this study (‘FULL’ dataset; Table 7.19). MIG-2-REP-C is only the core which reports a value outside the 1.5x interquartile range of the dataset.

7.5 Carbon dynamics of Peat Accumulation Types

The five Peat Accumulation Types (PATs) established in Section 6.3.2 detail peatlands which have developed in a similar manner over the last *ca*. 150 years, at least according to patterns of vegetation change. These discrete classes were developed to detect the impacts of land use. One may also expect that peatlands characterised by differing vegetation may accumulate carbon at differing rates given the established control of plant community structure on carbon accumulation (Blodau, 2002; Holden, 2005a).

Average CAR profiles were created for each PAT to detect systematic differences in carbon accumulation regime between PATs. These PAT CAR profiles were calculated as a constant value for each ten-year time-slice within the SCP dating window (i.e. AD 1850-1860, 1860-1870 etc.) to minimise noise within the datasets to detect more consistent differences in carbon dynamics between PATs.

Yearly CAR values for each core were first averaged to create ten-year time-slices. A mean rate of CAR for each time-slice for each PAT was then calculated using the cores within each PAT. PAT CAR values were calculated from AD 1850-1990. The most recent *ca*. 20-25 years were not plotted due to the apparent recent decline in CAR in most cores that is likely due to the less consolidated upper portions of the profile (see above). *Sphagnum*-dominated cores (Type A) exhibit the greatest rate of pre-AD 1950 CAR and remain highest through the AD 1950-1990 period (Figure 7.18). Such a high rate of CAR relative to other states may present an 'optimum' condition for carbon accumulation; hence the average rate and confidence intervals for the *Sphagnum*-dominated PAT were overlaid onto those of remaining PATs in Figure 7.18 to allow comparison. Whilst the *Sphagnum*-dominated PAT exhibits the greatest rate of pre-AD 1950 CAR, this value only exceeds the confidence intervals of PATs 'increasing *Sphagnum*' and '*Molinia*'. Post-AD 1955, the deviation from '*Sphagnum*-dominated' becomes more apparent for PATs '*Calluna*' and '*Molinia*', though 'afforested' and '*Sphagnum*-dominated' confidence intervals overlap until AD 1990. 'Increasing *Sphagnum*' PAT CAR notably increases post-AD 1970 and its confidence band overlaps with that of '*Sphagnum*-dominated' for the AD 1980-1990 time-slice (Figure 7.18).

The same broad patterns in CAR can also be identified from the PAT carbon stock and RERCA values (Figure 7.19). '*Sphagnum*-dominated' exhibits the greatest median stock for all SCP horizons and hence the greatest RERCA. The median 'increasing *Sphagnum*' post-AD 1850 RERCA is the second lowest of all PATs. This PAT has the second highest median post-AD 1976 RERCA, reflecting a rapid increase in apparent carbon accumulation over the last *ca*. forty years (Figure 7.19). The '*Molinia*' stock and RERCA are the lowest of all PATs across all SCP horizons (Figure 7.19). A Dunn Test (DT) (see Section 6.3.2) was used to test for significant differences in RERCA values between PAT types. Two-tailed significant differences exist between '*Sphagnum* dominated' RERCA values and '*Calluna*' and '*Molinia*' values for the post-AD 1955 and 1976 horizons (Table 7.6). '*Molinia*' PAT RERCA values exhibit the greatest difference relative to other PAT types across all horizons, save for the comparison between '*Molinia*' and 'Increasing *Sphagnum*' for the post-AD 1850 horizon (Table 7.6). More significant differences may not have been detected due to the somewhat noisy data given the distribution of RERCA values for some PATs often overlap (e.g. post-AD 1955; Figure 7.19).

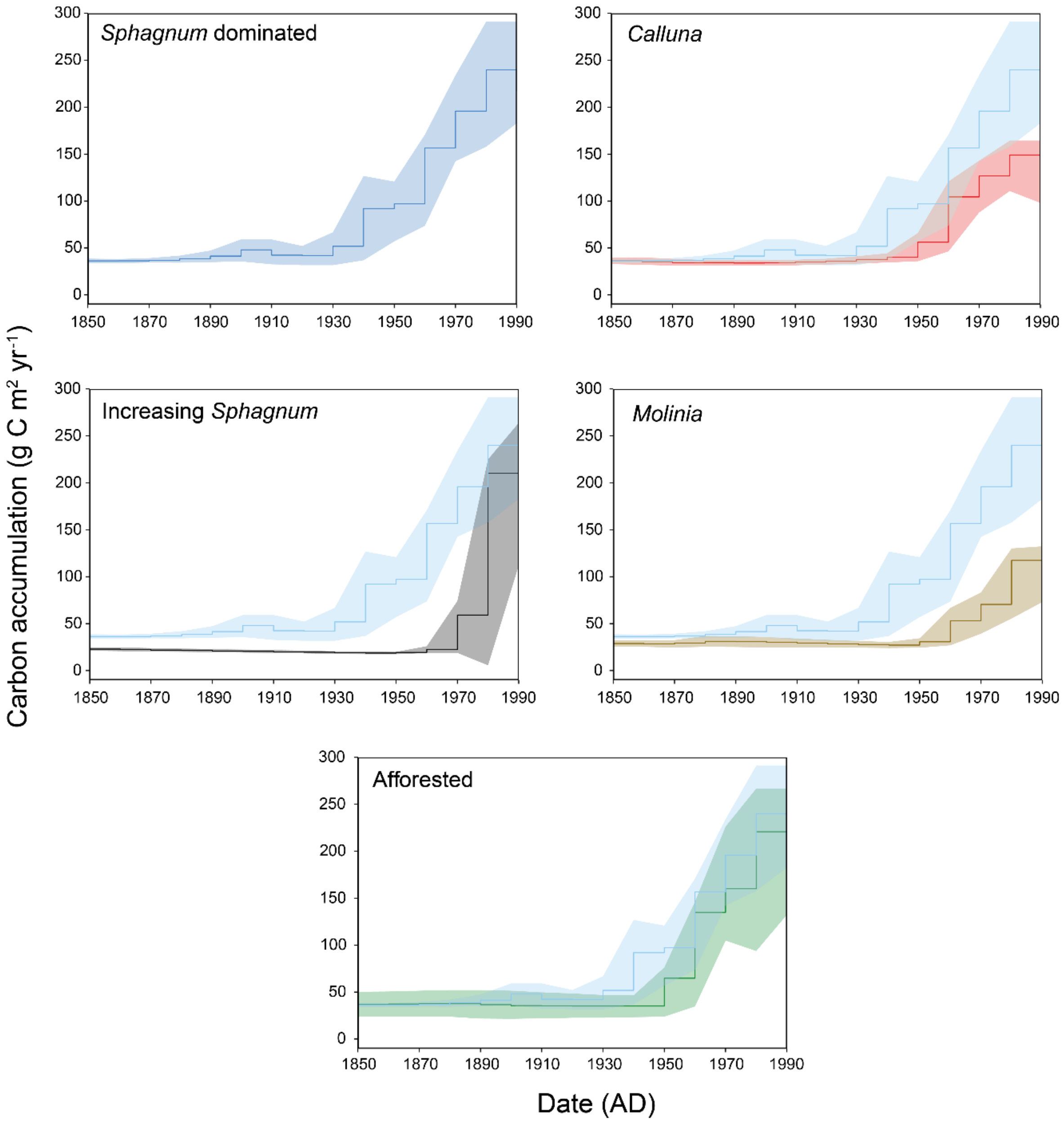


Figure 7.18: Average CAR values per 10-year time slice for each PAT. Only AD 1850-1990 is presented following reasoning established in the text. Confidence bands represent standard error of the mean used to create each average value for each ten-year time-slice. '*Sphagnum* dominated' is superimposed on all categories to emphasize deviation from this potential reference state. MIG-2-REP-C was excluded from the calculation given reasoning established in Section 7.4.2.

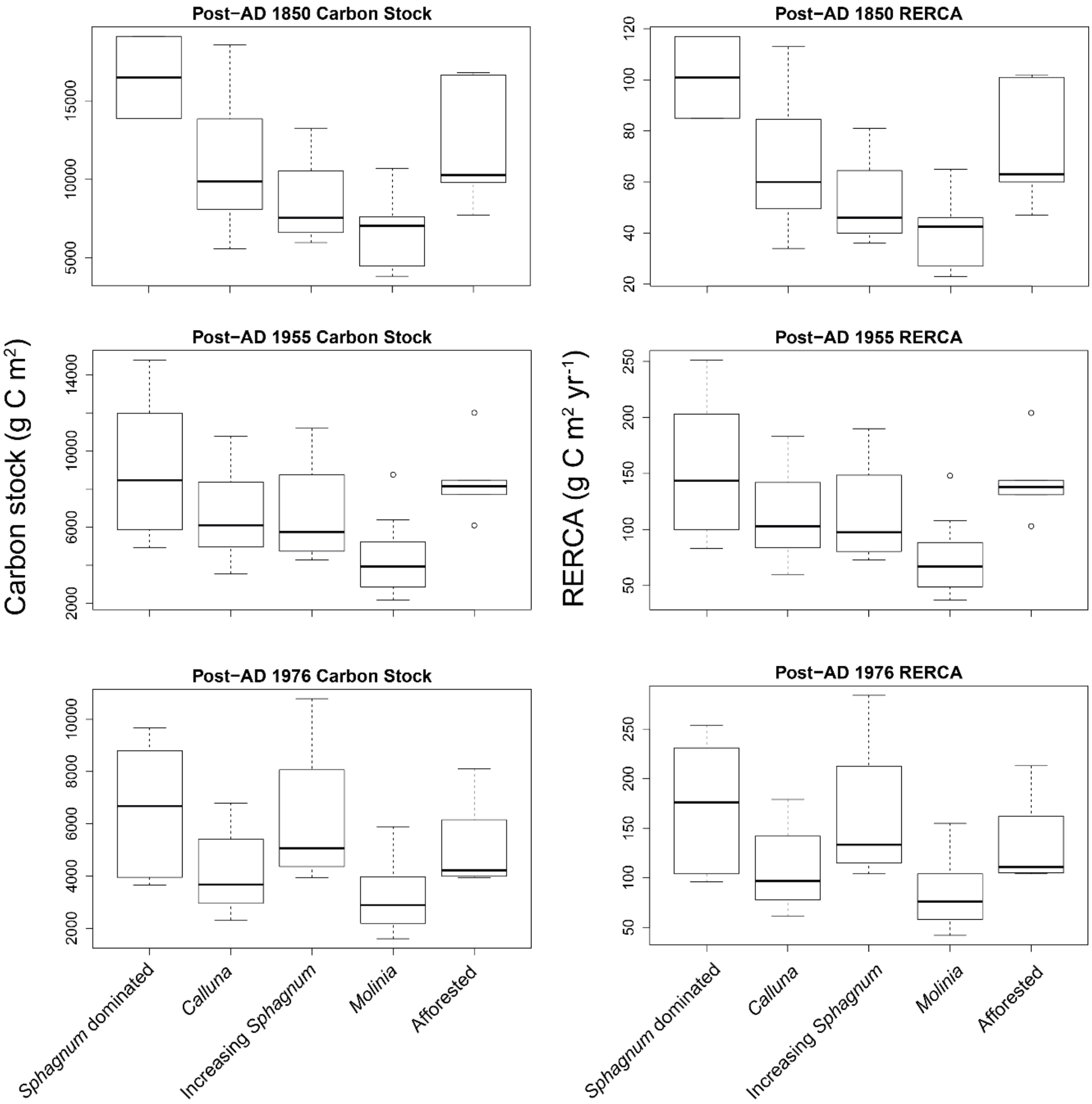


Figure 7.19: Distribution of carbon stock and RERCA values above each SCP horizon according to each PAT. Note scale changes for reporting carbon stock and RERCA value.MIG-2-REP-C was excluded from the calculation given reasoning established in Section 7.4.2.

Table 7.6: Dunn test results for pairwise comparisons between RERCA values for PATs A-E (lettering according to Table 6.18). Values above the diagonal indicate the z-test-statistic for each comparison, and values below the diagonal indicate significance of the result. Results significant at <0.05 in a two-tailed test indicated in italics. Results significant at <0.05 in a one-tailed test indicated by a *.

	A	B	C	D	E
Post-AD 1850					
A	/	1.4	1.5	2.7	0.8
B	0.16	/	0.4	2.5	-0.8
C	0.12	0.66	/	1.3	-1.0
D	0.007*	*<0.05*	0.19	/	-2.6
E	0.44	0.44	0.32	0.09*	/
Post-AD 1955					
A	/	2.0	1.4	3.4	1.1
B	*<0.05*	/	-0.3	2.4	-0.9
C	0.16	0.76	/	1.9	-0.4
D	*<0.001*	*<0.05*	0.56*	/	-2.6
E	0.28	0.38	0.68	*<0.01*	/
Post-AD 1976					
A	/	2.6	1.1	3.7	1.4
B	*<0.01*	/	-1.2	1.8	-1.0
C	0.26	0.24	/	2.3	0.2
D	*<0.001*	0.74*	*<0.05*	/	-2.2
E	0.16	0.32	0.82	*<0.05*	/

7.6 Conclusions

This chapter has presented the results of the chemical and physical analysis of the peat cores recovered in this study. The analyses conducted in this chapter have also demonstrated the uncertainty associated with palaeoenvironmental reconstructions of peatland carbon dynamics. Uncertainties in calculating carbon dynamics (be it carbon accumulation rate (CAR) or recent apparent rate of carbon accumulation (RERCA)) will always be present given inherent uncertainties with dating sedimentary sequences. Data here have been evaluated and Spheroidal Carbonaceous Particles (SCPs) were found to provide reasonable reconstructions of carbon dynamics when compared with Constant Rate of Supply ^{210}Pb (CRS) reconstructed carbon dynamics (Section 7.4.1). As such, data are deemed of a sufficient quality to have characterised carbon accumulation at the studied sites.

Analyses suggest that the Peat Accumulation Types (PATs) created in Section 6.3.2 may be useful in assessments of peatland carbon dynamics. There is interest in such vegetation based approaches to monitoring carbon dynamics given the relative ease with which peatland vegetation can be surveyed (Audet *et al*. 2013; Garnett *et al*. 2001). The averaging of carbon dynamics into discrete classes will inevitably mask some variability, with this issue explored in more detail in Chapter 8. However, significant differences in carbon dynamics between PATs (Section 7.5) suggests these classes are of use in subsequent discussions on the role of vegetation and land use in affecting peatland carbon accumulation.

The main conclusions from this chapter are:

- Peat carbon and nitrogen content may in part be controlled by bulk density. The combined physical and chemical metrics derived here may be used to approximate the acrotelm/catotelm transition.
- Continuous CAR determined using SCP age/depth models often displayed significant agreement with CRS CAR records. The ‘SCP Vis.’ method gave the best agreement and was adopted for investigating carbon dynamics.
- Post-AD 1955 and 1976 RERCA determined by SCPs may overestimate RERCA above the same horizons as determined using CRS chronologies.
- Significant differences exist between the carbon dynamics of PATs which suggests either that vegetation directly, or land use which causes vegetation to change, is affecting peatland carbon accumulation.

Chapter 8 Discussion

8.1 Introduction

Peatland land management is a contentious issue given the range of conflicting interests involved in land use and the equivocal evidence base regarding the impact of specific land use types (e.g. burning, grazing) on peatland integrity (Chapter 2). The primary aim of this thesis is to characterise the impact of land management on blanket peatland carbon accumulation (Section 1.2). In this chapter, the findings from results presented in Chapters 5, 6 and 7 are discussed and their implications for the peatland management debate (i.e. the degree to which land management alters peatland vegetation, carbon dynamics etc.) are highlighted.

The first section of this chapter (Section 8.2) discusses the development of Welsh blanket peatlands using vegetation records as indicators of long-term environmental change. Observed trends in vegetation are discussed in light of known and inferred changes in land management history. The second section builds on the initial discussion and attempts to characterise the impact of land management on blanket peatland carbon accumulation (Section 8.3). A final section (Section 8.4) considers the applied relevance of the thesis by determining its contribution to the peatland management debate.

8.2 Palaeoenvironmental evidence for environmental change across Welsh blanket peatlands

Many previous palaeoenvironmental records have been recovered from Welsh blanket peatlands (Payne *et al*. 2016; Table 2.2). A limited number of these have examined changes in peatland integrity over time (Section 2.7). This section of the discussion explores how the plant macrofossil records presented in this thesis may be used to improve understanding of the drivers behind long-term vegetation change on Welsh blanket peatlands.

8.2.1 Considerations when reconstructing environmental change using palaeoenvironmental approaches

An environmental change (e.g. climate change, a change in land use) can only be identified in a plant macrofossil record if the change was of a sufficient magnitude to affect peatland vegetation communities. Previous palaeoenvironmental investigations often focussed on the sensitivity of peatland plant communities to hydrological conditions (Barber *et al*. 2003; Hughes *et al*. 2006; Langdon and Barber, 2005; Roland *et al*. 2015), primarily shifts in water table depth (WTD) (Amesbury *et al*. 2012a). Analyses presented in Chapter 6 showed that hydrological conditions were likely the key control on blanket peatland vegetation assemblages at three of the four study sites. Hence, the reliability of hydrological inferences taken from plant macrofossil records are reviewed here.

Peatland plant communities exhibit threshold/step-change responses to variations in WTD (Barber *et al*. 2003) and temperature (Dieleman *et al*. 2015), hence plants do not respond instantaneously to environmental change. The perennial life cycles of plants mean that rapid environmental changes may not be recorded accurately in the palaeo-record unless the change persists (Hughes *et al*. 2006; Väliranta *et al*. 2012). Species affiliated with wetter environments exhibit a narrower hydrological tolerance than those affiliated with drier environments (Lamentowicz and Mitchell, 2005; Väliranta *et al*. 2012). A shift from wetter conditions may therefore be identified by the loss of hydric taxa, whilst signals of a system shifting from a dry state may be relatively muted due to the wider hydrological tolerances of xeric-mesic taxa. There may be a time lag from a change in land use to its manifestation in the palaeo-record (Belyea, 2009; Page and Baird, 2016). Where a known date is linked to a management change, chronological uncertainty (Chapter 5) may ‘suck-in’ in unrelated related changes or ‘smear’ changes over a wider period (*sensu* Baillie, 1991, 1993; Blaauw *et al*. 2006). Peatland autogenic processes (Malhotra *et al*. 2016), e.g. secondary decomposition from water table variability (Morris *et al*. 2015b), may mask the impacts of external forcings capable of altering peatland conditions (Belyea and Baird, 2006), as documented for climate reconstructions (Swindles *et al*. 2012). Homeostatic regulation, e.g. declining hydraulic conductivity following drainage as the substrates dries (Morris *et al*. 2015a), may mean that macrofossil records indicate no change in peatland management regime even if one had occurred.

Plant macrofossil records remain a useful method for investigating the impacts of land management despite such caveats. Surface vegetation is a widely used indicator of peatland condition (Section 2.2.4.1), hence a vegetation-based approach to identifying the impacts of land management is congruent with established peatland monitoring doctrine. Changes in plant macrofossil records are also likely to indicate a notable change in peatland condition (i.e. of interest to land managers) given plants require sustained environmental forcing to induce community turnover (*sensu* Mackay, 2015; see above).

Determining direct quantitative links between specific management interventions and changes in plant phytosociological assemblages is not possible using macrofossil analysis (Section 3.4.5). Assuming such a relationship would be in any regard flawed. It is unlikely that all blanket peatlands respond to management along some linear continuum (Section 2.8) in much the same way that changes in WTD may not be linearly related to climatic phenomena (Swindles *et al*. 2012). A major vegetation change may in cases be linked to a land use change. As such, this thesis does attribute changes in vegetation to be an expression of the impact of land management but does not provide a direct quantitative link between specific management regimes and peatland plant communities. Comparisons between multiple sites may be sufficient for detecting the expression of management, thereby excluding noise related to autogenic processes, site specific characteristics and/or proxy uncertainty (see above).

Section 2.7 demonstrated the limited understanding of long-term changes in Welsh blanket peatland integrity. Chronological control here is constrained to the previous *ca.* 150-180 years, hereafter termed the ‘dateable period’. Temporally uncertain inferences can be made about longer term development using macrofossil evidence situated ‘below’ the earliest dated point of each record.

8.2.2 The role of climate in driving recent blanket peatland development

The expression of climate change over the dateable period in previously published Welsh blanket peatland records suggests a warming and drying trend relative to the colder and wetter Little Ice Age which ended *ca*. AD 1800-1850 (Section 2.7). Instrumental data can be used to determine whether palaeoenvironmental data indicate past climate change (Jones and Thompson, 2003). Previous studies which compare palaeoecological data with instrumental records originate from an assortment of peatland habitats and from several climatic settings. To date (AD 2017), no direct comparison of palaeoecological records with instrumental data has been attempted for Welsh blanket peatlands. AD 1912-2009 mean annual temperature and precipitation for the UK Climate Projections 2009 (UKCP09) gridded datasets (Section 4.3) show that mean annual air temperature appears to be increasing at all sites, whilst changes in precipitation are often not significant (Appendix D). There were several significant correlations between climatic variables and master core plant macrofossil Detrended Correspondence Analysis (DCA) sample scores over the AD 1912-2009 period, predominantly between precipitation and Axis two sample scores (Appendix D). Axis two was identified in three cases to be some environmental gradient capable of forcing a change in habitat status from blanket bog to humid heath (Section 6.4) and precipitation is a key variable in modelling the future integrity of the blanket peatland habitat under projected 21st Century climate change (Gallego-Sala *et al*. 2010; Gallego-Sala and Prentice, 2013).

It is uncertain whether blanket peatland vegetation communities are sensitive enough to respond to changes in climate over decadal to sub-decadal timescales. Amesbury *et al*. (2012b) showed plant macrofossil records were responding to changes in climate at inter-decadal scales in four north-west European mires, though none were blanket bogs. They determined a peat accumulation rate >1mm yr^{-1} would be required for detecting rapid shifts in plant community structure in response to climate. Seven of the master cores in this thesis exhibited rates of apparent peat accumulation >1mm yr^{-1} over the AD 1912-2009 period. As such, peat is accumulating sufficiently fast to detect inter-decadal climate variability following Amesbury *et al*. (2012b).

Interpretations of vegetation change on DCA Axis two should be treated with caution given the eigenvalue may indicate an insufficient ecological gradient was sampled in response to the supposed ‘climatic’ driver (e.g. Migneint Axis two eigenvalue of 0.2534; Figure 6.20). Significant correlations may therefore represent statistical noise in the macrofossil datasets rather than climatic control on vegetation. Chronological uncertainty may also confound the relationship between the macrofossil record and climate (Amesbury *et al*. 2012a). Each macrofossil sample has an uncertainty associated with the most likely date (i.e. ± error). DCA values for a given sample may be ‘sucked-in’ to represent a single point in time (*sensu* Baillie, 1991), hence either falsely confirming or refuting a link between vegetation and climate. There is therefore limited evidence to suggest climate is the primary driver of vegetation change at any of these sites.

No previously published Mid- and North-Welsh blanket peat climatic reconstructions make inferences regarding precipitation and/or temperature changes post-AD 1900 (Blackford, 1990; Bostock, 1980; Ellis and Tallis, 2001). It is not explicitly acknowledged whether this was due to a concern that another driver may have displaced the climate signal, a lack of chronological control, or whether this recent period was simply not of interest. Human activities through a combination of pollution, grazing and/or burning have been the primary drivers of blanket peatland development at Drygarn Fawr since *ca.* AD 1900 (Chambers *et al.* 2007b) and the Brecon Beacons since *ca.* AD 1800 (Chambers *et al.* 2007a, 2013). In other blanket peat records, interpretations regarding climatic change are explicitly limited to pre-AD 1700 (Blundell *et al.* 2016) and pre-AD 650 (McCarroll *et al.* 2017) because of concerns regarding a human impact signal on these macrofossil records. Anderson (1998) and Chambers *et al.* (1997) both comment on the difficulty in differentiating between a climatic and human driver of change in the wetness record *ca.* 1,200 BC at Scottish sites, suggesting humans have been affecting peatland development for thousands of years. In previously published work there is clearly a concern that human activities, either by point (e.g. burning) or diffuse (e.g. pollution) means, have affected blanket peatland vegetation development for some time. One may therefore expect to see no robust link between vegetation development and changes in climate over the dateable period. This is the case in the records in this thesis.

8.2.3 The role of humans in driving recent blanket peatland development

Anthropogenic land-use pressure is not constant, but fluctuates over annual to centennial time scales according to motivations, knowledge and technology (Foley *et al.* 2005). Humans have been active in the Welsh uplands for several thousand years (Bostock, 1980; Woodbridge *et al.* 2012). The last 2,000 years of Welsh history incorporate the Iron Age (700 BC - AD 43), Roman (AD 43-450), early-Medieval (AD 450-1066), Medieval (AD 1066-1547) and post-Medieval (post-AD 1547) archaeological periods (British and Irish Archaeological Bibliography, 2017). The signal of human impact over the dateable period is clear in the Welsh uplands (Chambers *et al.* 2013) and elsewhere in the British Isles (Blundell *et al.* 2016). It is therefore important to assess the degree to which human activities have been the dominant control over vegetation change on blanket peatlands.

Over the last two millennia Welsh people's motivations, knowledge and technology changed, as did their relationship with the uplands (Figure 2.21). In

Chapter 2, evidence of management impacts focussed primarily on land use that occurred during the 20th Century. This temporal focus does not imply an assumption that these impacts have had a greater impact on peatland integrity than earlier human activities. It rather reflects the inability of contemporary disciplines (e.g. biogeochemistry, ecology) to acknowledge changes caused by management that pre-dates 20th Century interventions. This bias is exacerbated by a lack of palaeoenvironmental records that address the condition and integrity of Welsh blanket peats (Section 2.7). The records in this thesis will address this issue both by increasing the number of records from Welsh peats but also by assessing the state of blanket peatlands over the dateable period and comparing changes to earlier, albeit undated, peatland development.

In Chapter 2, four distinct land management practices were highlighted, these being burning, drainage, forestry and grazing. The impacts of these on peatland vegetation communities are now discussed sequentially. Where appropriate, their potential interaction is also highlighted.

8.2.3.1 The effect of burning on blanket peatland development

The range of stakeholders and its prominence in the media make burning arguably the most contentious management practice employed on contemporary blanket peatlands (Brown *et al*. 2016; Davies *et al*. 2016a, b, c, d; Douglas *et al*. 2016a, b; IUCN, 2017b; Yorkshire Post, 2017). Macroscopic charcoal is recorded in thirty-eight of the cores recovered in this study. The presence of macroscopic charcoal is a reasonable indicator for burning at that point on the peatland given its relatively large mass inhibits aeolian and hydrological transport (Patterson *et al*. 1987). Abundant charred wood and charcoal may indicate periods where the peat itself was burned (New *et al*. 2016).

Firing of blanket peatlands is currently primarily performed as rotational burning to manage *Calluna vulgaris* growth on grouse estates, though firing is also used to regulate graminoid growth for grazing. One may therefore expect an association between charcoal and *C. vulgaris* in peat records*. Calluna* and charcoal plot within two standard deviation units (SDs) of each other in the ALL CORE DCA (Figure 6.27), indicating there is an overlap in their distributions toward the drier end of the hydrological gradient that Axis one represents. Of the twenty-two records where *Calluna* is recorded at the surface, seventeen have at least a single ‘abundant’ incidence of charcoal prior to the recording of *Calluna* in the profile. Fires are more likely to occur on peatlands where *C. vulgaris* is present given these drier substrates tend to be more susceptible to burning (Glaves *et al*. 2013; IUCN, 2017b). It is therefore unclear as to whether fire causes or is caused by *Calluna* presence (Davies *et al*. 2016a). Whilst there may be an association between *Calluna* and the use of firing, it is overly simplistic to link the two in this thesis in isolation of other factors.

A key concern regarding the effect of rotational burning is the potential that it may reduce *Sphagnum* coverage (Bain *et al*. 2011; IUCN, 2017b). Whilst this is a view often cited in land management debates, evidence supporting such a notion is mixed. There is evidence that firing has displaced prior *Sphagnum* communities in records recovered in this thesis. Abundant charcoal at LVW-1-REP-B occurs alongside *E. vaginatum* and appears to displace an *S.* s. *Sphagnum* community. There is though evidence that *S. papillosum* can persist through periods of burning (e.g. MIG-3-REP-B and C, MIG-5-REP- A and C). *S. papillosum* is a relatively resistant species, at least relative to *S. austinii* and *S. magellanicum* (McCarroll *et al*. 2016b). It expanded following phases of burning on West Arkengarthdale bog in Yorkshire (McCarroll *et al*. 2016a). As such, its ability to persist through phases of burning in this thesis is unsurprising.

S. s. *Acutifolia* is the most frequently recorded type of *Sphagnum* in surface samples (*ca*. 10-0cm deep) across the dataset. In several cases the *S.* s. *Acutifolia* which expanded following burning phases was identified as *S. rubellum* or *S. capillifolium* subsp. *capillifolium*. Not all work follows this taxonomic framework of Daniels and Eddy (1985) and simply classifies both subspecies as *S. capillifolium*. *S. capillifolium* undiff. has responded favourably to rotational burning at Hard Hill (Lee *et al*. 2013) and managed to return to a near pre-fire coverage <10 years following a wildfire at Glasson Moss NNR (Lindsay and Ross, 1994). In experimental settings *S. capillifolium* has also been shown to persist through/recover soon after burns conducted according to industry practice (Taylor *et al*. 2017). Many *Sphagnum* forms are apparently capable of persisting through burning. Sites where a *S. capillifolium* subspecies persists alongside *Calluna* may indicate a peatland where previous burning has occurred.

The most consistent response of a species to firing seems to be from *E. vaginatum*. Where macrofossil remains are identifiable, *E. vaginatum* often persists alongside abundant charcoal. *E. vaginatum* is resistant to burning as its tussocks protect it from flames and it grows rapidly following fire by using phosphorus deposited in the ash (Anderson *et al*. 2006; Shaw *et al*. 1997). *E. vaginatum* plots as a mid-gradient taxon in all ordinations, predominantly one to two SDs apart from charcoal on Axis one, because of its relative ubiquity through most records. It may still be promoted by fire, but it would not necessarily plot closely to charcoal in a DCA as it is so common.

There is a tendency across sites where burning has occurred for charcoal abundance to decrease toward the surface, indicative of a decline in burning intensity and/or frequency (Ohlson and Tryterud, 2000). Many sites where charcoal is recorded exhibit greater charcoal abundance prior to the onset of the dateable period. The use of fire in the Welsh uplands is by no means constrained to the 20th and 21st Centuries (Section 2.7). Charcoal records from the defunct grouse estate at Lake Vyrnwy indicate a decline in firing intensity/frequency once the estate was established relative to the pre-estate landscape.

The finding here of an apparent decline in burning intensity under grouse estate management is not entirely novel. At Keighley Moor in northern England, charcoal abundance *ca*. 800 BC either matched or exceeded abundances recorded under management for grouse (post-AD 1900) (Blundell *et al*. 2015), indicating heavy burning in prehistory which may exceed contemporary management intensity. Macroscopic charcoal is completely absent from samples on Mossdale Moor which has been managed for grouse for several hundred years (no dates provided) (McCarroll *et al*. 2017). Lascelles (1995) showed that microscopic charcoal concentrations were much greater *ca*. 1,000 BC – AD 0 at Cefn Mawr than concentrations when the peatland was managed for grouse in the 20th Century. Microscopic charcoal is not as preferable for identifying *in situ* burning as macroscopic charcoal (MacDonald *et al*. 1991; Rhodes, 1996), but is still useful for identifying temporal changes in burning activity. Blanket peatland records from elsewhere in Wales also record much greater microscopic charcoal concentrations pre-AD 0 (Mighall and Chambers, 1993; Mighall *et al*. 2002), AD 0-500 (Mighall and Chambers, 1995), AD 0-1400 (Woodbridge *et al*. 2012), AD 800-1200 (Bostock, 1980; Mighall *et al*. 2013) and AD 1200 (Grant, 2007, 2009) relative to the last five hundred years.

The charcoal records in this thesis, alongside the assembled work of others, suggest that firing for grouse represents a decline in burning intensity relative to the historic use of fire. Burning conducted according to ‘industry standards’ require that burns should be relatively ‘cool’ and should not damage the moss/*Sphagnum* layer (Allen *et al*. 2016; Sotherton *et al*. 2017). If such burns could be carried out to this specification, then industry proponents argue *Sphagnum* could persist (Davies *et al*. 2016d, e). Some challenge this notion

(Thompson *et al*. 2016) given the prevalence of 'bare peat' on blanket peatlands managed for grouse (Brown *et al*. 2015a; Yallop and Clutterbuck, 2009). Thesis findings lend credence to the notion that rotational burning may not be as intense as firing of peatlands practiced in earlier times. However, the records have no chronology. Whilst fire may therefore have played a role in the development of many Welsh blanket peatlands, it cannot be definitively stated that such firing was employed as management for grouse (*ca*. post-AD 1800) as opposed to an earlier motivation e.g. Medieval pastoral agriculture.

8.2.3.2 The effect of drainage on blanket peatland development

Most ditches were installed in UK blanket peatlands to increase water table depth (WTD) during the post-World War II expansion of upland agriculture (Section 2.4.2). Ten cores on the Migneint were recovered from within drained peatland. *C. vulgaris* is present in the surface samples of all ten cores, *Scirpus cespitosus* in six, *E. vaginatum* in eight and assorted *Ericaceae* (often *Empetrum nigrum*) in four cores (Figure 6.13 and Figure 6.15). These taxa together indicate a predominantly dry substrate. One may expect these sites to be characterised by such taxa if peats had been 'successfully' drained (i.e. an increase in WTD).

There is a potential signal of drainage in the macrofossil assemblages at some Migneint sites. *S. papillosum* is present at MIG-3-REP-C through the dateable period until its disappearance *ca.* AD 1976. This may relate to the (AD 1970s) drain installation at Lyn Serw (Roberts, A. 2015 pers. comm.). The earlier drainage at Llyn Serw (AD 1918-1939) apparently had little impact on *S. papillosum* cover as the species remains common or abundant across this period. *S. papillosum* declines at MIG-5-REP-A (*ca.* AD 1900-1955) and MIG-5-REP-C (*ca.* AD 1930-1955) may be an effect of the drainage at Llyn Conwy between World War I and II. Davies *et al.* (2016a) suggest that gullying and an associated change in WTD may have contributed to the loss of *S. papillosum* at Keighley Moor in Blundell and Holden (2015). Stewart and Lance (1991) also recorded a significant decline in *S. papillosum* cover at Moor House downslope from drains.

S. papillosum favours intermediate microhabitats and its ability to compete with other *Sphagnum* (e.g. *S. capillifolium* subsp. *capillifolium* or *S. rubellum*) is reduced where water tables are lowered as this inhibits *S. papillosum* capillary action and photosynthesis (Robroek *et al*. 2007b). *S. rubellum* expands to dominance alongside the decline of *S. papillosum* at MIG-3-REP-C and MIG-5-REP-A. *S. rubellum* may have outcompeted *S. papillosum* as it can maintain capitula growth when WTD increases whilst *S. papillosum* cannot (Robroek *et al*. 2007a). Inter-specific competition may be important when determining *Sphagnum* response to drainage, and work focusing simply at genus level (e.g. Green *et al*. 2017; Williamson *et al*. 2017) may not provide adequate understanding of the effects of ditching on *Sphagnum*.

Constraining vegetation response time is important for assessing the impacts of drainage (Bellamy *et al*. 2012). Declines in *S. papillosum* associated with drainage occur over eleven years at MIG-5-REP-A (*ca.* AD 1960-1971) and MIG-5-REP-C (*ca.* AD 1955-1966), these declines starting *ca*. fifteen-twenty years after drainage at Lyn Conwy. The data therefore imply a tendency for *S. papillosum* to decline gradually following drainage. There is limited evidence from blanket bogs to test such a finding. A monitoring study showed *Sphagnum* declines were spatially variable and in cases take twenty years to occur following drainage (Stewart and Lance, 1991). *Sphagnum* declines here are therefore broadly consistent with Stewart and Lance (1991).

Evidence of an impact of drainage on blanket peatland vegetation is present in certain cores from the Migneint. It appears that previous communities of *S. papillosum* may have been displaced by *C. vulgaris*, *E. vaginatum* and *S.* s. *Acutifolia* communities. *S. papillosum* was not prevalent in the records of the remaining drained cores. Instead, xeric *C. vulgaris*/*E. vaginatum* communities were present pre-drainage and appear to have persisted in the presence of abundant charcoal. The effects of fire on these sites pre-dated drain installation (Section 8.2.3.1) and, as such, the drains may have had less of an impact than they would have if they had been installed on intact bog. Limited expressions of drainage in vegetation records may also have occurred due to internal homeostatic mechanisms limiting changes in WTD. Evidence suggests that the low hydraulic conductivity of blanket peatlands (Holden and Burt, 2003) means that ditch installation may have a limited effect (*sensu* Morris *et al*. 2015a; Waddington *et al*. 2015). As such, it is interesting that not all sites in this thesis exhibit a notable response to drain installation.

8.2.3.3 The effect of forestry on blanket peatland development

Forestry arguably causes the greatest change in peatland vegetation of the four management practices considered. Six cores were recovered from a forty-five year old *Picea sitchensis* plantation (Blaenycwm) on the Migneint. All cores record bark and needles alongside shade-affiliated bryophytes, primarily *Rhytidiadelphus* *loreus*. A similar dominance of *Rhytidiadelphus* undiff. was reported from several afforested UK blanket bogs twenty-six years after planting (Anderson, 2001). *Plagiothecium laetum* and *P. schreberi*, other taxa recorded at afforested blanket bogs (Anderson, 2000; Atherton *et al*. 2010) and on forested arctic peatlands (Lachance and Lavoie, 2004) are also present on the forest floor (Figure 6.14). Forest establishment displaced the pre-existing mixed *C. vulgaris*, *E. vaginatum* and *Sphagnum* communities. Declining water levels and increased nutrient inputs from trees in what were waterlogged, nutrient-poor environments reduce the competitive advantage of *Sphagnum* (Anderson, 2001; Wallace *et al*. 1992). *Sphagnum* is also outcompeted in shaded conditions by other bryophytes which can photosynthesize more effectively in low light availability (Hájek *et al*. 2009). Fenton *et al*. (2007) found *Sphagnum* required ≥20% canopy openness to outcompete feather mosses on boreal forest floors.

In most afforested cores there is a *ca.* five-ten year 'transition period' during which *Sphagnum* declines before *R. loreus* becomes established (Figure 6.19). The transition period may reflect poor macrofossil preservation as the peat is characterised by high UOM. *P. sitchensis* roots, which promote oxidation and decomposition (Ohlson *et al*. 2001), are also present in the peat. There is limited ecological monitoring information on the replacement of 'open-bog' vegetation with forest species with which to compare this transition. *Racomitrium lanuginosum*, *S. papillosum* and *S. capillifolium* declined in cover seven years after afforestation on a Caithness blanket bog (Usher, 1996, cited in Anderson, 2001). A survey at Coom Rigg Moss in the Kielder forest also showed little to no *Sphagnum* survived *ca.* thirty years after afforestation (Chapman and Rose, 1991). The decline in *Sphagnum* species at the expense of shade-tolerant mosses is reported for Scandinavian mires (Laine *et al*. 1995, b; Vasander *et al*. 1993), with a mixed *Sphagnum* assemblage displaced over a five-year period on a Swedish bog (Gustafson, 1988). As such, the five-ten year transition period reported in this thesis seems similar to previous observations.

8.2.3.4 The effect of grazing on blanket peatland development

The impact that grazing exerts on peatlands is hard to quantify. The effect of a given stocking regime is spatially heterogeneous across small scales (<1 ha^{-1}) according to antecedent vegetation coverage (Oom *et al*. 2008; Ross *et al*. 2016). Difficulties in quantifying the impact of grazing are compounded when using palaeoenvironmental techniques (Section 2.6.1). Qualitative inferences about grazing pressure may be made here because grazers are capable of altering vegetation communities (Section 2.6.2), hence their agency is theoretically detectable in plant macrofossil records.

Stocking intensity of sheep in Wales was consistently greatest post-AD 1962, following the introduction of the European Union Common Agricultural Policy (CAP), until late 21st Century agri-environment schemes encouraged a reduction in upland grazing intensity (Holden *et al.* 2007a). A key concern about the impact of grazing is its detrimental impact on *Calluna* coverage, this being one of the rationale behind the introduction of agri-environment schemes (Martin *et al.* 2013). *Calluna* is frequently recorded (i.e. the five-point abundance scale) in the upper 10cm of twenty-two of the recovered cores. Where *C. vulgaris* remains are present in the profile but not at the surface, there are intermittent above-ground remains in eight cores. The only cases where there are notable *Calluna* declines occur at Blaenycwm which relate to the impacts of afforestation (Section 8.2.3.3).

Therefore, there appears to be no systematic decline in *Calluna* in any of the cores that relates to the post-AD 1962 increases in stocking under the CAP. Grazing has occurred in the Welsh uplands since *ca.* 4,000 BC, as identified by an increase in pastoral pollen indicators (Bostock, 1980). Pollen studies have circumstantially suggested grazing has driven declines in *Calluna ca*. AD 1650 in Snowdonia (Woodbridge *et al*. 2012), *ca*. AD 1780 on the Migneint (Stevenson and Rhodes, 2000) and *ca*. AD 1800 at Mynydd Hiraethog (Grant, 2007). No declines in *Calluna* occurred at Plynlimmon (Mighall *et al*. 2013) and Cefn Mawr (Lascelles, 1995). Ellis and Tallis (2001) cautiously attribute a *ca*. BC 1,100 decline in *Ericaceae* macrofossils and pollen to be a result of increased grazing pressure. Berwyn pollen and macrofossil records show several non-synchronous phases of *Calluna* decline and expansion over the last *ca.* 2,000 years (Bostock, 1980). Many of these recorded declines in *Calluna*, where they occur, occur prior to the dateable period. Declines in *Calluna* may therefore relate to Medieval activity and 20th Century grazing regimes may not be related to *Calluna* declines. It is therefore unsurprising that data in the thesis do not support the notion that declines in *Calluna* occurred alongside the introduction of the CAP.

The abundance of *Calluna* remains typically increases in recently accumulated (*ca*. 20 years) peat relative to older macrofossil samples. Contemporary ecological studies show that declines in sheep stocking (as warranted under agri-environment agreements) promote an increase in *Calluna* coverage (Anderson and Radford, 1994; Grant *et al*. 1985; Rawes and Welch, 1969). The recent increased abundance of *Calluna* may therefore indicate a decline in grazing pressure. The decline in stocking may also explain the expansion of *S. cespitosus* in six cores as this taxon increased alongside *C. vulgaris* following stocking declines in Grant *et al*. (1985).

It is uncertain whether the apparent increase in *Calluna* and *S. cespitosus* abundance can be reliably attributed to stocking declines. Increased *Calluna* and *S. cespitosus* abundance may relate to preferable preservation of macrofossils in the acrotelm (Chambers *et al*. 2007a). Differential preservation to the catotelm may exaggerate changes in vegetation assemblages, a theme explored in Section 8.4.2. The expansion of *S. cespitosus* may also relate to changes in WTD related to drainage as it can be indicative of dry conditions (Pearsall, 1956; Stewart and Lance, 1991; van der Molen *et al*. 1994). Inferences about ditching using *S. cespitosus* abundance are complicated given its multi-modal distribution regarding WTD tolerance (Boatman and Armstrong, 1969; Chambers and Mauquoy, 1998; Dupont, 1986; Malmer, 1962; Økland, 1990). Uncertainty around the cause of the *S. cespitosus* increase highlights issues with ascribing causality to a single management intervention (see below).

8.2.4 Uncertainties in identifying a single driver of environmental change

The discussion thus far has highlighted the challenge in identifying a coherent impact of a given management practice as data are fragmentary and equivocal. Challenges relate to 1) the difficulty in detecting a systematic response to management (e.g. *Calluna* response to drainage), 2) identifying which land use caused an observed change (e.g. whether *S. cespitosus* increases occur due to drainage or grazing) and 3) identifying whether earlier land uses affect peatland response to more recent land uses (e.g. xeric plant communities being promoted by fire rather than drainage). Save for afforestation, management approaches have often been used in conjunction as fire and drainage have been used in herbivore management (Martin *et al*. 2013). It is therefore unsurprising that all ten drained cores exhibit macroscopic charcoal prior to drainage occurring. Simplifying peatland management into discrete land use categories (i.e. is it a ‘drained’ or a ‘grazed’ peatland) is therefore a flawed exercise as it is not appropriate to assume peatland states are the result of a single land use. This is especially pertinent where management records are sparse/lacking in detail.

The importance of understanding peatland management history as opposed to simply current management is stressed by Lindsay *et al*. (2014). Assuming blanket peatland condition to be a result of 20th Century management is a misstated approach to understanding land use impacts. Impacts of 19th and 20th Century land use were superimposed upon potentially more 'severe' legacy effects of management occurring hundreds or thousands of years earlier. The notion that 19th and 20th Century human impacts on peatlands have been less severe than earlier ones has implications for the Anthropocene debate. Attempts to identify the point where humans became the dominant driver of environmental change (Crutzen, 2002) typically endorse *ca*. AD 1950 as representative of the acceleration of human impacts on the environment (Dean *et al*. 2014; Steffen *et al*. 2015; Zalasiewicz *et al*. 2014). Evidence in this thesis and from previously published work suggests that Welsh blanket peatlands do not fit this horizon, as humans appear to have been the dominant driver of environmental change prior to AD 1950. A 'Blanket Peatland Anthropocene' should likely be defined during the Medieval period. This is in concordance with the 'anthropogenic soils' argument of Certini and Scalenghe (2011) who suggest the alteration of the pedosphere is the best indicator of the degree to which humans have altered the biosphere. Management interventions be they ditching, firing for grouse or post EU CAP induced stocking increases, were not applied to 'pristine' landscapes but were rather impressed upon landscapes showing the effects of previous human actions – the concept of palimpsest landscapes (*sensu* Bailey, 2007).

