THE USE OF MICROPROPAGATED FOUNDER MATERIAL WITH TOP-DOWN IRRIGATION:

A NOVEL APPROACH FOR *SPHAGNUM* FARMING IN A LOWLAND ENVIRONMENT

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“See the opportunities for adventures, not the constraints that get in the way.”

Alastair Humphreys

Abstract

Paludiculture is the cultivation of wetland crops on rewetted peat soils. It represents a new agricultural paradigm offering solutions to peatland degradation by restoring Ecosystem Services (ES) and maintaining land productivity. *Sphagnum* moss is a paludiculture crop, produced via *Sphagnum* farming (SF), which has several uses, particularly peat replacement in growing media.

The conventional SF approach uses the Moss Layer Transfer Technique (MLTT) to provide founder material from a donor site with surface irrigation via raised water tables. This method is challenging in the UK lowland agricultural context as donor *Sphagnum* sites are scarce, and there is reluctance to raise water levels within a conventional agricultural landscape.

This thesis presents a novel alternative SF option, the Micropropagated-Irrigation-From-Above 'MIFA' approach. Micropropagated *Sphagnum* requires very small amounts of donor material and overhead irrigation removes the need for active water table management, removing some current SF challenges.

The effectiveness of the MIFA approach on *Sphagnum* hydrology and growth was monitored across three pilot studies and two experimental field sites in the UK via hydrological and growth measurements.

Sphagnum water availability was assessed via pore water pressure (PWP) measurements. Across the field sites, the MIFA approach resulted in PWP measurements broadly equivalent to literature values for natural peatland systems and better than those recorded for a drained peatlands capable of supporting *Sphagnum* growth.

Sphagnum growth was assessed up to 24 months post-establishment via Terrestrial Laser scanning (TLS), with *Sphagnum* carpet height increases of up to 16cm recorded. Suggesting in some cases, growth under the MIFA approach was comparable or better than *Sphagnum* grown under conventional SF approach.

The results demonstrate that the MIFA approach produces a good *Sphagnum* crop across two contrasting sites, offering a viable alternative to the MLTT approach for *Sphagnum* farming in areas where the conventional SF approach is problematic.

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Glossary of Terms

Agriculturalised peat: A former peatland habitat that has undergone conversion to agriculture. This conversion removes all peatland vegetation, and agricultural processes such as drainage, fertilisation, and grazing/cultivation. Agriculturalised peat reduces positive ecosystem services compared to natural peatlands and increases negative aspects.

BeadaGel™ (gel/gels): A micropropagated *Sphagnum* founder material produced by the commercial partner MicroPropagation Services LTD. This consists of immature *Sphagnum* fragments suspended in a colloidal gel.

BeadaHumok™ (plug/plugs): A micropropagated *Sphagnum* founder material produced by the commercial partner MicroPropagation Services LTD. This consists of mature *Sphagnum* moss individuals bundled together into a plug plant.

Bulk density: Bulk density is the mass of a material divided by the total volume it occupies. Through the thesis dry bulk density (DBD) is used for the bulk density of dry *Sphagnum*, peat or other material. Occasional fresh mass bulk density (fm bd) is used for undried material. Bulk density is not an intrinsic property of a material; it can change depending on how the material is handled so this must be kept in mind.

Commercial peat extraction site: A former raised bog peatland that has undergone peat extraction. The conversion process removes all peatland vegetation, induces drainage and physical removal of peat soil. This process reduces positive ecosystem services compared to natural peatlands and increases negative aspects.

Compost: compost is technically the by-product of a composting operation, whereas composting involves the aerobic decomposition of organic solid wastes. Compost is rich in organic matter and nutrients. A composted material may be a component of a growing medium, but peat is not a product of composting.

Dripline or 'Drip' irrigation: a top-down method of irrigation. Water is delivered as small droplets via a hollow tape with evenly spaced water emitters. Drip tape utilises pressure compensation. This ensures a uniform flow rate across all the water emitters when the length and diameter of each dripline is uniform across the system. Water is delivered directly to the *Sphagnum* surface.

Growing media (GM): a term used to describe the material used in a container to grow a plant. Often formulated from a blend of different raw materials to achieve the correct balance of air, water holding capacity and nutrient mix for a chosen plant species. Materials can be both inorganic or organic.

Sprinkler or 'Spray' irrigation: A top-down method of irrigation. Water is delivered via an above-ground set of sprinkler tubes. Sprinkler systems are versatile and can utilise a range of water delivery patterns, from fine mist to a heavier spray. In this study, the system utilised a **spray** pattern, which aimed to ensure even water delivery across the *Sphagnum* surface. However, like all spray systems, water can be lost due to wind and run-off.

Surface irrigation: The use of irrigation canals, dug into the peat surface. These aim to maintain the water table, via lateral seepage of water through the peat soil.

Top-down irrigation: The practice of applying water directly to the growing *Sphagnum* moss. These distribute water laterally and vertically through the growing moss layer. Water may also spread vertically through the peat soil.

Pore water pressure (PWP): Pore water pressure refers to the pressure of groundwater held within a soil or rock, in gaps between particles. Due to its unique structure, *Sphagnum* moss exhibits pores as inter, intra and inner plant pores. *Sphagnum* farming occurs in the unsaturated vadose zone, where pore water pressure is used as a measure of capillarity. *Sphagnum* relies on capillary force to obtain water, so PWP is of great interest.