8.2.5 Peat Accumulation Types as indicators of peatland condition

A more holistic approach to monitoring long-term change is needed to understand the 'state' of Welsh blanket peatlands. Vegetation responds to both fast and slow drivers of peatland development. Grouping sites according to similarities in vegetation development may be useful for understanding the long-term impacts of management which have led sites to develop along similar trajectories. Such a rationale formed the premise for the development of Peat Accumulation Types (PATs) in Section 6.3.2. The PAT approach identifies trends in vegetation change and is useful for applied peatland management because vegetation cover is already used as an indicator of peatland condition (Section 2.2.4.1). There is of course variability within PATs, but the presented scheme does offer a more appropriate tool for understanding the effect of land use on peatland systems. It may also be used to investigate peatland carbon dynamics (Section 7.5). The possible environmental drivers of PATs are discussed below.

8.2.5.1 PAT A: *Sphagnum* dominated

These sites are typically characterised by a mixed *Sphagnum* assemblage alongside indicators of wetter peat conditions, such as *Eriophorum angustifolium* and *Erica tetralix* (Figure 6.29). These records are likely relatively undisturbed given none were recovered from drained areas, and there is a general lack of, or a complete absence of, charcoal. These sites may represent a relatively 'intact' peatland state (*sensu* Section 1.4.2) relative to other PATs, though they may have been intensely managed at some point beyond the 50cm core detection window.

8.2.5.2 PAT B: *Calluna*

These sites are typically characterised by a mixed *C. vulgaris*, *E. vaginatum* and *S.* s*. Acutifolia* assemblage for much of the 50cm record, plus intermittent to abundant xeric bryophytes (Figure 6.30). Many cores within this category have evidence of abundant charcoal and were recovered from drained sub-sites. Some cores are characterised by greater *Sphagnum* abundance earlier in the record, e.g. *S. papillosum* at MIG-3-REP-C and MIG-5-REP-C, but many cores within this category do not have a history of *Sphagnum* dominance. *Calluna* appears to have persisted through most records alongside *E. vaginatum* and frequent to abundant UOM. *Calluna* was often present prior to the onset of the dateable period, suggesting fire may have promoted its occurrence pre-AD 1850. As such, this PAT is likely indicative of a peatland that has been disturbed by prior management.

8.2.5.3 PAT C: Increasing *Sphagnum*

These sites are typically characterised by a shift from *Sphagnum*-poor *Eriophorum* assemblages, to *Sphagnum*-rich assemblages which persist to the present (Figure 6.31). Many species appear in both PATs A and C in their post-AD 1976 time slices. The difference in the development of these two PATs has important implications. First, it shows the difficulty in interpreting peatland condition using only surface vegetation cover. Second, it contradicts the conclusions of Bragg and Tallis (2001) who suggested a 'healthy *Sphagnum*' cover indicates a peatland subject to relatively little management, given many PAT C cores have evidence of abundant charcoal in the record prior to their shift to *Sphagnum*-rich cover.

PAT C sites may act as partial analogues to restored sites which aim to encourage the cover of *S. papillosum*, as this is often a restoration target species (Gillingham *et al*. 2016). *S. papillosum* is relatively tolerant of high N loading (Payne, 2014; van der Heijden *et al*. 2000) and burning and grazing (McCarroll *et al*. 2016a). *S.*

papillosum can establish itself following peat accumulation hiatuses (Swindles *et al*. 2016a) or on cutover surfaces (Hughes, P, D, M. pers. comm. 2017). Three of the four PAT C cores record slow apparent peat accumulation rates (PAR) prior to the expansion of *S. papillosum*. These slow apparent PARs may suggest peat accumulation stopped. *S. papillosum* may have re-initiated peat accumulation at these sites, or followed soon after another colonising *Sphagnum* species as recorded on degraded raised bogs by McMullen *et al*. (2004).

8.2.5.4 PAT D: *Molinia*

These sites are characterised by the expansion of *Molinia caerulea* (Figure 6.35). The dominance of *M. caerulea* at Bryniau Pica (BRP) (Section 4.3.1) is a recent feature of landscape development as the palaeoenvironmental record does not reflect its current abundance. *M. caerulea* arrival at BRP and MIG-6-REP-B and C ranges across a twenty-eight year period, with the average arrival date being *ca*. AD 1986. AD 1986 is more recent than most reported rises of *M. caerulea* on peatlands across the British Isles which typically suggest an AD 1900-1950 rise (Chambers *et al*. 1979, 1999, 2007a, b, 2013; Swindles *et al*. 2016a). The ca. AD 1986 *M. caerulea* rise is similar to Mynydd Llangatwg, Brecon Beacons (Chambers *et al*. 2007a). One may infer that evidence from BRP-2-M and BRP-2-REP-A support the notion that *M. caerulea* is not a completely ‘alien’ species to blanket bogs (*sensu* Chambers *et al*. 1999), as it has been a previous community constituent *ca.* AD 1850-1975 and prior to AD 1850 respectively. It may be that *M. caerulea* was locally present in previous phases of peat accumulation, but some force caused it to expand as the primary peat constituent (*sensu* Swindles *et al*. 2016a).

Determining the driver of the *M. caerulea* rise is of interest to peatland managers (Marrs *et al*. 2004). Efforts to reduce its prevalence, if indeed such efforts are wise (*sensu* Chambers *et al*. 1999), may require the initial driver to be addressed/counteracted. Frequent burning may promote *M. caerulea* given its fire resistance and ability to rapidly assimilate fresh nutrient inputs (Damblon, 1992; Grant *et al*. 1963; Taylor *et al*. 2001; Todd *et al*. 2000). There is no evidence of burning through the BRP-1-M, BRP-1-REP-A and BRP-2-M records, yet *M. caerulea* has risen and replaced *E. vaginatum* at these sampling points. Where charcoal is recorded, sustained *M. caerulea* increases do not occur during the firing, but after burning has apparently stopped for some time (>30 years in most cases). Assembled evidence does not therefore suggest that fire promotes the expansion of *M. caerulea*, or if it does, then the effect of firing is lagged. These findings concur with Chambers *et al*. (2007a, b) who found no association between charcoal and *M. caerulea* on South and Mid-Wales blanket peat.

Pollution from nitrogen (N) deposition can alter nutrient dynamics in bogs, thereby permitting the expansion of *M. caerulea* into previously nutrient poor environments. Reconstructed N deposition suggests a mean average of 20-30 kg N ha^{-1} yr^{-1} was deposited across BRP and the Migneint between AD 1900-2000, (Fowler *et al*. 2004). UK N deposition declined following the AD 1999 Gothenburg Protocol (Fowler *et al*. 2005), but total N deposition remains *ca.* 15 kg N $ha^{-1}yr^{-1}$ for BRP and MIG-6 at present (Air Pollution Information System, 2016). N loading over the last *ca.* 110 years has therefore exceeded the 5-10 kg N ha^{-1} yr^{-1} blanket bog critical loading limit (Bobbink and Roelofs, 1995). The N content of BRP samples are not expressly different to other sites. Peat N content at MIG-6 is higher relative to the N content of other Migneint samples. As such, N loading may have played a role in *M. caerulea* expansion at MIG-6. This explanation may constitute circular reasoning given MIG-6 sample N content may be higher than other Migneint sites because of the high N content of *M. caerulea* litter (Smith *et al*. 2014b; van Heerwaarden *et al*. 2005). Reconstructed N deposition was also not different at MIG-6 relative to other Migneint sites (Fowler *et al*. 2004).

The *M. caerulea* rise at BRP and MIG-6 has been spatially and temporally variable. A variety of forcings acting in unison (e.g. changes in grazing, nutrient loading) have likely driven the increase in *M. caerulea* over the last *ca.* forty years. The evidence here supports others in supposing the dominance of *M. caerulea* is not attributable to a single ‘trigger’ (Chambers *et al*. 2013; Swindles *et al*. 2016a).

8.2.5.5 PAT E: Afforested

These sites are characterised by a shift from mixed *C. vulgaris*, *E. vaginatum* and *Sphagnum* communities to one characterised by *P. sitchensis* and shade affiliated bryophytes. All cores were recovered from inside the Blaenycwm plantation and an explanation for their development was presented in Section 8.2.3.3.

8.2.6 Summary of the palaeoenvironmental evidence for environmental change

This section of the discussion has outlined the understanding that can be gained from the macrofossil records presented in this thesis. The notion of ascribing environmental changes to a sole management practice has been critiqued and a more appropriate framework for assessing peatland development (the PATs) was presented. The relevance of the PATs to understanding peatland carbon dynamics forms the focus of the next section of the discussion (see below).

8.3 Recent changes in blanket peatland carbon accumulation

Blanket peatlands are the most efficient British terrestrial habitat for carbon storage and their protection is essential to meet greenhouse gas reduction targets. The scientific community (Bain *et al.* 2011; Hughes *et al.* 2014; Lindsay, 2010) and Government bodies (DEFRA, 2009; Environmental Audit Committee, 2016; HM Government, 2011; Welsh Government 2013) recognise the need to improve the way in which peatlands are managed with respect to their carbon dynamics. Peatland carbon accumulation is a complex process affected by external/internal controls and non-linear drivers (Belyea and Baird, 2006; Lindsay, 2010). Section 8.2 highlighted the lack of a consistent peatland response to a given management regime and demonstrated that assessing the condition of peatlands according to plant communities provided a promising framework for examining the effect of land use. Vegetation cover is often used as an indicator of peatland condition (JNCC, 2006, 2011; Stewart *et al.* 2005; Williamson *et al.* 2017). As such, investigating carbon dynamics according to this metric provides the most effective applied use of data in this thesis.

This section of the discussion examines the difficulty in calculating reliable rates of long-term carbon accumulation where acrotelm material is sampled (Section 8.3.1), before assessing the carbon dynamics of PATs (Section 8.3.2) and the relevance of the data to the land management debate (Section 8.3.3). Two types of carbon accumulation are used in this part of the discussion when reporting carbon accumulation values, namely ‘carbon accumulation rate’ (CAR) and ‘classic recent apparent rate of carbon accumulation’ (RERCA) as defined in Section 2.2.3.

8.3.1 The relationship of PAT CAR and RERCA to long-term carbon accumulation

CAR and RERCA values that include material recovered from the acrotelm (e.g. cores in this thesis) do not account for carbon lost in decomposition as material passes from the acrotelm to the catotelm. CAR and RERCA may therefore overestimate long-term rates of carbon accumulation (i.e. carbon accumulated in the catotelm) because catotelm peat has undergone much more decomposition than acrotelm peat (Belyea and Clymo, 2001). Apparent rates of carbon accumulation based solely/primarily on peat recovered from the catotelm (e.g. long-term apparent rate of carbon accumulation (LORCA)) contain improved estimates of the amount of carbon a given peatland is sequestering over decadal-millennial timescales, and are comparable with contemporary carbon budgets (Parry, 2011). The acrotelm/catotelm transition (ACT) is therefore an important boundary when measuring carbon accumulation (Figure 7.10) as it is the catotelm which is the "*true site of peat accumulation*" (Clymo, 1996, page 209). Distinguishing the peatland into two functional layers is fraught with uncertainty as it obscures much ecohydrological variability (Morris *et al*. 2011), but does at least acknowledge the impact the ACT may have on carbon accumulation values reported in this thesis.

Following the Clymo (1992) acrotelm/catotelm model, bulk density (BD) is the most appropriate metric for identifying the ACT, in that peat BD is significantly greater in the catotelm relative to the acrotelm. Data in this thesis suggest there is a point in the assembled peatland profiles where a significant increase in BD occurs (Section 7.3), i.e. the matrix resembles Clymo's (1992) conceptual model of catotelm peat. Lindsay (2010) reviewed the basis for Clymo's model using published British peat data and determined that using BD data alone may produce uncertain ACT estimates, but he did recognise the value of an 'instantaneous' measure of the ACT using BD. As such, the use of BD to detect the ACT seems appropriate for assessing carbon dynamics in this thesis. Potential positions of the ACT were identified as being *ca.* 25-19cm depth, meaning peat would transfer to the catotelm after *ca*. 11-38 years' worth of accumulation (i.e. *ca.* AD 1976-2003 on an age/depth model).

Some identify the ACT as a depth within a peatland (e.g. Clymo, 1992; Evans *et al*. 1999) which is logical given the boundary can be identified as the deepest annual WTD (i.e. a 'depth' based measurement) (Charman 2002). Contemporary monitoring studies demonstrate that WTD is spatially and temporally variable (Laine *et al*. 2007; Lewis *et al*. 2012 Waddington *et al*. 2015), hence the usefulness of an ACT depth metric is uncertain. As such, subsequent discussions refer to the approximate age of peat at the ACT. The average age of the ACT was AD 1982 according to Davies' tests (Table 7.3) and is used in subsequent discussions. Using an average date is useful for making assessments of rates of carbon accumulation across the dataset assembled in this thesis, but an average date does mask variability in the age of peat at the ACT between cores.

Pre-AD 1982 CAR values likely reflect rates of carbon accumulation in the catotelm rather than the acrotelm, and therefore provide an improved estimate of long-term rates of carbon sequestration (*sensu* Blodau, 2002). *Ca.* AD 1950-1980 CAR remains much greater than pre-AD 1950 CAR (Figure 7.18) suggesting that CAR *ca*. AD 1950-1980 may still not approximate long-term carbon accumulation. Decomposition does not halt in the catotelm, it merely slows relative to the higher rates of acrotelm decomposition (Belyea, 2006). Peat sequestered into the catotelm does not therefore instantaneously represent the effect of long-term decay, hence the apparent rate of carbon accumulation at time of core recovery is likely greater in the upper catotelm relative to deeper peats (Clymo *et al*. 1998). It may take *ca.* thirty years of peat accumulation within the catotelm for CAR to represent long-term carbon accumulation (i.e. the difference between the AD 1982 ACT estimate and the *ca*. AD 1950 increase in CAR referred to above). The gradual decline in carbon accumulation below the ACT proposed here is akin to the concave LORCA model (Clymo *et al*. 1998; Figure 7.10). As such, pre-AD 1950 CAR data in this thesis is likely broadly representative of long term rates of carbon accumulation across the surveyed peatlands (Figure 8.1). RERCA was calculated between the AD 1850 and 1955 Spheroidal Carbonaceous Particle (SCP) horizons to detect a value more representative of long-term rates of carbon accumulation (Table 8.1) because acrotelm peat was included in RERCA values calculated in Section 7.5.

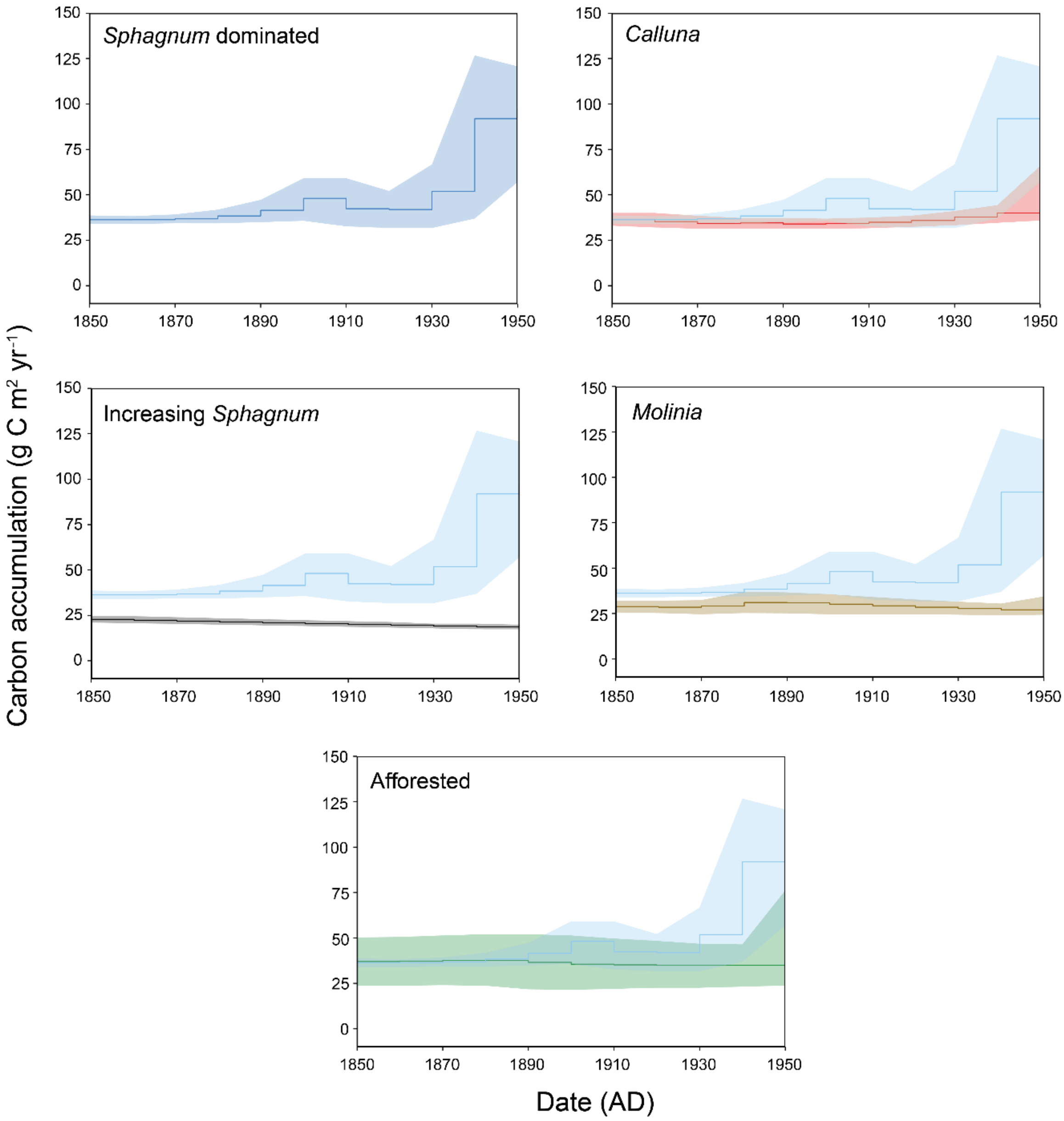


Figure 8.1: Average CAR values per 10-year time slice for each PAT established in Section 6.3.2. Only AD 1850-1950 is presented following reasoning in the text. Confidence bands represent standard error of the mean used to create each average value for each ten-year time-slice. '*Sphagnum*' dominated is superimposed on all categories to emphasize deviations from this potential reference state. MIG-2-REP-C was excluded from the calculation given reasoning established in Section 7.3.6.

Table 8.1: AD 1850-1955 RERCA statistics (g C m^2 yr^{-1}) for all PATs. Maximum and minimum values calculated using the effect of chronological errors on peat accumulation rates.

Statistic	PAT A	PAT B	PAT C	PAT D	PAT E
Mean	37	35	17	25	36
Median	37	30	18	22	20
S.D.	6	19	2	10	28
Max.	54	121	26	57	106
Min.	27	13	12	13	12

The attempt here to define when CAR becomes representative of long-term carbon accumulation is an improvement on previous palaeoenvironmental research. The effect of the ACT on CAR has previously been acknowledged with the boundary determined in semi-arbitrary manners (e.g. Garneau *et al*. 2014; Magnan and Garneau, 2014; Panait *et al*. 2017; Roulet *et al*. 2007) or work has not addressed the effect of the ACT when highlighting ‘recent’ CAR increases (e.g. Hapsari *et al*. 2017; Lourençato *et al*. 2017; Milecka *et al*. 2017).

8.3.2 The effect of vegetation on peatland carbon accumulation

The peat matrix is composed of vegetation that grew at, or near to, the surface of a peatland. PATs appear to display differing carbon dynamics when assessed using the CAR or RERCA method. RERCA values which include acrotelm peat (i.e. Section 7.5) are also discussed here as they may be used to determine differences in the relative rate of carbon accumulation between sites (*sensu* Parry, 2011).

Peatlands which have been predominantly characterised by *Sphagnum* over the dateable period (PAT A) exhibit the greatest CAR (Figure 8.1) and AD 1850-1955 RERCA values (Table 8.1). It is logical that *Sphagnum*-dominated sites accumulate carbon at the greatest rate, as *Sphagnum* is characterised by decay-resistant polymers which both inhibit its own decomposition (Dorrepaal *et al*. 2005; Pinsonneault *et al*. 2016; Verhoeven and Liefveld, 1997) and suppress decomposer activity in the surrounding substrate (Opelt *et al*. 2007; Turetsky *et al*. 2012). Hence, researchers view *Sphagnum*-dominated peat as having a greater capacity for carbon accumulation over longer timescales relative to peat formed from vascular plants (Fenner *et al*. 2004; Limpens *et al*. 2012; Malmer *et al*. 2003; Panait *et al*. 2017). It is interesting to note that *Sphagnum*-dominated sites still exhibit the greatest rates of carbon accumulation in the acrotelm over shorter time-scales. PAT A exhibits the greatest rates of post-AD 1850 (Table 8.2), 1955 (Table 8.3) and 1976 (Table 8.4) RERCA across all datasets. CAR values are also the greatest post-AD 1950 for PAT A (Figure 7.18). Where peat accumulation is rapid (i.e. the apparently rapid rate in the acrotelm), assemblages dominated by vascular plants may exhibit greater rates of CAR because they have high carbon content and BD values (Loisel *et al*. 2014; Mathijssen *et al*. 2017). Findings in this thesis suggest *Sphagnum*-dominated sites are accumulating carbon at the greatest rate of all PATs regardless of carbon accumulation value reported.

Table 8.2: Post-AD 1850 RERCA statistics (g C m^2 yr^{-1}) for all PATs. Mean, median and standard deviation PAT values were determined using mean RERCA values for each core. Maximum and minimum values calculated using the effect of chronological errors on peat accumulation rates.

Statistic	PAT A	PAT B	PAT C	PAT D	PAT E
Mean	101	66	52	41	75
Median	101	60	46	43	63
S.D.	23	22	20	14	26
Max.	138	108	95	77	121
Min.	73	29	32	20	41

Peatlands characterised by *Calluna* (PAT B) exhibit a reduced CAR relative to *Sphagnum*-dominated sites (Figure 8.1), though confidence bands for both PATs overlap and CAR appears similar pre-AD 1890. Assessments of peatland carbon stock which account for standing surface vegetation typically view *Calluna* as favourable for carbon accumulation (e.g. Alday *et al*. 2015), primarily because of the high carbon content of above-ground growth (Ward *et al*. 2007). Where the peat matrix is considered, as is the case here, researchers typically deem *Calluna* to have a negative association with carbon accumulation due to its promotion of peat desiccation (Armstrong *et al*. 2012; Dixon *et al*. 2015; Lindsay, 2010) and initiation of peat piping (Holden, 2005a; Holden *et al*. 2012). Thesis data support the findings that *Calluna* may be detrimental to long-term carbon sequestration. The variability of carbon dynamics values may be important in determining the effect of *Calluna* on carbon accumulation. Mean PAT B AD 1850-1955 RERCA is 35 g C m^2 yr^{-1} whilst the median value is 30 g C m^2 yr^{-1} (Table 8.1), suggesting some high rates may be affecting the mean. An example for is MIG-3-REP-C which exhibits AD 1850-1955 RERCA of 93 g C m^2 yr^{-1} which may be high because *S. papillosum* dominates during this period rather than *Calluna*. PAT B CAR also appears lower than PAT A CAR post-AD 1955 (Figure 7.18).

Table 8.3: Post-AD 1955 RERCA statistics (g C m^2 yr^{-1}) for all PATs. Maximum and minimum values calculated using the effect of chronological errors on peat accumulation rates.

Statistic	PAT A	PAT B	PAT C	PAT D	PAT E
Mean	154	114	114	74	144
Median	144	107	97	106	138
S.D.	66	36	52	32	37
Max.	274	200	207	162	222
Min.	77	56	67	34	95

Carbon dynamics of sites which have shifted from a degraded state to one characterised by *S. papillosum* cover (PAT C) change over time. PAT C exhibits the lowest AD 1850-1955 RERCA (17 g C m^2 yr^{-1}; Table 8.1) and CAR values for the AD 1850-1950 period (Figure 8.1) of all PATs. Sites within PAT C were typically characterised by a mixed assemblage of *E. vaginatum* and *C. vulgaris*, alongside evidence of plentiful charcoal, during this period. Some evidence of a pre-existing *S.* s. *Sphagnum* cover is present at LVW-1-REP-D and PLY-2-REP-B, though firing may have suppressed the peat accumulation rate and/or burnt directly into the peat itself (Section 8.2.5.3), reducing the apparent rate of carbon accumulation. Low pre-ACT carbon accumulation values of this PAT are indicative of either a very slow carbon accumulation rate, or potentially that peat accumulation at these sites may have ceased (see above). The SCP sampling resolution is too coarse to reliably infer a hiatus. In contrast, apparent carbon accumulation above the AD 1976 SCP marker suggests contemporary rates of carbon accumulation greater than PATs B, D and E (Figure 7.18; Table 8.4). These findings may suggest that the potential for long-term carbon accumulation at these sites has increased relative to their previous condition. As discussed for PAT A (see above), it is logical for sites dominated by *Sphagnum* to accumulate carbon at a rate greater than sites dominated by vascular plants.

Table 8.4: Post-AD 1976 RERCA statistics (g C m^2 yr^{-1}) for all PATs. Maximum and minimum values calculated using the effect of chronological errors on peat accumulation rates.

Statistic	**PAT A**	**PAT B**	**PAT C**	**PAT D**	**PAT E**
Mean	171	109	163	83	139
Median	176	104	133	76	111
S.D.	70	35	82	34	48
Max.	276	194	308	168	232
Min.	89	59	96	39	96

There is concern over the expansion of *M. caerulea* on British blanket peatlands from a biodiversity standpoint (Chambers *et al*. 2007a), though the effect of *M. caerulea* on long-term carbon dynamics is largely uncertain. PAT D exhibits the second lowest rate of AD 1850-1955 RERCA (Table 8.1) and CAR (Figure 8.1), and the lowest rate of post-AD 1850 (Table 8.2), 1955 (Table 8.3) and 1976 (Table 8.4) RERCA. The standard deviation (SD) for PAT D RERCA is consistently lower than for other PATs when acrotelm material is included (Table 8.2; Table 8.3; Table 8.4), suggesting a consistent signal of low rates of apparent carbon accumulation in the acrotelm for *Molinia*-dominated sites. Sites currently characterised by *M. caerulea* therefore appear to be least favourable for carbon accumulation. The negative implications for carbon accumulation associated with the presence of *M. caerulea* on peatlands have previously been raised (Gogo *et al*. 2011) but never quantified over the timescales presented in this thesis.

PAT D peatlands have been accumulating carbon at an apparently reduced rate relative to other PATs prior to the expansion of *Molinia*, which occurred primarily post-AD 1980 (Section 8.2.5.4). The decomposability of *M. caerulea* litter and root exudation may stimulate microbial community activity (Leroy *et al*. 2017) and thus the decomposition of peat which accumulated prior to the expansion of *Molinia* (Ritson *et al*. 2016). Walker *et al*. (2016) recently showed that graminoid organic inputs could promote the release of carbon accumulated into a peatland several thousand years earlier, but they did not specify the taxa included within their 'graminoid' class. Dieleman *et al*. (2017) presented similar evidence in that graminoid-dominated peats exhibit greater carbon lability and microbial activity than bare or *Sphagnum*-dominated peat. The thesis data suggest that *M. caerulea* expansion onto peatlands has had a similar effect, with its presence both promoting the release of stored carbon (Table 8.1) and limiting the capacity for future carbon sequestration (Table 8.4).

Afforested peatlands (PAT E) have accumulated carbon at a rate broadly similar to *Sphagnum*-dominated sites prior to the ACT (Figure 8.1; Table 8.1). As with PAT B, there is variation within the afforested PAT. Mean AD 1850-1955 RERCA is 36 g C m^2 yr^{-1} whilst the median is 20 g C m^2 yr^{-1}. SD is the highest for all PATs (28 g C m^2 yr^{-1}) which reflects the range of values reported for AD 1850-1955 RERCA (12-106 g C m^2 yr^{-1}) (Table 8.1). MIG-2B-REP-C appears to have accumulated carbon at a greater rate prior to AD 1955 than other PAT E cores, and when excluded, mean AD 1850-1955 RERCA declines to 25 g C m^2 yr^{-1} (Table 8.5). MIG-2B-REP-C was a rapidly accumulating *R. lanuginosum* hummock prior to conversion to forestry, and oxidation appears to not have affected the carbon it had sequestered. Post-AD 1955 afforested CAR remains high with confidence intervals overlapping with PAT A (Figure 7.18). It is questionable whether the traditional acrotelm/catotelm model (Figure 7.10) remains applicable to an afforested peatland. RERCA values, which theoretically include acrotelm peat, suggest PAT E has accumulated less carbon than *Sphagnum*-dominated sites post-AD 1850 (Table 8.2), 1955 (Table 8.3) and 1976 (Table 8.4).

Table 8.5: AD 1850-1955 RERCA statistics (g C m^2 yr^{-1}) for PAT E. Maximum and minimum values calculated using the effect of chronological errors on peat accumulation rates.

Statistic	PAT E	PAT E (excl. MIG-2B-REP-C)
Mean	36	25
Median	20	19
S.D.	28	14
Max.	106	46
Min.	12	15

The net impact of forestry on peatland carbon dynamics depends upon whether oxidative losses from the peat are offset by carbon accumulated within trees (Section 2.5.2). Modelled carbon accumulation in a *P. sitchensis* plantation was used to estimate carbon accumulating in trees at Blaenycwm (Vanguelova *et al.* 2012). The cumulative value for biomass (105 t C ha^{-1}) is calculated over a 62-year felling cycle, so the average accumulation per year (169 g C m^2 yr^{-1}) negates within-life cycle variability (Section 2.5.2). The impact of forestry on the Blaenycwm cores was assessed according to the 47-year age of the plantation as well as the 37-years' worth of growth following the ten-year transition period (Section 8.2.3.3). Estimated carbon losses/reduced potential for carbon accumulation were calculated according to the difference between PAT A and E RERCA values above each horizon. Assuming the Vanguelova *et al.* (2012) estimates reflect local site conditions, forestry appears to have had a net positive effect on the carbon balance, because carbon accumulated in tree biomass offsets carbon presumed 'lost' from the matrix as ongoing carbon accumulation is reduced (Table 8.6).

Table 8.6: Estimated net effect of afforestation on peatland carbon stock above each SCP horizon.

Horizon (AD)	'Peat matrix' carbon lost per year (g C m^2 yr^{-1})	'Peat matrix' carbon lost post horizon (g C m^2)	Net impact of forestry on carbon balance (g C m^2)	
			47 years growth	37 years growth
1850	26	4264	3696	2002
1955	66	3894	4066	2372
1976	84	3192	4768	3074

8.3.3 Identifying the impact of land management on carbon dynamics

A theme of this discussion has been to highlight the difficulties in identifying a consistent peatland response to a given land management regime. The exception is perhaps forestry, where it is much easier to spatially constrain the limits of the activity, date its onset and assess its impact on carbon dynamics. Whilst there is some uncertainty in the values of Vanguelova *et al*. (2012) used to explore the effect of forestry, it is of interest that afforestation at Blaenycwm may have had a net positive impact on the carbon balance of the site (Table 8.6). Identifying the impact of the other three land use activities (burning, drainage and grazing) in palaeoenvironmental records is harder (Section 8.2.4), hence, it is not appropriate to use data here to provide a 'standard' carbon accumulation value for these three land uses. The developed PATs may be used to explore the effect of these land uses on carbon accumulation. There is a 'general expression' of land management in the macrofossil records of some PATs, hence, the undifferentiated effect of burning, drainage and grazing on peatland carbon dynamics is discussed below.

Sphagnum-dominated sites appear to have accumulated carbon at the greatest rate over the dateable period compared with other PATs (Section 8.3.2). These sites are likely analogous to a relatively intact peatland, given limited/absent burning through the dateable period and the lack of obvious drainage near the core locations. All of these cores were recovered from sites which have a history of grazing (Roberts, A. pers. comm. 2017; Walker, M. pers. Comm. 2015), and at times this grazing has likely exceeded the ≤ 0.5 ewes ha^{-1} (Roberts, A. pers.

comm. 2014) that is recommended to avoid degradation of blanket peatland habitats (DEFRA, 2008; Shaw *et al*. 1996). These data therefore provide yet more evidence that the notion of a 'standard' stocking level for all blanket peatlands to avoid degradation is simply not practical (Section 2.6.1). Sites which are now *S. papillosum*-rich have also recovered from apparently low rates of carbon accumulation where burning has occurred (Section 8.2.5.3). Stopping the use of fire may have increased the ability of these sites to sequester carbon into the future (Table 8.4), but how carbon accumulation in the acrotelm relates to long-term accumulation is uncertain as these sites appear to still be exhibiting patterns of recovery.

Where *Calluna* is present on a blanket peatland it seems likely that it is a result of some form of disturbance (e.g. burning and/or drainage) (Section 8.2.5.2). Sites dominated by *Calluna* sequester *ca*. 2-7 g C m^2 yr^{-1} less to the catotelm than *Sphagnum* dominated sites (Table 8.1) and accumulate carbon in the acrotelm at a much-reduced rate as well (Figure 7.18). In cases (e.g. MIG-3-REP-C) it appears that drainage and the subsequent expansion of *Calluna* has had a negative impact on carbon accumulation at the sampling point (Section 8.2.3.2). An expansion of *Calluna* that can be clearly linked to a change in land management (e.g. drain installation or grouse estate management) could be quantified in terms of its impact on carbon dynamics using the values reported above.

While it is not possible to reliably identify the management driver of *M. caerulea* expansion, it does seem likely that *Molinia* has become established on peatlands due to some interference by human activity. This expansion is certainly of concern for peatland managers who seek to promote carbon accumulation (Section 8.4.1).

8.4 The relevance of this investigation to the land use debate

The effect of land use on blanket peatland carbon accumulation has long-been poorly understood, meaning that carbon accumulation could not be reliably incorporated as a factor (e.g. aesthetic, economic, social) in a land management decision making process. Linking changes to poorly defined management categories (e.g. burning, grazing) has led to an equivocal evidence base and an often ill-informed discussion (Davies *et al*. 2016a, c; IUCN, 2017b). This thesis has shown that partitioning peatlands into discrete management types is not an appropriate way to approach carbon-conscious peatland management. This sub-section considers the ways in which data and ideas generated in this thesis may aid decision making about upland land use (Section 8.4.1) and identifies limitations to consider if findings are to be applied in a policy-relevant context (Section 8.4.2).

8.4.1 Applicability of research findings to decision making in peatland management

Blanket peatlands are managed for a variety of reasons. There is an increasing focus on the benefits of managing peatlands specifically for their carbon sequestration function (Bain *et al.* 2011; Environmental Audit Committee, 2016, 2017), given concerns over contemporary climatic change. It is beyond the scope of this thesis to fit presented carbon accumulation values into a Payment for Ecosystem Services economic framework (Section 2.2.1.3). Nevertheless, research findings still contribute to the land management debate and can act as a precursor to the integration of carbon accumulation as a service into the Glastir scheme. PATs demonstrate that blanket peatlands across Mid- and North-Wales have developed differently, and these differences can (in some cases) be related to management interventions. Each PAT, save for the afforested PAT (E), is recorded at more than one study site. Though classification into PATs obscures some variability in peatland development, the PATs may be broadly applicable across differing blanket peatland sites through Mid- and North-Wales. This is a novel palaeoecological approach. Prior work has focussed on reconstructions which provide management targets at a site-specific level (e.g. McCarroll, 2014).

Sites with a rich *Sphagnum* cover through the dateable period appear to be favourable for carbon accumulation (PAT A). *S. papillosum* also appears to be effective at sequestering carbon in cases where it expands following a previous degraded peatland state where burning was frequent (PAT C). The impact of fire on *Sphagnum* is often the most contentious point in the debate surrounding rotational burning (Brown *et al.* 2016; Davies *et al.* 2016a; IUCN, 2017b). If PAT A does represent a relatively ‘intact’ blanket peatland (Section 8.2.5.1) it is interesting that these sampling points are characterised by little or no charcoal. *S. papillosum* apparently persisted through burning prior to its expansion in some of the PAT C cores. It appears, therefore, that whilst this taxon can tolerate burning, its cover is greatest where this stressor is removed. These findings suggest *S. papillosum* is favourable for carbon accumulation and that this taxon should be encouraged where *Sphagnum* cover is currently poor, particularly as it is relatively resistant to contemporary pressures such as N deposition (McCarroll, 2014; Payne, 2014). *S. papillosum* may tolerate rotational burning if management for grouse is required as part of a management plan. Burning should, however, cease if a manager is seeking to maximise the potential for carbon accumulation.

C. vulgaris is a taxon affiliated to drier peatland environments and its dominance at a site can be considered indicative of a degraded blanket peatland (Lindsay, 2010). Swaths of *Calluna* coverage are considered aesthetically pleasing to some upland stakeholders (Allen *et al*. 2013; Lee *et al*. 2013; Marrs *et al*. 2007) as heather is considered a long-standing component of the uplands (Simmons, 1990; Stevenson and Thompson, 1993). The taxon is also of international importance under the UK Biodiversity Action Plan given its relative rarity overseas (JNCC, 2008). Over the dateable period, sites characterised by *Calluna* (PAT B) have accumulated less carbon than sites where it is predominately absent over the dateable period. Seventeen of the twenty-one cores within PAT B have plentiful charcoal in their record. *Calluna* is also present at all drained sites. As such, *C. vulgaris* appears to indicate a disturbed peatland.

The use of *Calluna* as a ‘positive indicator’ species for blanket bog within the Common Standards Monitoring (CSM) framework (based on the National Vegetation Classification (NVC)) should be re-assessed. The monitoring term ‘positive indicator’ does not imply that *Calluna* is indicative of a ‘good’ bog, but rather that it can be used to identify a blanket bog as it is a ‘typical’ species found within this habitat (JNCC, 2008). Data here support the view that dominance of *Calluna* at a site is unlikely to be representative of an intact peatland (see above), but rather represents a shift from a previous mosaic of upland plant types due to land use (Davies and Bunting, 2010). The use of *Calluna* as a positive indicator of blanket bog may therefore be hampering carbon-conscious management as an observer can record heather as ‘typical’ of the habitat, hence its removal is not warranted as the taxon is not viewed as indicative of a degraded site. This will be the case as long as *Calluna* remains part of blanket bog NVC communities (e.g. Bruneau and Johnson, 2014; O’Brien *et al*. 2007). The limitations of the NVC system for understanding long-term peatland development have previously been highlighted (Chambers and Daniell, 2011), with data here supporting a re-assessment of *Calluna* within the NVC system.

The timing of these *Calluna* shifts must be determined to greater detail as *Calluna* establishment in most cases pre-dates the onset of the dateable period.

As such, the disturbance which caused the dominance of *Calluna* at many sites pre-dates late 19th and 20th Century land use. Other work suggests *Calluna* has been present in the Welsh uplands for thousands of years (Bostock, 1980; Mighall *et al*. 2013). Data across a suite of northern England sites suggests the onset of *Calluna* dominance in many cases occurred during the 19th Century (Chambers *et al*. 2017). The relationship between management and *Calluna* rises may therefore be spatially variable across the British Isles. Appropriate *Calluna* management should consider that its dominance may 1) not be indicative of an intact blanket peatland and 2) not be favourable for long-term carbon accumulation.

The presence/dominance of *M. caerulea* on a blanket peatland should be of concern if managing the system from a carbon-conscious standpoint. The impact of *Molinia* dominance on long-term blanket peatland carbon dynamics has been largely uncertain to date (AD 2017). The thesis data clarify this uncertainty. The expansion of *M. caerulea* onto Welsh blanket peats may reduce long-term rates of carbon accumulation by *ca*. 12 g C m^2 yr^{-1} (Table 8.1). This value is likely a conservative estimate and supports the observations elsewhere that *Molinia* root exudates promote the decomposition of previously sequestered carbon (Section 8.3.2). Low rates of acrotelm carbon accumulation suggest the long-term carbon-sequestering capacity of PAT D sites has been reduced, hence their long-term rate of carbon accumulation may be lower than the estimate in Table 8.1. The removal of *M. caerulea* should be a priority for carbon-conscious management.

New forestry plantations are not permitted on UK blanket peatlands because of their detrimental impact on peatland functioning (Reed *et al*. 2009; Worrall *et al*. 2011b). Established plantations are, however, allowed to be felled and re-stocked to help meet domestic requirements for timber supply (Forestry Commission Scotland, 2015; Forestry Commission Wales, 2011). Data presented here suggest that forestry establishment (PAT E) causes a reduction in apparent carbon accumulation within the peat matrix, but afforestation may offer a net gain for system carbon dynamics when carbon sequestered within trees is accounted for (Table 8.6). It is uncertain how representative these findings are given data are noisy. Furthermore, trees will not offset oxidative losses from peat in perpetuity. Productivity of the first crop of trees may not be matched by subsequent rotations. Any decline in productivity would affect the estimates of tree carbon biomass of Vanguelova *et al*. (2012) (Table 8.6), and would alter estimates of the net impacts of oxidative losses across multiple felling and planting cycles.

Thesis data suggest the fixation of the peatland management debate on the impact of 19th and 20th Century land use is misplaced. These relatively recent changes have imprinted upon the effects of land use which occurred over the previous hundreds to thousands of years (Section 8.2.3). Charcoal records suggest burning intensity/frequency declined during the *ca*. AD 1800-1850 period when upland burning intensity is typically viewed to have increased for grouse and sheep management (Hester and Sydes, 1992; Lee *et al*. 2013). Future work should consider that pre-historic and Medieval impacts were more severe relative to current land use and that the impacts of these actions may affect the response of peatlands to more recent management interventions.

Determining a pre-human peatland 'state' is at least theoretically important for restoration. The typical concept of restoration is to return a system to some idealized pre-human disturbance state (Barnosky *et al*. 2017; Davis and Wilkinson, 2004), assuming the peatland would remain in that state in the absence of human activity (McCarroll, 2014). The Society for Ecological Restoration uses a more appropriate restoration definition "*the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed*" (SER, 2004, page 2). PATs here may provide more realistic restoration targets by demonstrating favourable target states (i.e. PAT A). Future research should explore the development of blanket peatlands over a longer timeframe to determine in which archaeological/historical period humans exhibited the greatest effect on peatland vegetation and carbon dynamics. These investigations may be useful in determining whether detecting a blanket peatland 'baseline' condition is possible or even appropriate for ongoing habitat management.

8.4.2 Limitations in using palaeoenvironmental evidence

The robustness of any scientifically determined land management intervention is inherently reliant on the quality of the data used to formulate guidance/advice (Jeffers *et al*. 2015). Whilst uncertainties associated with palaeoenvironmental data do not preclude its use, these uncertainties must be addressed here.

Each method used to reconstruct peat accumulation rates has uncertainties associated with their application (Sections 3.4.3 and 3.4.4). These uncertainties affect conclusions regarding carbon dynamics given peat accumulation rate is a strong control on carbon accumulation rate (Section 3.4.2). Somewhat regardless of specific uncertainties in the dating techniques, there is the question as to whether results in this thesis provide realistic estimates of carbon accumulation.

Analyses within Section 8.3.1 produced carbon accumulation rates analogous to long-term apparent rate of carbon accumulation (LORCA) (Figure 8.1; Table 8.1). LORCA accounts for the impact of decay within the catotelm. Lindsay (2010) collated LORCA values for blanket bogs across the British Isles to produce a range of 30-70 g C m^2 yr^{-1}. Mean AD 1850-1955 RERCA for PATs A, B and E fall within the lower bounds of this range (Table 8.1). AD 1850-1955 RERCA for PATs C and D are below this range, but these PATs have been affected by degradation and *M. caerulea* encroachment (Section 8.3.2) so it is unsurprising that long-carbon accumulation is lower.

PATs B and E exhibit maximum and minimum AD 1850-1955 RERCA values outside the 30-70 g C m^2 yr^{-1} range (Table 8.1). LORCA values negate changes in peat accumulation rate (Section 2.2.3). CAR profiles from a range of peatlands demonstrate periods of high (>100 g C m^2 yr^{-1}) and low CAR (<20 g C 20 m^2 yr^{-1}) within the same core (Hapsari *et al*. 2017; Hughes *et al*. 2013; Lourençato *et al*. 2017; Peros *et al*. 2016; Silva-Sánchez *et al*. 2016). These high and low CAR phases may last for several hundred years and may be masked if a single LORCA value was reported for the core. None of these studies are from a British blanket peatland. Their use is necessitated to illustrate potential discrepancies between CAR and LORCA given only two studies have published British blanket peatland CAR profiles to date (AD 2017) using reliable BD and carbon content measurements.

Anderson (2002) presented an average LORCA value of 21 g C m^2 yr^{-1} for three Scottish blanket peats. Masked within this value were periods of CAR ranging from 4-73 g C m^2 yr^{-1}. Ratcliffe *et al*. (2017) presented an average LORCA value for three Scottish blanket peats (16 g C m^2 yr^{-1}) which masked periods of CAR ranging from *ca*. 5-30 g C m^2 yr^{-1}. Maximum AD 1850-1955 RERCA values in PATs B and E may therefore be recording increased CAR over a *ca.* 100-year period which may be masked should LORCA be calculated. That mean AD 1850-1955 RERCA for most PATs agree with the range of Lindsay (2010) suggests presented data are reflective of long-term carbon accumulation rates.

The identification of land use impacts and creation of PATs has relied on the assumption that macrofossil assemblages are adequate representations of peat-forming communities. Many cores exhibited phases of UOM ≥3 on the five-point abundance scale. Abundant UOM inhibits understanding of phytosociological relationships of plant communities growing at these times because species specific information has been lost. The absence of poorly preserved taxa (e.g.

lichens and/or liverworts) does not mean that they may not be useful for assessing peatland conditions. The applicability of PATs in relation to land management also implies that the relation between vegetation, land use and carbon dynamics is constant over time. Identified PATs may therefore be primarily applicable to contemporary peatland management as they were determined over the dateable period. PATs may not be similar to blanket peatland vegetation communities which occurred prior to the industrial revolution and associated changes in atmospheric pollutant deposition.

It is uncertain whether some relatively recent changes in peatland plant communities (*ca.* post-AD 1982) are more indicative of the effect of the ACT as opposed to a true change in peatland vegetation assemblages. A greater ‘number’ of taxa are typically recorded in more recent samples (*ca*. post-AD 1982) which often coincide with a decline in UOM (Appendix A). One could interpret that the ACT is in part responsible for some changes in macrofossil abundance. The ACT cannot account for the majority of observed changes in recent decades. No taxon is recorded solely in the recent decades which reflect acrotelm peat (post-AD 1982). Notable changes in recent decades (e.g. the expansion of *M. caerulea*) are consistently recorded across many cores and taxa which characterise these changes are recorded in samples likely recovered from catotelm peat. Any ACT effect on abundance would have no effect on statistical analyses given these were calculated using presence/absence data.

The logic that underpins comparisons between differing peatlands should also be qualified (*sensu* Davies *et al*. 2016a). It may not be appropriate to make comparisons between the carbon dynamics of some of the sites examined in this thesis. A *Sphagnum* rich bog from North-Wales (most PAT A cores) may present more favourable conditions for carbon accumulation than a Mid-Wales bog, regardless of the latter being dominated by *M. caerulea* (PAT D). PAT A may therefore not provide a ‘true’ counterfactual to assess the impact of land use on carbon dynamics. The potential for peatlands to develop according to specific PATs may be constrained by climate, basin morphometric characteristics and the effects of human activities not detectable by methods employed in this study. As such, comparisons between PATs (Table 8.1) may provide overly positive/negative views of the effect of differing PATs and land use on carbon accumulation. The detection of all PATs, save for the afforested PAT, at multiple study sites suggests differing blanket peatlands may exhibit some similarities in behaviour.

A final uncertainty as to the usefulness of the PATs is the somewhat conspicuous lack of insight as to how PATs may be affected by projected 21st Century climatic change. Blanket bogs are amongst the most vulnerable UK soil types to changes in moisture availability given their reliance on atmospheric moisture inputs (Ostle *et al*. 2009). There is therefore concern that UK blanket peatland carbon stores may be de-stabilized under climate change (Cole *et al*. 2002). Bioclimatic-envelope models establish the geographical limits where a habitat can exist. Increased temperatures and changing rainfall patterns may mean fewer areas provide the required moisture for blanket bog to continue accumulating peat (Clark *et al*. 2010; Gallego-Sala and Prentice, 2012; Gallego-Sala *et al*. 2010). Bioclimatic envelope models assume that contemporary blanket peatland distributions accurately represent their ecological limits. Work by Heinemeyer (2017, unpublished data) suggests blanket peatlands may be able to persist under warmer/drier climate regimes than is currently acknowledged. The current absence of blanket peatlands from warmer/drier regions of the British Isles may be due to human interference and degradation. Bioclimatic-envelope models may therefore be over-estimating the sensitivity of the habit to projected climate change. Modelling peatland carbon dynamics in a changing climate cannot be addressed here. The presented PATs may though offer differing peatland 'states' (*sensu* Page and Baird, 2016) which can be used to test the resilience/vulnerability of blanket peatlands to contemporary climate change.

Chapter 9 Conclusions

9.1 Introduction

Blanket peatlands are of international importance and offer many benefits to society. Their value has not always been acknowledged, which has led to the progressive degradation of many sites. Anticipated changes to the UK's agricultural system, necessitated by the vote to leave the European Union, have cast doubt over how blanket peatlands will be safeguarded going forward. Evidence-based land management approaches offer the best chance to ensure these systems are managed appropriately in the coming decades. The applied relevance of this thesis to peatland management decision making was discussed in Section 8.4. This chapter presents the main scientific findings of this thesis, with commentary provided where a specific finding has land use policy relevance.

The primary aim of this thesis was to characterise the impact of land management on blanket peatland carbon accumulation. Conclusions from the literature review were used to develop three research questions to address the principle thesis aim. This chapter commences by using the primary findings of this thesis to answer these research questions (Section 9.2). Each research question is answered in turn, alongside a description of evidence used to answer the question and a commentary on how these findings relate to the thesis aim. A set of technical findings which were used to underpin these conclusions are next discussed (Section 9.3). Recommendations for future research are next presented (Section 9.4), before a final comment (Section 9.5) is offered in the hope that this thesis may contribute to wider progress in the peatland management debate.

9.2 Principle findings of this thesis

9.2.1 Research question one: Do differing land use practices affect blanket peatland carbon accumulation in a characteristic manner?

This research question was developed to identify whether the land management practices of burning, drainage, forestry and grazing produced characteristic rates of carbon accumulation (Section 2.8.1). It was concluded that save for forestry, the examined land use practices do not affect blanket peatland carbon accumulation in a characteristic manner. Research findings which led to this conclusion are detailed below:

- Burning, drainage and grazing management practices are not associated with a characteristic rate of carbon accumulation. Cores were recovered from peatlands which were believed to be managed according to these three discrete land use types. Differences in carbon accumulation rate between cores recovered close together, under the presumed same management history, suggests that the practices of burning, drainage and grazing do not affect the carbon dynamics of Welsh blanket peatlands in a consistent manner. An example is the cores MIG-3-REP-B and C which were recovered from the same drained site <10m apart, but exhibited carbon accumulation rates of *ca*. 25 g C m^2 yr^{-1} and *ca*. 75-350 g C m^2 yr^{-1} respectively over the AD 1850-1976 period.
- Of the examined land uses, forestry is the only practice which one could argue produces a characteristic impact on carbon accumulation. This was because five of the six forestry cores exhibited an AD 1850-1955 carbon accumulation rate of *ca*. 25-35 g C m^2 yr^{-1}. Variability is still exhibited within this ‘management category’ for AD 1850-1955 recent apparent rate of carbon accumulation where the mean was 36 g C m^2 yr^{-1}, but the median was 20 g C m^2 yr^{-1}. An estimate of the net effect of forestry suggested that the plantation may have increased the net carbon stock of the site (*ca*. >2300 g C m^2) as carbon accumulating in the trees offset potential oxidative losses in the peat.
- A consistent land use impact on carbon accumulation could not be recorded in many cases because it is not possible to reliably identify the effect of a single land use in palaeoenvironmental records. This finding is expanded upon in Section 9.2.2 below.

This research question is important for addressing the thesis aim because it shows that land use types do not affect carbon accumulation in a consistent manner, even if one knew a given peatland had only ever been subject to a single land use. This research question is relevant to peatland management as it shows that any attempt to manage peatlands from a ‘carbon conscious’ standpoint must acknowledge site specific variability in response to management, and not seek to implement a ‘standard’ management regime that is assumed to be beneficial for carbon accumulation (see Section 9.4).

9.2.2 Research question two: Is it appropriate to treat peatland management regimes as discrete entities, or is it more appropriate to recognise that multiple environmental drivers are responsible for peatland development?

This research question was developed to explore whether it was appropriate for previous peatland research to partition peatlands into those managed solely according to four discrete land use categories (Section 2.8.2). It was concluded that it is not appropriate to treat blanket peatland management regimes as discrete entities as it was not possible to reliably identify peatlands which have been managed solely according to the land uses of burning, drainage, forestry or grazing. It is therefore more appropriate to recognise that multiple environmental drivers are responsible for peatland development. Research findings which led to this conclusion are detailed below:

- The impacts of burning, drainage and grazing on blanket peatland development are hard to identify in isolation. This finding was based on there being little consistency in plant macrofossil records across cores recovered from the same presumed management category. An example is the MIG-1 sub-site where all cores were recovered from a ‘grazed’ area. No drains were present and there is no charcoal in any MIG-1 record hence this site may only have been managed by grazing over the periods which these cores have accumulated. Some forcing has though caused MIG-1-REP-B to be dominated by xeric taxa, notably different to the other MIG-1 cores.
- Fire has affected peat development at most sampling points studied in this thesis. This finding was based on macroscopic charcoal being recorded in thirty-eight of the fifty cores recovered in this study. The effect of fire was likely mixed with the impacts of grazing and/or drainage at many sampling points. An example is cores recovered from MIG-4 and 5. These cores were recovered from amongst drains, have evidence of plentiful charcoal across the 50cm coring window and were from within areas of the Migneint which have been managed for grazing. Drier favouring taxa (e.g. *Calluna vulgaris*) often dominate sites with a history of firing, suggesting the use of fire promotes the incidence of this taxon. Shifts to drier plant communities at many drained sites often occurred prior to drainage occurring (e.g. MIG-4 sampling points), suggesting it may be hard to distinguish the impact of 20th Century land use from the impacts of land use in earlier periods (see Section 9.2.3 below).

- The impact of forestry on blanket peatland development is possible to identify, but antecedent conditions may affect peatland response to the plantation. This finding was based on all forestry cores exhibiting notably different vegetation community development (dominance of *Picea sitchensis* and *Rhytidiadelphus loreus*) following conversion to forestry compared with the rest of the cores recovered from the Migneint. Some forestry cores exhibited macroscopic charcoal prior to the site being converted to a plantation and *Sphagnum* also persisted at some sampling points within the forest. Whilst the establishment of a plantation can therefore be identified, antecedent land use may affect peatland response to the creation of the plantation.
- The impact of land use on blanket peatland development could be identified by categorising peatlands according to those which exhibited similar patterns of vegetation change over time. This finding was based on the detection of five Peat Accumulation Types (PATs), these being peatlands which exhibited similar vegetation community composition in the time periods of pre-AD 1850, AD 1850-1955, AD 1955-1976 and post-AD 1976. As vegetation is known to respond to the impacts of land use, differing vegetation patterns exhibited between the PATs were interpreted as being driven primarily by differing management regimes. An example is PAT B: *Calluna vulgaris*. These sites were characterised by the expansion/persistent presence of the shrub and often exhibited plentiful records of macroscopic charcoal. The expansion of *Calluna* at these sites was therefore interpreted as these peatlands expressing the impact of land use. The PATs have an applied relevance to the peatland land use debate because vegetation is already used in peatland monitoring, hence if contemporary PAT assemblages (i.e. post-AD 1976 vegetation communities) are identified on Welsh blanket peatlands then land managers may use them to identify the degree to which land management has disturbed the site.
- The developed PATs provide a framework to identify the impact of land use on carbon accumulation. This finding was based on the five PATs exhibiting distinct carbon accumulation rates and/or recent apparent rates of carbon accumulation. As differing PATs were interpreted as having developed under differing management regimes (see above), inferences were made about the effect of land use on carbon dynamics where management had forced a site to develop according to a specific PAT. An

example is land use which has caused an expansion of *Calluna vulgaris* (PAT B) has reduced carbon accumulation by *ca.* 2-7 g C m^2 yr^{-1} relative to a *Sphagnum* dominated state (PAT A). Likewise, the expansion of *Molinia caerulea* on Welsh blanket peatlands may have reduced carbon accumulation by *ca.* 12 g C m^2 yr^{-1}. PATs can therefore be used to identify the impact of land use on carbon dynamics in a more appropriate manner than simplistic approaches of determining the impact of 'grazing' on carbon accumulation relative to 'burning'.

This research question is important for addressing the thesis aim because it shows that previous attempts by policy makers/land mangers/researchers to categorise sites into discrete management types was an inappropriate method to examine the impacts of land use. Discussions which attribute the current state of peatlands to be a result of a specific and overly defined land use impact should therefore be avoided. Anyone seeking to characterise the impact of land management on blanket peatland carbon accumulation must recognise that it is unlikely that a given management regime would ever cause a consistent response. Instead, the PATs presented here show that the general disturbance caused by land use is expressed in changes in peatland vegetation over time. As these changes in vegetation can be identified (e.g. the ordinations of the ALL CORE dataset) the impact of this disturbance/management on carbon accumulation can be quantified.

This research question is relevant to peatland management as it shows that carbon conscious peatland management must focus on promoting specific vegetation types (e.g. mixed *Sphagnum* communities). The removal of vegetation which is not favourable for carbon accumulation (e.g. *Calluna vulgaris* and *Molinia caerulea*) is therefore a more appropriate carbon conscious management strategy than the removal of *a priori* assumed 'detrimental' management regimes (e.g. grazing).

9.2.3 Research question three: Did late-19th and 20th Century management impacts affect blanket peatland integrity to a greater degree than pre-AD 1850 impacts?

This research question examined the accuracy of the assertion that peatland management impacts have intensified since the mid-19th Century (Section 2.8.3). It was concluded that pre-AD 1850 land use through burning was often identifiable in peatland records, and post-AD 1850 burning regimes were likely less intense than pre-AD 1850 regimes. Research findings which led to this conclusion are detailed below:

- Burning on Welsh blanket peatlands occurred more intensely/frequently prior to *ca*. AD 1850. This finding was based on twenty-seven of the thirty-eight records where macrofossil charcoal was recorded exhibiting greater charcoal abundance below the AD 1850 horizon than above it. There is no chronology for pre-AD 1850 burning patterns, hence their occurrence cannot be linked to a specific period.
- Some contemporary vegetation assemblages likely developed because of pre-AD 1850 land use rather than the effects of late 19th and 20th Century land use. This finding was based on most Peat Accumulation Type (PAT) B (*Calluna vulgaris*; see Section 9.2.2 above) sites exhibiting plentiful macroscopic charcoal below the AD 1850 horizon and that *Calluna* often became established prior to the AD 1850 marker. Where PAT B sampling points had also been drained, *Calluna* had therefore become established prior to drainage occurring. This PAT is therefore likely influenced by the effect of pre-AD 1850 land use.
- Some peatlands exhibit resilience to intense land management which occurred pre-AD 1850 and they can recover from this impact. This finding was based on results presented for PAT C: Increasing *Sphagnum*. Carbon accumulation rate was low (*ca*. 17 g C m^2 yr^{-1}) at these sites *ca*. AD 1850-1976. These low carbon accumulation rates are likely related to the effects of pre-AD 1850 land use as plentiful charcoal is often recorded below the AD 1850 horizon. Post-AD 1976, carbon accumulation rates increase to *ca*. 200 g C m^2 yr^{-1} as *Sphagnum papillosum* expands, similar to the supposed reference carbon accumulation rate of PAT A: *Sphagnum*-dominated. In some cases, therefore, the effect of pre-AD 1850 land use on carbon dynamics may not linger as much as others (e.g. PAT B (see above)).

This research question is important for addressing the thesis aim because it shows that the impacts of pre-AD 1850 land management may linger to this day as their impact is recorded in vegetation communities which persist to present. The PATs were used to assess the impacts of land use on carbon accumulation. As these PATs were classified by vegetation communities, it appears the effect of pre-AD 1850 land use may affect current carbon accumulation rates at some Welsh peatland sites.

9.3 Technical findings of this thesis

This section presents technical findings which were used to support the conclusions reported in Section 9.2. These findings are detailed below:

- Both radiometric (^{210}Pb, ^{137}Cs and ^{241}Am) and Spheroidal Carbonaceous Particle (SCPs)-determined rates of peat accumulation may be used to provide reliable estimates of carbon accumulation rate (CAR) and recent apparent rate of carbon accumulation (RERCA). This finding was based on estimates of CAR analogous to long-term rates of carbon accumulation (see below) being within the 30-70 g C m^2 yr^{-1} range expected for blanket peatlands (Lindsay, 2010). Whilst the differing chronological techniques (radiometric and SCPs) produced statistically similar rates of catotelm carbon accumulation, there was discrepancy between the techniques in their estimates of acrotelm carbon dynamics (see below). Where possible, multiple chronological techniques should be used to reconstruct recent rates of peat accumulation for investigations of carbon dynamics. The evidence presented shows that SCPs can be used to produce reliable estimates of carbon accumulation where the use of alternative chronological techniques is precluded by financial or other constraints.
- It is possible to identify the position of the acrotelm/catotelm transition using measurements of peat bulk density, carbon and nitrogen content. The identification of this transition means that improved estimates of long-term rate of carbon accumulation can be calculated because rates of carbon accumulation in the acrotelm overestimate long-term carbon accumulation. This finding was based on the detection of a significant difference in peat properties (primarily in bulk density) between peat above and below the *ca*. AD 1982 horizon in a peat profile. This horizon was used to identify the acrotelm/catotelm transition, but carbon accumulation rates still exhibited a notable decline below this horizon, indicative of the effect of ongoing decomposition as peat passes into the catotelm. These results therefore support the previously proposed model of ‘concave’ long-term apparent rate of carbon accumulation in that peat must be present in the catotelm for some time (*ca*. thirty years) before apparent rates of carbon accumulation become analogous to long-term carbon accumulation rates. Identifying the effect of the acrotelm/catotelm transition in this way was a novel palaeoenvironmental approach and produced realistic estimates of long-term rates of carbon accumulation (see above).

9.4 Recommendations for future research

- Vegetation communities were used effectively in this thesis to examine the impacts of land use on blanket peatland development. Peat Accumulation Types (Section 9.2.2) were determined using only presence/absence plant macrofossil data. Future work should examine macrofossil assemblages at a more detailed level (e.g. Quadrat and Leaf count approach) to refine the Peat Accumulation Types. Barber (1994) highlighted that the sensitivity of plant macrofossil records to climatically driven hydrological changes are best expressed where the *Sphagnum* assemblage contains a diverse mix of species. This thesis has shown that statistically delineating the genus only to section level was a valid approach in the formation of Peat Accumulation Types. Results may be improved if the *Sphagnum* genus is examined at a finer taxonomic resolution.
- Investigations into the effect of land use on blanket peatland management provide an arena to 'bridge the gap' between palaeoecology and ecology (*sensu* Chambers and Daniell, 2011; Davies and Bunting, 2010). Ecological knowledge is required to understand the effects of land management on blanket peatland vegetation. Future interdisciplinary projects could use ecological experiments to better constrain the response of key peatland flora to a range of land management pressures (e.g. using enclosure/exclosure experiments to explore how the effect of grazing on peatland vegetation varies according to herbivore species). This understanding could be applied to paleoecological datasets using weighted averaging techniques to create a 'disturbance index' in a similar manner to the 'Dupont' Hydroclimatic Index (Dupont, 1986). Indicator values assigned to key peatland vegetation could therefore be used to assess the relative effect of management on peatland development and make semi-quantitative between site 'disturbance' comparisons.
- This thesis has commented on the challenges of relating short-term rates of carbon accumulation in the acrotelm to long-term rates of carbon accumulation in the catotelm (Section 9.3). The number of studies which assess carbon sequestration using both contemporary carbon fluxes and peatland carbon accumulation profiles should be increased to build upon this work. Several recent studies have identified that relating contemporary carbon flux measurements to long-term rates of blanket peat carbon accumulation can be problematic (Levy and Gray, 2015; Ratcliffe *et al.*

2017). The problem may be that these studies focussed on ‘classic’ long term apparent rate of carbon accumulation in their comparisons with flux data. Hence, it may not be surprising that inter-annual variability in carbon dynamics detected using flux measurements do not reflect an averaged millennial scale carbon accumulation value (Frolking *et al*. 2014; Ratcliffe *et al*. 2017). Resolving differences between carbon flux and peat core carbon dynamics would improve our understanding of the relationship between British blanket peatland vegetation and carbon accumulation. The relationship is unlikely to be completely accounted for using paleoenvironmental approaches given land management histories may have affected the apparent rate of carbon accumulation of peat deposited prior to the impact of land use occurring.

- Future palaeoenvironmental work on the impact of land use should use additional techniques to those used within this thesis:
    - Testate amoebae may be useful in improving understanding of the effects of land use given they have been shown to be sensitive to drainage (Stivrins *et al*. 2017; Warner and Chmielewski, 1992), restoration attempts (Davis and Wilkinson, 2004; Secco *et al*. 2016; Swindles *et al*. 2016b; Valentine *et al*. 2013), cut-over peatland regeneration (Laggoun-Défarge *et al*. 2008), the effects of fire (Qin *et al*. 2017; Turner and Swindles, 2012) and dust deposition (Ireland and Booth, 2012; Swindles *et al*. 2016a). The rapid nature of testate community change, which can be resolved to a seasonal scale (Booth *et al*. 2008), would help constrain temporal understanding of land use change. A weighted averaging disturbance index (see above) could also be developed for testate amoebae using contemporary ecological experiments.
    - Recently published work on soil monoliths from the Alaskan tundra suggests that charcoal reflectance can be quantitatively linked to fire duration and intensity (Hudspith *et al*. 2017). Testing of the approach in blanket peat deposits may improve our understanding of the relationship between macroscopic charcoal records and fire intensity.
    - Coprophilous dung spores (e.g. *Sordaria* and *Sporormiella*) have been used in palaeolimnological investigations to attempt to reconstruct past records of herbivore abundance (e.g. Etienne *et al*. 2013). Studies suggest the link may be affected by within lake

  transport and deposition issues (Parker and Williams, 2011; Raper and Bush, 2009) with more research needed to validate the proxy in lacustrine settings (Baker *et al.* 2016). A quantitative link between spore abundance and herbivore numbers may be possible for peatlands as herbivores graze directly on the sedimentary sequence of interest (*sensu* Gill *et al.* 2013). Proximity of the peat to dung does not intrinsically mean that spore records in peat sequences will be robust. Wood and Wilmshurst (2011) showed that spore abundance in saturated soils can fluctuate according to hydrological change. The use of coprophilous spores warrants investigation as it may offer a means to quantify grazing pressure at specific peatland sampling points over decadal-millennial timescales.
  - Ancient DNA (*a*DNA) may be used for investigating peatland flora which are not preserved in the stratigraphy and for reconstructing the history of those animals (Brown and Barnes, 2015) which grazed on a peatland. To date (AD 2017), peat based *a*DNA studies (Boessenkool *et al.* 2013; Gould *et al.* 2010; Parducci *et al.* 2015) are scarce in comparison to lacustrine *a*DNA reconstructions (Birks and Birks, 2015). The reliability of the approach is therefore more uncertain relative to its application in lakes (Brown and Barnes, 2015; Sjögren *et al.* 2016). No *a*DNA studies have occurred in blanket peat hence it is uncertain whether as yet unidentified taphonomic characteristics (Haile *et al.* 2007; Seddon *et al.* 2014) preclude its usage in such sedimentary systems. A successful application of the method would permit the investigation of the effect of changing grazing patterns in the Welsh uplands (i.e. the large-scale replacement of cattle by sheep) over the last millennia.

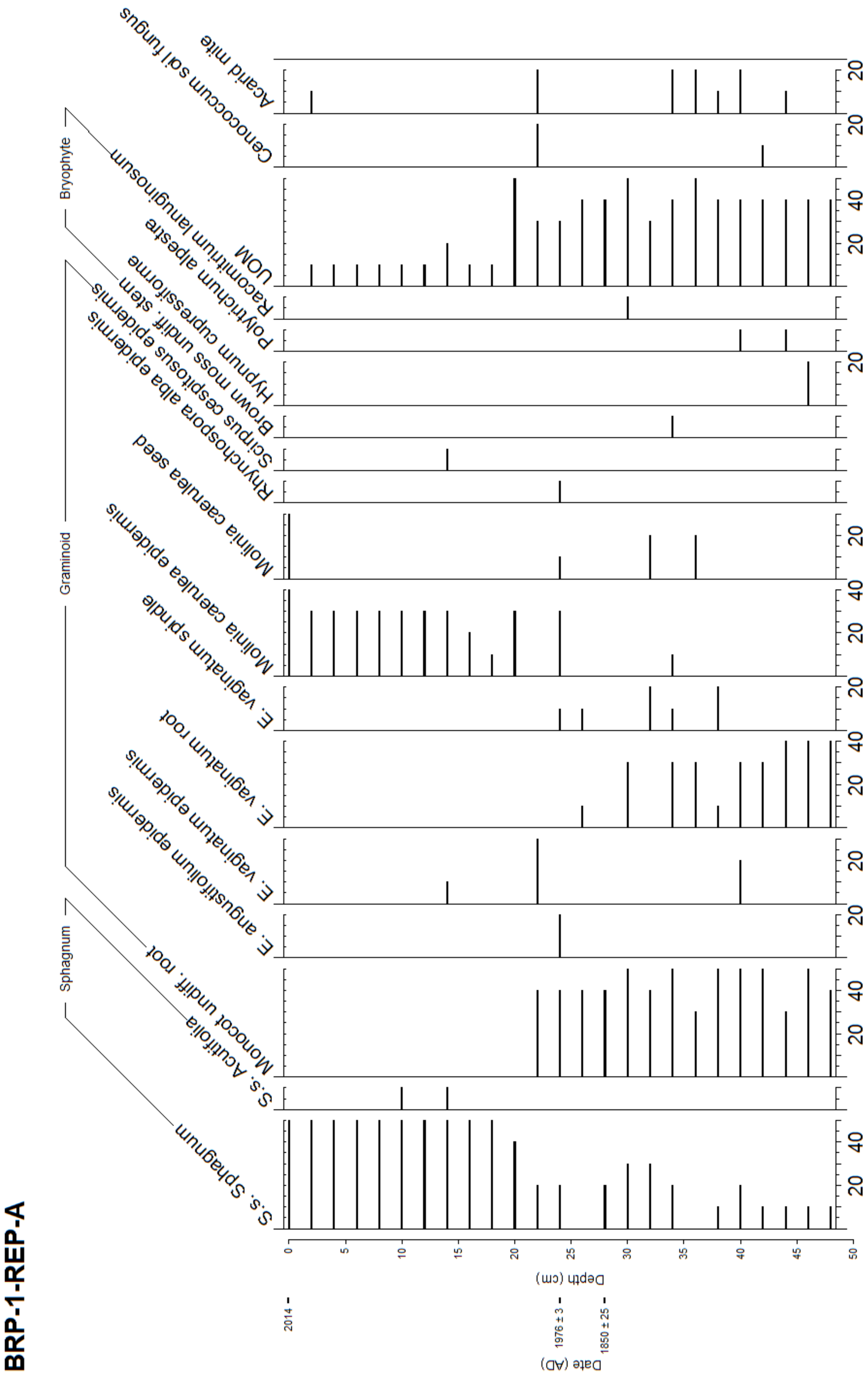
- Peat Accumulation Types (PATs) presented here may provide contextual information to those seeking to manage blanket peatlands from a carbon conscious standpoint. PATs may also act as conditions to test the resilience of differing ‘states’ of a peatland (*sensu* Page and Baird, 2016) or carbon accumulation data could be used in a future Payment for Ecosystem services (PES) based land management model (*sensu* Reed *et al.* 2013, 2014). Any developed PES scheme must recognise that a given management regime may not affect peatland carbon accumulation in a consistent manner (Section 9.2.1). PES schemes which seek to promote carbon accumulation must therefore focus on promoting specific vegetation communities (e.g. PAT A), rather than simply focussing on the assumption that a given stocking regime (e.g. ≤0.5 ewes ha^{-1}) is beneficial for carbon accumulation. Data concerning the relationship between vegetation, land use and carbon cycling may be used to develop 'safe and just' operating spaces (*sensu* Dearing *et al.* 2014) to guide more appropriate peatland management by identifying trade-offs between differing management rationale (e.g. biodiversity vs. carbon sequestration). Data here may act as a foundation to explore such approaches.

9.5 Closing comments on the management debate

The scientific debate concerning the impact of land use on blanket peatlands is characterised by equivocal data, biased comment and a lack of consensus. Even those who call for "*informed and unbiased debate*" (Davies *et al.* 2016a, page 3) conflate blanket bogs and heaths under the 'peatland' moniker. This thesis has attempted to contribute to the land use debate in an unbiased manner and suggested a change in the way the debate is conducted. There needs to be less focus on investigating semi-arbitrary land use types and more focus on the variable ways management affects peatlands. In this way, it is hoped that the scientific community can appropriately engage with land owners to promote the sustainable use of these systems rather than alienating swathes of upland stakeholders (e.g. grouse moor managers) as often occurs (Davies *et al.* 2016a).

Science is ultimately uncertain and probabilistic, hence land management devised using scientific methods may not conform to the narrow niches of certainty and precision which policy makers favour (Freudenburg *et al.* 2008; Jamieson *et al.* 2015; Jasanoff, 2007; Rayner, 2006). Uncertainty should be no excuse for inaction (Holmes and Clark, 2008; Lewandowsky *et al.* 2015; Oreskes, 2004, 2015) and removing the allure of certainty may better accommodate the idiosyncratic nature of blanket peatland ecosystems (*sensu* Hiers *et al.* 2016) which exhibit site-specific variability in their response to land use. By embracing uncertainty regarding the impacts of land use on peatland functioning, the debate may move toward more valuable and appropriate conclusions. Further analysis of the Welsh blanket peatland palaeoenvironmental archive will add valuable insight into the ongoing debate about how best to manage these resources, both for the good of the land owner, but also for the benefit of wider society.

Appendix A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E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
Molinia caerulea epidermis
Molinia caerulea seed
Rhynchospora alba epidermis
Scirpus cespitosus epidermis
Brown moss undiff. stem
Hypnum cupressiforme
Polytrichum alpestre
Racomitrium lanuginosum
UOM
Cenococcum soil fungus
Acarid mite
20
40
Depth (cm)
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15
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25
30
35
40
45
50
Date (AD)
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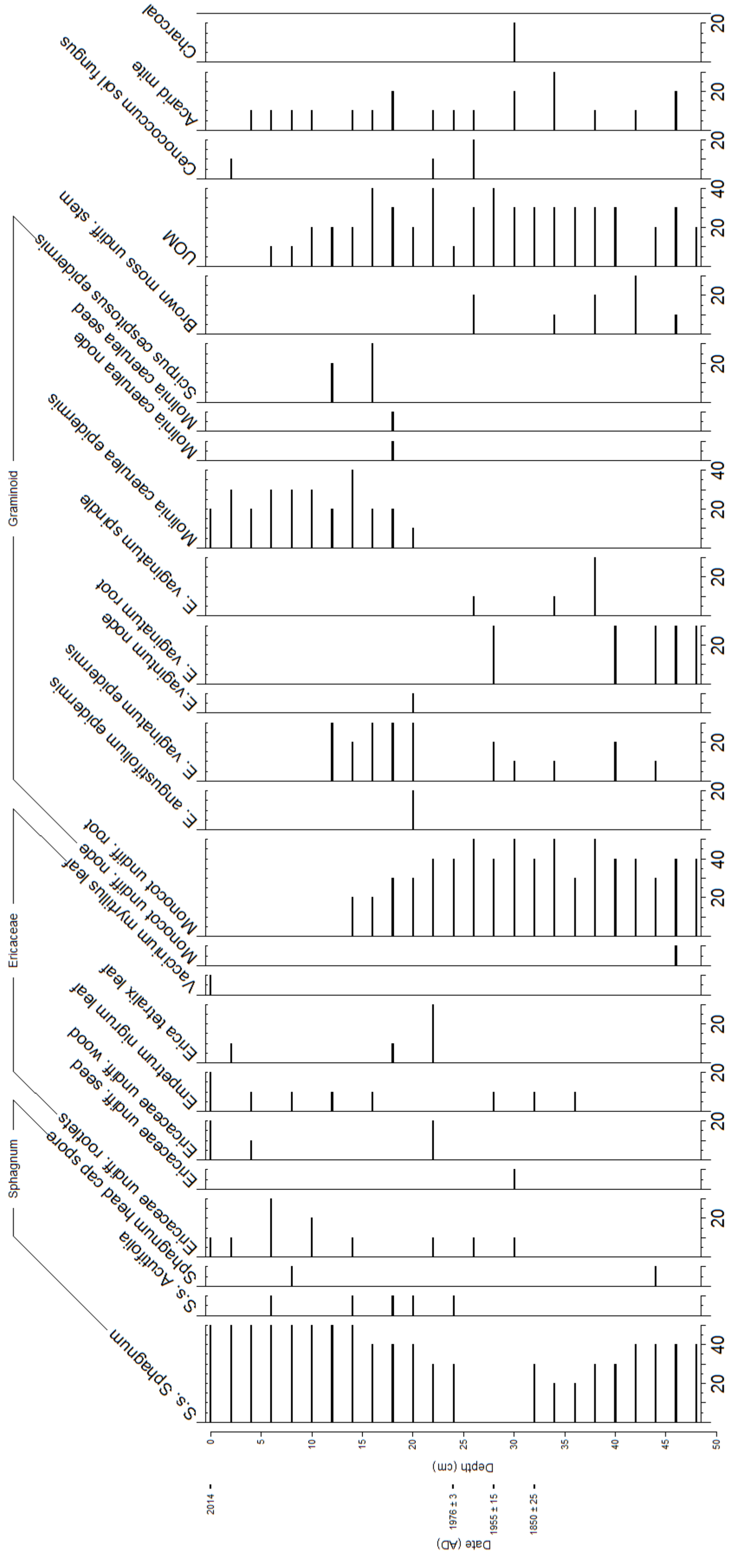
1850 ± 25

BRP-1-REP-B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S.s. Acutifolia
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Ericaceae undiff. seed
Ericaceae undiff. wood
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Erica tetralix leaf
Vaccinium myrtillus leaf
Monocot undiff. node
Monocot undiff. root
E. angustifolium epidermis
E. vaginatum epidermis
E.vaginatum node
E. vaginatum root
E. vaginatum spindle
Molinia caerulea epidermis
Molinia caerulea node
Molinia caerulea seed
Scirpus cespitosus epidermis
Brown moss undiff. stem
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
Sphagnum
Ericaceae
Graminoid
20
40
Depth (cm)
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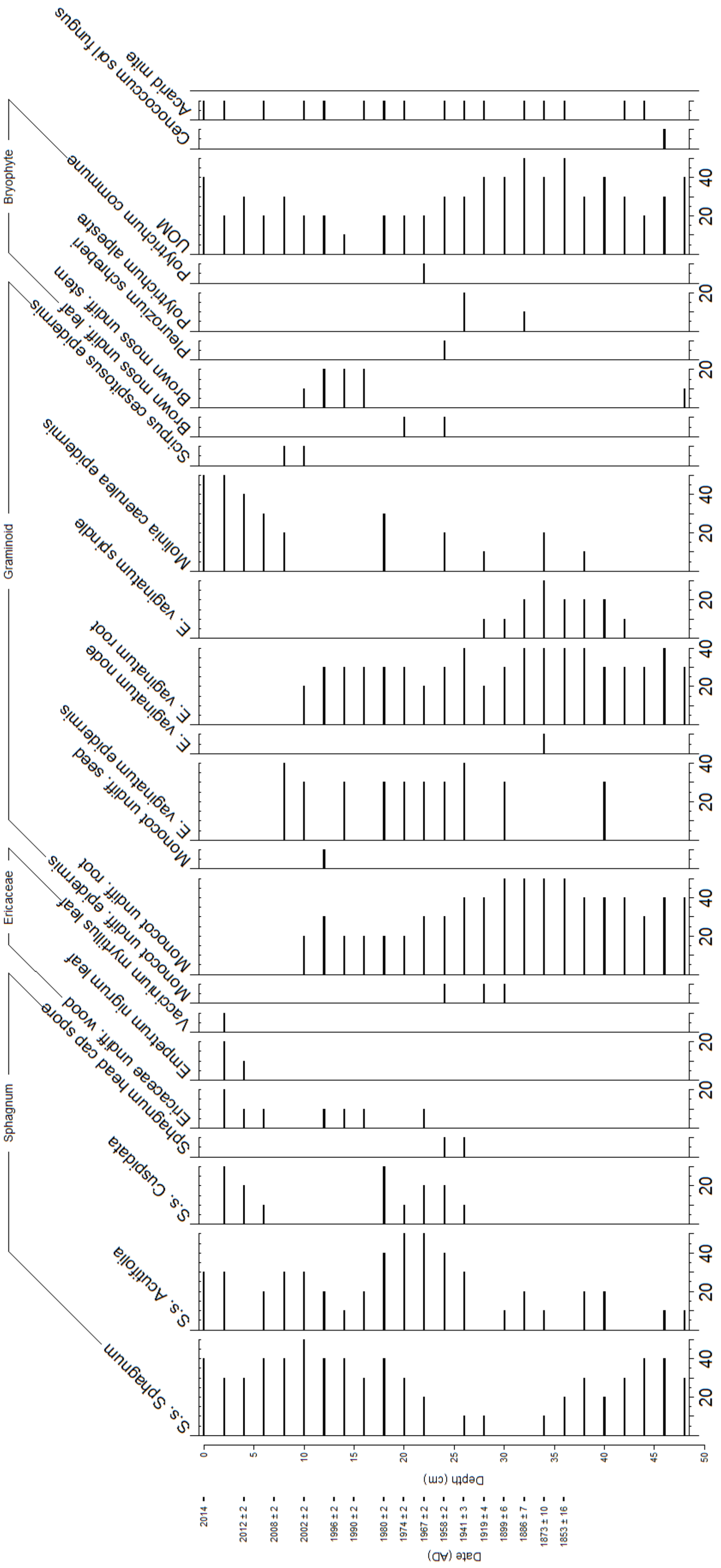
1850 ± 25

BRP-2-M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Empetrum nigrum leaf
Vaccinium myrtillus leaf
Monocot undiff. epidermis
Monocot undiff. root
Monocot undiff. seed
E. vaginatum epidermis
E. vaginatum node
E. vaginatum root
E. vaginatum spindle
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Scirpus cespitosus epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
UOM
Cenococcum soil fungus
Acarid mite
20
40
0
5
10
15
20
25
30
35
40
45
50
Depth (cm)
2014
2012 ± 2
2008 ± 2
2002 ± 2
1996 ± 2
1990 ± 2
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1873 ± 10
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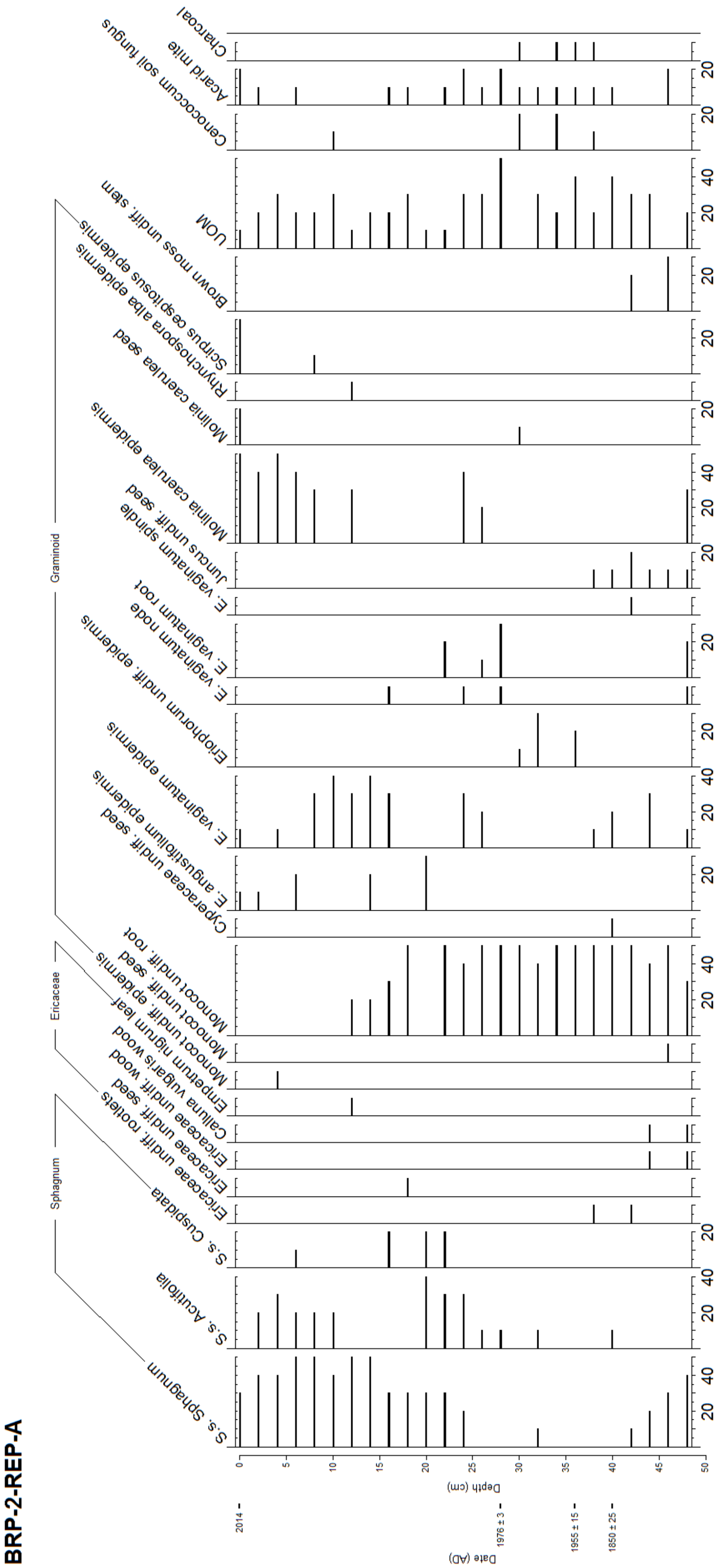
Date (AD)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Ericaceae undiff. seed
Ericaceae undiff. wood
Calluna vulgaris wood
Empetrum nigrum leaf
Monocot undiff. epidermis
Monocot undiff. seed
Monocot undiff. root
Cyperaceae undiff. seed
E. angustifolium epidermis
E. vaginatum epidermis
Eriophorum undiff. epidermis
E. vaginatum node
E. vaginatum root
E. vaginatum spindle
Juncus undiff. seed
Molinia caerulea epidermis
Molinia caerulea seed
Rhynchospora alba epidermis
Scirpus cespitosus epidermis
Brown moss undiff. stem
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
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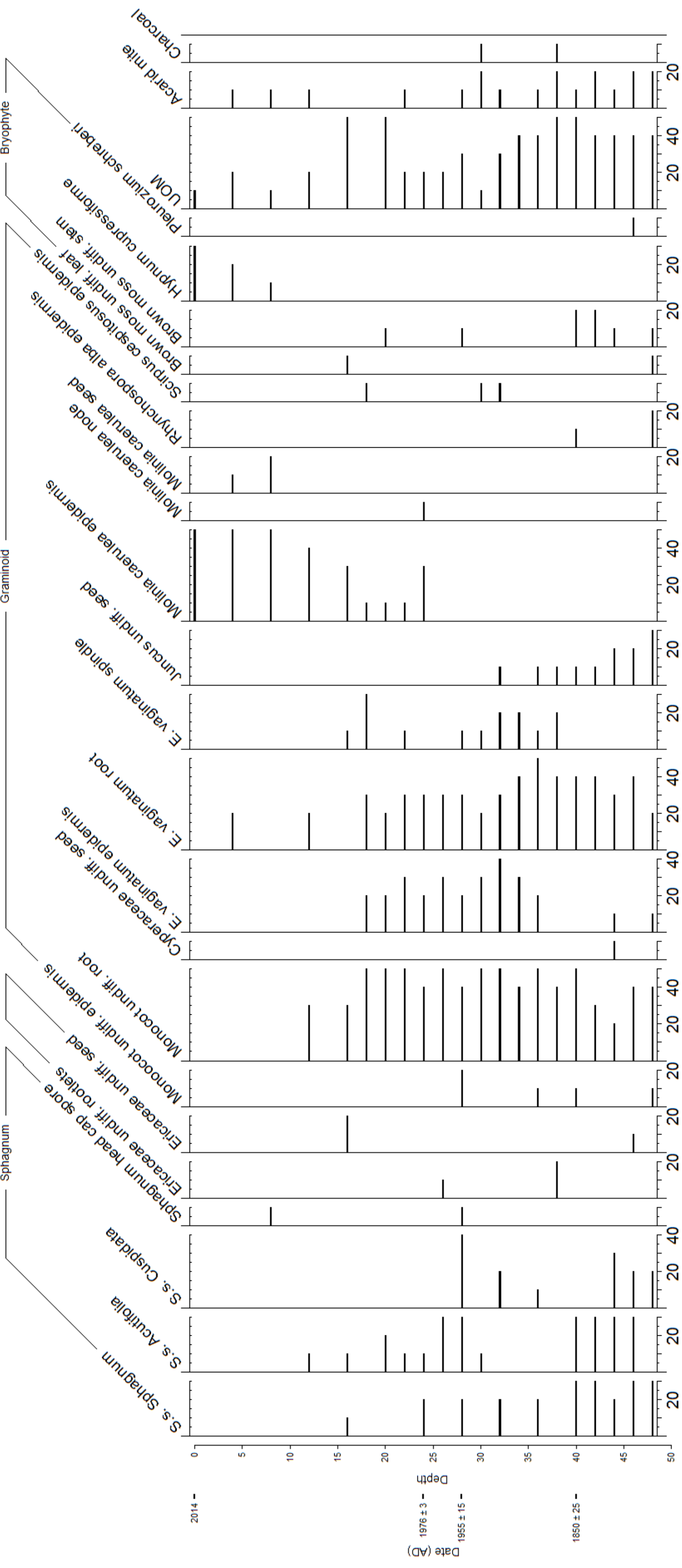
Depth (cm)
2014
1976 ± 3
1955 ± 15
1850 ± 25
Date (AD)