Acronyms

DBD - dry bulk density

ES - ecosystem services

PWP - pore water pressure

GHG - greenhouse gas

SF – *Sphagnum* farming

General introduction

Peatlands are a major form of wetland ecosystem now recognised for their extraordinary carbon storage, as well as a range of other ecosystem services (ES). Until recently, their significance had gone largely unrecognised, leaving them unrecognised as a 'Cinderella habitat'. However various factors, in particular climate change, have resulted in a profound shift in the way in which peatland ecosystems are perceived around the world. Consequently, this has resulted in calls to action around the world at all levels of decision making in relation to peatland use and exploitation. This will involve everyone, from members of the public to world leaders.

This PhD focuses on one aspect of this new call to action – namely the potential for a new form of land use on lowland peat soils which have been subject to intensive exploitation and land conversion. This new land use is *Sphagnum* bog moss cultivation, which can be cultivated as a direct replacement for commercially mined peat. Peat is currently used by the horticultural industry as the dominant growing media constituent. The industrial extraction of peat has led to many environmental problems and is inherently unsustainable as extraction rates exceed natural peat formation and remove peat forming vegetation.

Adoption of *Sphagnum* cultivation on severely damaged peat soils has the potential to provide multiple benefits across a wide range of ES. It also represents one of the most rapidly developing sectors in a new form of farming called 'paludiculture'. Under paludiculture, former or damaged wetland sites are re-wetted to grow wetland species in a new guise as commercially viable agricultural crops.

To set the context for this new approach to agricultural production and make clear the urgent need for adoption of *Sphagnum* farming and other forms of paludiculture, it is necessary to review the history of wetland (and in particular peatland) exploitation and its consequences.

It is also instructive to review previous responses to the consequences of such wetland exploitation and consider the various reasons why these responses have so far largely failed to achieve more sustainable ‘wise-use’ of peatland soils and their associated habitats.

Chapter 1. Wetland ecosystems – character and services

1.1 Introduction

While this PhD is focused on certain aspects of peatland ecology and most particularly on a group of species central to that ecology, it is impossible to place this research into a wider context without considering the wider picture of wetland ecosystems and their history of use. Peatlands are one particular group of wetlands within this over-arching categorisation. In part, this broader perspective is necessary because much relevant literature refers only to the broad category of 'wetland' rather than specifically highlighting peatland systems. Furthermore, many of the characteristics of, and issues impacting on, wetlands as a whole apply directly to peatland ecosystems. Finally, certain aspects of the present research have the potential to extend beyond the strict definition of peatlands and be applicable to a wider range of wetland conditions.

1.2 Wetlands – their definition and characterisation

Wetlands are ecosystems dominated by water either permanently or periodically (Maltby and Acreman, 2011). Wetlands are defined by the Ramsar convention as *"…areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres."* (Ramsar, 1971).

According to Ramsar, wetlands can be grouped into 5 major types:

1. Marine – coastal wetlands such as lagoons and coral reefs
2. Estuarine – deltas and mangrove swamps
3. Lacustrine – wetlands associated with lakes
4. Riverine – wetlands along rivers and streams
5. Palustrine – bogs, marshes and swamps.

Wetlands are one of the most globally widespread ecosystems, being found on every continent (Junk *et al.*, 2013) including Antarctica (Moorhead *et al.*, 2003). The global extent is difficult to quantify due to widespread inconsistency in the use of wetland definitions as well as current dependence on incomplete datasets. Global wetland extent estimations vary from 5.70 million km^2 (Thorsell, Levy and Sigaty, 1997), 14.86 million km^2 (Hu *et al.*, 2017) and 16.2 million km^2 (Davidson *et al.*, 2018; Davidson, Fluet-Chouinard and Finlayson, 2018). It is highly likely that these are all nevertheless underestimates because a thorough global survey and mapping effort remains to be undertaken. Until such an exercise has been undertaken it is not possible to state the true global peatland extent, but it is at least possible to regard existing published estimates as likely *minimum* estimates of extent.

1.3 Peatlands – peat-forming wetlands

Peatlands, which are the focus of this thesis, are possibly the most extensive type of wetland globally, having been identified by Rubec (1996) as providing somewhere between 3.4 million km^2 and 4.2 million km^2 of the 6.5 million km^2 total area of global wetland area – i.e. between 52% and 65% of the mapped global wetland area. Despite this, they are frequently regarded as merely a subcategory of palustrine wetlands, whereas in fact peatlands can form in all five categories of Ramsar definitions, only being absent from coral reef wetlands.

Peatlands are formed in areas where the provision of water exceeds the amount of water lost from the system and conditions permit the accumulation of waterlogged, dead plant material. Specifically, peatlands are wetlands "*with a naturally accumulated layer of peat at the surface*", while 'peat' is defined as "*a sedentarily accumulated material consisting of at least 30% (dry mass) of dead organic material*" (Joosten *et al.*, 2017). The water surplus allows anaerobic conditions to dominate, which results in a build-up of semi-decomposed plant and sometimes animal material.