BRP-2-REP-B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S.s. Acutifolia
S.s. Cuspidata
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Ericaceae undiff. seed
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Monocot undiff. root
Cyperaceae undiff. seed
E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
Juncus undiff. seed
Molinia caerulea epidermis
Molinia caerulea node
Molinia caerulea seed
Rhynchospora alba epidermis
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Hypnum cupressiforme
Pleurozium schreberi
UOM
Acarid mite
Charcoal
Sphagnum
Ericaceae
Graminoid
Bryophyte
Depth
0
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1850 ± 25
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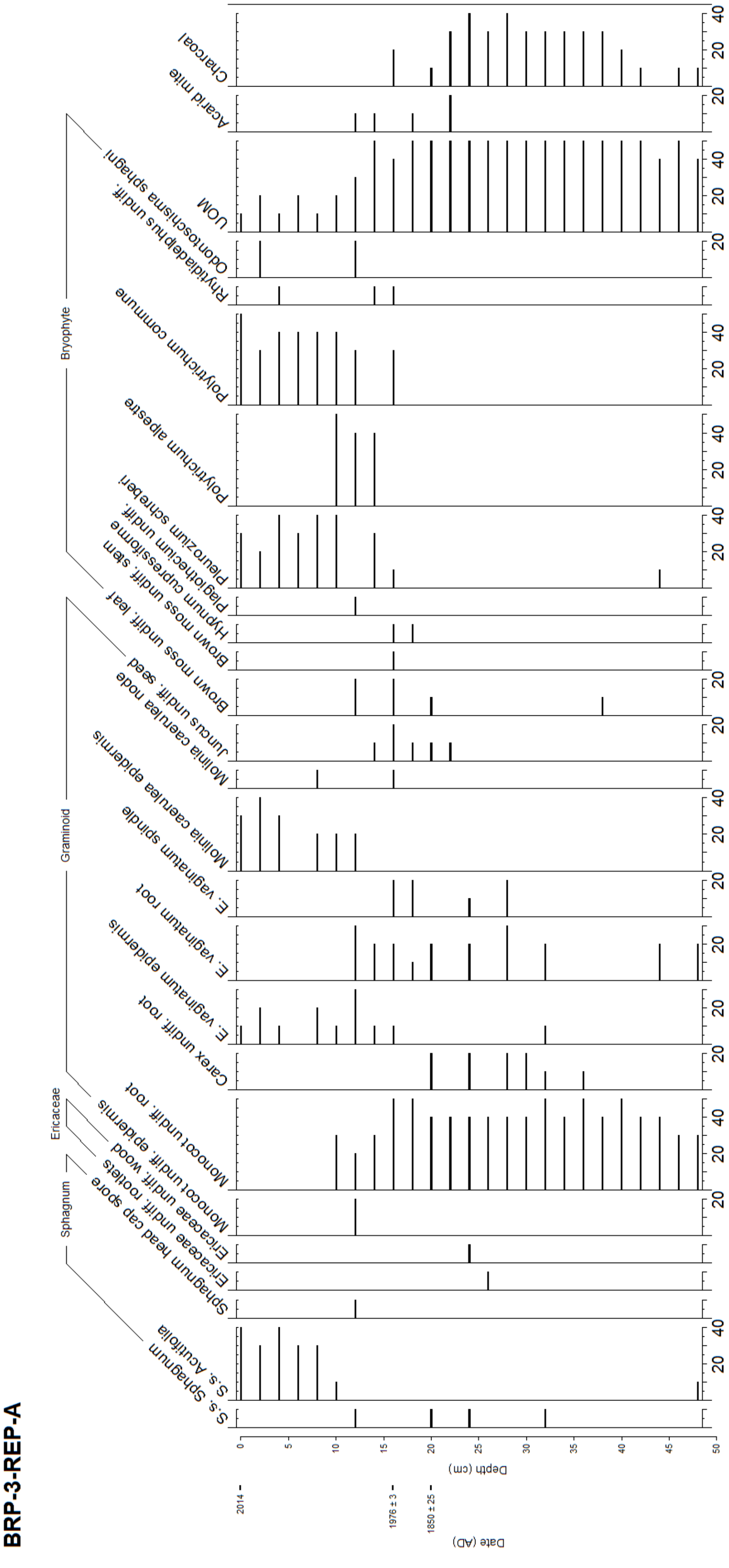
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Bryophyte
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Ericaceae undiff. wood
Monocot undiff. epidermis
Monocot undiff. root
Carex undiff. root
E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
Molinia caerulea epidermis
Molinia caerulea node
Juncus undiff. seed
Brown moss undiff. leaf
Brown moss undiff. stem
Hypnum cupressiforme
Plagiothecium undiff.
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
Rhytidiadelphus undiff.
Odontoschisma sphagni
UOM
Acarid mite
Charcoal
20
40
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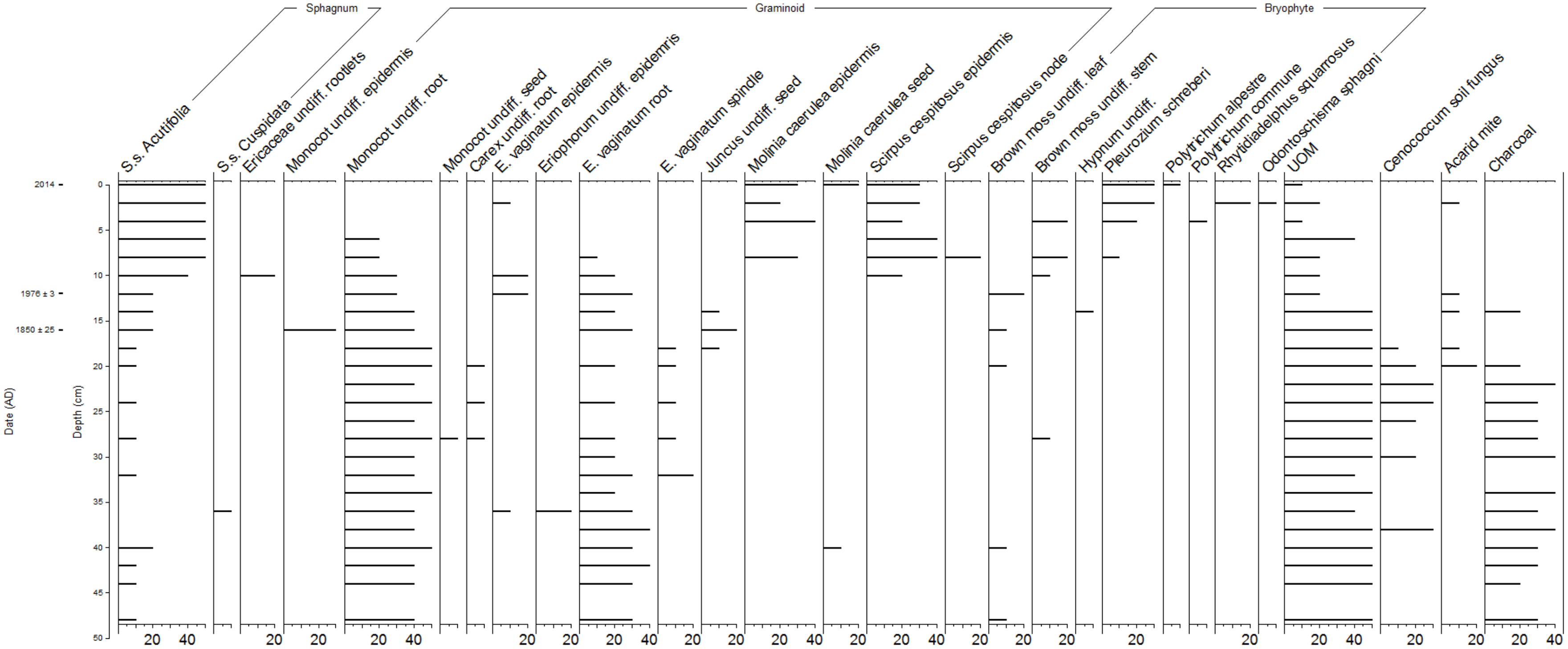
Depth (cm)
2014
1976 ± 3
1850 ± 25
Date (AD)

BRP-4-REP-A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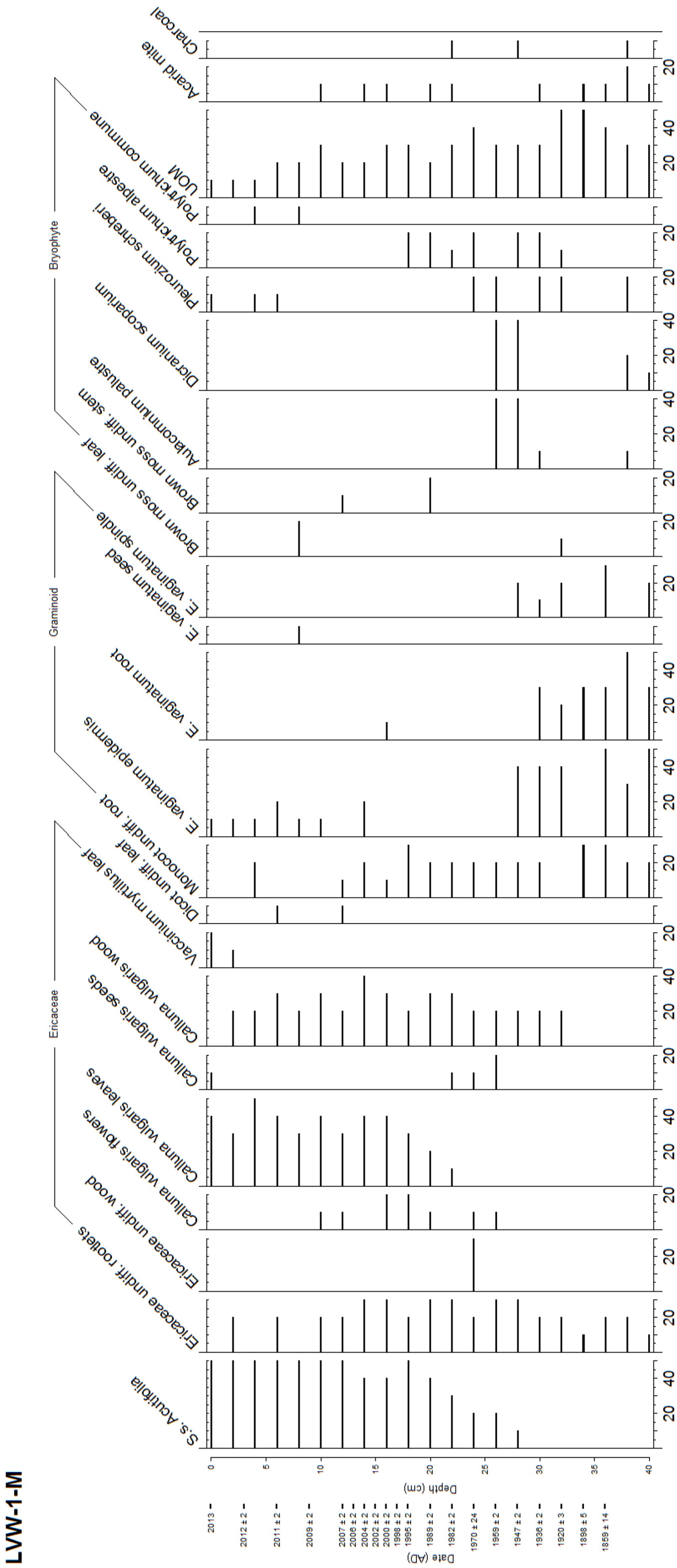
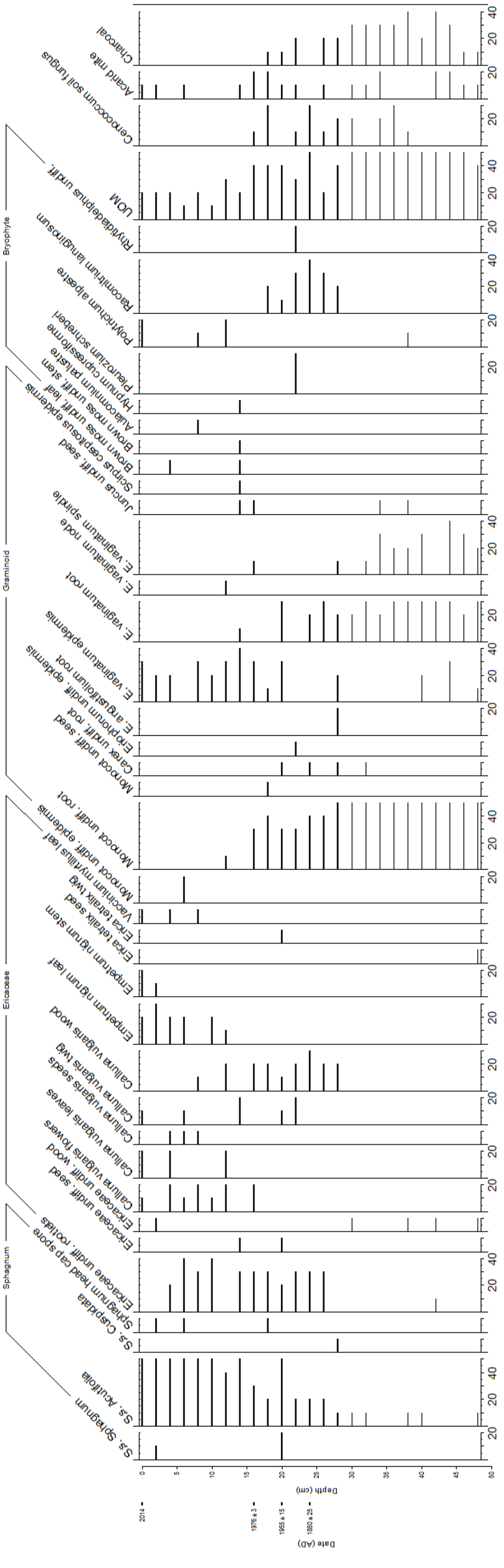
Graminoid
Bryophyte
S.s. Acutifolia
S.s. Cuspidata
Ericaceae undiff. rootlets
Monocot undiff. epidermis
Monocot undiff. root
Monocot undiff. seed
Carex undiff. root
E. vaginatum epidermis
Eriophorum undiff. epidermis
E. vaginatum root
E. vaginatum spindle
Juncus undiff. seed
Molinia caerulea epidermis
Molinia caerulea seed
Scirpus cespitosus epidermis
Scirpus cespitosus node
Brown moss undiff. leaf
Brown moss undiff. stem
Hypnum undiff.
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
Rhytidiadelphus squarrosus
Odontoschisma sphagni
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
Date (AD)
2014
1976 ± 3
1850 ± 25
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
20
40

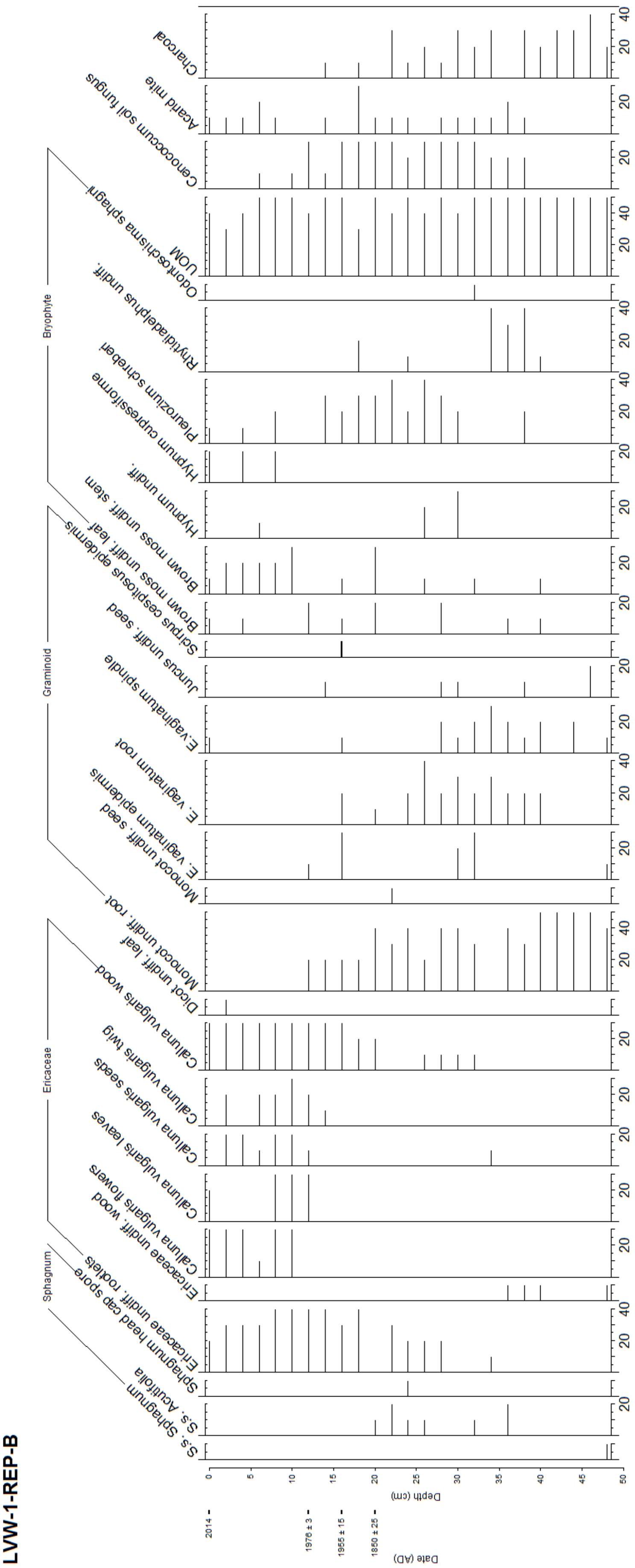

LVW-1-M
Ericaceae
Graminoid
Bryophyte
S.s. Acutifolia
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seeds
Calluna vulgaris wood
Vaccinium myrtillus leaf
Dicot undiff. leaf
Monocot undiff. root
E. vaginatum epidermis
E. vaginatum root
E. vaginatum seed
E. vaginatum spindle
Brown moss undiff. leaf
Brown moss undiff. stem
Aulacomnium palustre
Dicranium scoparium
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
UOM
Acarid mite
Charcoal
20
40
0
5
10
15
20
25
30
35
40
Depth (cm)
2013
2012 ± 2
2011 ± 2
2009 ± 2
2007 ± 2
2006 ± 2
2004 ± 2
2002 ± 2
2000 ± 2
1998 ± 2
1995 ± 2
1989 ± 2
1982 ± 2
1970 ± 24
1959 ± 2
1947 ± 2
1936 ± 2
1920 ± 3
1898 ± 5
1859 ± 14
Date (AD)



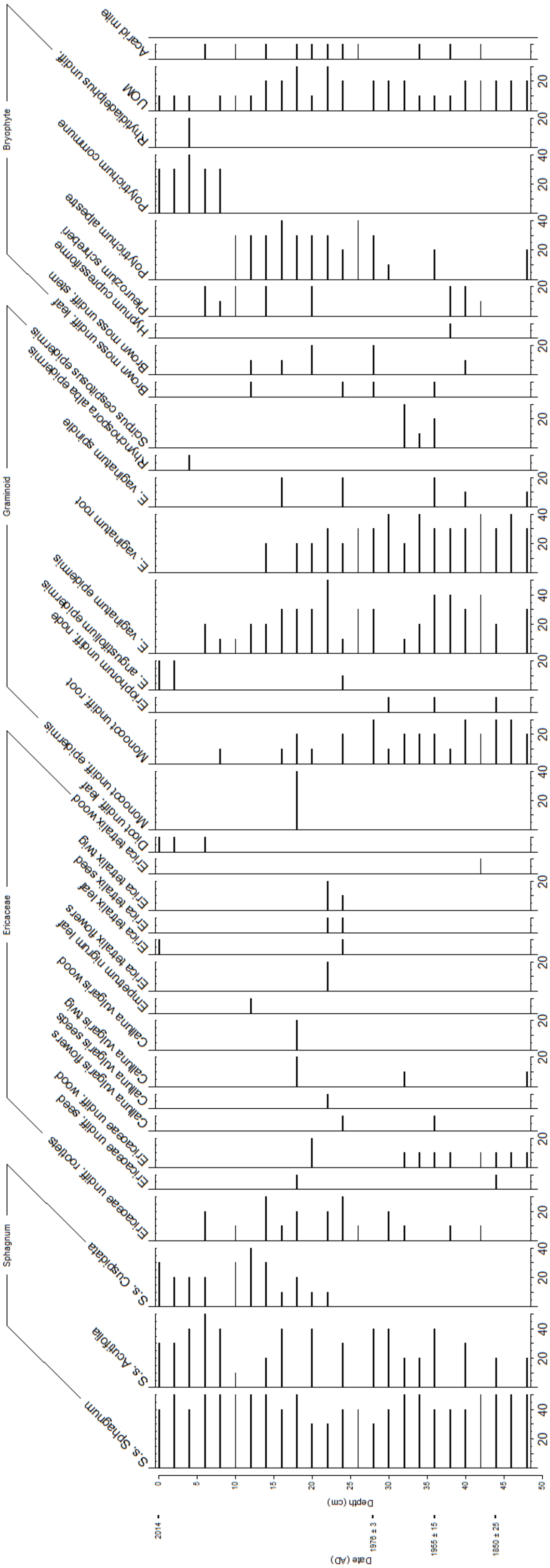
LVW-1-REP-A

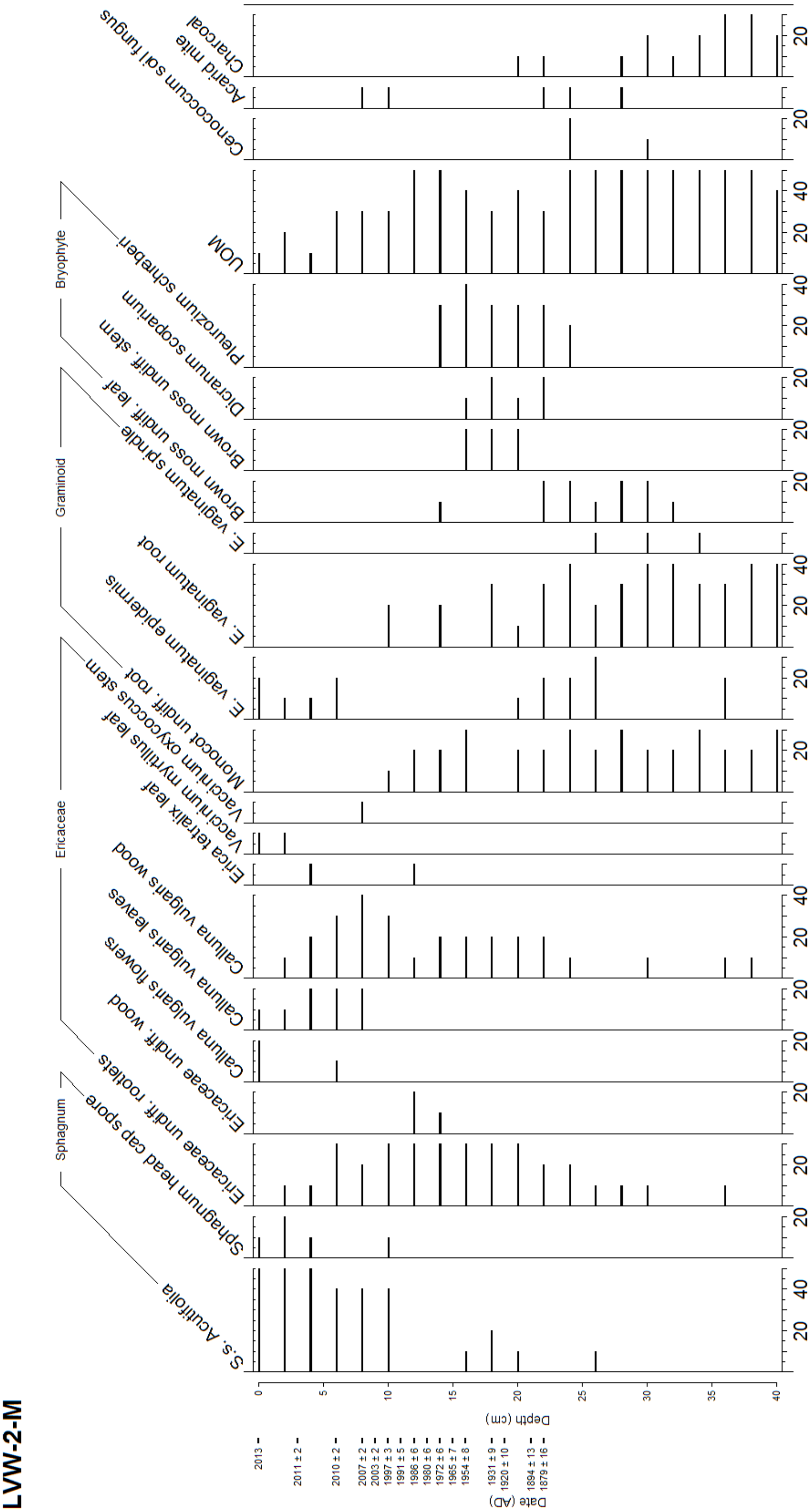
S.s. Sphagnum
S.s. Acutifolia
S.s. Cuspidata
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. seed
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seeds
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Empetrum nigrum stem
Erica tetralix seed
Erica tetralix twig
Vaccinium myrtillus leaf
Monocot undiff. epidermis
Monocot undiff. root
Monocot undiff. seed
Carex undiff. root
Eriophorum undiff. epidermis
E. angustifolium root
E. vaginatum epidermis
E. vaginatum root
E. vaginatum node
E. vaginatum spindle
Juncus undiff. seed
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Aulacomnium palustre
Hypnum cupressiforme
Pleurozium schreberi
Polytrichum alpestre
Racomitrium lanuginosum
Rhytidiadelphus undiff.
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
Sphagnum
Ericaceae
Graminoid
Bryophyte
20
40
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Date (AD)
2014
1976 ± 3
1965 ± 15
1850 ± 25

Appendix A

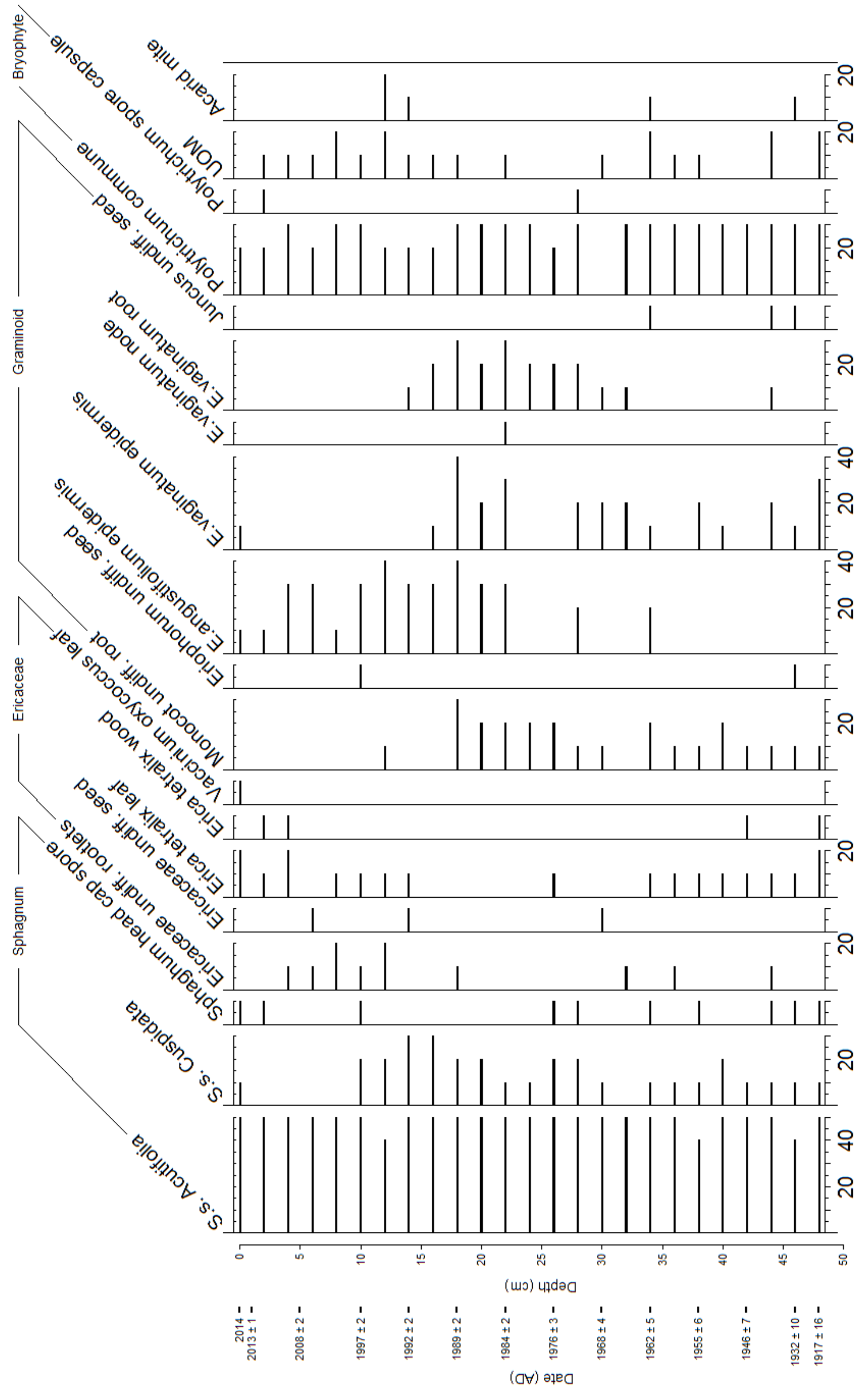

LVW-1-REP-B
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seeds
Calluna vulgaris twig
Calluna vulgaris wood
Dicot undiff. leaf
Monocot undiff. root
Monocot undiff. seed
E. vaginatum epidermis
E. vaginatum root
E.vaginatum spindle
Juncus undiff. seed
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Hypnum undiff.
Hypnum cupressiforme
Pleurozium schreberi
Rhytidiadelphus undiff.
Odontoschisma sphagni
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Date (AD)
2014
1976 ± 3
1955 ± 15
1850 ± 25
20
40

LVW-1-REP-C

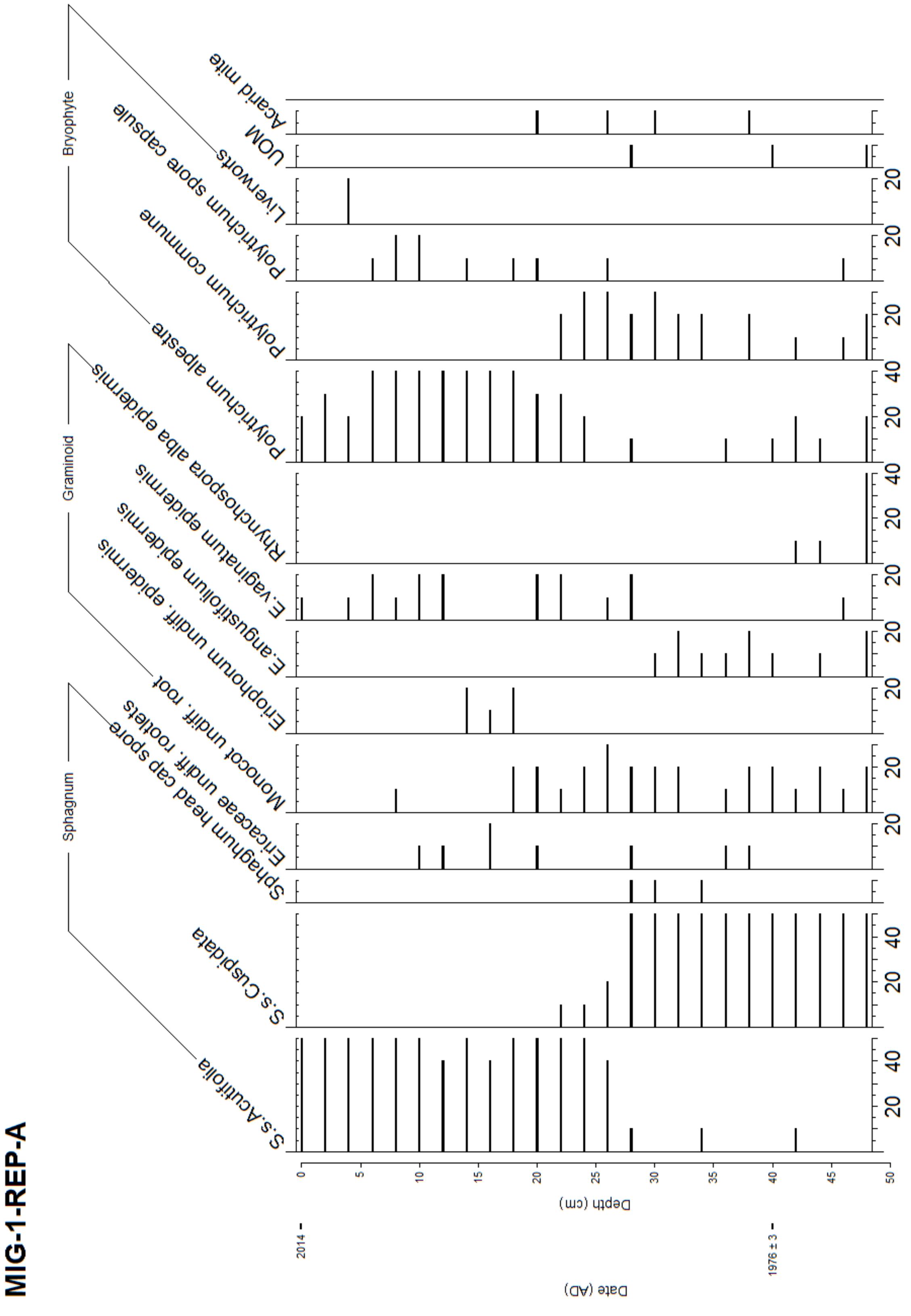

Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
S.s. Cuspidata
Ericaceae undiff. rootlets
Ericaceae undiff. seed
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris seeds
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Erica tetralix flowers
Erica tetralix leaf
Erica tetralix seed
Erica tetralix twig
Erica tetralix wood
Dicot undiff. leaf
Monocot undiff. epidermis
Monocot undiff. root
Eriophorum undiff. node
E. angustifolium epidermis
E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
Rhynchospora alba epidermis
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Hypnum cupressiforme
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
Rhytidiadelphus undiff.
UOM
Acarid mite
20
40
0
5
10
15
20
25
30
35
40
45
50
Depth (cm)
Date (AD)
2014
1976 ± 3
1955 ± 15
1850 ± 25


LVW-2-M
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Acutifolia
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris wood
Erica tetralix leaf
Vaccinium myrtillus leaf
Vaccinium oxycoccus stem
Monocot undiff. root
E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
Brown moss undiff. leaf
Brown moss undiff. stem
Dicranum scoparium
Pleurozium schreberi
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40
Depth (cm)
0
5
10
15
20
25
30
35
40
Date (AD)
2013
2011 ± 2
2010 ± 2
2007 ± 2
2003 ± 2
1997 ± 3
1991 ± 5
1986 ± 6
1980 ± 6
1972 ± 6
1965 ± 7
1954 ± 8
1931 ± 9
1920 ± 10
1894 ± 13
1879 ± 16

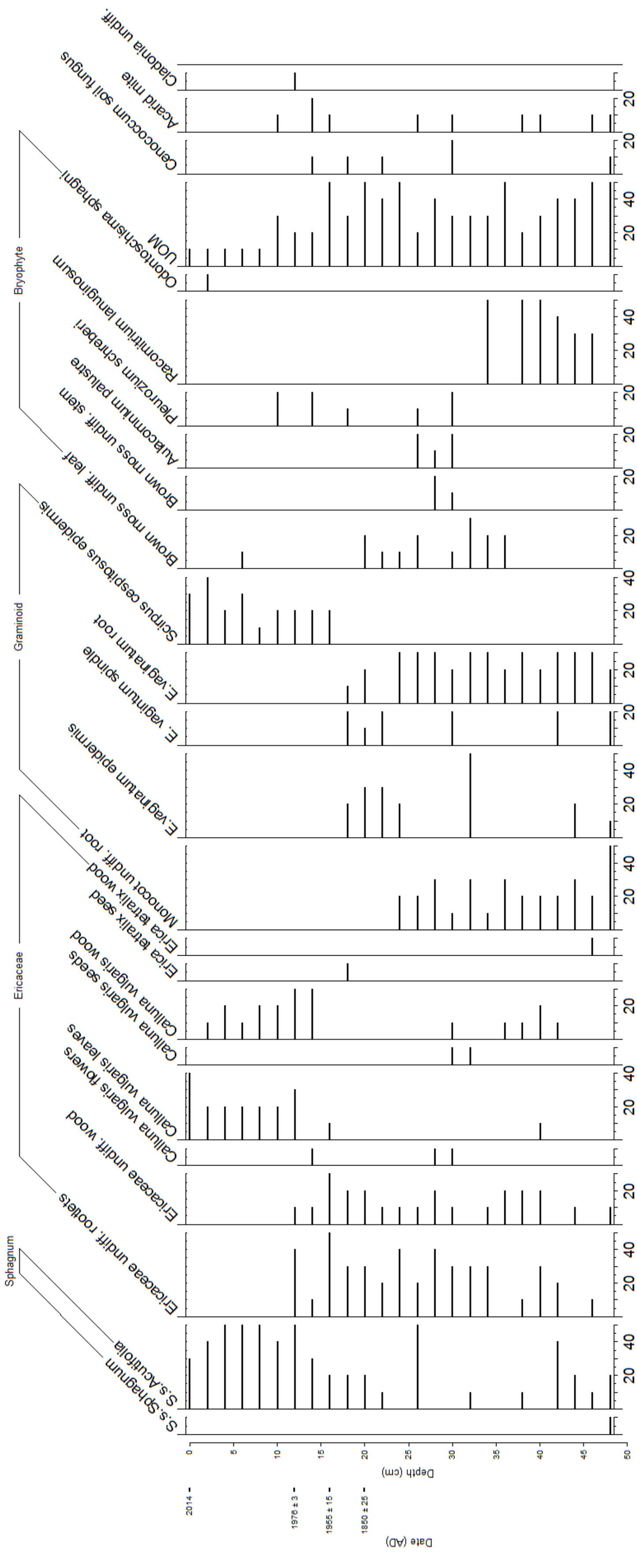
MIG-1-M


Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Acutifolia
S.s. Cuspidata
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. seed
Erica tetralix leaf
Erica tetralix wood
Vaccinium oxycoccus leaf
Monocot undiff. root
Eriophorum undiff. seed
E.angustifolium epidermis
E.vaginatum epidermis
E.vaginatum node
E.vaginatum root
Juncus undiff. seed
Polytrichum commune
Polytrichum spore capsule
UOM
Acarid mite
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Date (AD)
2014
2013 ± 1
2008 ± 2
1997 ± 2
1992 ± 2
1989 ± 2
1984 ± 2
1976 ± 3
1968 ± 4
1962 ± 5
1955 ± 6
1946 ± 7
1932 ± 10
1917 ± 16
20
40

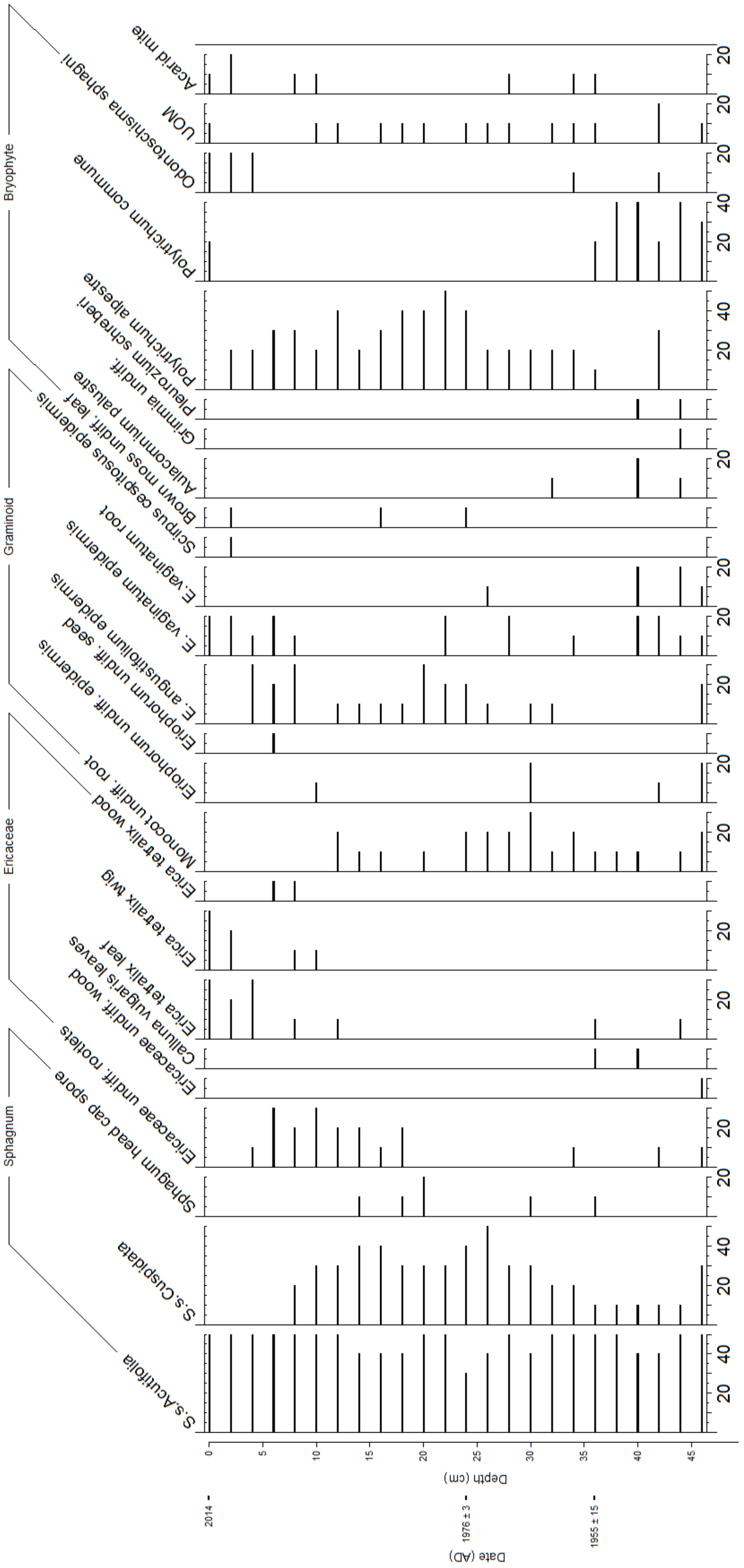
Appendix A

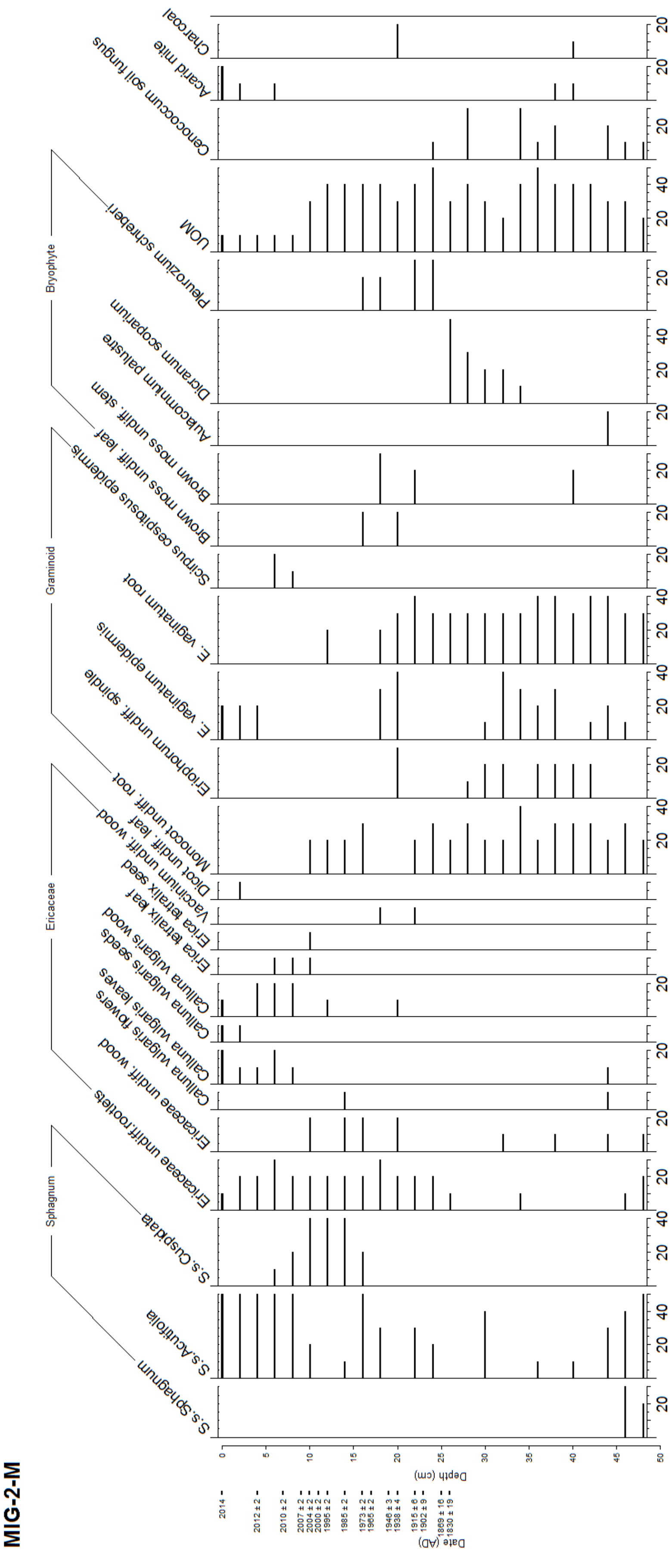

MIG-1-REP-A
Sphagnum
Graminoid
Bryophyte
S.s.Acutifolia
S.s.Cuspidata
Sphagnum head cap spore
Ericaceae undiff. rootlets
Monocot undiff. root
Eriophorum undiff. epidermis
E.angustifolium epidermis
E.vaginatum epidermis
Rhynchospora alba epidermis
Polytrichum alpestre
Polytrichum commune
Polytrichum spore capsule
Liverworts
UOM
Acarid mite
20
40
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Date (AD)
2014
1976 ± 3

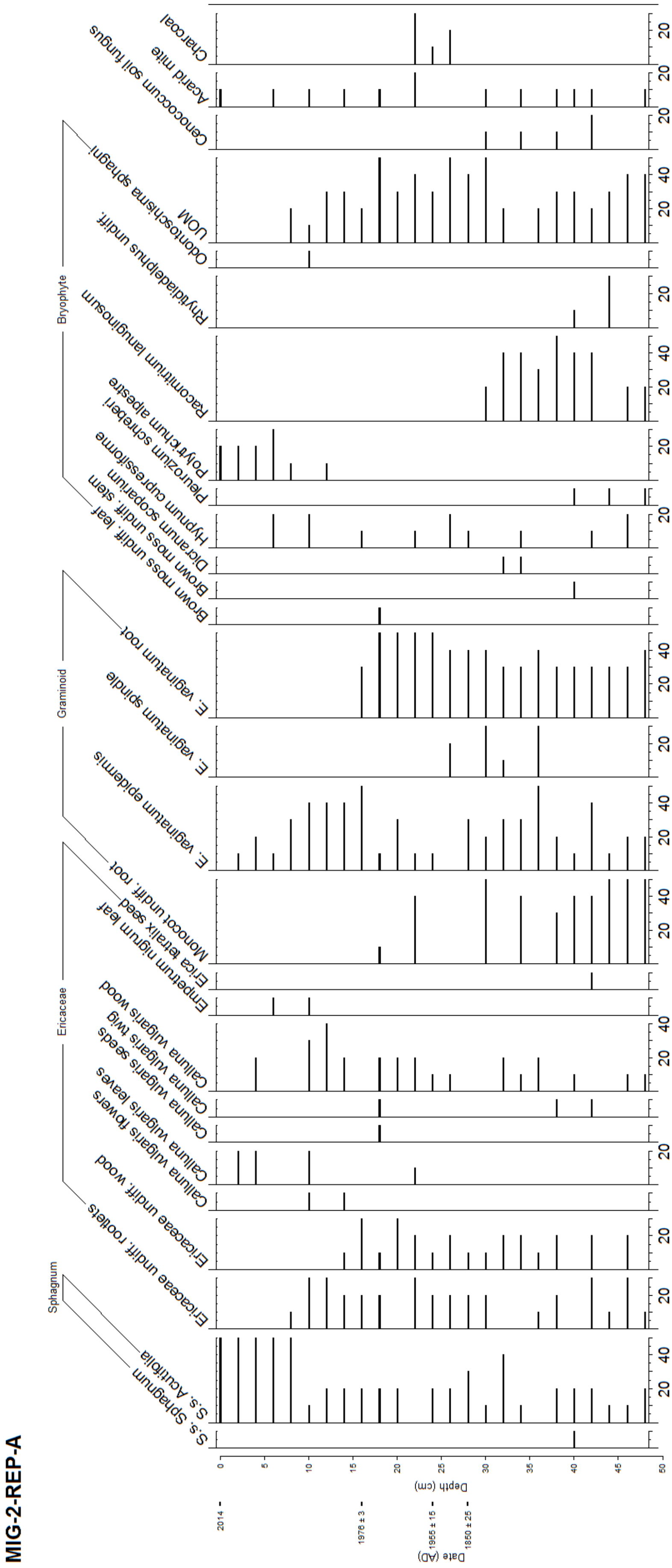
MIG-1-REP-B


Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s.Sphagnum
S.s.Acutifolia
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seeds
Calluna vulgaris wood
Erica tetralix seed
Erica tetralix wood
Monocot undiff. root
E.vaginatum epidermis
E. vaginatum spindle
E.vaginatum root
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Aulacomnium palustre
Pleurozium schreberi
Racomitrium lanuginosum
Odontoschisma sphagni
UOM
Cenococcum soil fungus
Acarid mite
Cladonia undiff.
20
40
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Date (AD)
2014
1976 ± 3
1955 ± 15
1850 ± 25

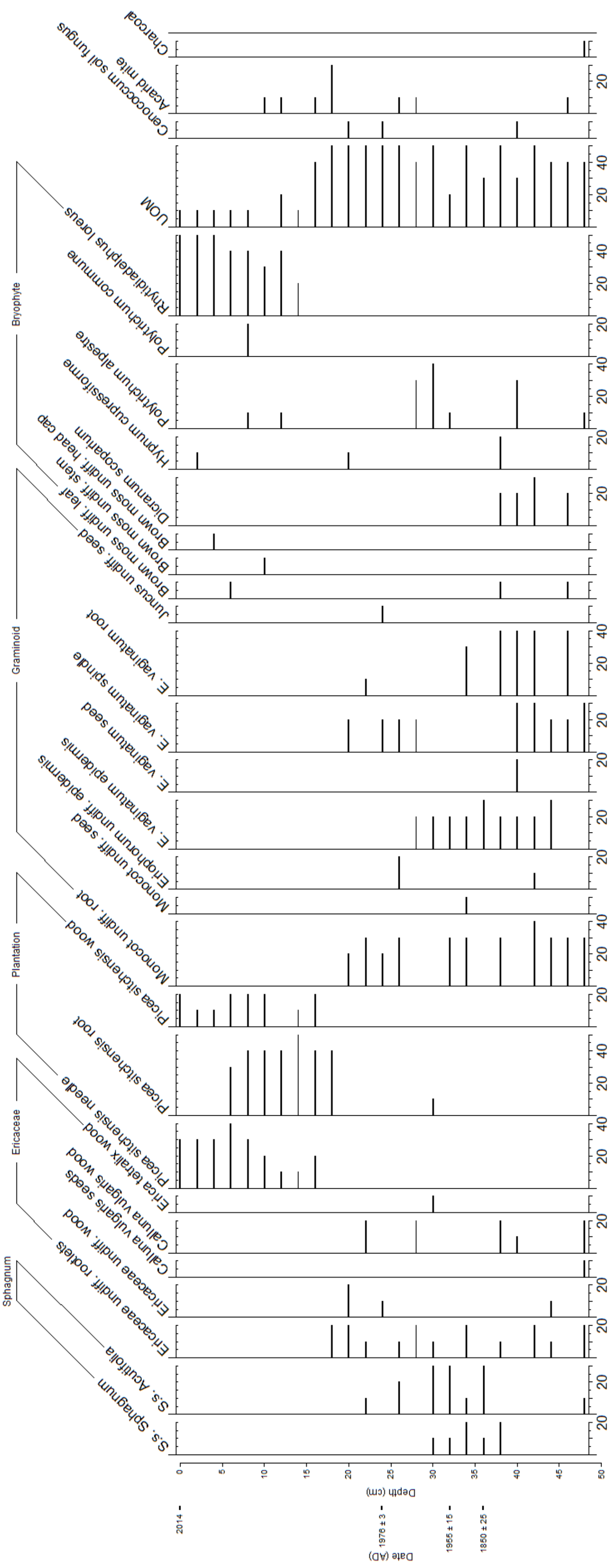
MIG-1-REP-C

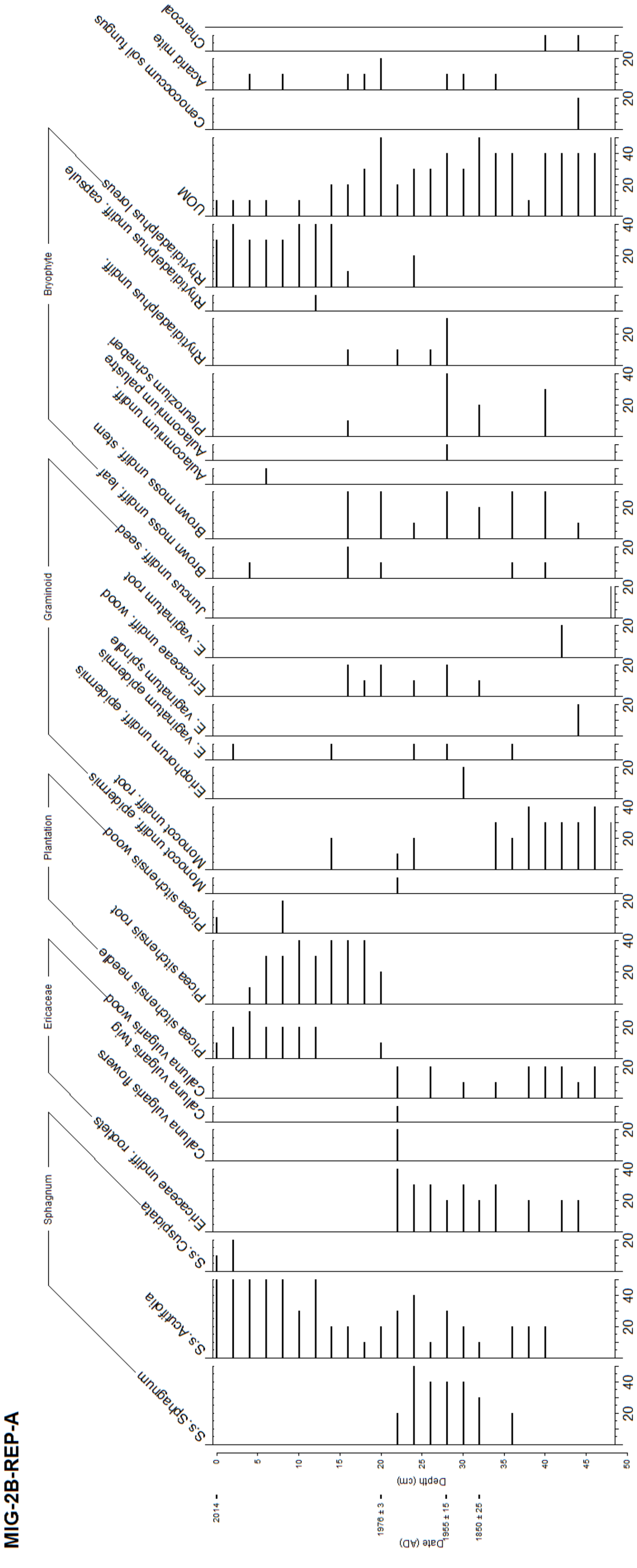

Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s.Acutifolia
S.s.Cuspidata
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris leaves
Erica tetralix leaf
Erica tetralix twig
Erica tetralix wood
Monocot undiff. root
Eriophorum undiff. epidermis
Eriophorum undiff. seed
E. angustifolium epidermis
E. vaginatum epidermis
E.vaginatum root
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Aulacomnium palustre
Grimmia undiff.
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
Odontoschisma sphagni
UOM
Acarid mite
20
40
Depth (cm)
0
5
10
15
20
25
30
35
40
45
Date (AD)
2014
1976 ± 3
1955 ± 15

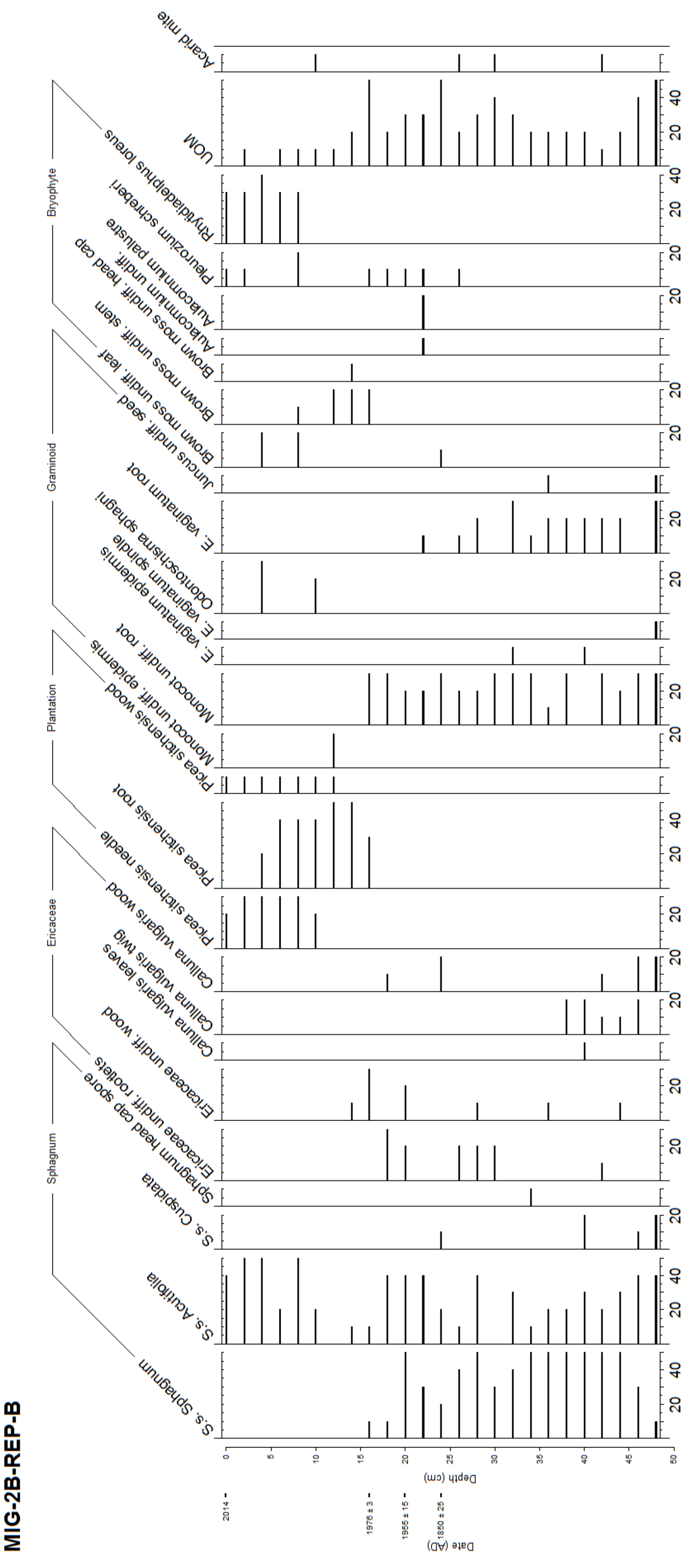

MIG-2-M
Date (AD)
2014
2012 ± 2
2010 ± 2
2007 ± 2
2004 ± 2
2000 ± 2
1995 ± 2
1985 ± 2
1973 ± 2
1965 ± 2
1946 ± 3
1938 ± 4
1915 ± 6
1902 ± 9
1869 ± 16
1830 ± 19
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s.Sphagnum
S.s.Acutifolia
S.s.Cuspidata
Ericaceae undiff.rootlets
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seeds
Calluna vulgaris wood
Erica tetralix leaf
Erica tetralix seed
Vaccinium undiff. wood
Dicot undiff. leaf
Monocot undiff. root
Eriophorum undiff. spindle
E. vaginatum epidermis
E. vaginatum root
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Aulacomnium palustre
Dicranum scoparium
Pleurozium schreberi
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40


MIG-2-REP-A
S.s. Sphagnum
S.s. Acutifolia
Sphagnum
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seeds
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Erica tetralix seed
Ericaceae
Monocot undiff. root
E. vaginatum epidermis
E. vaginatum spindle
E. vaginatum root
Graminoid
Brown moss undiff. leaf
Brown moss undiff. stem
Dicranum scoparium
Hypnum cupressiforme
Pleurozium schreberi
Polytrichum alpestre
Racomitrium lanuginosum
Rhytidiadelphus undiff.
Odontoschisma sphagni
UOM
Bryophyte
Cenococcum soil fungus
Acarid mite
Charcoal
20
40
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Date (AD)
2014
1976 ± 3
1955 ± 15
1850 ± 25

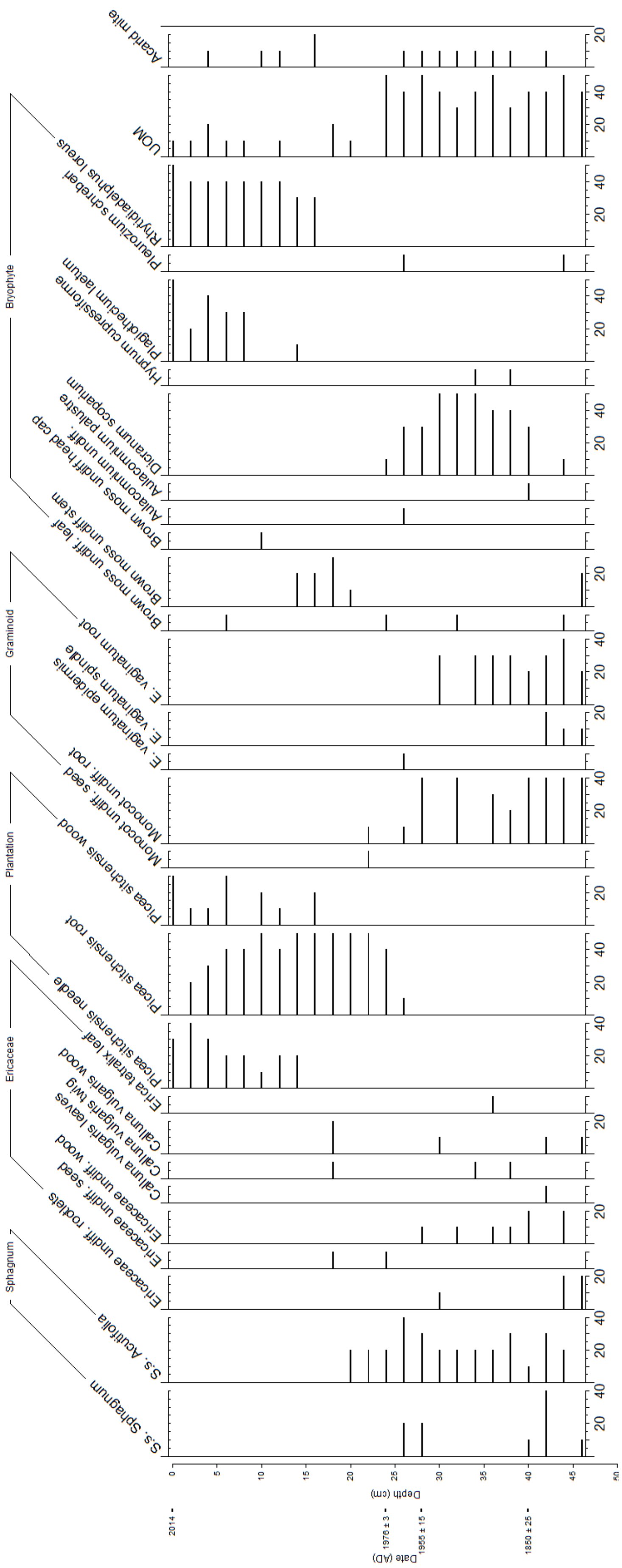
MIG-2-REP-C

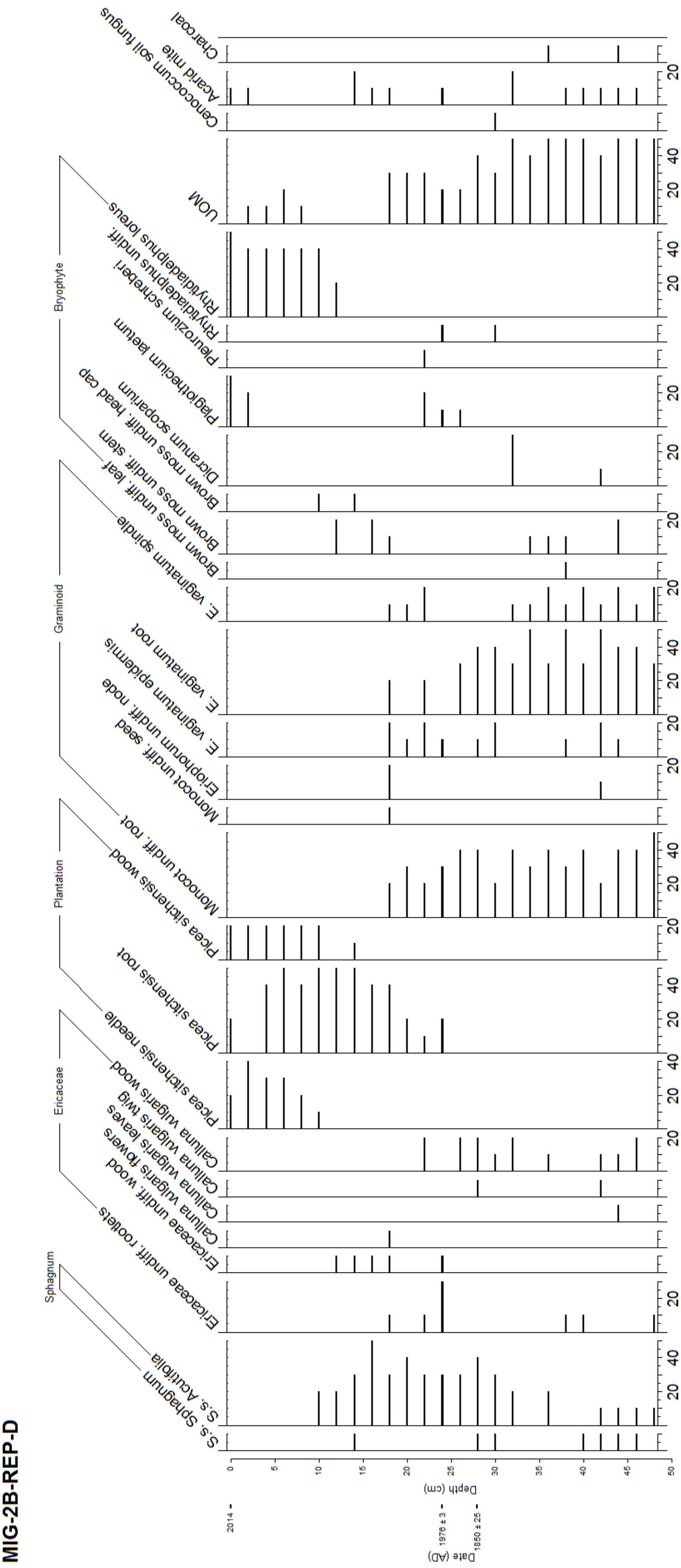

Sphagnum
Ericaceae
Plantation
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris seeds
Calluna vulgaris wood
Erica tetralix wood
Picea sitchensis needle
Picea sitchensis root
Picea sitchensis wood
Monocot undiff. root
Monocot undiff. seed
Eriophorum undiff. epidermis
E. vaginatum epidermis
E. vaginatum seed
E. vaginatum spindle
E. vaginatum root
Juncus undiff. seed
Brown moss undiff. leaf
Brown moss undiff. stem
Brown moss undiff. head cap
Dicranum scoparium
Hypnum cupressiforme
Polytrichum alpestre
Polytrichum commune
Rhytidiadelphus loreus
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
0
5
10
15
20
25
30
35
40
45
50
Depth (cm)
2014
1976 ± 3
1955 ± 15
1850 ± 25
Date (AD)
20
40


MIG-2B-REP-A
Sphagnum
Ericaceae
Plantation
Graminoid
Bryophyte
S.s.Sphagnum
S.s.Acutifolia
S.s.Cuspidata
Ericaceae undiff. rootlets
Calluna vulgaris flowers
Calluna vulgaris twig
Calluna vulgaris wood
Picea sitchensis needle
Picea sitchensis root
Picea sitchensis wood
Monocot undiff. epidermis
Monocot undiff. root
Eriophorum undiff. epidermis
E. vaginatum epidermis
E. vaginatum spindle
Ericaceae undiff. wood
E. vaginatum root
Juncus undiff. seed
Brown moss undiff. leaf
Brown moss undiff. stem
Aulacomnium undiff.
Aulacomnium palustre
Pleurozium schreberi
Rhytidiadelphus undiff.
Rhytidiadelphus undiff. capsule
Rhytidiadelphus loreus
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Date (AD)
2014
1976 ± 3
1955 ± 15
1850 ± 25

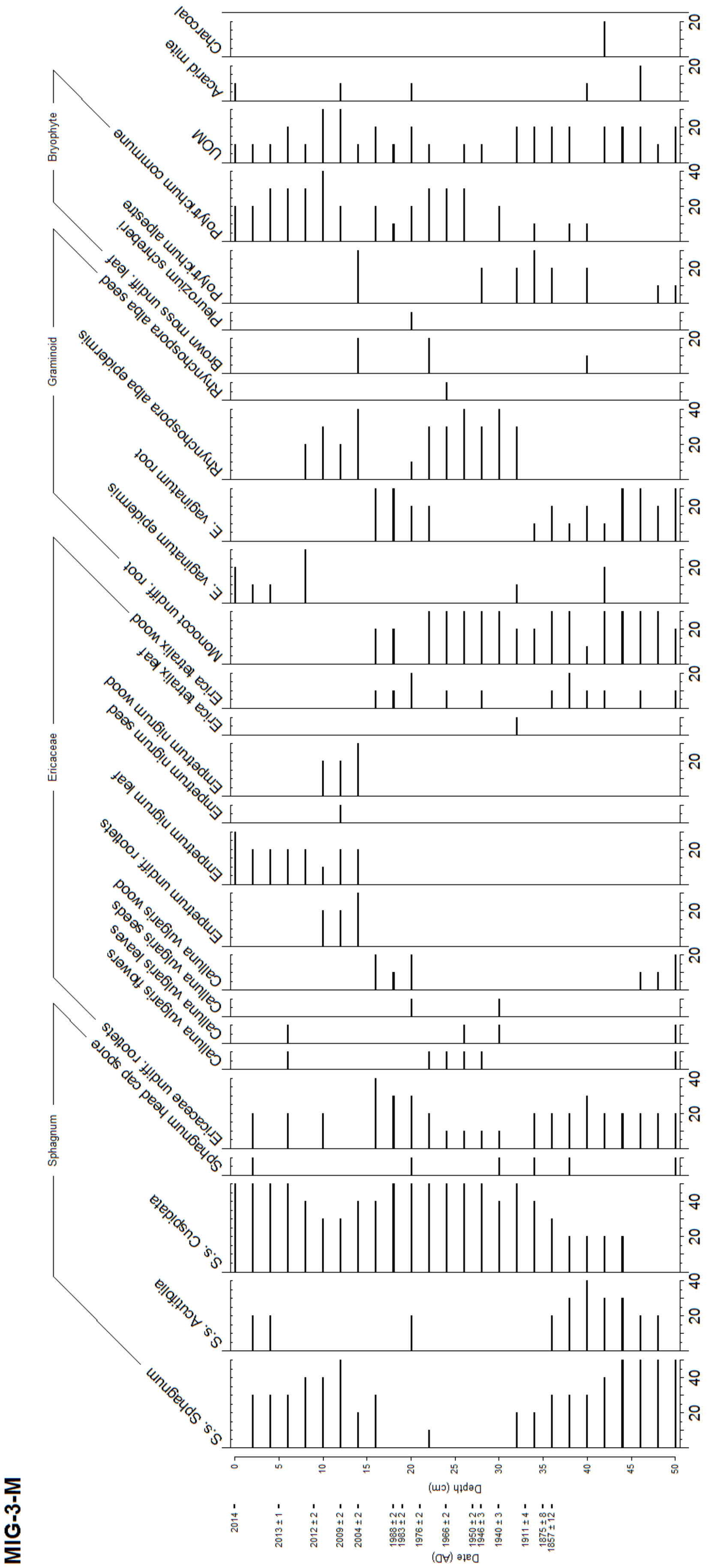

MIG-2B-REP-B
Sphagnum
Ericaceae
Plantation
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
S.s. Cuspidata
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris leaves
Calluna vulgaris twig
Calluna vulgaris wood
Picea sitchensis needle
Picea sitchensis root
Picea sitchensis wood
Monocot undiff. epidermis
Monocot undiff. root
E. vaginatum epidermis
E. vaginatum spindle
Odontoschisma sphagni
E. vaginatum root
Juncus undiff. seed
Brown moss undiff. leaf
Brown moss undiff. stem
Brown moss undiff. head cap
Aulacomnium undiff.
Aulacomnium palustre
Pleurozium schreberi
Rhytidiadelphus loreus
UOM
Acarid mite
20
40
0
5
10
15
20
25
30
35
40
45
50
Depth (cm)
2014
1976 ± 3
1955 ± 15
1850 ± 25
Date (AD)

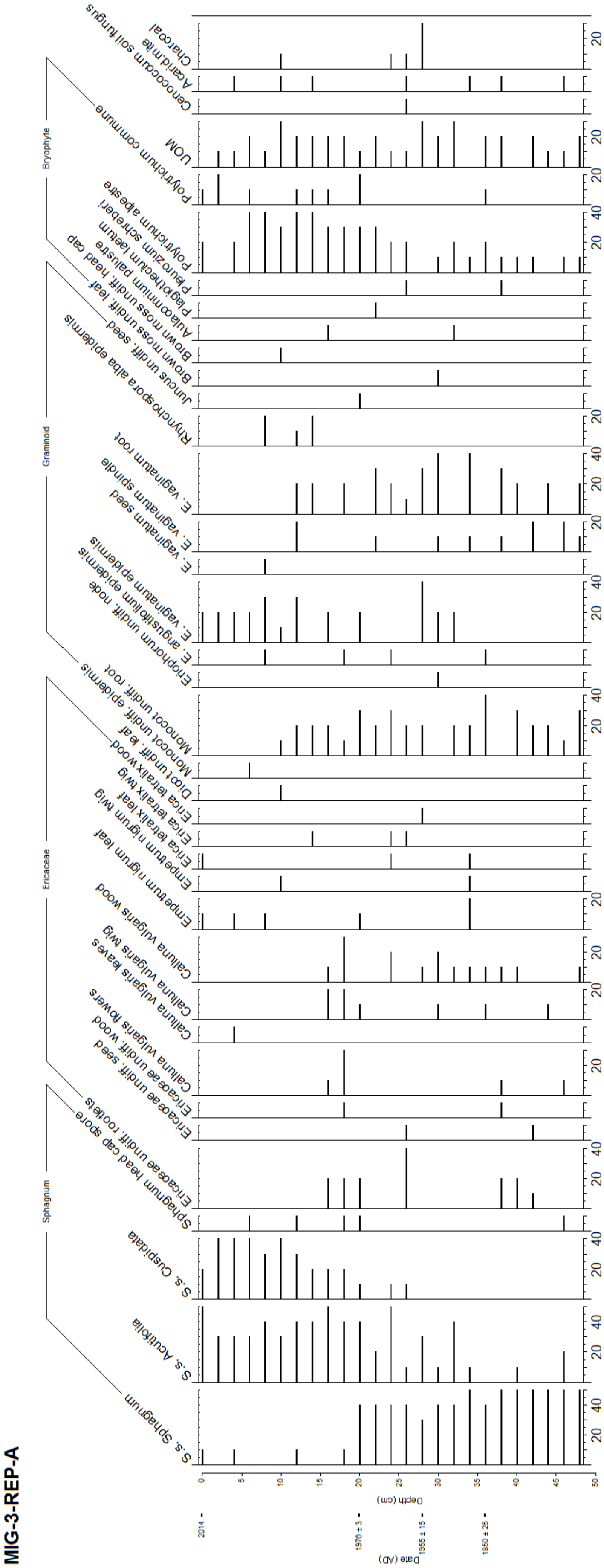
MIG-2B-REP-C

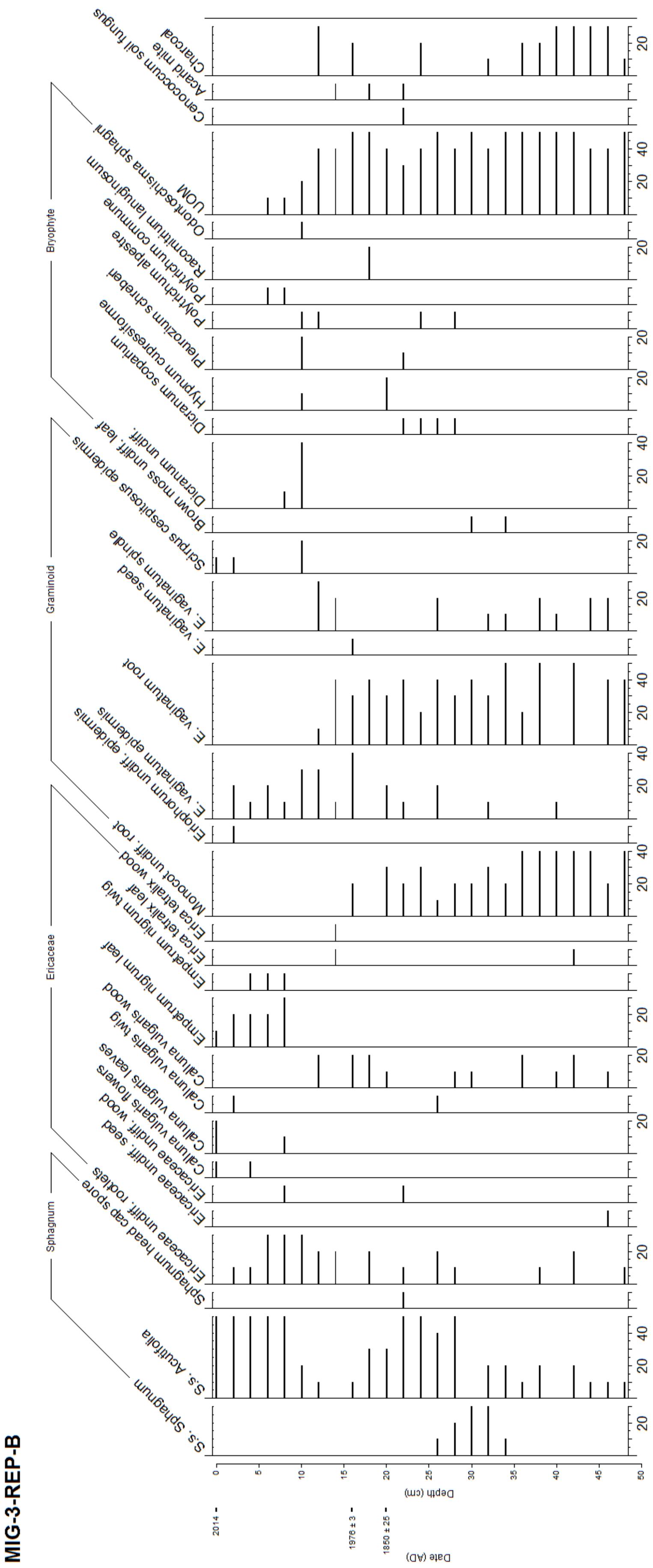

Date (AD)
2014
1976 ± 3
1955 ± 15
1850 ± 25
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Sphagnum
Ericaceae
Plantation
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
Ericaceae undiff. rootlets
Ericaceae undiff. seed
Ericaceae undiff. wood
Calluna vulgaris leaves
Calluna vulgaris twig
Calluna vulgaris wood
Erica tetralix leaf
Picea sitchensis needle
Picea sitchensis root
Picea sitchensis wood
Monocot undiff. seed
Monocot undiff. root
E. vaginatum epidermis
E. vaginatum spindle
E. vaginatum root
Brown moss undiff. leaf
Brown moss undiff. stem
Brown moss undiff. head cap
Aulacomnium undiff.
Aulacomnium palustre
Dicranum scoparium
Hypnum cupressiforme
Plagiothecium laetum
Pleurozium schreberi
Rhytidiadelphus loreus
UOM
Acarid mite
20
40


MIG-2B-REP-D
Sphagnum
Ericaceae
Plantation
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris twig
Calluna vulgaris wood
Picea sitchensis needle
Picea sitchensis root
Picea sitchensis wood
Monocot undiff. root
Monocot undiff. seed
Eriophorum undiff. node
E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
Brown moss undiff. leaf
Brown moss undiff. stem
Brown moss undiff. head cap
Dicranum scoparium
Plagiothecium laetum
Pleurozium schreberi
Rhytidiadelphus undiff.
Rhytidiadelphus loreus
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40
0
5
10
15
20
25
30
35
40
45
50
Depth (cm)
2014
1976 ± 3
1850 ± 25
Date (AD)

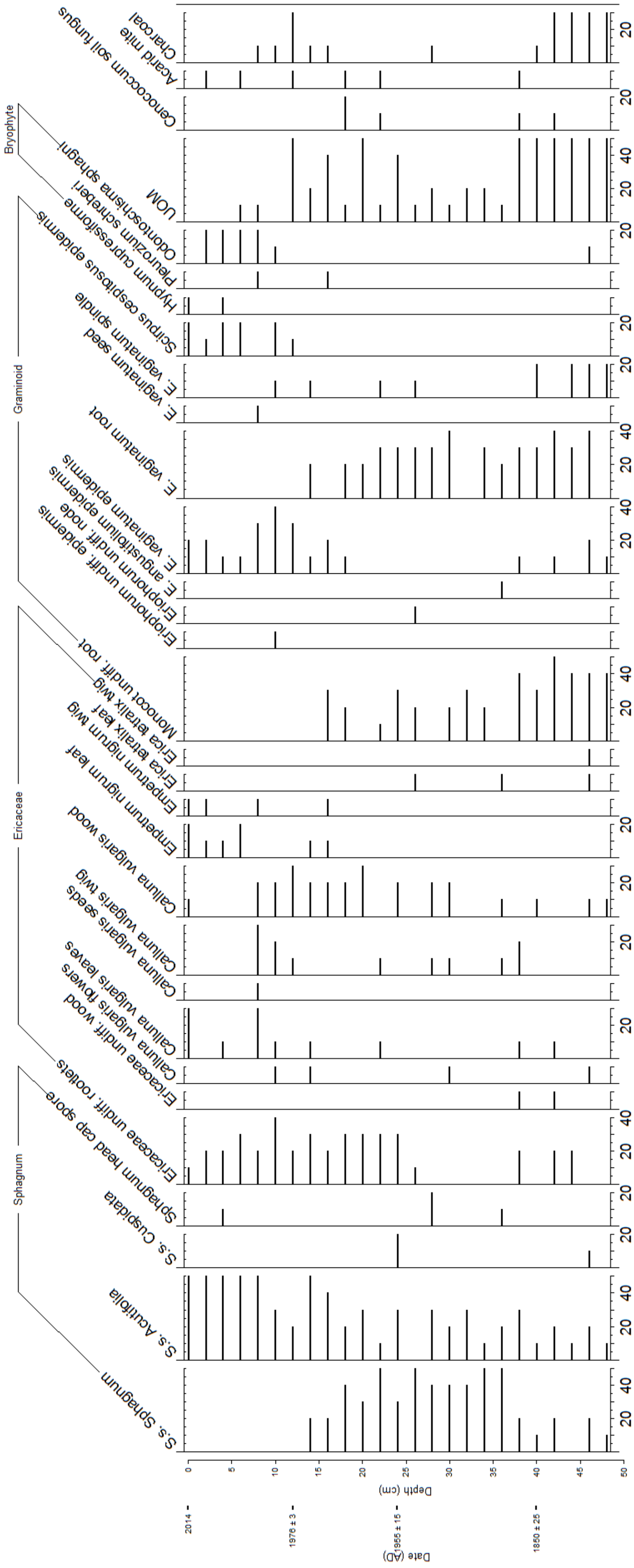
Appendix A


MIG-3-M
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
S.s. Cuspidata
Sphagnum head cap spore
Ericaceae undiff. rootlets
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seeds
Calluna vulgaris wood
Empetrum undiff. rootlets
Empetrum nigrum leaf
Empetrum nigrum seed
Empetrum nigrum wood
Erica tetralix leaf
Erica tetralix wood
Monocot undiff. root
E. vaginatum epidermis
E. vaginatum root
Rhynchospora alba epidermis
Rhynchospora alba seed
Brown moss undiff. leaf
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
UOM
Acarid mite
Charcoal
20
40
0
5
10
15
20
25
30
35
40
45
50
Depth (cm)
2014
2013 ± 1
2012 ± 2
2009 ± 2
2004 ± 2
1988 ± 2
1983 ± 2
1976 ± 2
1966 ± 2
1950 ± 2
1946 ± 3
1940 ± 3
1911 ± 4
1875 ± 8
1857 ± 12
Date (AD)

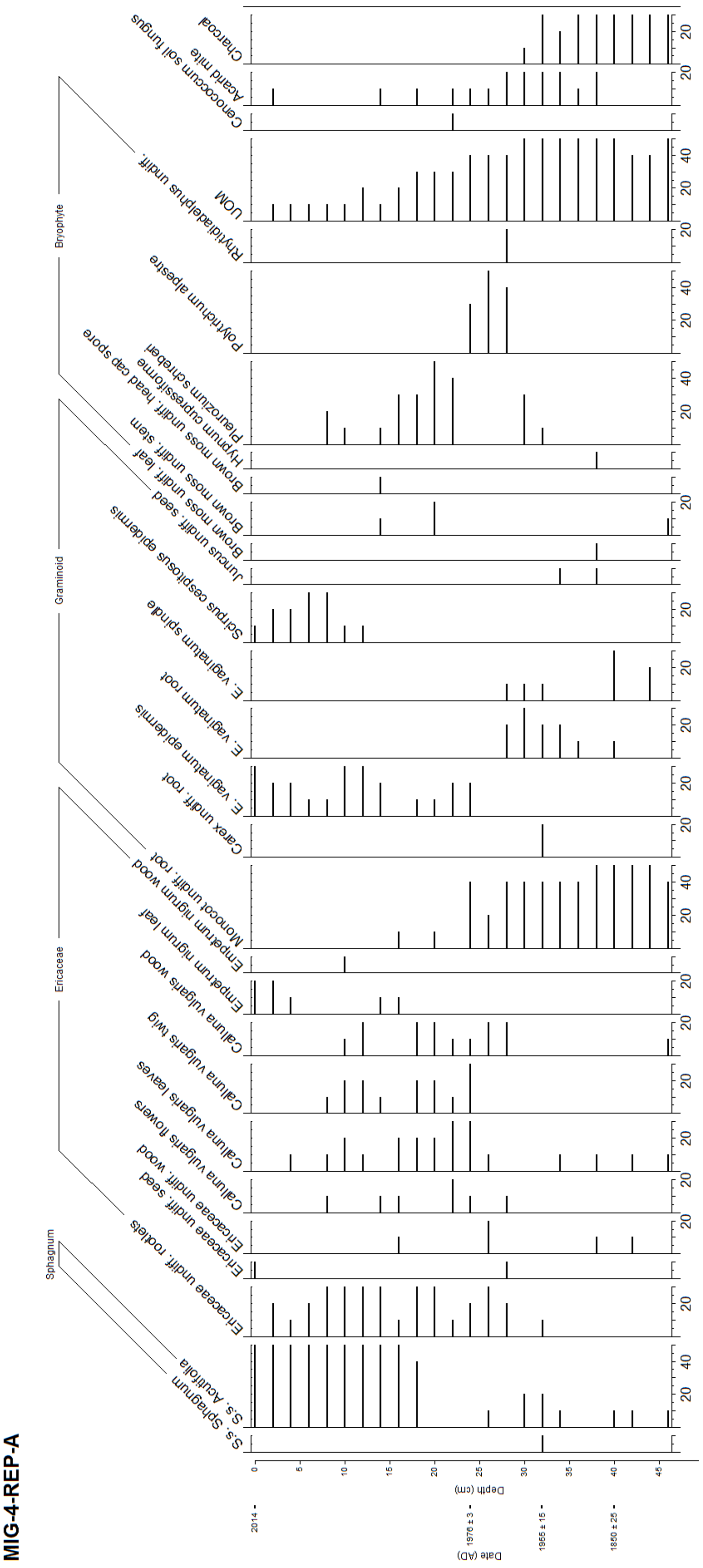

MIG-3-REP-A
Date (AD)
2014
1978 ± 3
1965 ± 15
1850 ± 25
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
S.s. Cuspidata
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. seed
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Empetrum nigrum twig
Erica tetralix leaf
Erica tetralix twig
Dicot undiff. wood
Monocot undiff. leaf
Monocot undiff. epidermis
Monocot undiff. root
Eriophorum undiff. node
E. angustifolium epidermis
E. vaginatum epidermis
E. vaginatum seed
E. vaginatum spindle
E. vaginatum root
Rhynchospora alba epidermis
Juncus undiff. seed
Brown moss undiff. leaf
Brown moss undiff. head cap
Aulacomnium palustre
Plagiothecium laetum
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
UOM
Cenococcum soil fungus
Acari/mite
Charcoal
20
40


MIG-3-REP-B
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. seed
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Empetrum nigrum twig
Erica tetralix leaf
Erica tetralix wood
Monocot undiff. root
Eriophorum undiff. epidermis
E. vaginatum epidermis
E. vaginatum root
E. vaginatum seed
E. vaginatum spindle
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Dicranum undiff.
Dicranum scoparium
Hypnum cupressiforme
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
Racomitrium lanuginosum
Odontoschisma sphagni
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Date (AD)
2014
1976 ± 3
1850 ± 25

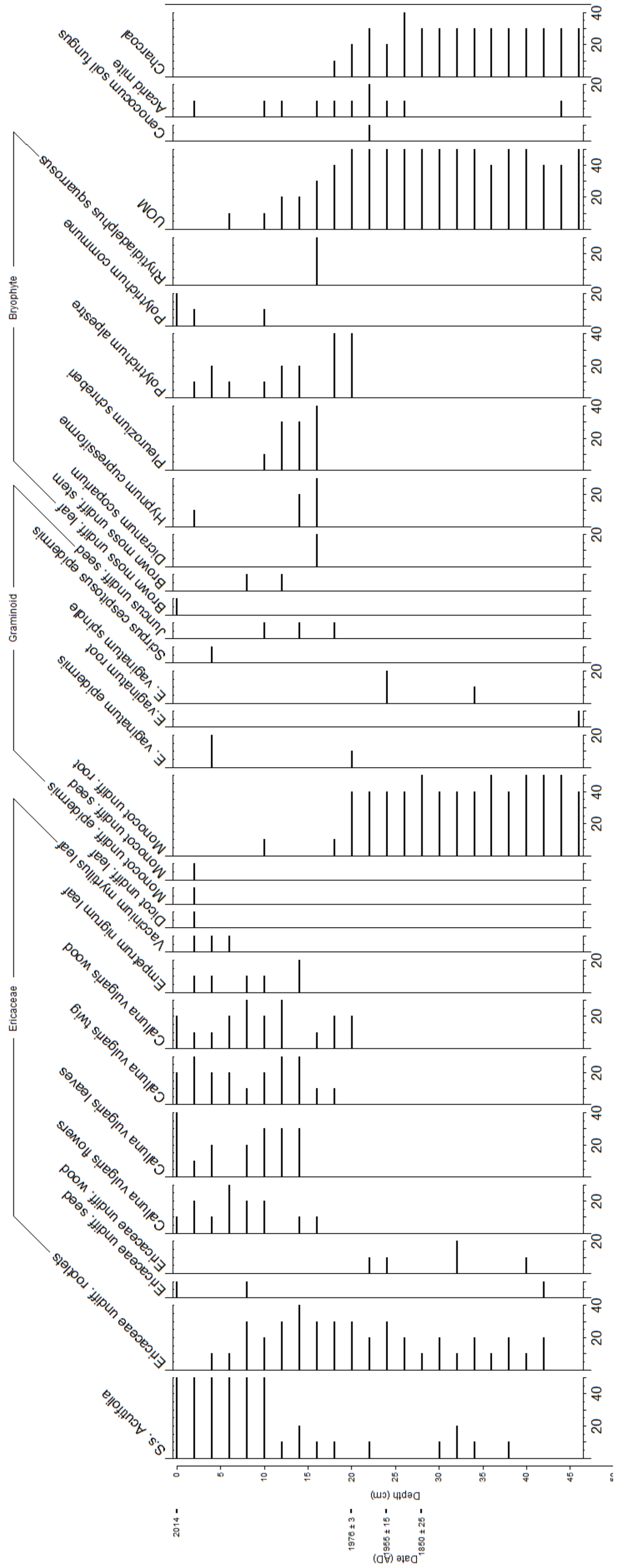
MIG-3-REP-C


Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
S.s. Cuspidata
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seeds
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Empetrum nigrum twig
Erica tetralix leaf
Erica tetralix twig
Monocot undiff. root
Eriophorum undiff. epidermis
Eriophorum undiff. node
E. angustifolium epidermis
E. vaginatum epidermis
E. vaginatum root
E. vaginatum seed
E. vaginatum spindle
Scirpus cespitosus epidermis
Hypnum cupressiforme
Pleurozium schreberi
Odontoschisma sphagni
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Date (AD)
2014
1976 ± 3
1955 ± 15
1850 ± 25
20
40