Crucially all types of peatland can only exist in a ‘near natural’ or ‘healthy’ condition while they remain in a waterlogged state, with peat-forming vegetation present. The functioning of a peat bog system depends on the presence of peat forming vegetation and adequate water retention within the system to enable peat formation. Peat formation and associated carbon sequestration occur due to an imbalance between net primary production or biomass production and decomposition. The hydrological and chemical variation across peatland types affects the range of species found in each habitat, but whatever these conditions and resulting botanical composition of the vegetation, waterlogging results in this imbalance between production and decay. A peatland system still supporting some peat-forming vegetation is termed a ‘mire’ - as opposed to a peatland system where the characteristic peat-forming vegetation has been replaced by a vegetation which does not form peat (e.g. a rye grass ley), although the peat soil remains.

Peatlands typically fall into two broad categories – fen and bog (Ingram, 1967; Gore, 1983). Fens are formed where minerotrophic water inputs are delivered via groundwater, surface water, or both, as well as by rainfall. Minerotrophic peatlands are strongly influenced by their underlying geology and are generally more base-rich and nutrient-rich than bogs, though extremely base-poor fens such as the New Forest valley mires have traditionally (if strictly incorrectly) been termed ‘bogs’ because of their nutrient poverty (Tansley, 1939; Lindsay, 2016).

Bogs receive water only via precipitation (rain, mist, fog, snow) and are therefore termed ‘ombrotrophic’. Bogs are therefore also largely independent of underlying geology and surrounding mineral groundwater chemistry. Their distinctive and restricted water supply results in acidic and nutrient-poor conditions because rain is slightly acidic and because the only nutrient supply to the surface vegetation is via direct precipitation.

A fundamental aspect of a peat bog has for many decades been recognised as the diplotelmic (two layer) structure of a bog ecosystem (Ivanov, 1948), whereby the system has an ‘active’ and ‘inactive’ layer. This concept was developed further into the more widely used acrotelm – catotelm model as described by Ingram, (1978). The acrotelm is a shallow, periodically aerated ‘living’ surface layer between 10 and 40 cm deep. The acrotelm represents a zone of high hydraulic conductivity, allowing water to move vertically and laterally with relative ease. *Sphagnum* mosses are one of the most characteristic living plant components of a healthy acrotelm in ombrotrophic peatlands.

Plants forming the acrotelm complete their cycle of growth, decay and regrowth but due to the waterlogged anaerobic conditions, total decay of the biomass does not occur. The semi-decomposed remains of peat-forming vegetation accumulate beneath the acrotelm layer as a sedentate. This underlying sedentate layer is known as the ‘catotelm’ (Ingram, 1978). Unlike the acrotelm, the catotelm is permanently saturated. Hydraulic conductivity here is poor, with water flow many times slower than that of the acrotelm layer above it (Lindsay, 2010)

The accumulated sedentate forming the catotelm layer has many names within the literature, depending on geographical location, discipline, and age of publication. However, this soil is most often referred to as ‘peat’ (Clymo, 1983). Peat is created as a result of the steady supply of dead plant material from the acrotelm. Biomass production occurs in the acrotelm layer but so does relatively rapid oxygen-fuelled decomposition. Decomposition in the waterlogged catotelm does occur but at a much lower rate determined by much less efficient anaerobic metabolic processes (Clymo and Fogg, 1984; Malmer and Wallén, 1996; Belyea and Malmer, 2004). The slightly greater rate of accumulation over decomposition results in only 2 – 16% of the produced biomass being stored as peat (Päivänen and Vasander, 1994).

1.4 Extent of the wetland/peatland resource in Britain

Figures produced by, or on behalf of, DEFRA provide an indication of the amount of land having a natural tendency to support wetland conditions based on location within a floodplain or in locations subject to coastal or riverine flooding (see Table 1).

It can be seen from Table 1 that approximately 1.5 million ha of land in England alone has a natural tendency to form wetland habitat, and some 1.2 million ha of this is currently subject to agricultural use. The remainder being largely urban or industrial areas such as low-lying parts of London or the industrialised coastal plain of the Thames Estuary. It is also worth noting that surface-water flooding is an additional source of wetland formation but figures for the total area subject to such flooding are not currently available, although indications of extent for any given location can be obtained from the online Environment Agency flood-risk maps. The area at flood risk from surface water although listed as unknown is has the potential to add very substantially to the area of land having a natural tendency to support wetland habitat.

Table 1: Extent of land in England with a natural tendency to form wetland habitat taken from (Roca *et al.*, 2011).

Land category	Area (ha)
Area at flood risk from river or sea	1,655,400
Area lying within floodplains	1,564,740
Area in agricultural use within floodplains	1,224,900
Area at flood risk from surface water	unknown

While flood risk is an indicator of conditions likely to favour wetland formation, the presence of peat is a sure indication of relatively constant waterlogged conditions. It is therefore instructive to compare figures for the extent of peat in the British lowlands with those indicating a tendency to the wetland condition.

Fen peatlands can still be found throughout the British landscape, albeit now generally as fragments of what were once much larger systems, because all they require is a groundwater or surface-water supply within a suitable landform.