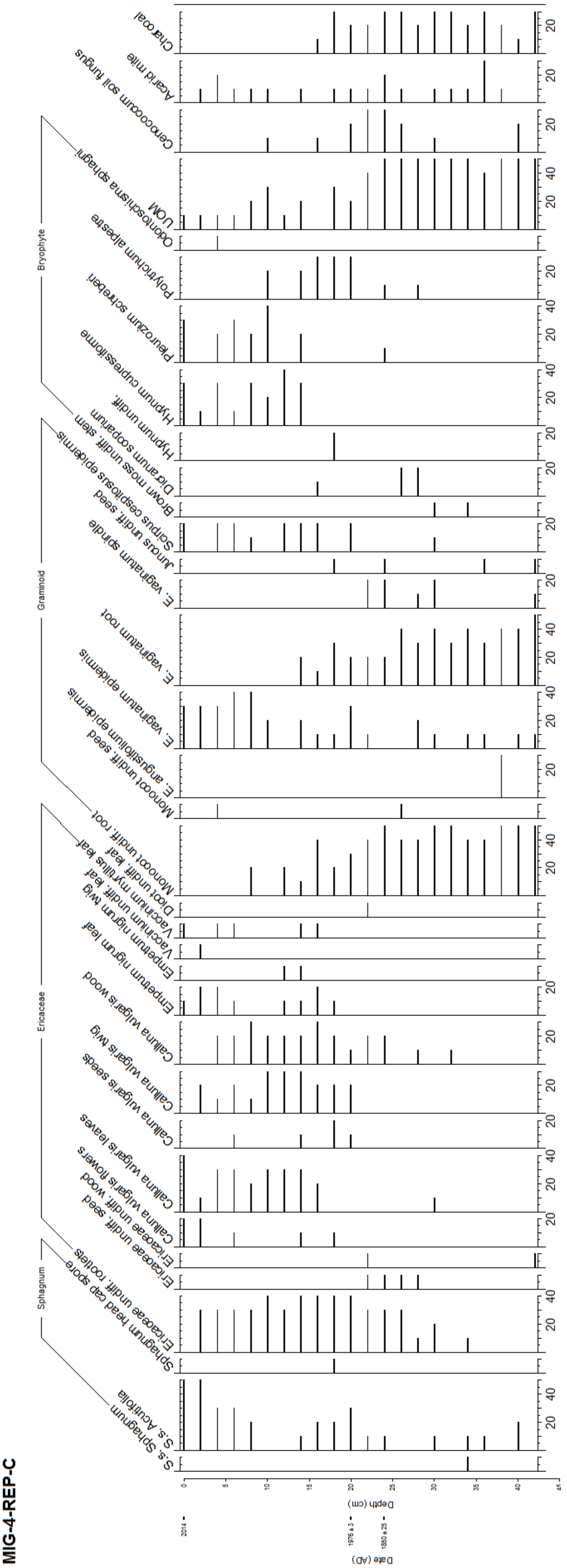
Appendix A


MIG-4-REP-A
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
Ericaceae undiff. rootlets
Ericaceae undiff. seed
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Empetrum nigrum wood
Monocot undiff. root
Carex undiff. root
E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
Scirpus cespitosus epidermis
Juncus undiff. seed
Brown moss undiff. leaf
Brown moss undiff. stem
Brown moss undiff. head cap spore
Hypnum cupressiforme
Pleurozium schreberi
Polytrichum alpestre
Rhytidiadelphus undiff.
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40
Depth (cm)
0
5
10
15
20
25
30
35
40
45
Date (AD)
2014
1976 ± 3
1955 ± 15
1850 ± 25

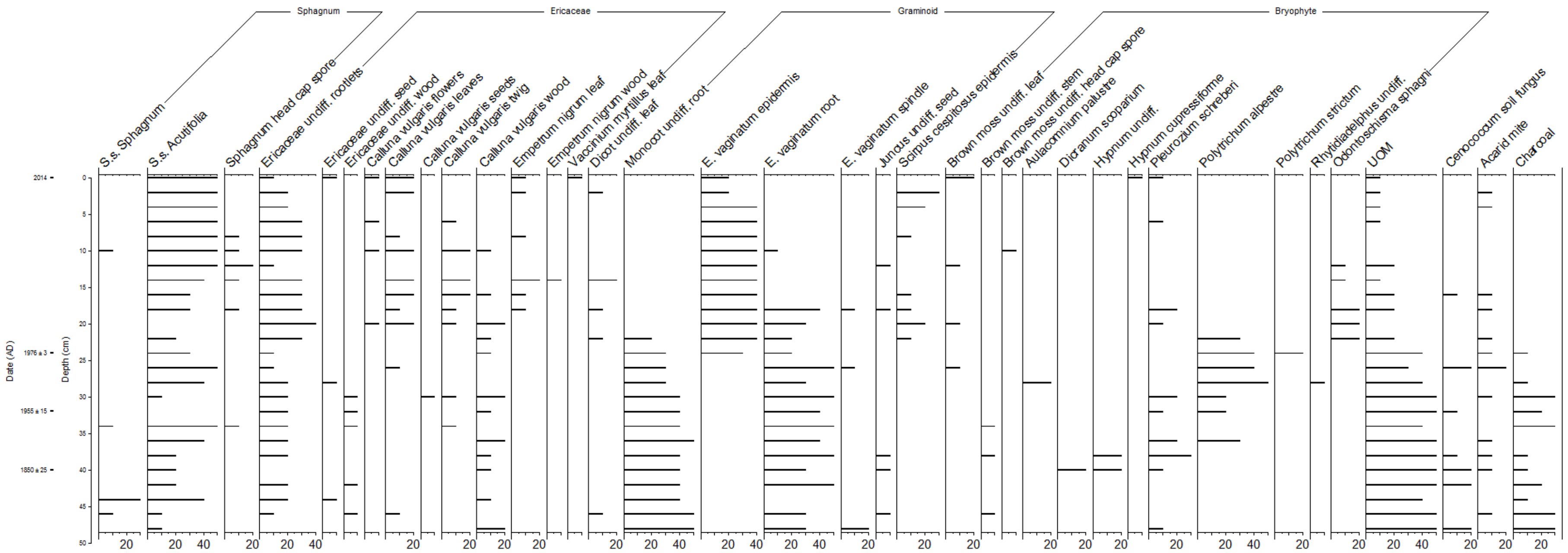
MIG-4-REP-B


Ericaceae
Graminoid
Bryophyte
S.s. Acutifolia
Ericaceae undiff. rootlets
Ericaceae undiff. seed
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Vaccinium myrtilus leaf
Dicot undiff. leaf
Monocot undiff. leaf
Monocot undiff. epidermis
Monocot undiff. seed
Monocot undiff. root
E. vaginatum epidermis
E.vaginatum root
E. vaginatum spindle
Scirpus cespitosus epidermis
Juncus undiff. seed
Brown moss undiff. leaf
Brown moss undiff. stem
Dicranum scoparium
Hypnum cupressiforme
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
Rhytidiadelphus squarrosus
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Date (AD)
2014
1976 ± 3
1955 ± 15
1850 ± 25

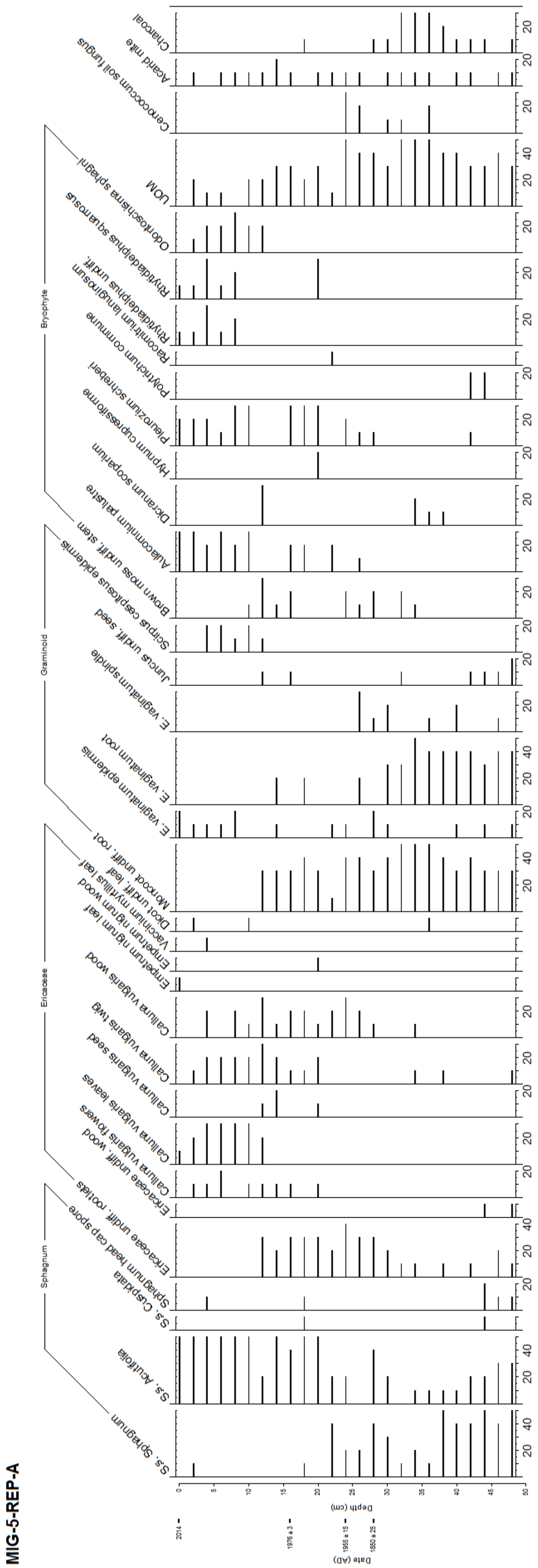
Appendix A

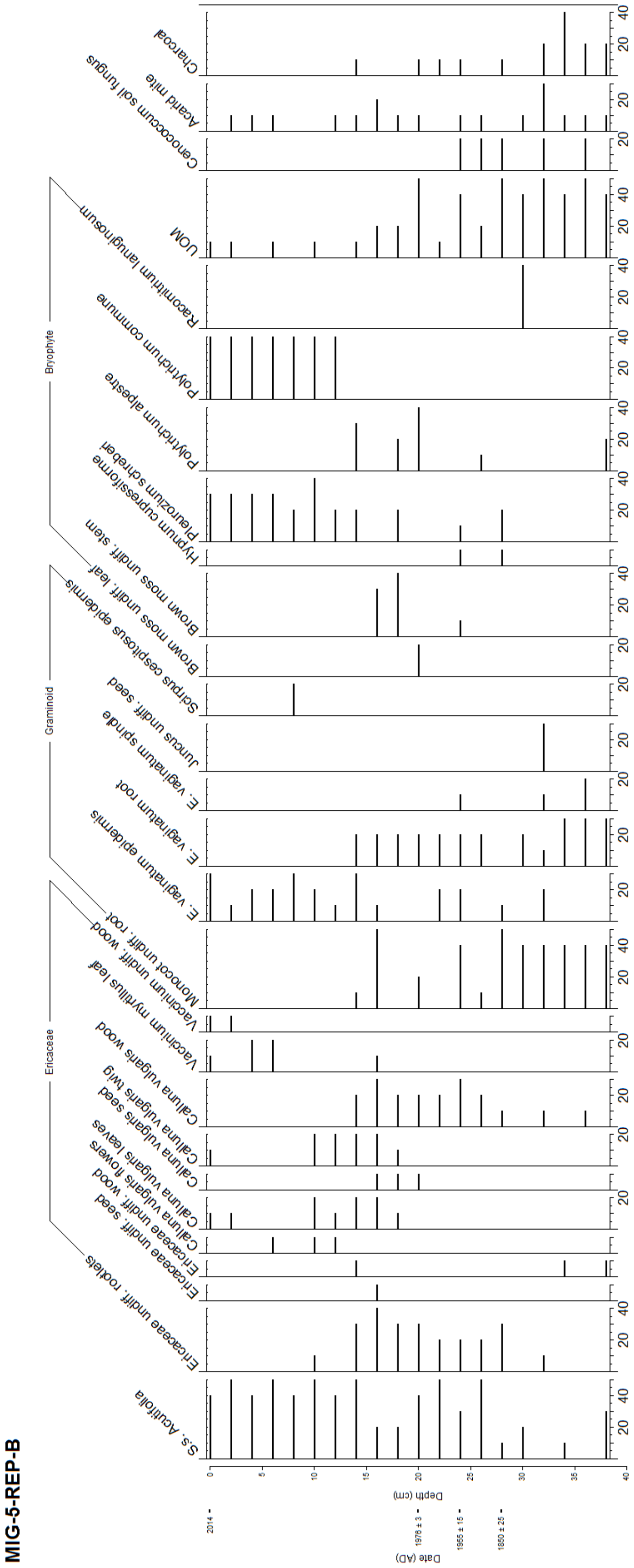

MIG-4-REP-C
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. seed
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seeds
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Empetrum nigrum twig
Vaccinium undiff. leaf
Vaccinium myrtillus leaf
Dicot undiff. leaf
Monocot undiff. root
Monocot undiff. seed
E. angustifolium epidermis
E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
Juncus undiff. seed
Scirpus cespitosus epidermis
Brown moss undiff. stem
Dicranum scoparium
Hypnum undiff.
Hypnum cupressiforme
Pleurozium schreberi
Polytrichum alpestre
Odontoschisma sphagni
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40
0
5
10
15
20
25
30
35
40
45
Depth (cm)
2014
1976 ± 3
1850 ± 25
Date (AD)

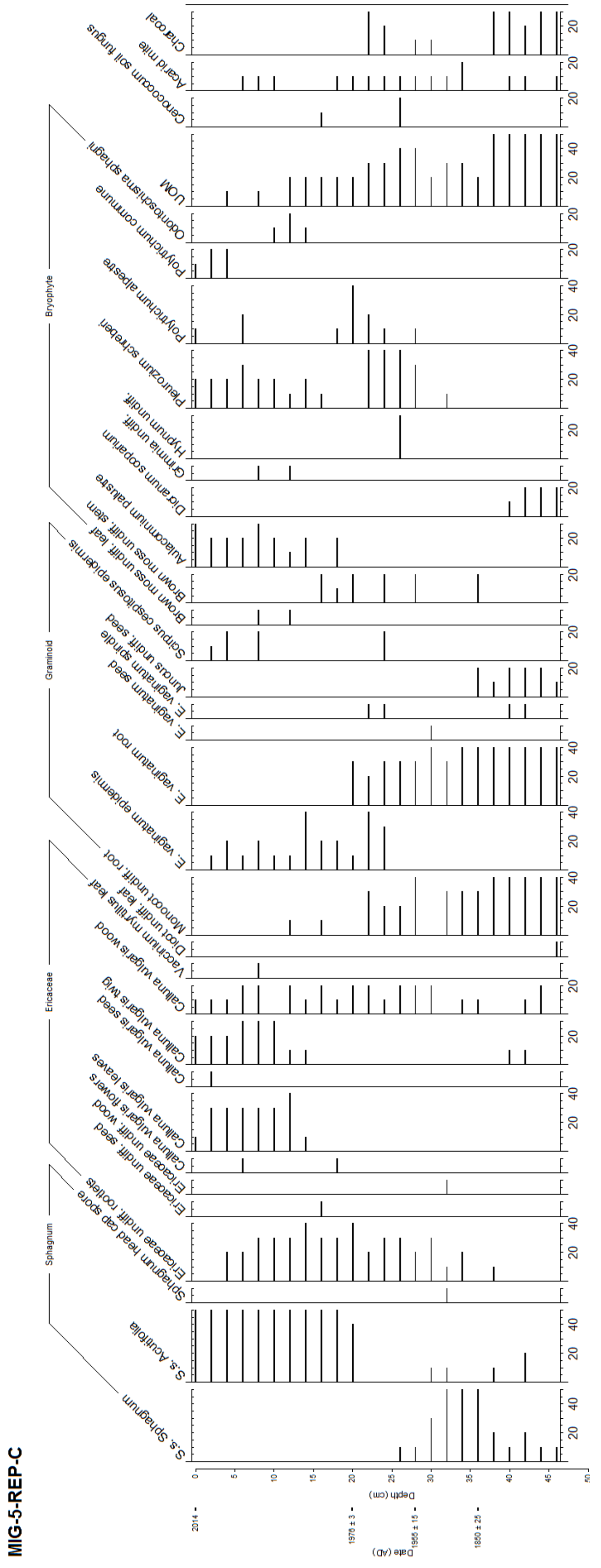
MIG-4-REP-D


Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. seed
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seeds
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Empetrum nigrum wood
Vaccinium myrtillus leaf
Dicot undiff. leaf
Monocot undiff. root
E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
Juncus undiff. seed
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Brown moss undiff. head cap spore
Aulacomnium palustre
Dicranum scoparium
Hypnum undiff.
Hypnum cupressiforme
Pleurozium schreberi
Polytrichum alpestre
Polytrichum strictum
Rhytidiadelphus undiff.
Odontoschisma sphagni
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
Date (AD)
2014
1976 ± 3
1955 ± 15
1850 ± 25
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
20
40

Appendix A


MIG-5-REP-A
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
S.s. Cuspidata
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seed
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Empetrum nigrum wood
Vaccinium myrtillus leaf
Dicot undiff. leaf
Monocot undiff. root
E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
Juncus undiff. seed
Scirpus cespitosus epidermis
Brown moss undiff. stem
Aulacomnium palustre
Dicranum scoparium
Hypnum cupressiforme
Pleurozium schreberi
Polytrichum commune
Racomitrium lanuginosum
Rhytidiadelphus undiff.
Rhytidiadelphus squarrosus
Odontoschisma sphagni
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40
0
5
10
15
20
25
30
35
40
45
50
Depth (cm)
2014
1976 ± 3
1955 ± 15
1850 ± 25
Date (AD)

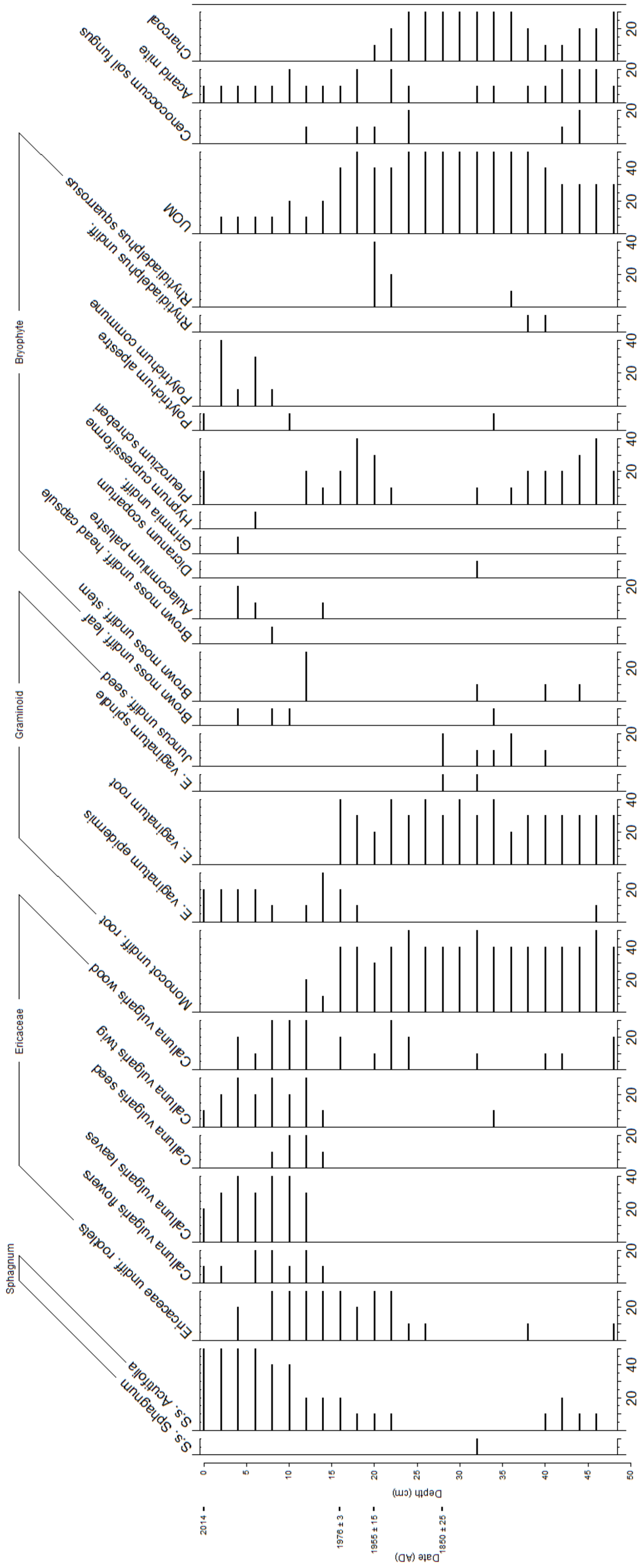

MIG-5-REP-B
Date (AD)
2014
1976 ± 3
1955 ± 15
1850 ± 25
Depth (cm)
0
5
10
15
20
25
30
35
40
Ericaceae
Graminoid
Bryophyte
S.s. Acutifolia
Ericaceae undiff. rootlets
Ericaceae undiff. seed
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seed
Calluna vulgaris twig
Calluna vulgaris wood
Vaccinium myrtillus leaf
Vaccinium undiff. wood
Monocot undiff. root
E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
Juncus undiff. seed
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Hypnum cupressiforme
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
Racomitrum lanuginosum
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40



MIG-5-REP-C

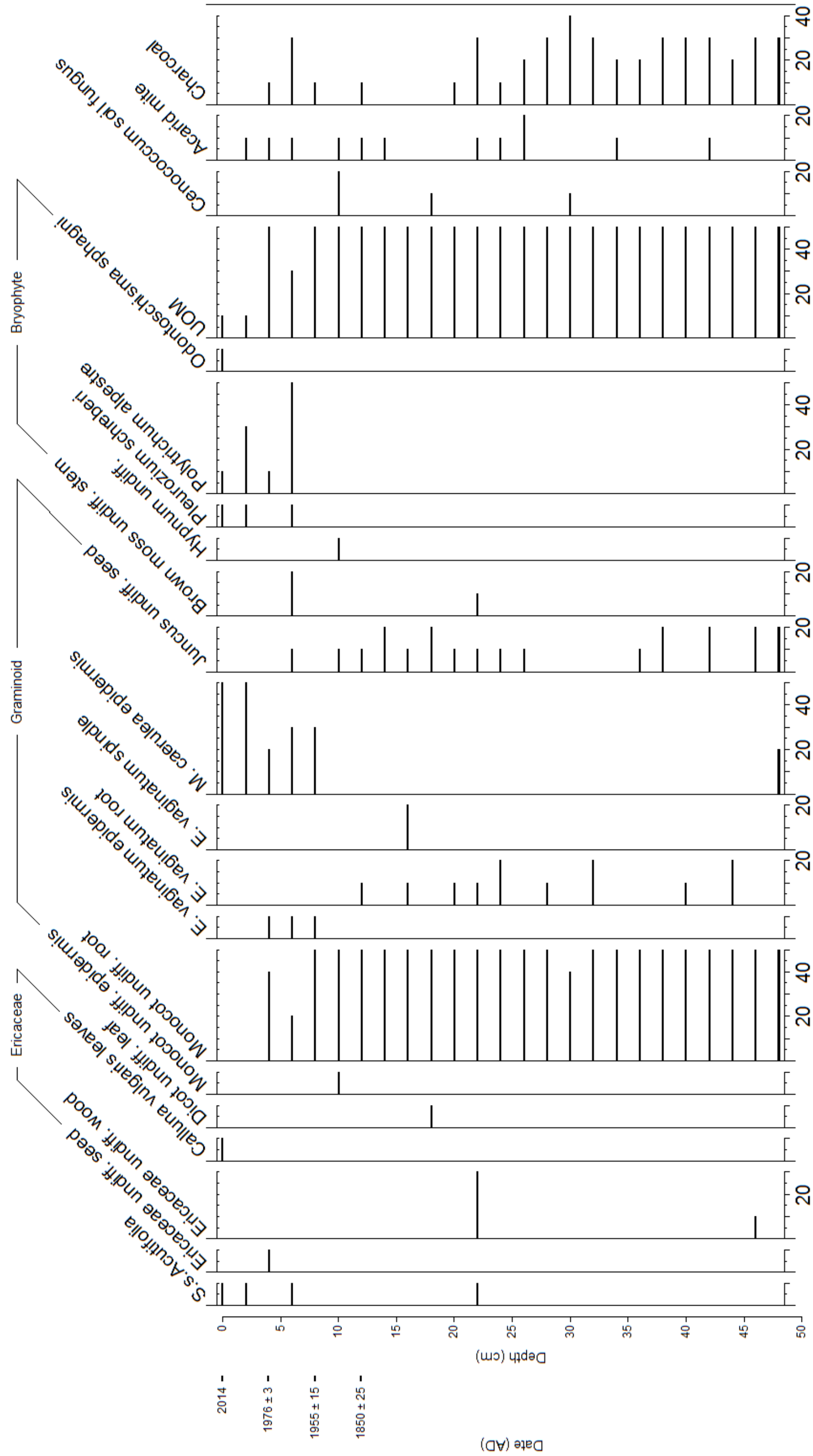
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. seed
Calluna undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seed
Calluna vulgaris twig
Calluna vulgaris wood
Vaccinium myrtillus leaf
Dicot undiff. leaf
Monocot undiff. root
E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
E. vaginatum seed
Juncus undiff. seed
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Aulacomnium palustre
Dicranum scoparium
Grimmia undiff.
Hypnum undiff.
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
Odontoschisma sphagni
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40
0
5
10
15
20
25
30
35
40
45
50
Depth (cm)
2014
1976 ± 3
1955 ± 15
1850 ± 25
Date (AD)

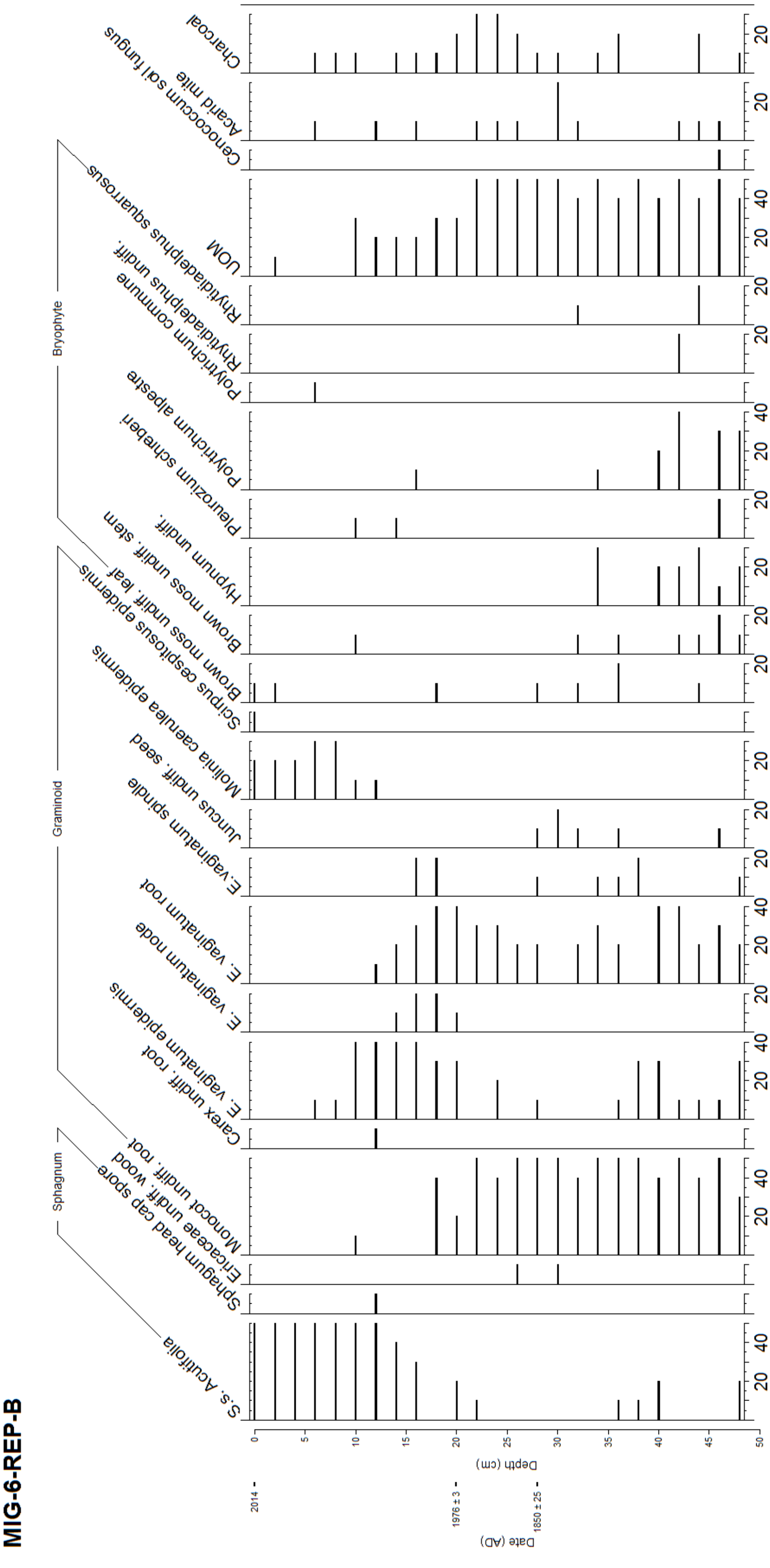
MIG-5-REP-D


S.s. Sphagnum
S.s. Acutifolia
Ericaceae undiff. rootlets
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seed
Calluna vulgaris twig
Calluna vulgaris wood
Monocot undiff. root
E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
Juncus undiff. seed
Brown moss undiff. leaf
Brown moss undiff. stem
Brown moss undiff. head capsule
Aulacomnium palustre
Dicranum scoparium
Grimmia undiff.
Hypnum cupressiforme
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
Rhytidiadelphus undiff.
Rhytidiadelphus squarrosus
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
Sphagnum
Ericaceae
Graminoid
Bryophyte
20
40
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Date (AD)
2014
1976 ± 3
1955 ± 15
1850 ± 25

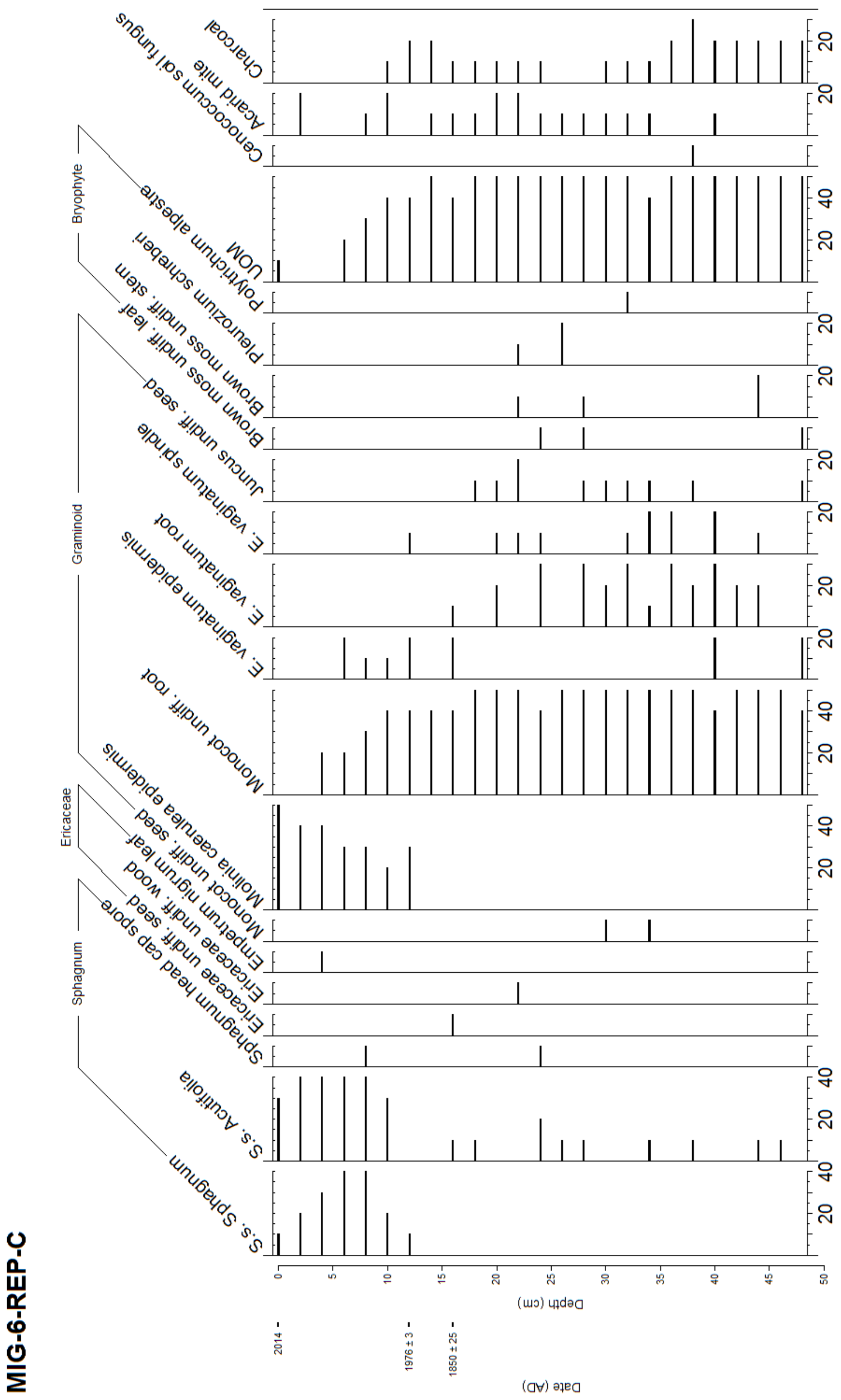
Appendix A

MIG-6-REP-A

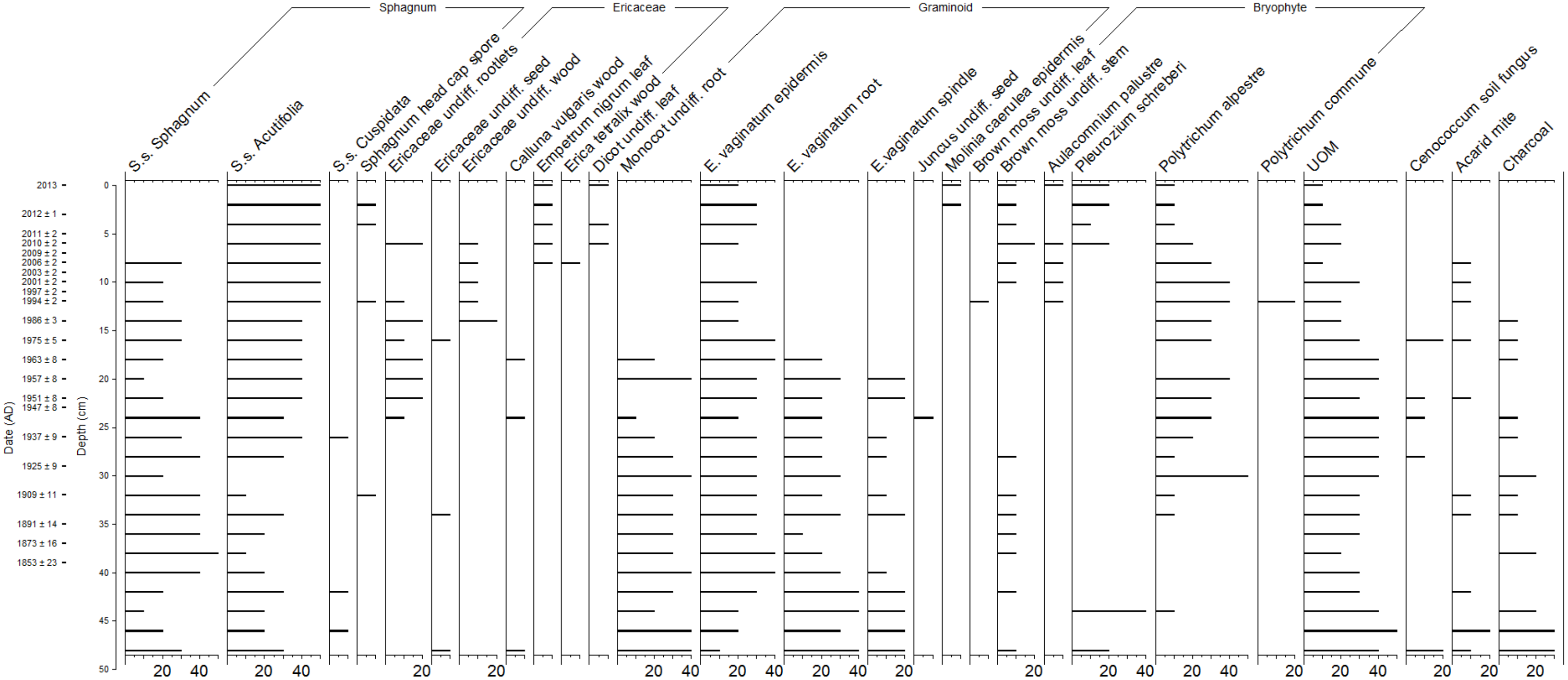



MIG-6-REP-B
Sphagnum
Graminoid
Bryophyte
S.s. Acutifolia
Sphagnum head cap spore
Ericaceae undiff. wood
Monocot undiff. root
Carex undiff. root
E. vaginatum epidermis
E. vaginatum node
E. vaginatum root
E.vaginatum spindle
Juncus undiff. seed
Molinia caerulea epidermis
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Hypnum undiff.
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
Rhytidiadelphus undiff.
Rhytidiadelphus squarrosus
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Date (AD)
2014
1976 ± 3
1850 ± 25
20
40

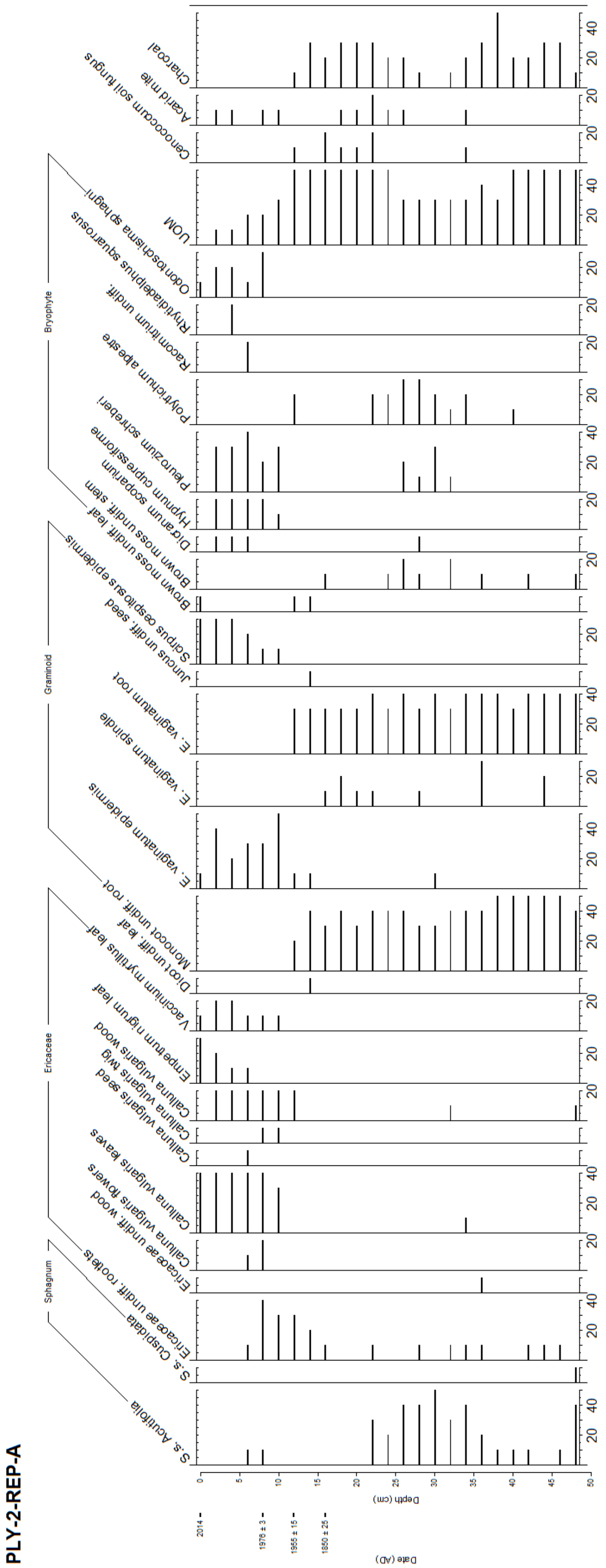
Appendix A

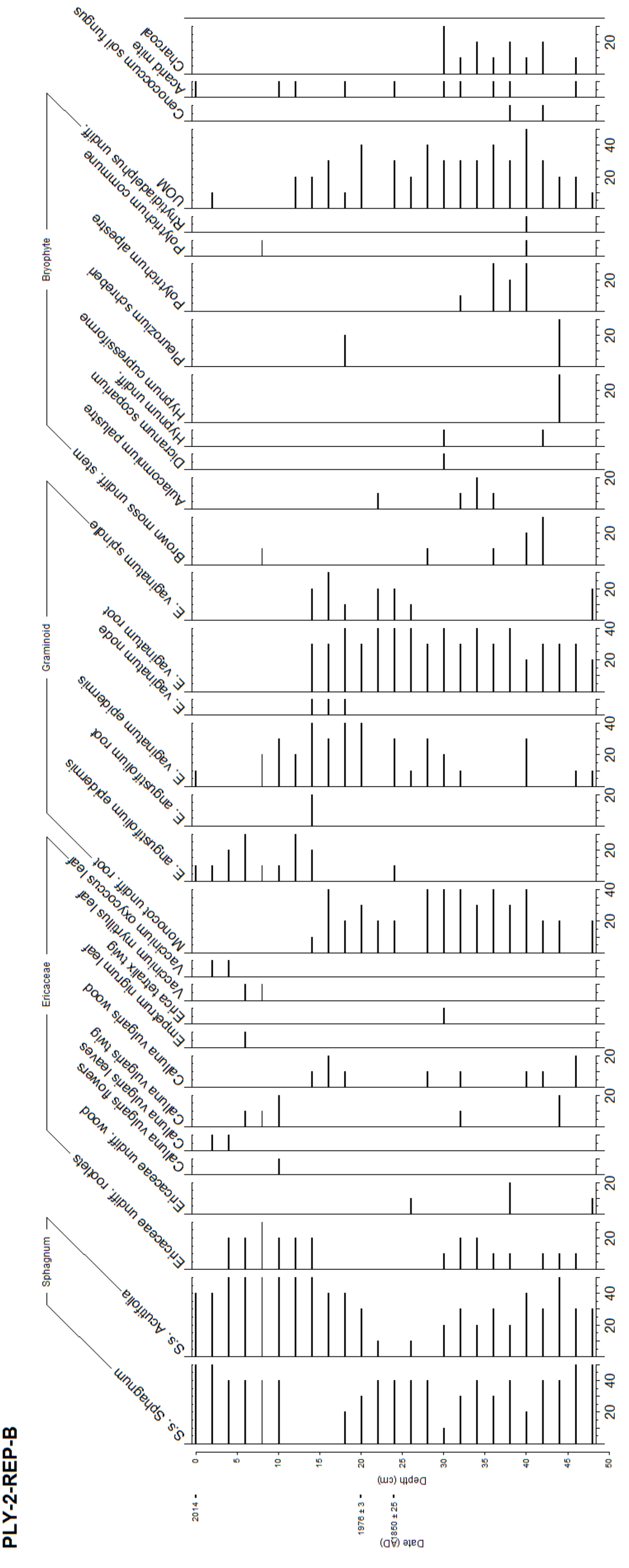

MIG-6-REP-C
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
Sphagnum head cap spore
Ericaceae undiff. seed
Ericaceae undiff. wood
Empetrum nigrum leaf
Monocot undiff. seed
Molinia caerulea epidermis
Monocot undiff. root
E. vaginatum root
E. vaginatum epidermis
E. vaginatum spindle
Juncus undiff. seed
Brown moss undiff. leaf
Brown moss undiff. stem
Pleurozium schreberi
Polytrichum alpestre
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
Date (AD)
2014
1976 ± 3
1850 ± 25