Bogs, on the other hand, are more constrained by climatic conditions because they are wholly dependent upon a regular supply of precipitation. In the lowlands, such bogs are typically formed within basins varying from relatively deep to extremely shallow in the landscape but now rising as gentle domes which attain heights of as much as 10 m above the surrounding landscape (and local groundwater table). For obvious reasons these are termed ‘raised bogs’. Although there is evidence to indicate that such bogs could once be found as far south as Romney Marsh (Lindsay and Clough, 2017), their main distribution across lowland Britain has always been centred on the north and west of Britain (Lindsay and Immirzi, 1996).

The climate of upland Britain favours peat formation to such an extent that much of upland Britain is draped in a complex mosaic of bog and fen termed ‘blanket bog’ or more accurately ‘blanket mire’ (Tansley, 1939; Charman, 2002; Lindsay and Clough, 2017). This is by far the most extensive peatland type in the UK and is considered to be one of the UK’s most extensive semi-natural habitats. Although as the focus of this PhD is lowland Britain, the UK blanket mire resource lies outside the main focus of the present work.

Lindsay and Clough (2017) provide estimates, based on a variety of sources, for the current estimated extent of fen peat, lowland bog peat and blanket peat soils in the UK. As will become evident later, this distinction between the extent of peat *soils* as opposed to the extent of *mire habitat* (i.e. surviving peat-forming ecosystem) represents an important distinction in the context of the present study.

Table 2: Extent of peat soils and remaining peatland habitat in Britain (Source: Lindsay and Clough, 2017)

Soil/habitat type	**England** (ha)	**Scotland** (ha)	**Wales** (ha)	**Britain total** (ha)
All peat soils [a]	679,926	1,726,900	70,600	2,477,426
All peaty soils [a]	738,618	3,461,200	359,200	4,559,018
Blanket bog soils [d]	1,092,841	4,922,208	136,722	6,151,771
Raised bog soils [b/c]	**37,694**	**27,892**	**4,078**	**69,664**
Fen soils [e]	**288,009**	**238,000**	**289,000**	**815,009**
Total peat and peaty soils	1,418,544	5,188,100	429,800	7,036,444

[a] derived from Joint Nature Conservation Committee (2011), Tables 7, 12, 16, 19; [b] from Lindsay & Immirzi (1996); [c] from Hammond (1981); [d] from 'Total peat and peaty soils' minus 'raised bog soils' and 'fen soils'; [e] derived from Joint Nature Conservation Committee (2011), Table 8 and Tables 15, 18, 21 which give a possible proxy minimum area.

It can be seen from Table 2 that by far the largest extent of peat soil occurs as upland blanket peat, with some 70,000 ha is recorded as lowland raised bog peat while more than 810,000 ha is identified as fen peat. All of which are found in the lowlands, giving a total lowland peat resource in Britain of almost 900,000 ha. Specifically for England, Table 2 indicates a total extent for fen and raised bog peat soils as some 325,000 ha, approximately a quarter of the area identified for England as having conditions likely to favour wetland formation. It should be noted, however, that 'peaty soils' are recorded as extending across almost 740,000 ha of England. Some of these peaty soils occur as the fringes of extensive tracts of blanket peat, but significant areas also occur in the lowlands. Peaty soils in the lowlands are sometimes found in association with areas with a tendency to surface flooding but also in areas where, for various reasons, the peat has become thin or, in agricultural terms, 'wasted'.

In summary, therefore, some 1.2 million ha of lowland England has the potential to support wetland habitat, of which at least 25% currently possesses a peat soil and thus indicates consistent wetland conditions. While across lowland Britain as a whole, there is probably 1 million ha of peat and peaty soils which have sufficiently consistent natural waterlogging to support wetland conditions, though the area capable of supporting some form of wetland environment is almost certainly much greater.

1.5 Valuing ecosystems through Ecosystem Services

The driving force behind the line of research described in the present thesis has arisen because of a recent rather abrupt shift in perception with respect to wetlands and more specifically of peatlands in terms of their value to society. Such approaches to ecosystem valuation are now well established as the concept of 'ecosystem services' – a concept which has driven much that is relevant to the present study in both positive and negative ways.

Ecosystems provide intrinsic benefits through Regulating, Supporting, Provisioning and Cultural services (MEA, 2005). Ecosystem Services (ES) are incredibly powerful for illustrating environmental importance as they 'contribute to making human life both possible and worth living' (UK NEA, 2011). Ecosystems are supported by primary process such as nutrient recycling and soil creation. This enables ecosystems to function and provide service such as climate and water regulation. Ecosystems provide resources such as food, water, and raw materials (Costanza *et al.*, 1997) and go beyond our basic functional needs, providing cultural meaning which provides humanity with a sense of place, education and inspiration (Milcu *et al.*, 2013).

The ES concept was refined further by The Economics of Ecosystems and Biodiversity initiative (TEEB). The TEEB definition of ES is "the direct and indirect contributions of ecosystems to human well-being" (Kumar, 2012). The

MEA definition and TEEB definition are similar, but a key point of difference is that the TEEB framework integrates governance and decision-making pathways that show how human choices can impact ES and demonstrates that external factors such as climate change will impact ES in multiple and indirect ways. The TEEB framework also restructures supporting services as defined by the MEA, positioning supporting services within ecological structures and processes (as seen in fig. Figure 1). As of 2010 the TEEB framework represents the most current framework to link ecosystems, biodiversity, and human wellbeing (Kumar, 2012).