PLY-1-M

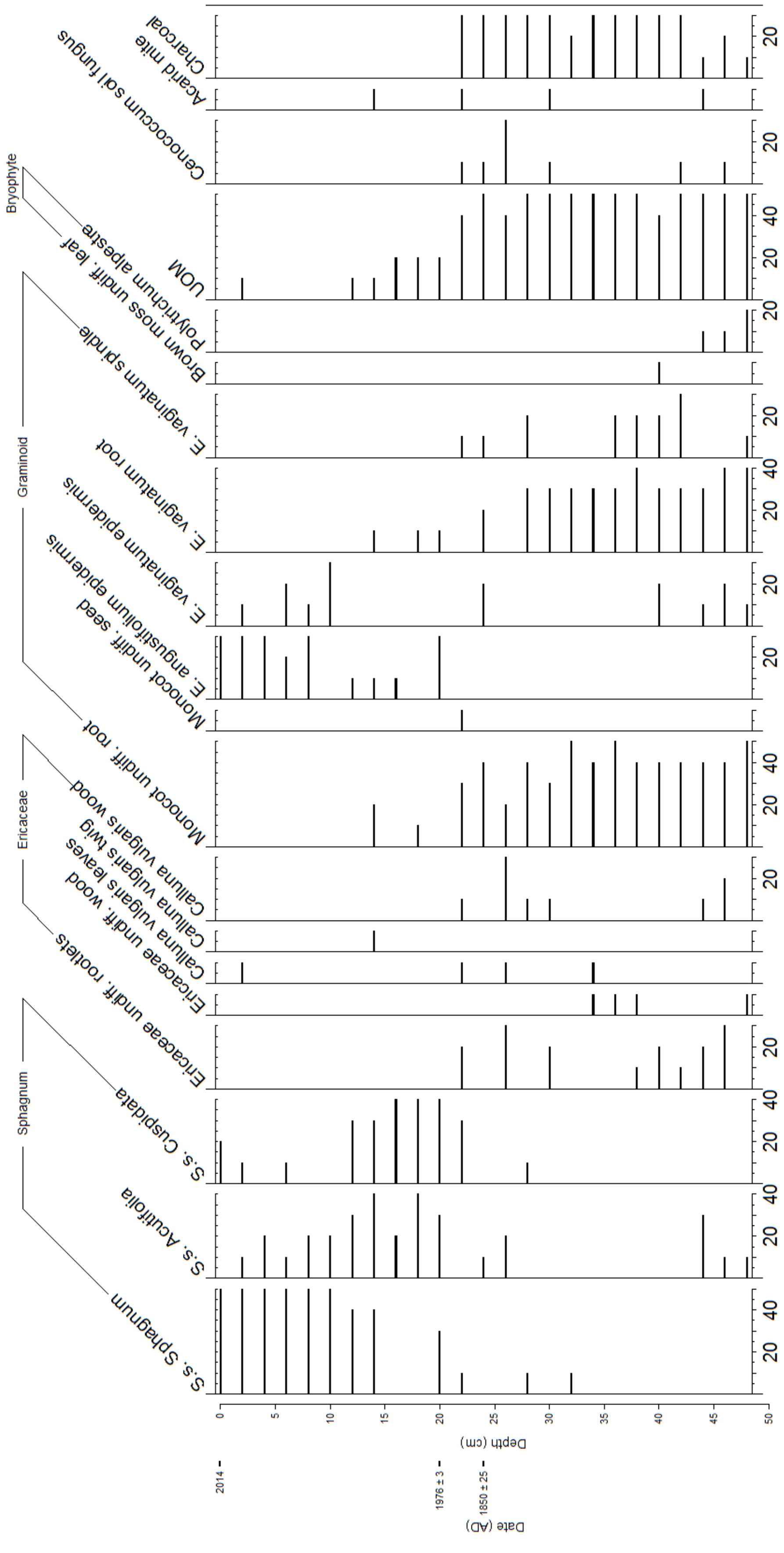

Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
S.s. Cuspidata
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. seed
Ericaceae undiff. wood
Calluna vulgaris wood
Empetrum nigrum leaf
Erica tetralix wood
Dicot undiff. leaf
Monocot undiff. root
E. vaginatum epidermis
E. vaginatum root
E.vaginatum spindle
Juncus undiff. seed
Molinia caerulea epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Aulacomnium palustre
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
Date (AD)
2013
2012 ± 1
2011 ± 2
2010 ± 2
2009 ± 2
2006 ± 2
2003 ± 2
2001 ± 2
1997 ± 2
1994 ± 2
1986 ± 3
1975 ± 5
1963 ± 8
1957 ± 8
1951 ± 8
1947 ± 8
1937 ± 9
1925 ± 9
1909 ± 11
1891 ± 14
1873 ± 16
1853 ± 23
Depth (cm)
0
5
10
15
20
25
30
35
40
45
50
20
40

Appendix A

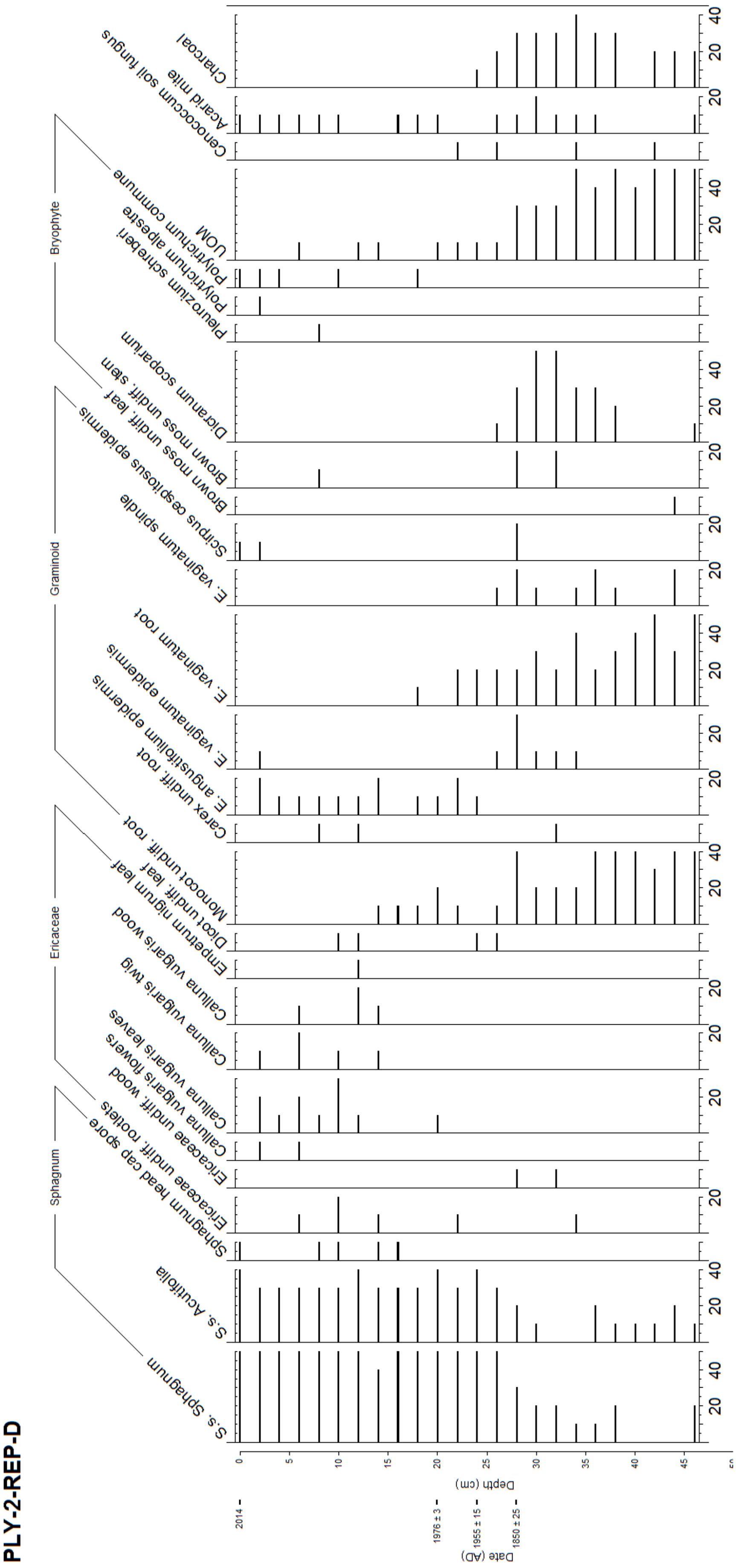

PLY-2-REP-A
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Acutifolia
S.s. Cuspidata
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris seed
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Vaccinium myrtilus leaf
Dicot undiff. leaf
Monocot undiff. root
E. vaginatum epidermis
E. vaginatum spindle
E. vaginatum root
Juncus undiff. seed
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Dicranum scoparium
Hypnum cupressiforme
Pleurozium schreberi
Polytrichum alpestre
Racomitrium undiff.
Rhytidiadelphus squarrosus
Odontoschisma sphagni
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40
0
5
10
15
20
25
30
35
40
45
50
Depth (cm)
2014
1976 ± 3
1955 ± 15
1850 ± 25
Date (AD)


PLY-2-REP-B
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Erica tetralix twig
Vaccinium myrtillus leaf
Vaccinium oxycoccus leaf
Monocot undiff. root
E. angustifolium epidermis
E. angustifolium root
E. vaginatum epidermis
E. vaginatum node
E. vaginatum root
E. vaginatum spindle
Brown moss undiff. stem
Aulacomnium palustre
Dicranum scoparium
Hypnum undiff.
Hypnum cupressiforme
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
Rhytidiadelphus undiff.
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
Depth (cm)
Date (AD)
2014
1976 ± 3
1850 ± 25
0
5
10
15
20
25
30
35
40
45
50
20
40

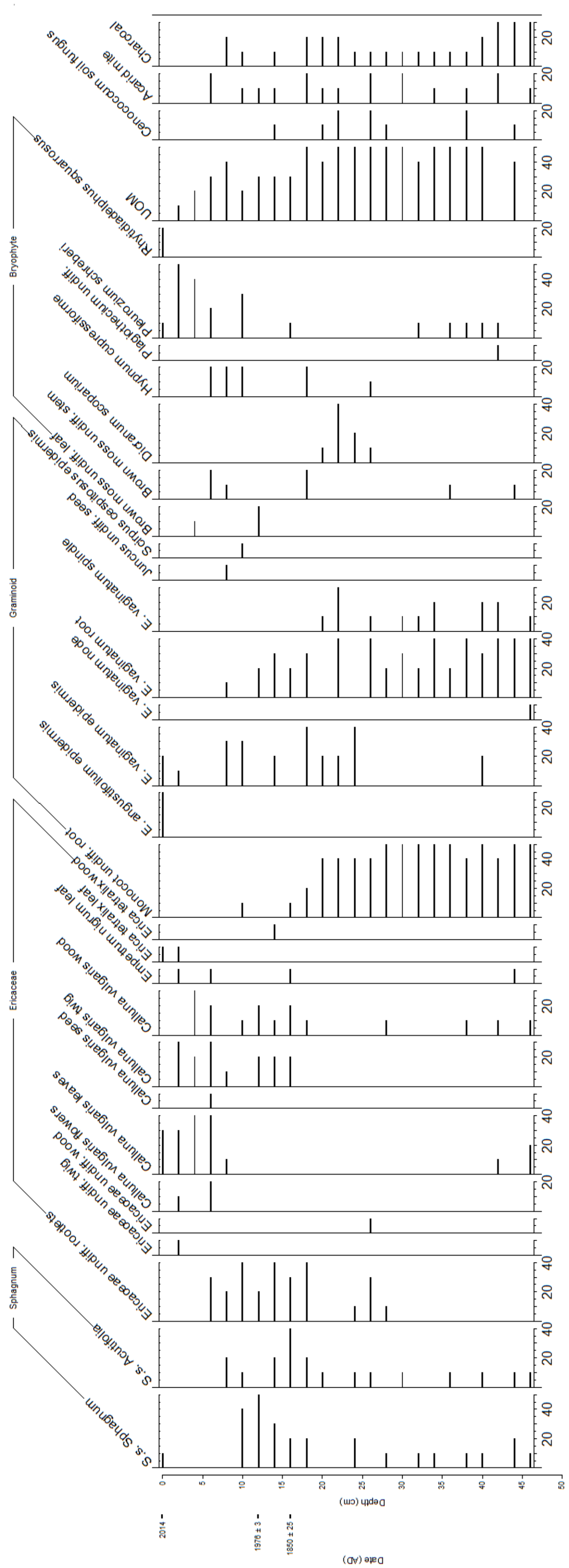
Appendix A



PLY-2-REP-C

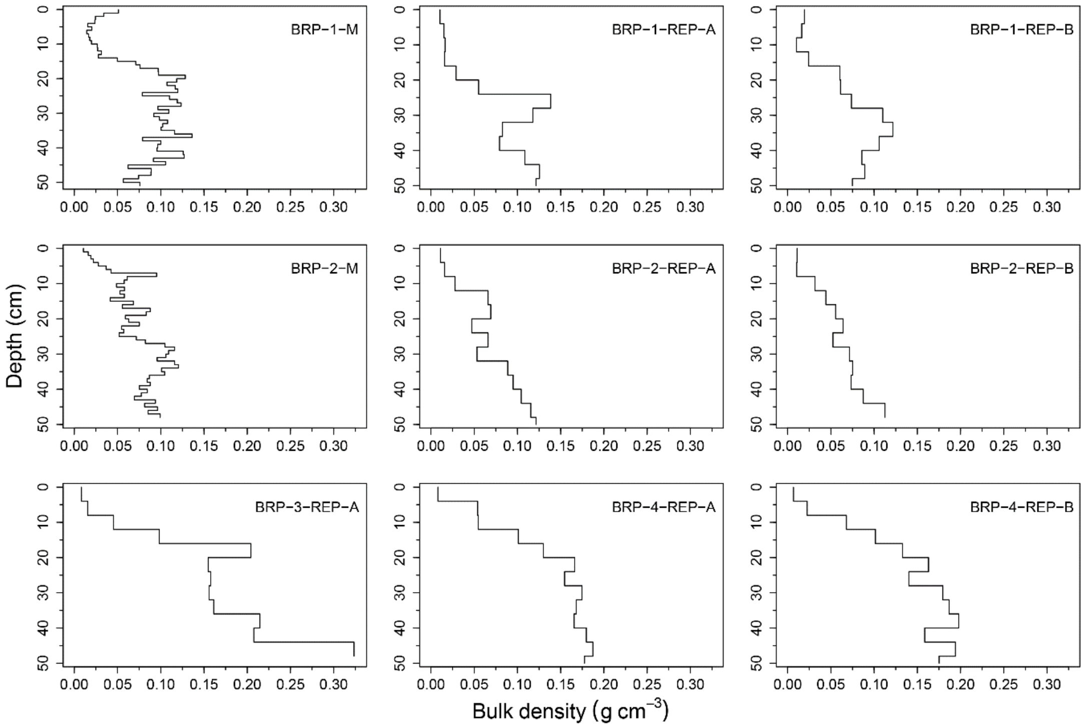

PLY-2-REP-D
Sphagnum
Ericaceae
Graminoid
Bryophyte
S.s. Sphagnum
S.s. Acutifolia
Sphagnum head cap spore
Ericaceae undiff. rootlets
Ericaceae undiff. wood
Calluna vulgaris flowers
Calluna vulgaris leaves
Calluna vulgaris twig
Calluna vulgaris wood
Empetrum nigrum leaf
Dicot undiff. leaf
Monocot undiff. root
Carex undiff. root
E. angustifolium epidermis
E. vaginatum epidermis
E. vaginatum root
E. vaginatum spindle
Scirpus cespitosus epidermis
Brown moss undiff. leaf
Brown moss undiff. stem
Dicranum scoparium
Pleurozium schreberi
Polytrichum alpestre
Polytrichum commune
UOM
Cenococcum soil fungus
Acarid mite
Charcoal
20
40
Depth (cm)
0
5
10
15
20
25
30
35
40
45
Date (AD)
2014
1976 ± 3
1955 ± 15
1850 ± 25

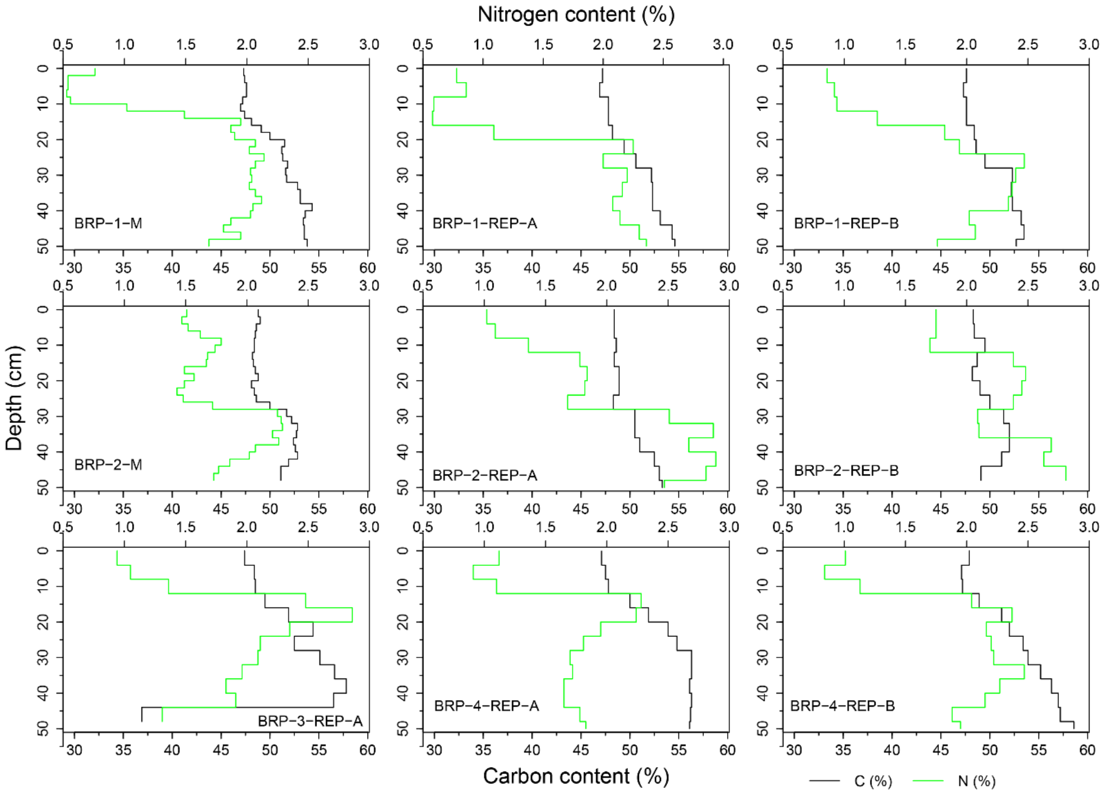
Appendix A



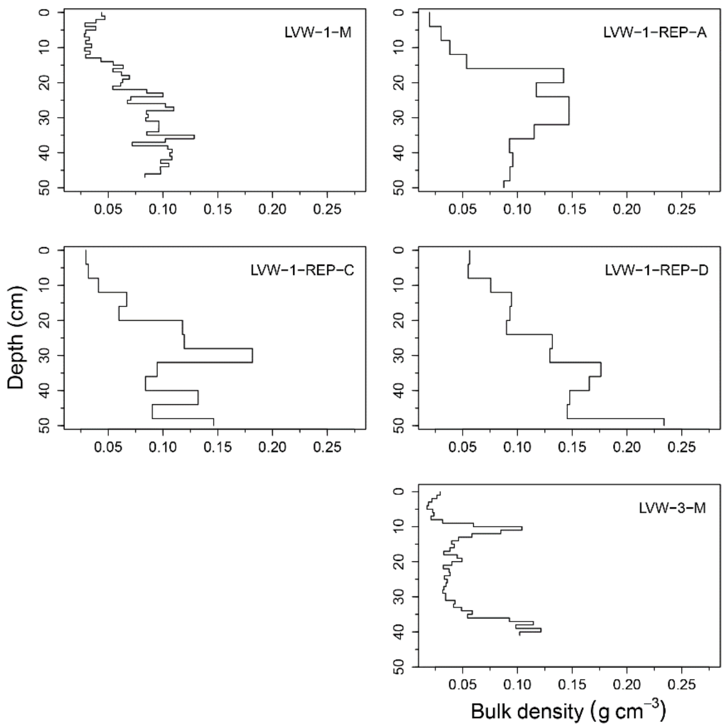
PLY-2-REP-E

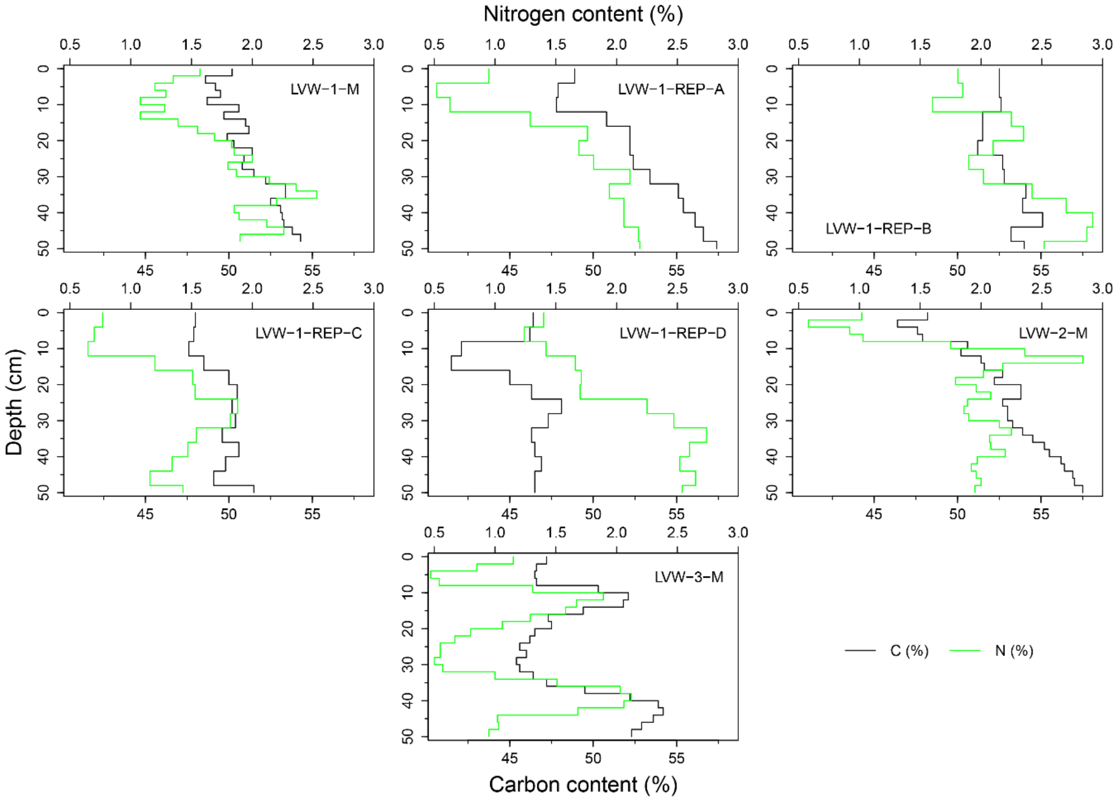
Appendix B

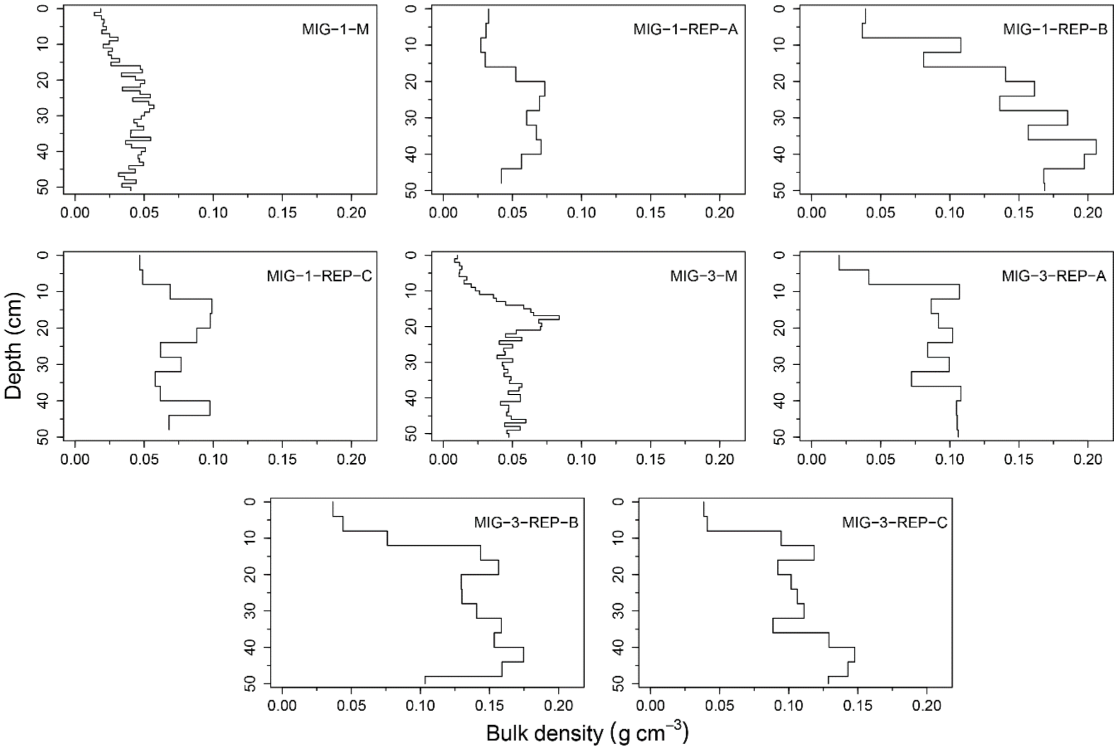

BRP-1-M
BRP-1-REP-A
BRP-1-REP-B
BRP-2-M
BRP-2-REP-A
BRP-2-REP-B
BRP-3-REP-A
BRP-4-REP-A
BRP-4-REP-B
Depth (cm)
0
10
20
30
40
50
0.00
0.05
0.10
0.15
0.20
0.25
0.30
Bulk density (g cm^{-3})

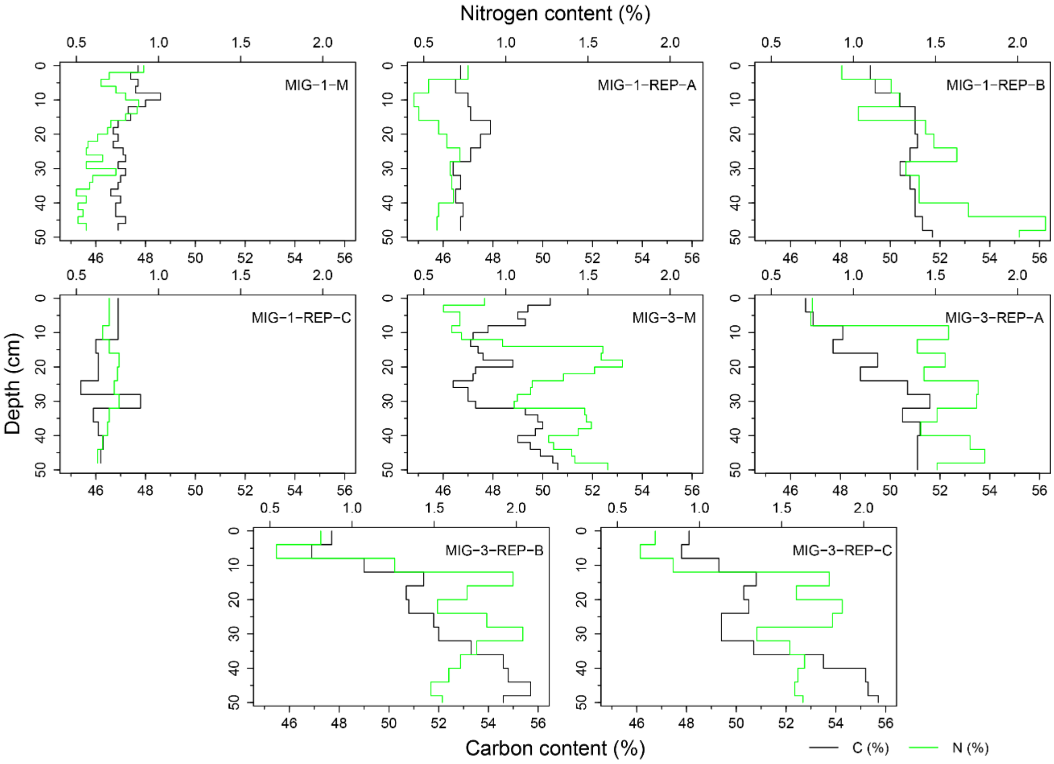

Nitrogen content (%)
0.5
1.0
1.5
2.0
2.5
3.0
BRP-1-M
BRP-1-REP-A
BRP-1-REP-B
BRP-2-M
BRP-2-REP-A
BRP-2-REP-B
BRP-3-REP-A
BRP-4-REP-A
BRP-4-REP-B
Depth (cm)
0
10
20
30
40
50
30
35
40
45
50
55
60
Carbon content (%)
C (%)
N (%)

Appendix B

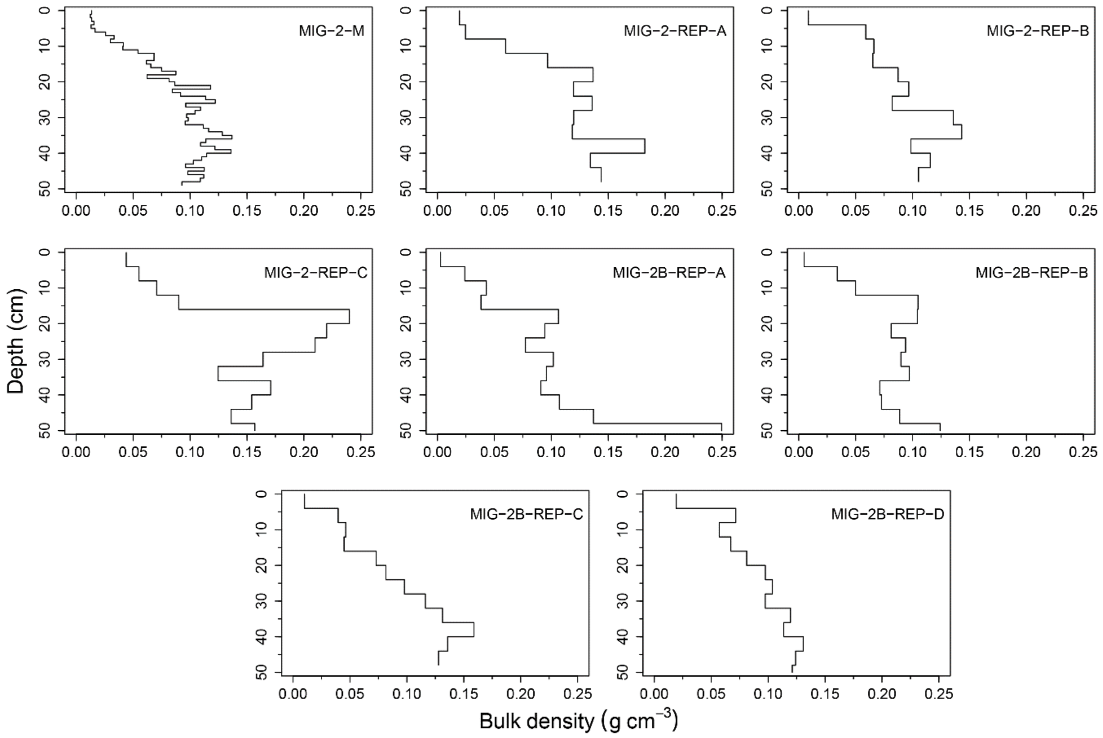

LVW−1−M
LVW−1−REP−A
LVW−1−REP−B
LVW−1−REP−C
LVW−1−REP−D
LVW−2−M
LVW−3−M
Depth (cm)
0
10
20
30
40
50
0.05
0.10
0.15
0.20
0.25
Bulk density (g cm^{-3})

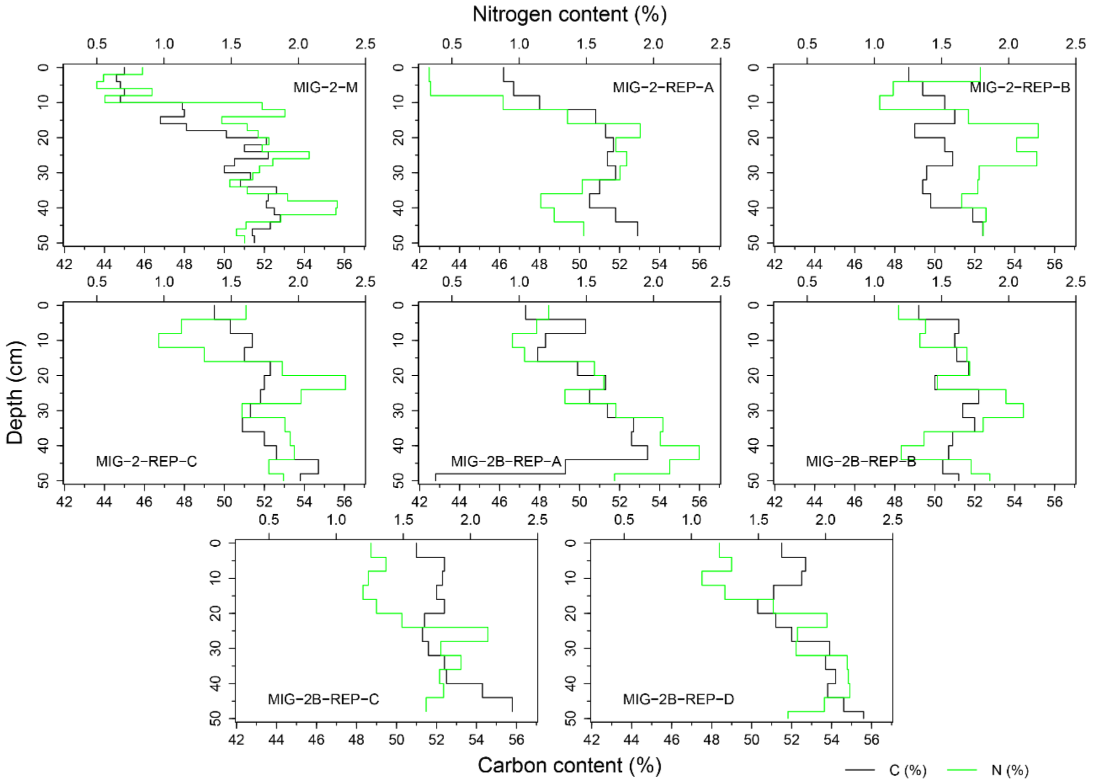

Nitrogen content (%)
0.5
1.0
1.5
2.0
2.5
3.0
LVW−1−M
LVW−1−REP−A
LVW−1−REP−B
LVW−1−REP−C
LVW−1−REP−D
LVW−2−M
LVW−3−M
Depth (cm)
0
10
20
30
40
50
45
50
55
Carbon content (%)
C (%)
N (%)

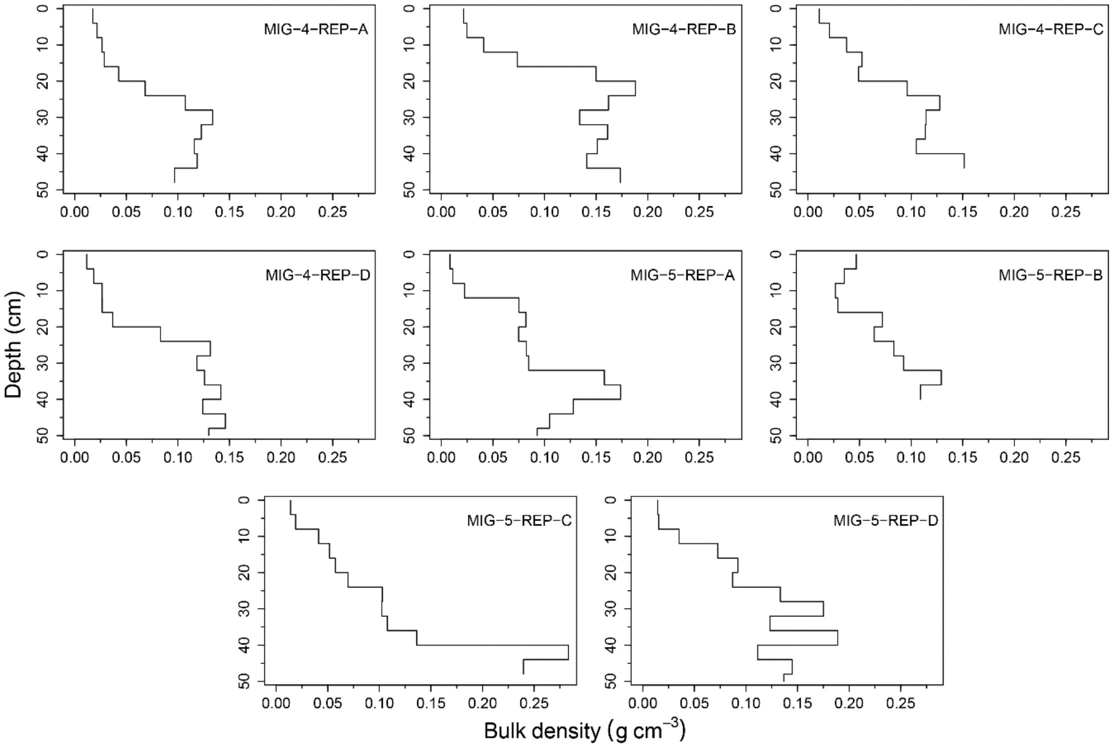

MIG−1−M
MIG−1−REP−A
MIG−1−REP−B
MIG−1−REP−C
MIG−3−M
MIG−3−REP−A
MIG−3−REP−B
MIG−3−REP−C
Depth (cm)
0
10
20
30
40
50
0.00
0.05
0.10
0.15
0.20
Bulk density (g cm^{-3})

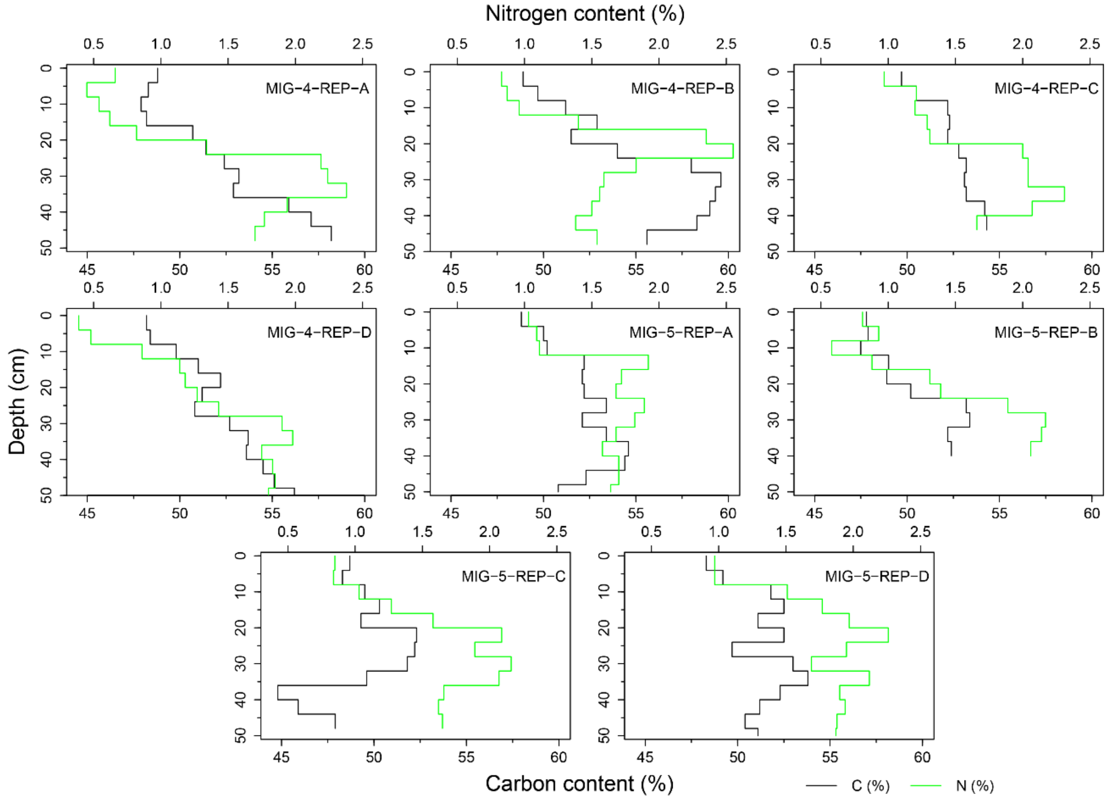

Nitrogen content (%)
0.5
1.0
1.5
2.0
MIG−1−M
MIG−1−REP−A
MIG−1−REP−B
MIG−1−REP−C
MIG−3−M
MIG−3−REP−A
MIG−3−REP−B
MIG−3−REP−C
Depth (cm)
0
10
20
30
40
50
46
48
50
52
54
56
Carbon content (%)
C (%)
N (%)

Appendix B

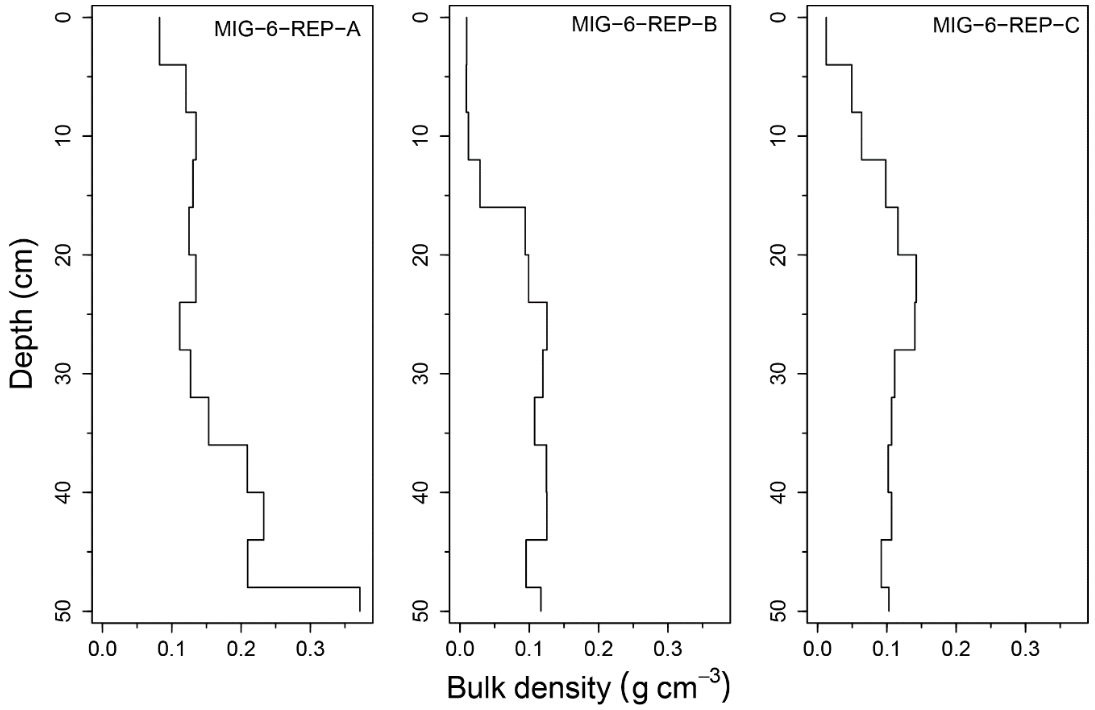

MIG−2−M
MIG−2−REP−A
MIG−2−REP−B
MIG−2−REP−C
MIG−2B−REP−A
MIG−2B−REP−B
MIG−2B−REP−C
MIG−2B−REP−D
Depth (cm)
0
10
20
30
40
50
0.00
0.05
0.10
0.15
0.20
0.25
Bulk density (g cm−3)

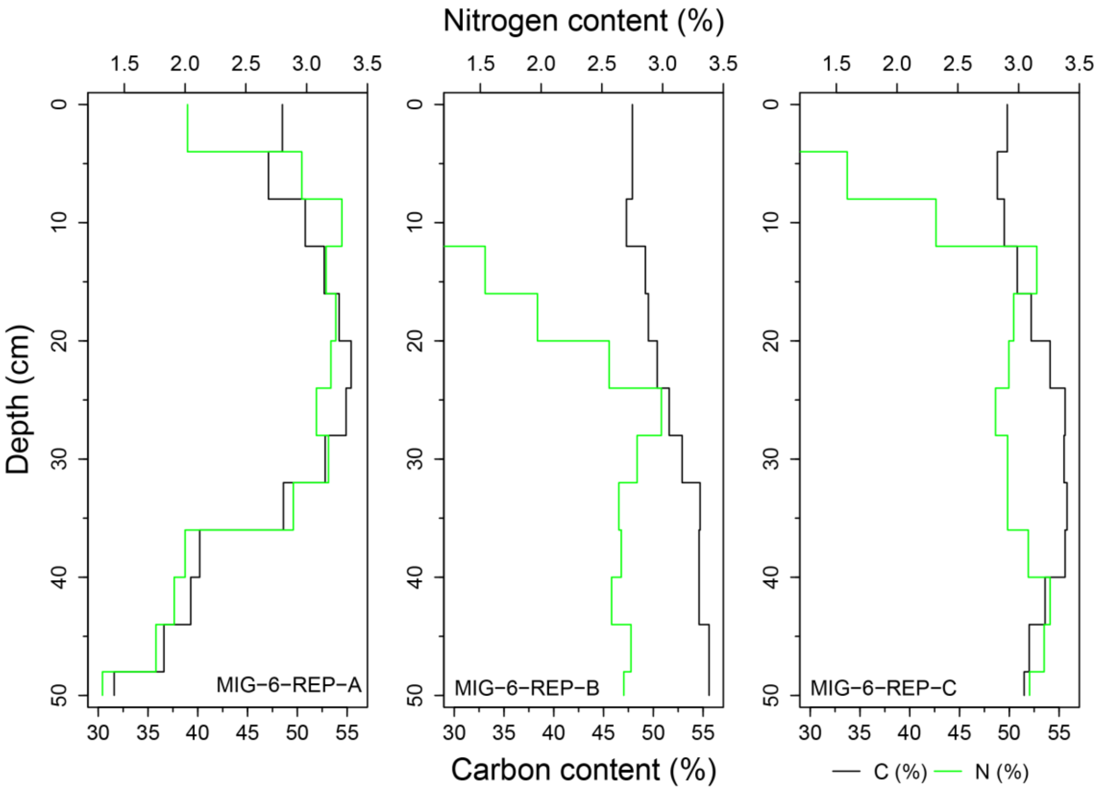

Nitrogen content (%)
0.5
1.0
1.5
2.0
2.5
MIG−2−M
MIG−2−REP−A
MIG−2−REP−B
MIG−2−REP−C
MIG−2B−REP−A
MIG−2B−REP−B
MIG−2B−REP−C
MIG−2B−REP−D
Depth (cm)
0
10
20
30
40
50
42
44
46
48
50
52
54
56
Carbon content (%)
C (%)
N (%)

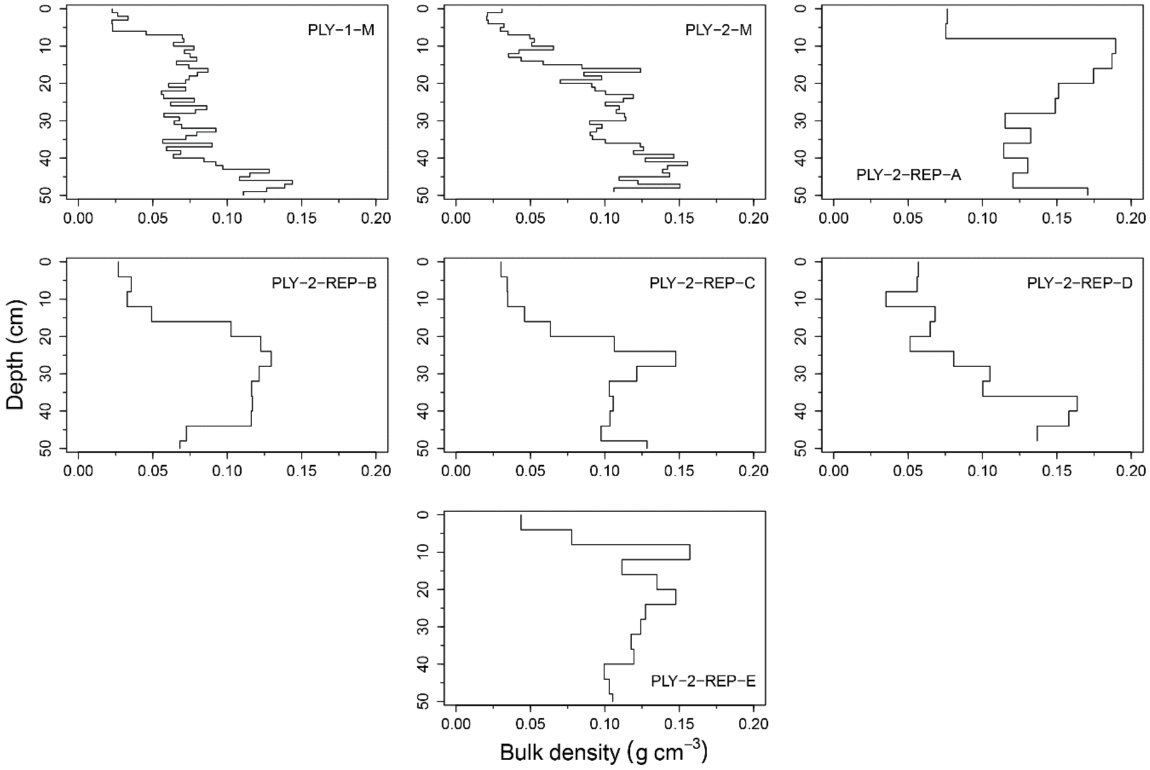

MIG-4-REP-A
MIG-4-REP-B
MIG-4-REP-C
MIG-4-REP-D
MIG-5-REP-A
MIG-5-REP-B
MIG-5-REP-C
MIG-5-REP-D
Depth (cm)
0
10
20
30
40
50
0.00
0.05
0.10
0.15
0.20
0.25
Bulk density (g cm^{-3})

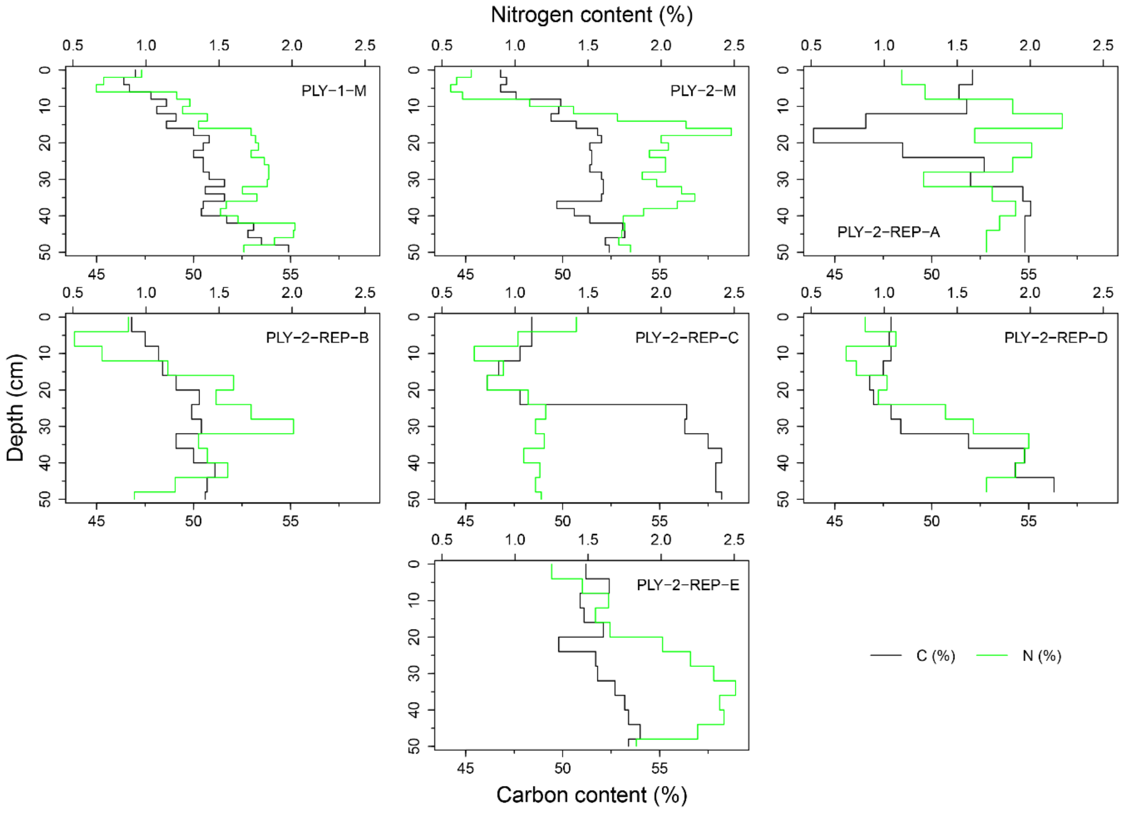

Nitrogen content (%)
0.5
1.0
1.5
2.0
2.5
MIG-4-REP-A
MIG-4-REP-B
MIG-4-REP-C
MIG-4-REP-D
MIG-5-REP-A
MIG-5-REP-B
MIG-5-REP-C
MIG-5-REP-D
Depth (cm)
0
10
20
30
40
50
45
50
55
60
Carbon content (%)
C (%)
N (%)

Appendix B


MIG−6−REP−A
MIG−6−REP−B
MIG−6−REP−C
Depth (cm)
0
10
20
30
40
50
0.0
0.1
0.2
0.3
Bulk density (g cm^{-3})

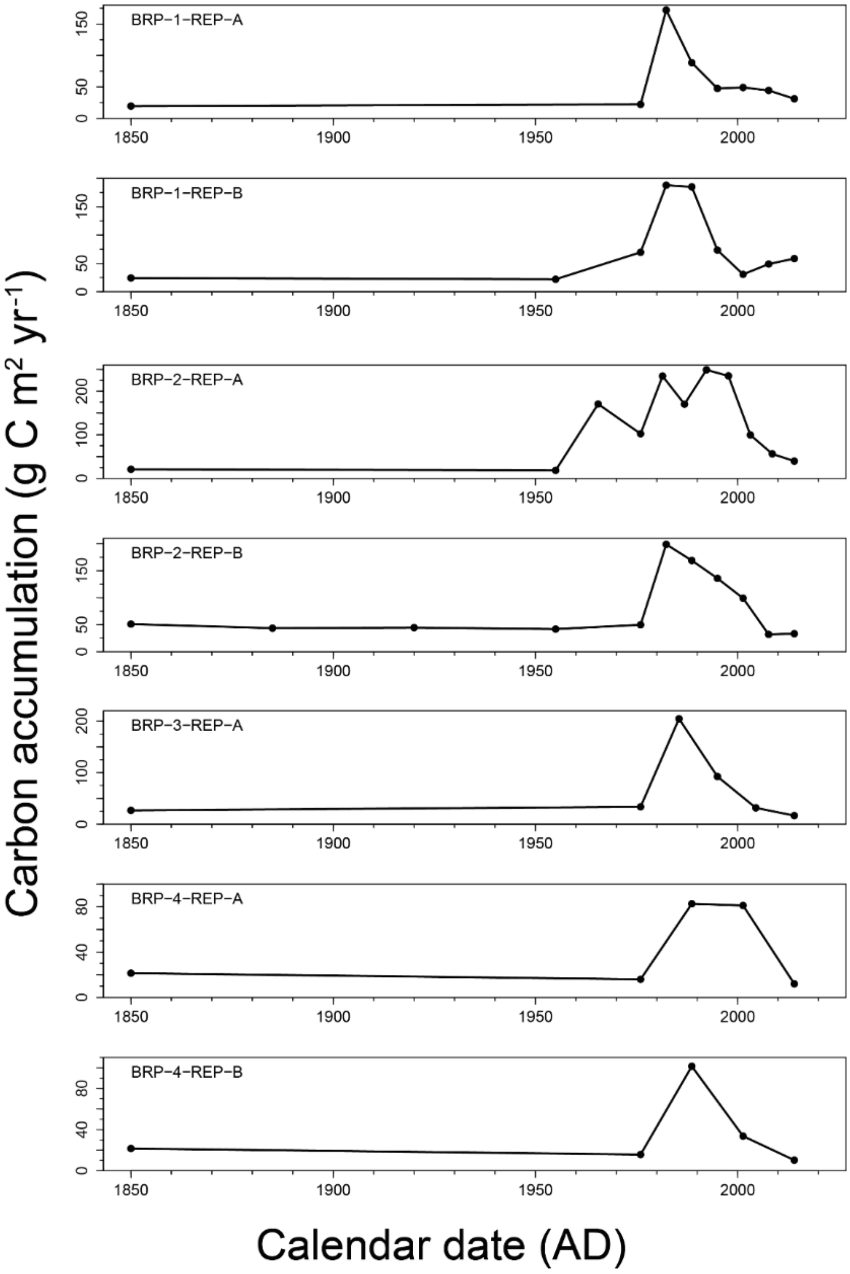

Nitrogen content (%)
1.5
2.0
2.5
3.0
3.5
Depth (cm)
0
10
20
30
40
50
MIG−6−REP−A
MIG−6−REP−B
MIG−6−REP−C
30
35
40
45
50
55
Carbon content (%)
C (%)
N (%)


PLY−1−M
PLY−2−M
PLY−2−REP−A
PLY−2−REP−B
PLY−2−REP−C
PLY−2−REP−D
PLY−2−REP−E
Depth (cm)
0
10
20
30
40
50
0.00
0.05
0.10
0.15
0.20
Bulk density (g cm^{-3})

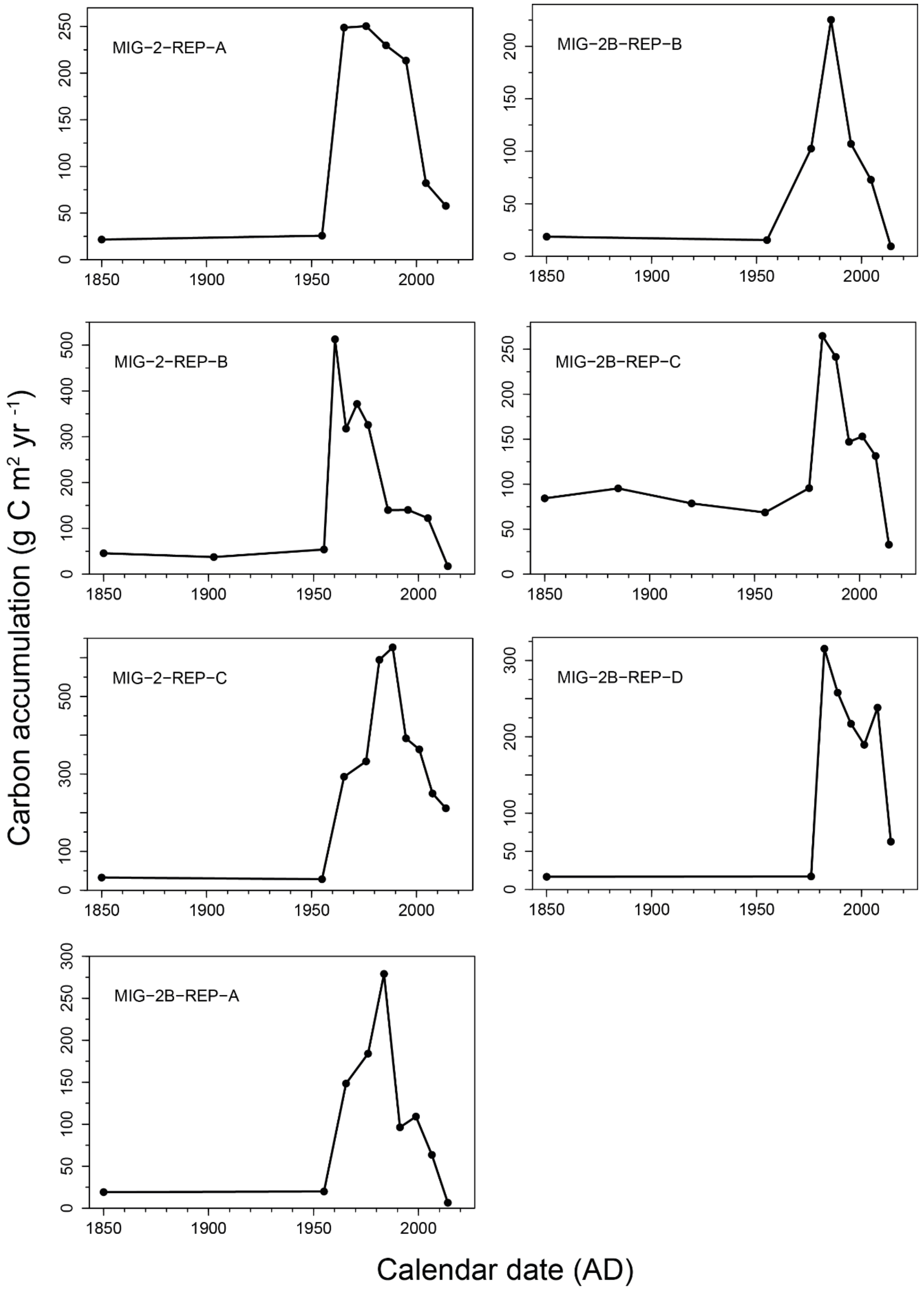

Nitrogen content (%)
0.5
1.0
1.5
2.0
2.5
PLY−1−M
PLY−2−M
PLY−2−REP−A
PLY−2−REP−B
PLY−2−REP−C
PLY−2−REP−D
PLY−2−REP−E
Depth (cm)
0
10
20
30
40
50
45
50
55
Carbon content (%)
C (%)
N (%)

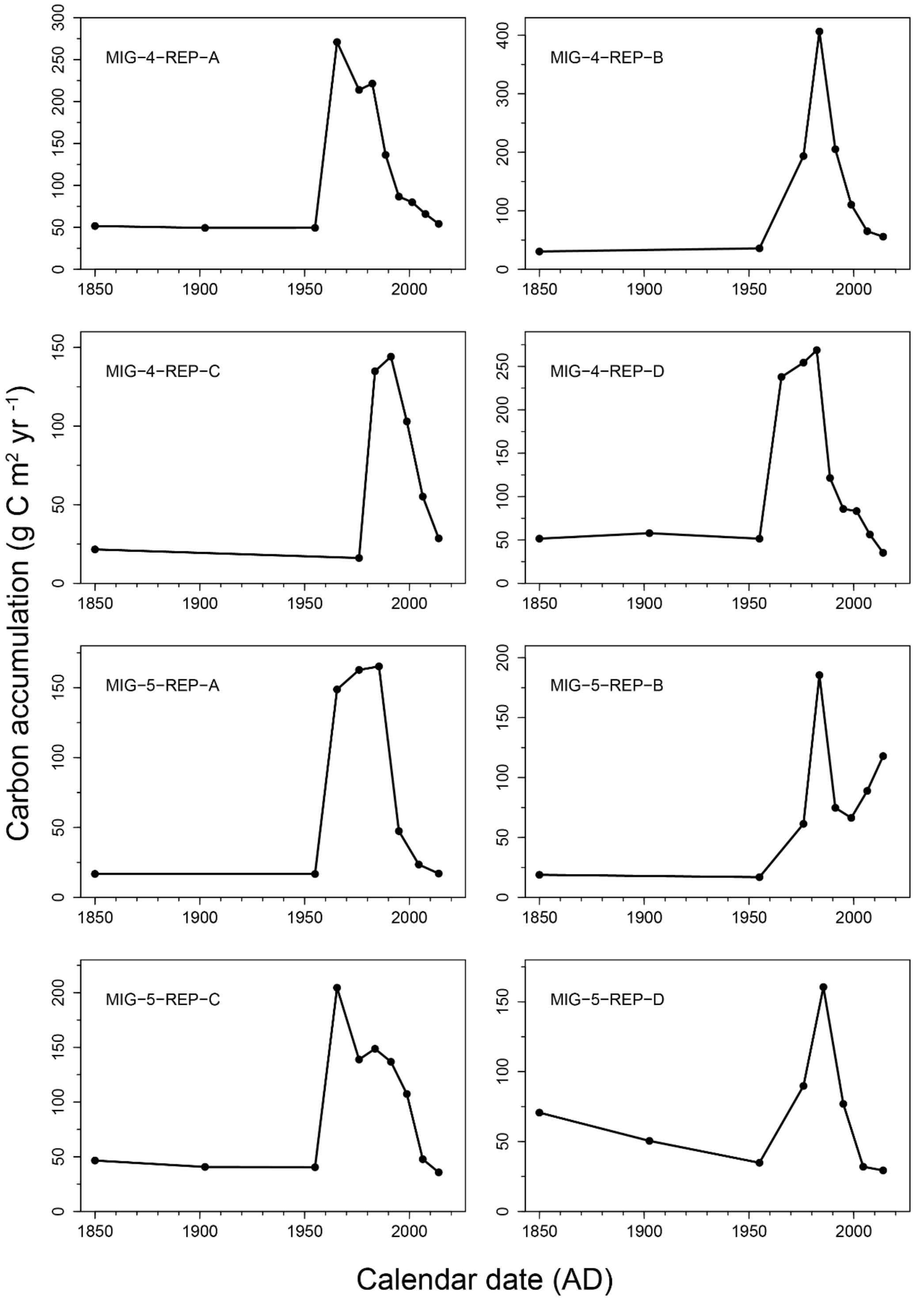
Appendix C

C.1 Replicate CAR profiles

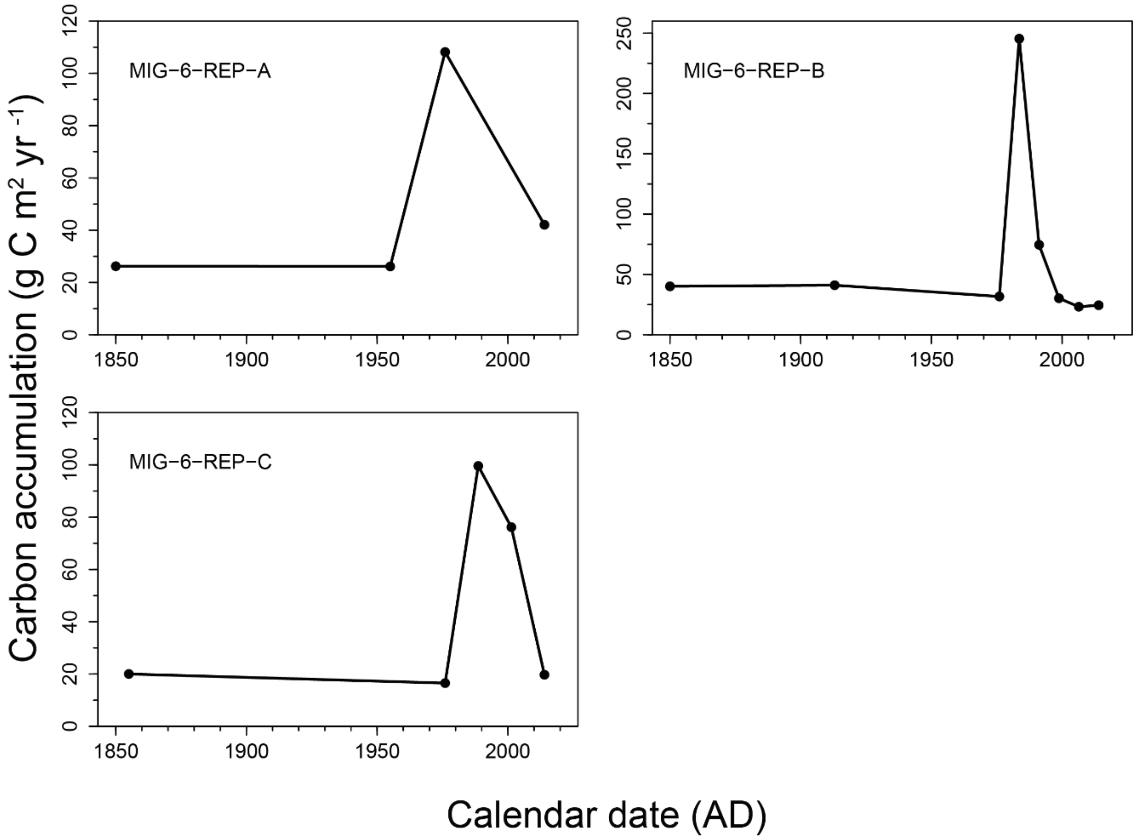


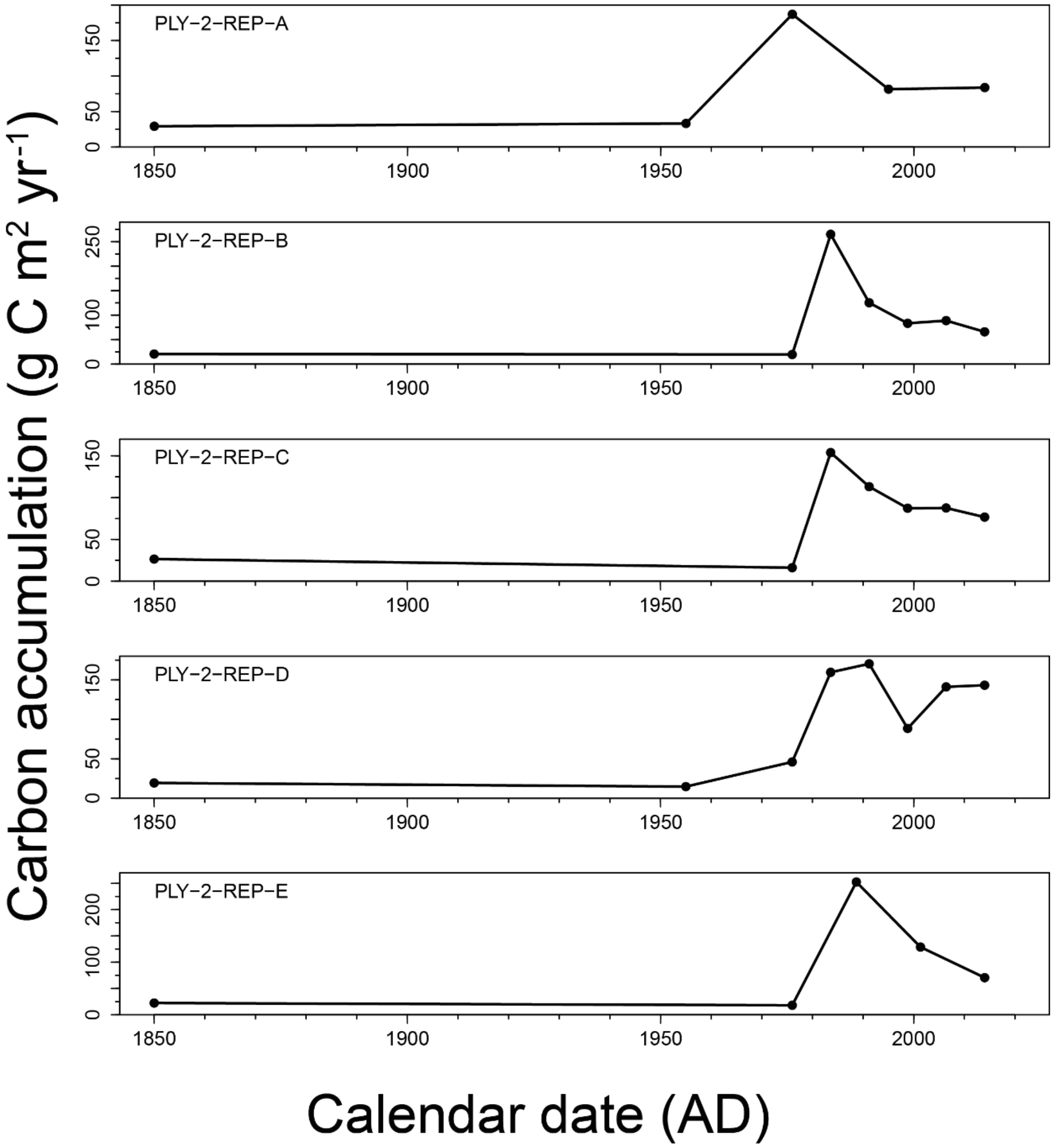
Appendix C


MIG−2−REP−A
MIG−2B−REP−B
MIG−2−REP−B
MIG−2B−REP−C
MIG−2−REP−C
MIG−2B−REP−D
MIG−2B−REP−A
0
50
100
150
200
250
300
400
500
1850
1900
1950
2000
Carbon accumulation (g C m² yr⁻¹)
Calendar date (AD)


MIG−4−REP−A
MIG−4−REP−B
MIG−4−REP−C
MIG−4−REP−D
MIG−5−REP−A
MIG−5−REP−B
MIG−5−REP−C
MIG−5−REP−D
Carbon accumulation (g C m2 yr -1)
Calendar date (AD)
1850
1900
1950
2000
0
50
100
150
200
250
300
400

Appendix C


MIG−6−REP−A
MIG−6−REP−B
MIG−6−REP−C
0
20
40
60
80
100
120
0
50
100
150
200
250
1850
1900
1950
2000
Carbon accumulation (g C m^2 yr^{-1})
Calendar date (AD)


PLY−2−REP−A
PLY−2−REP−B
PLY−2−REP−C
PLY−2−REP−D
PLY−2−REP−E
0
50
100
150
200
250
1850
1900
1950
2000
Carbon accumulation (g C m2 yr-1)
Calendar date (AD)

C.2 Replicate RERCA results

Core	Horizon (AD)	Carbon stock(g C m^2)	RERCA (g C m^2 yr^{-1})		
			Minimum	Estimated	Maximum
BRP-1-REP-A	1850	5544	29	34	40
	1955	3207	50	54	59
	1976	2740	67	72	78
BRP-1-REP-B	1850	7459	39	45	54
	1955	5158	81	87	96
	1976	3698	90	97	106
BRP-2-REP-A	1850	10696	57	65	77
	1955	8757	137	148	162
	1976	5889	144	155	168
BRP-2-REP-B	1850	9841	52	60	71
	1955	5281	83	90	98
	1976	4234	103	111	121
BRP-3-REP-A	1850	9206	49	56	66
	1955	7520	117	127	139
	1976	3280	80	86	94
BRP-4-REP-A	1850	5594	30	34	40
	1955	4245	66	72	79
	1976	2227	54	59	64
BRP-4-REP-B	1850	5189	27	32	37
	1955	3828	60	65	71
	1976	1841	45	48	53

Core	Horizon (AD)	Carbon stock (g C m^2)	RERCA (g C m^2 yr^{-1})		
			Minimum	Estimated	Maximum
LVW-1-REP-A	1850	8206	43	50	59
	1955	5759	90	98	107
	1976	2791	68	73	80
LVW-1-REP-B	1850	12543	66	76	90
	1955	8365	131	142	155
	1976	4597	112	121	131
LVW-1-REP-C	1850	19117	101	117	138
	1955	14783	231	251	274
	1976	9242	225	243	264
LVW-1-REP-D	1850	13238	70	81	95
	1955	11193	175	190	207
	1976	10784	263	284	308

Core	**Horizon (AD)**	**Carbon stock (g C m^2)**	**RERCA (g C m^2 yr^{-1})**		
			Minimum	**Estimated**	**Maximum**
MIG-1-REP-A	1850	N/A	N/A	N/A	N/A
	1955	N/A	N/A	N/A	N/A
	1976	9671	236	254	276
MIG-1-REP-B	1850	8186	43	50	59
	1955	5322	83	90	99
	1976	3669	89	97	105
MIG-1-REP-C	1850	N/A	N/A	N/A	N/A
	1955	11989	187	203	222
	1976	8336	203	219	238
MIG-3-REP-A	1850	13887	73	85	100
	1955	10373	162	176	192
	1976	6678	163	176	191
MIG-3-REP-B	1850	9140	48	56	66
	1955	6495	101	110	120
	1976	5966	146	157	170
MIG-3-REP-C	1850	18565	98	113	134
	1955	9711	152	165	180
	1976	3392	83	89	97

Core	Horizon (AD)	Carbon stock (g C m^2)	RERCA (g C m^2 yr^{-1})		
			Minimum	Estimated	Maximum
MIG-2-REP-A	1850	13484	71	82	97
	1955	10778	168	183	200
	1976	5538	135	146	158
MIG-2-REP-B	1850	16798	89	102	121
	1955	12009	188	204	222
	1976	3988	97	105	114
MIG-2-REP-C	1850	24987	132	152	180
	1955	21999	344	373	407
	1976	15434	376	406	441
MIG-2B-REP-A	1850	9798	52	60	70
	1955	7707	120	131	143
	1976	4215	103	111	120
MIG-2B-REP-B	1850	7715	41	47	56
	1955	6093	95	103	113
	1976	3939	96	104	113
MIG-2B-REP-C	1850	16641	88	101	120
	1955	8154	127	138	151
	1976	6146	150	162	176
MIG-2B-REP-D	1850	10269	54	63	74
	1955	8471	132	144	157
	1976	8112	198	213	232

Core	Horizon (AD)	Carbon stock(g C m^{2})	RERCA (g C m^{2} yr^{-1})		
			Minimum	Estimated	Maximum
MIG-4-REP-A	1850	14361	76	88	103
	1955	9173	143	155	170
	1976	4082	100	107	117
MIG-4-REP-B	1850	14228	75	87	102
	1955	10470	164	177	194
	1976	6406	156	169	183
MIG-4-REP-C	1850	5571	29	34	40
	1955	3878	61	66	72
	1976	3540	86	93	101
MIG-4-REP-D	1850	15028	80	92	108
	1955	9289	145	157	172
	1976	6792	166	179	194
MIG-5-REP-A	1850	7435	39	45	53
	1955	5676	89	96	105
	1976	2406	59	63	69
MIG-5-REP-B	1850	7117	38	43	51
	1955	5343	83	91	99
	1976	4054	99	107	116
MIG-5-REP-C	1850	11483	61	70	83
	1955	7225	113	122	134
	1976	3620	88	95	103
MIG-5-REP-D	1850	9201	49	56	66
	1955	4724	74	80	87
	1976	2838	69	75	81

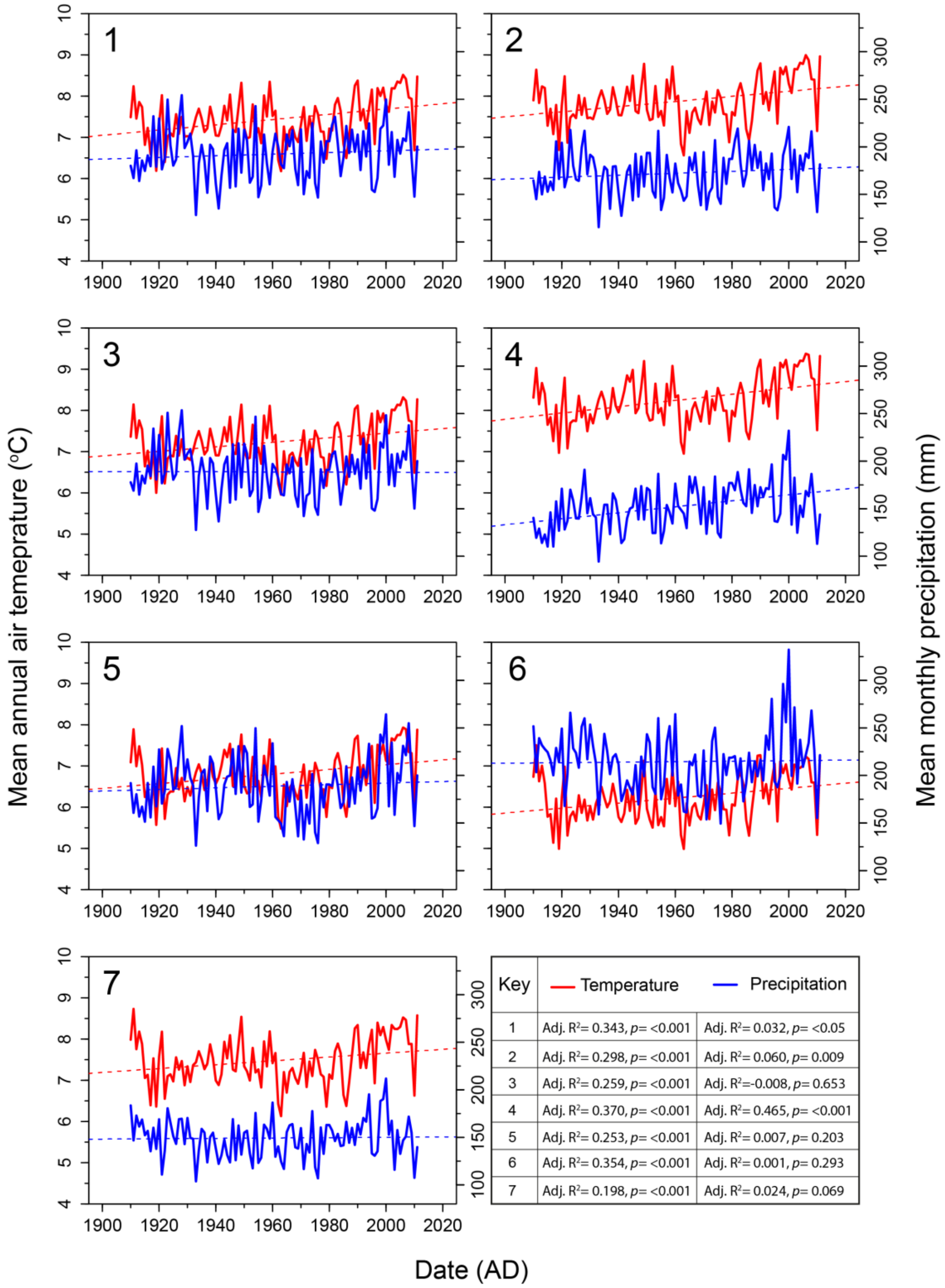
Core	**Horizon (AD)**	**Carbon stock (g C m^2)**	**RERCA (g C m^2 yr^{-1})**		
			Minimum	**Estimated**	**Maximum**
MIG-6-REP-A	1850	6615	35	40	48
	1955	3870	60	66	72
	1976	1599	39	42	46
MIG-6-REP-B	1850	7612	40	46	55
	1955	5023	78	85	93
	1976	3024	74	80	86
MIG-6-REP-C	1850	4476	24	27	32
	1955	2809	44	48	52
	1976	2476	60	65	71

Core	Horizon (AD)	Carbon stock ($g\ C\ m^{2}$)	RERCA ($g\ C\ m^{2}\ yr^{-1}$)		
			Minimum	Estimated	Maximum
PLY-2-REP-A	1850	10552	56	64	76
	1955	7066	110	120	131
	1976	3139	77	83	90
PLY-2-REP-B	1850	7243	38	44	52
	1955	5187	81	88	96
	1976	4776	116	126	136
PLY-2-REP-C	1850	5975	32	36	43
	1955	4281	67	73	79
	1976	3942	96	104	113
PLY-2-REP-D	1850	7854	42	48	57
	1955	6308	99	107	117
	1976	5342	130	141	153
PLY-2-REP-E	1850	8000	42	49	58
	1955	6102	95	103	113
	1976	5723	140	151	164

Appendix D

Easting and northing coordinates for the centre of each 5x5km UKCP09 grid cell of interest in this study (Met Office, 2017b). ‘Number’ refers to the number the grid was assigned in this thesis as presented in the following figure.

Number	Easting	Northing	Relevant Sites
1	277500	347500	MIG-1, MIG-5
2	282500	347500	MIG-6
3	277500	342500	MIG-2, MIG-3, MIG-4
4	297500	322500	LVW-3
5	292500	322500	LVW-1, LVW-2
6	282500	287500	PLY-1, PLY-2
7	282500	267500	BRP-1, BRP-2, BRP-3, BRP-4



Key	Temperature	Precipitation
1	Adj. R^2= 0.343, p= <0.001	Adj. R^2= 0.032, p= <0.05
2	Adj. R^2= 0.298, p= <0.001	Adj. R^2= 0.060, p= 0.009
3	Adj. R^2= 0.259, p= <0.001	Adj. R^2=-0.008, p= 0.653
4	Adj. R^2= 0.370, p= <0.001	Adj. R^2= 0.465, p= <0.001
5	Adj. R^2= 0.253, p= <0.001	Adj. R^2= 0.007, p= 0.203
6	Adj. R^2= 0.354, p= <0.001	Adj. R^2= 0.001, p= 0.293
7	Adj. R^2= 0.198, p= <0.001	Adj. R^2= 0.024, p= 0.069

UKCP 09 temperature and precipitation data (AD 1910-2011) for each 5x5 km grid of relevance to this thesis. Letters refer to grids detailed in previous table. Solid lines represent yearly mean whilst dashed lines indicate a linear model. Correlation statistics refer to a five-year moving average for each parameter (AD 1912-2009).

Correlation coefficients for relationships between normalised five-year moving average climatic variables and DCA Axis one and two sample scores. *R*-value precedes the '/', *P*- value follows the '/'. Correlations significant at <0.05 are given in bold.

Core	Data	Annual		Spring		Summer		Autumn		Winter	
		Temp.	**Precip.**	**Temp.**	**Precip.**	**Temp.**	**Precip.**	**Temp.**	**Precip.**	**Temp.**	**Precip.**
BRP-1-M	Axis one	-0.121 / 0.398	-0.564 / 0.094	-0.120 / 0.342	-0.637 / 0.062	-0.141 / 0.381	-0.610 / 0.073	0.397 / 0.811	0.388 / 0.805	-0.368 / 0.209	-0.668 / 0.05
	Axis two	-0.561 / 0.095	**-0.716 / <0.05**	-0.647 / 0.058	**-0.8 / <0.005**	-0.539 / 0.106	**-0.744 / <0.05**	-0.012 / 0.489	0.423 / 0.828	-0.485 / 0.135	**-0.772 / <0.05**
BRP-2-M	Axis one	-0.032 / 0.460	-0.128 / 0.346	-0.106 / 0.372	-0.379 / 0.112	-0.066 / 0.420	-0.037 / 0.454	-0.151 / 0.320	0.141 / 0.331	0.120 / 0.356	-0.051 / 0.437
	Axis two	**0.655 / <0.05**	0.434 / 0.079	0.453 / 0.070	-0.397 / 0.101	0.665 / 0.009	0.442 / 0.075	0.071 / 0.413	0.420 / 0.087	**0.625 / <0.05**	0.308 / 0.165
LVW-1-M	Axis one	-0.090 / 0.391	0.194 / 0.273	-0.078 / 0.405	-0.051 / 0.437	-0.212 / 0.254	-0.233 / 0.233	**-0.530 / <0.05**	-0.187 / 0.280	0.403 / 0.097	0.112 / 0.365
	Axis two	-0.255 / 0.212	-0.267 / 0.121	-0.296 / 0.175	-0.352 / 0.131	**0.719 /<0.005**	-0.113 / 0.363	-0.404 / 0.096	**-0.581 / <0.05**	0.108 / 0.370	0.076 / 0.407
LVW-2-M	Axis one	-0.224 / 0.335	-0.532 / 0.139	-0.181 / 0.365	**-0.95/<0.005**	-0.093 / 0.430	0.082 / 0.439	-0.560 / 0.119	-0.378 / 0.230	0.198 / 0.353	-0.473 / 0.172
	Axis two	-0.215 / 0.341	-0.389 / 0.223	-0.191 / 0.359	**-0.842 / <0.05**	-0.095 / 0.429	0.245 / 0.320	-0.480 / 0.168	-0.339 / 0.256	-0.152 / 0.387	-0.340 / 0.255
LVW-3-M	Axis one	0.279 / 0.167	0.632 / 0.008	0.160 / 0.293	**0.609 / <0.05**	0.375 / 0.093	0.066 / 0.411	0.375 / 0.093	0.392 / 0.083	0.044 / 0.441	**0.575 / <0.05**
	Axis two	-0.112 / 0.352	**0.560 / <0.05**	-0.159 / 0.293	**0.769 /<0.001**	-0.109 / 0.355	-0.013 / 0.482	-0.109 / 0.355	**0.540 / <0.05**	-0.328 / 0.126	0.192 / 0.255
MIG-1-M	Axis one	**-0.353 / <0.05**	-0.283 / 0.095	-0.252 / 0.123	-0.013 / 0.476	-0.133 / 0.273	0.071 / 0.373	**-0.383 / <0.05**	**-0.359 / <0.05**	-0.117 / 0.298	-0.316 / 0.071
	Axis two	0.202 / 0.178	0.033 / 0.441	0.177 / 0.209	0.068 / 0.379	0.121 / 0.291	-0.193 / 0.189	-0.048 / 0.414	0.064 / 0.385	0.225 / 0.151	0.202 / 0.177
MIG-2-M	Axis one	-0.258 / 0.269	0.002 / 0.498	-0.337 / 0.207	**-0.628 / <0.05**	-0.339 / 0.205	0.437 / 0.139	-0.621 / 0.050	-0.113 / 0.395	0.282 / 0.249	0.099 / 0.408
	Axis two	0.433 / 0.142	0.195 / 0.322	**0.656 / <0.05**	**0.0762 /<0.05**	0.200 / 0.317	0.442 / 0.136	0.374 / 0.181	-0.413 / 0.155	0.310 / 0.228	0.049 / 0.455
MIG-3-M	Axis one	0.008 / 0.491	0.234 / 0.244	-0.064 / 0.425	0.712 / 0.007	-0.286 / 0.197	-0.103 / 0.381	0.209 / 0.269	0.026 / 0.469	0.061 / 0.430	0.034 / 0.461
	Axis two	-0.209 / 0.269	-0.173 / 0.306	-0.013 / 0.484	0.335 / 0.157	-0.353 / 0.144	**-0.533 / <0.05**	0.066 / 0.423	-0.375 / 0.128	-0.314 / 0.174	0.360 / 0.139
PLY-1-M	Axis one	-0.232 / 0.234	-0.311 / 0.162	-0.226 / 0.240	-0.453 / 0.069	-0.075 / 0.408	0.093 / 0.387	0.307 / 0.166	0.041 / 0.450	-0.484 / 0.055	-0.427 / 0.083
	Axis two	-0.438 / 0.078	0.119 / 0.357	-0.299 / 0.173	0.011 / 0.487	-0.462 / 0.065	**0.584 / <0.05**	-0.310 / 0.164	0.016 / 0.280	-0.119 / 0.356	-0.134 / 0.339
PLY-2-M	Axis one	-0.529 / 0.089	-0.371 / 0.183	**-0.703 / <0.05**	-0.362 / 0.189	0.091/ 0.416	-0.099 / 0.408	**-0.660 / <0.05**	-0.294 / 0.240	-0.345 / 0.201	-0.169 / 0.345
	Axis two	0.321 / 0.219	0.475 / 0.117	0.527 / 0.090	-0.043 / 0.560	0.287 / 0.246	0.488 / 0.110	-0.104 / 0.403	0.381 / 0.176	0.296 / 0.238	0.372 / 0.182

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