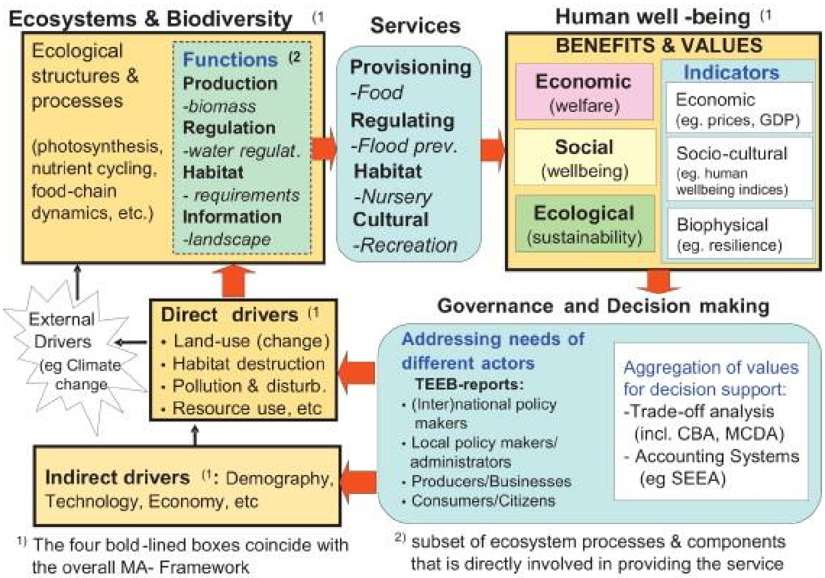


Figure 1: The TEEB conceptual framework for Ecosystem Services and Human wellbeing. (Source: Kumar, 2012).

Environmental economics has been used to quantify ES. This involves quantifying both the material and non-material benefits, which is challenging as non-material benefits are often external to the market. These externalities are benefits that we obtain for free, making the full monetary value of ES challenging to quantify. However, a landmark study (Costanza *et al.*, 1997) estimated a global annual value for ES of $US 16 – 54 trillion/yr, with an average ES value of $US 33 trillion per year. In an updated study using the same methodology as the original study, the value of ES was estimated at $US

125 – 145 trillion per year (Costanza *et al.*, 2014). The global GDP for 2020 was reported as $US 85.34 trillion (Statista, 2023), so the global monetary contribution of ES is exceptionally large.

However, demonstrating the two-way interactions of humanity and ES as shown in the TEEB framework, anthropogenic impacts have degraded ES through global land use change from 1997 – 2011, leading to a decline in the value of ES. The decline equated to an estimated $US 4.3 – 20.2 trillion lost per year (Costanza *et al.*, 2014) . The ES value change across the 1997 and 2014 Costanza studies was driven by an increased availability of data, not a change in methods suggesting these figures are underestimates. Indeed, the authors conclude “many ecosystem services are literally irreplaceable”.

1.6 Wetland Ecosystem Services

Wetlands are consistently regarded as one of the most valuable terrestrial ecosystems in terms of ES (Costanza *et al.*, 1997, 2014; de Groot *et al.*, 2012). The latest estimate is that natural wetlands provide an ES value of $US 47.4 trillion annually (Davidson *et al.*, 2019). This high value is due to wetlands delivering broad ES as shown in Table 3.

An obvious ES of wetlands is their contribution to the global hydrological cycle. Wetlands can control flood regulation, groundwater levels, and seasonal flows (Bullock and Acreman, 2003). Wetlands contribute to water quality (Verhoeven *et al.*, 2006) and supply (Davidson *et al.*, 2019). The contribution of peatlands to the global freshwater supply is less clear. Hotspot ‘water-supply peatlands’ which provide potable water to major population centres is a concept proposed by Xu *et al.* (2018), who calculated that water supply peatlands provide c.3.3% of all reservoir water globally. The study also found that the localised importance of potable water from peatlands can be extremely high. For example, up to 85% of the potable water supplied by peatlands directly is consumed within the UK and the Republic of Ireland showcasing the importance of good condition peatlands in ensuring sustainable water security.

Almost all ES provided by wetlands, as shown in Table 3, are maximised when an appropriate hydrological regime is maintained (Okruszko *et al.*, 2011) and the direct and indirect impact of human drivers on the natural ecosystem structure and function are minimised (Russi *et al.*, 2013). Maximising the ES potential of wetlands is essential to providing long term sustainable human wellbeing (Barbier, 2011). However, when wetlands are impacted by land use change, this ES potential is reduced or ceases entirely.

Wetlands play a key role in freshwater nutrient cycles. The regulation of nitrogen (Saunders and Kalff, 2001) and phosphorous (Reddy *et al.*, 1999) as wetland ES are well documented. More recently the ES of silica regulation in wetlands has been identified (Struyf and Conley, 2009). The ES benefits of nutrient cycling are widespread, as benefits are observed within geographically isolated wetland ecosystems (Marton *et al.*, 2015) and wetland ecosystems connected to downstream ecosystems where benefits are found downstream (Wolf, Noe and Ahn, 2013).

The carbon sequestration services provided globally by wetlands are significant, although carbon sequestration rates vary by wetland type and location. A general carbon sequestration range for wetlands globally has been reported as 20 - 140 g C m^{-2} yr^{-1} (Mitra, Wassmann and Vlek, 2005). Temperate Northern hemisphere peatlands are estimated to sequester 29 g C m^{-2} yr^{-1} (Gorham, 1991), while cooler boreal peatlands may achieve 12 - 26 g C m^{-2} yr^{-1} (Kuhry and Turunen, 2006). Tropical wetlands have the largest carbon sequestration potential, as mangrove swamps may achieve 90 - 230 g C m^{-2} yr^{-1} (Suratman, 2008) with tropical forest and floodplain wetlands achieving a sequestration rate of 42 - 306 g C m^{-2} yr^{-1} (Mitsch *et al.*, 2013).

There is scientific consensus that anthropogenic degradation of wetland habitats is reducing the carbon sequestration potential of wetlands worldwide (Lal, 2003, 2004; Lal *et al.*, 2018). Wetlands can be a source of methane, which like carbon emissions can affect radiative forcing. However, modelling suggests that even accounting for methane emissions, wetlands may sequester 1.28 Pg of C per year, while releasing 0.448 Pg of C via methane release. The overall balance is that wetlands act as a net sink of 0.83 Pg of C per year (Mitsch *et al.*, 2013).

It is important to note here that many of the figures for carbon sequestration quoted above have increasingly come to be recognised as specific to peatlands. Whereas in the past, peatlands have repeatedly been overlooked, mis-characterised or simply subsumed within the broad term 'wetland', their very particular capacity for carbon storage and sequestration has become increasingly recognised as a key component of the whole climate change debate. While peatlands provide a wide range of ES, it is perhaps the recognition of their role in the carbon cycle which has led to renewed interest in peatland ecosystems.

1.7: Peatland Ecosystem Services

Peatlands provide examples of every ES referred to in Table 3. However, the peatland ES which has galvanised so much recent interest and activity is undoubtedly the role of peatlands in climate regulation and carbon sequestration. Known peatlands are acknowledged to occupy 2 - 3% of the global land area (Turetsky *et al.,* 2015) but store more carbon than is stored in the biomass of all terrestrial ecosystems combined, including rainforests. The largest peatland carbon store is concentrated in the Northern hemisphere 500-600Gt (Yu *et al.,* 2010). An additional 100Gt of carbon is thought to be found elsewhere in the world most commonly in tropical peatlands (Page, Rieley and Banks, 2011) and southern peatlands (Yu *et al.*, 2010) giving an overall range of 500-700Gt. This represents approximately 21% of the global terrestrial carbon store.

Table 3: ES provided by wetlands, (Source: Barbier, 2011 & Russi *et al.*, 2013).

Ecosystem services	**Ecosystem structure and function providing this service**
Coastal protection	Attenuates and/or dissipates waves, buffers wind
Erosion control	Provides sediment stabilization and soil retention
Flood protection	Water flow regulation and control
Water supply	Groundwater recharge/discharge
Water purification	Provides nutrient and pollution uptake, as well as retention, particle deposition
Carbon sequestration	Generates biogeochemical activity, sedimentation, biological productivity
Maintenance of temperature, precipitation	Climate regulation and stabilization
Raw materials and food	Generates biological productivity and diversity
Maintains fishing, hunting and foraging activities	Provides suitable reproductive habitat and nursery grounds, sheltered living space
Tourism, recreation, education, and research	Provides unique and aesthetic landscape, suitable habitat for diverse fauna and flora
Culture, spiritual and religious benefits, bequest values	Provides unique and aesthetic landscape of cultural, historic, or spiritual meaning

As suggested above, peatland carbon sequestration rate and carbon density exceed those of all other oceanic and terrestrial ecosystems (Temmink *et al.*, 2022). Based on the estimated peatland carbon store (500-700 Gt), peatlands store more carbon than all the above ground biomass from other biomes combined, which stores 310-422 Gt (Liu *et al.*, 2015).

Peatlands support a range of biodiversity due to the abiotic restrictions produced via peat formation. Abiotic restrictions in peatlands include a low soil oxygen content, poor nutrient availability, a high-water table and abnormal pH (Parish *et al.*, 2008; Minayeva and Sirin, 2012). Such restrictions drive competition for scarce resources. Due to competition most species associated with wetlands have evolved adaptive strategies to gain a competitive advantage in such unique habitats.

Peatlands support flora and fauna adapted to them, but also provide habitat for a range of other migratory and transitional species (Minayeva, Bragg and Sirin, 2016). By providing support for species not permanently associated with peatlands, the value of peatlands as an ES for biodiversity extends beyond peatland habitats and can have wider regional impact (Keddy *et al.*, 2009). Peatland biodiversity includes a range of rare plants, unique breeding bird assemblages and a species rich assemblage of invertebrates (Littlewood *et al.*, 2010).

ES can, however, be a double-edged sword if interpreted in particular and singular ways, most specifically in terms of provisioning services. This has been the approach applied to wetlands and peatlands for millennia, with enormous consequences for society and the environment. While this focus on provisioning services has undoubtedly generated much food for many and much wealth often for a few, this has been at considerable cost to the landscape, climate and society in terms of flood risk, infrastructure cost, biodiversity loss and, most pressingly, climate change.

Of the lowland peatland area listed previously in Table 1, less than half is still considered to support peatland habitat in various condition states (see Table 4), meaning that more than 0.5 million ha of peat soil in the UK lowlands is currently not peatland habitat. It has instead been transformed by a sole focus on provisioning services into some form of land use that treats such ground as something other than a peatland system.

To provide an understanding of how this area of peat soil, lacking existing peatland habitat, has come into being, but also to understand the challenges facing any proposal to bring about changes to the *status quo* on these soils, it is necessary to look at the long and inter-connected history of western agriculture and its relationship with peatland ecosystems.

Table 4: Extent of surviving peatland habitat in Britain. 'Habitat' refers to any form of peatland habitat whatever its condition, while 'mire' refers to examples of the relatively undisturbed habitat still considered to be peat-forming. (Source: Lindsay and Clough, 2017).

Soil/habitat type	**England** (ha)	**Scotland** (ha)	**Wales** (ha)	**UK total** (ha)
Raised bog habitat	17,411	13,000	1,830	53,317
Raised bog mire	3,295	6,500	9,000	25,795
Fen habitat	117,000	238,000	36,000	437,067
Fen mire	67,860	138,040	21,000	253,619
Total UK peat habitat	**389,719**	**2,010,000**	**107,830**	**2,701,541**

The next section will therefore consider the consequences of this single focus on provisioning services from wetlands and peatlands, which will then help to set the specific context for the particular research programme described in the remainder of the thesis.

1.8 The story of farming and its focus on 'dryland' agriculture

The Neolithic revolution began around 9,000-11,500 years ago in the Near East, and established agriculture as a key driver of human existence (Zeder, 2011). There are contrasting views on the precise date range (Lev-Yadun, Gopher and Abbo, 2000), location of origin (Willcox, 2005) and the dispersal route of agricultural practice across regions (Lazaridis *et al.*, 2016). However, it is broadly agreed that the Neolithic revolution was a step change towards modern day domestication of crops and livestock. This was achieved through concerted efforts to systematically modify local environments and biotic communities to promote species of interest and is evidenced throughout the archaeological record (Brown *et al.*, 2009).

The first wild crops that were domesticated are known as founder crops (Zohary, 1996). It was once believed that all founder crops began from a single point of origin and were distributed globally from this source. However, the dominant theory within the literature is one of several independent agricultural revolutions. Founder crops varied by region: maize and beans were domesticated in Central America, potatoes were cultivated in South America, rice & millet were adopted in Asia, and African millet and sorghum were domesticated in Africa (Bellwood, 2005).

Crops and cultivation knowledge expanded beyond their points of origin via trade or warfare. This expansion led to the cultivation of founder crops in regions with differing ecosystems to their point of origin. To maximise the productivity of founder crops in new ecosystems, the environment was manipulated to suit the crop species. Thus, dry land focused cultivation became the dominant paradigm for agriculture as the founder crop species expanded across new regions.

Within the arid, semi-desert conditions of what is termed the ‘fertile crescent’ region embracing North-eastern Egypt, Anatolia and Mesopotamia (Weiss and Zohary, 2011), eight founder species are thought to have been domesticated to form the basis of what is perhaps best termed ‘conventional Western agriculture’. These included barley (*Hordeum vulgare*), emmer wheat *(Triticum dicoccoides)* and einkorn wheat (*Triticum monococcum)* (Weiss and Zohary, 2011)*.* These founder crops evolved under selection pressures arising from edaphic factors typical for their region of origin. Consequently, the species selected for early agriculture experienced a competitive advantage in dry arid conditions.

Thus, dryland-focused cultivation became the dominant paradigm for conventional Western agriculture as the founder crop species and their accompanying agricultural practices expanded westwards across new regions, reaching westernmost parts of Europe around 7,000 BC in the wave of Neolithic cultural transformation, which increasingly altered society from semi-nomadic hunter-gatherers to settled pastoralists.

The legacy of founder crop expansion is evident today. The domesticated variants of founder crops and livestock still dominate the species used for modern conventional Western agriculture. Indeed, even though this shift from hunter-gatherer to settled agriculture took a different path in Asia, focusing instead on rice which was, and is cultivated under wetland conditions, 90% of the world’s calories are nevertheless provided by < 1% of the worlds known edible species (Vavilov *et al.*, 1992; Cheng, 2018).

The crop legacy of conventional Western agriculture has ensured that for thousands of years the majority of agricultural effort, at least within Europe and its subsequent colonies, has remained focused on these few founder species suited to drier soils. Whilst there are some sound reasons for this focus on 'dryland' agriculture, not least the difficulties of access, harvest and operating machinery on wet soils, wetlands and wetland species are some of the most productive ecosystems on Earth. Mulholland *et al.* (2020) compared the productivity of various typical crop species and wetland species growing naturally. They showed that at least some wetland species can perform as well as, and in some cases even out-perform, some of the best conventional crops (see Figure 2).

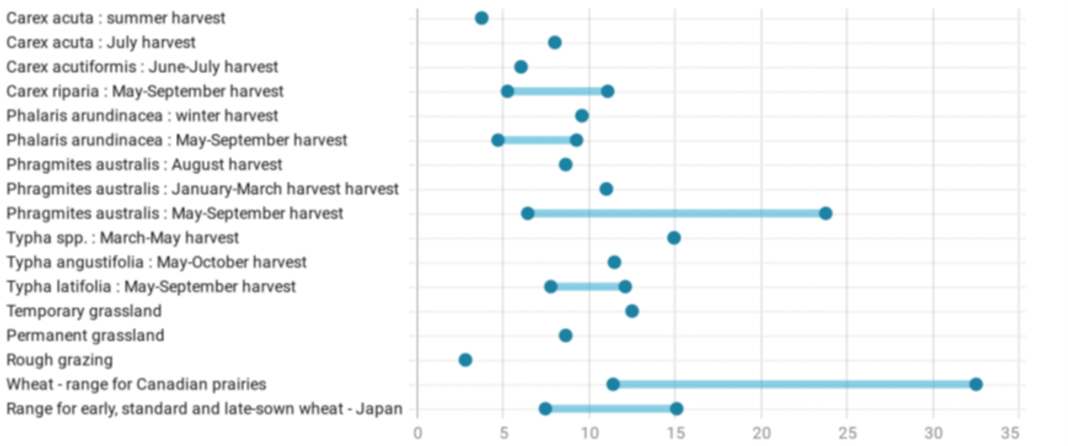


Figure 2: Average annual dry-matter yields (t ha^{-1} yr^{-1}) of selected wetland plants harvested at differing times for examples of natural or spontaneous vegetation stands, taken from Oehmke and Abel (2016), compared with dry-matter yields from grassland (Qi *et al.*, 2018) and wheat (Huffman *et al.*, 2015; Saweda *et al.*, 2019).

Some forms of wetland farming, combined with continued hunter-gathering, were practiced in certain parts of Britain until the mid-19th century (Darby, 1932; Darby *et al.*, 1979) and indeed common reed (*Phragmites australis*) continues to be harvested today as a profitable business despite the challenges of working in a wetland environment.

Nevertheless, ever-increasing demand for food production has, over the centuries, combined with opportunities for financial profitability to drive conventional Western agriculture steadily towards the belief that only drylands can be productive and that wetlands are economic wastelands.

1.9 The effect of 'dryland' agriculture on wetland systems of the world and the UK in particular

Global population change has increased agricultural demand significantly (Kopittke *et al.*, 2019). Some 2.65 million km^2 of land was used as cropland in 1700, but this area had increased to 20.39 million km^2 by 2014 (Ramankutty, Foley and Olejniczak, 2002). Globally, 14% and 26% of the ice-free land mass has been converted for crop and pasture respectively (FAO, 2014). In terms of primary crop area, cereal crop cultivation accounts for >50% of the world's primary crop area (FAO, 2021). Agricultural productivity is therefore a clear priority for human development.

The demand for agricultural productivity has resulted in large-scale conversion of natural ecosystems into agricultural ones. Agricultural land conversion has occurred across the globe and across a wide range of ecosystems. Wetlands are an ecosystem that have experienced a disproportionate ecosystem change as a result of agricultural practice. Globally, 50.9 million ha of wetlands have been converted to agriculture, forestry or grassland (Leifeld and Menichetti, 2018).

Wetland habitats are often in conflict with agriculture. The desire to convert wetlands into productive agricultural land has been the largest driver for wetland degradation across the globe (Asselen *et al.*, 2013). The global losses of wetland areas are high. It has been calculated that since the 18th century global wetland losses by region average 54 - 57%, with some regions experiencing up to 87% loss. The rate of loss has increased in the 20th and 21st centuries, with inland areas decreasing at a faster rate than coastal wetlands (Davidson, 2014).

It is estimated that by the 1980's, 48% of wetlands in the United States were lost, with at least 90% of this loss attributed to agricultural conversion (Dahl, 1990). In Western Europe many countries have lost over 90% of their original wetland peatland resource. The remaining wetland resource is typically greater in Central Europe with 10-50% of the original area remaining and Eastern Europe with less than 50% of the original area remaining (Bragg, Lindsay and Robertson, 2003).

Peatlands have generally experienced a very substantial decline in their extent, ecological condition and function. Much of the UK's peatland area is in a damaged or degraded state following anthropogenic land use change. It is estimated that 20% of the UK's remaining peatlands are undamaged (Bain *et al.*, 2011). However, calculating the extent of remaining peat soils remains problematic due to a lack of systematic surveys standardised definitions and universal protocols across the various regions of the UK. (Lindsay, 2010). The current best estimate provided by the Office for National Statistics (ONS) is that the UK has lost between 10-50% of its original peat area (Trenbirth and Dutton, 2019).

The best estimate for the remaining UK peatland extent is 3,227,197 ha, see Table 5 below, sourced from Evans *et al.* (2017),This represents around 13% of the total UK land area, but it should be recognised that the vast majority of this occurs in the uplands of the UK.

Table 5: Total estimated peat areas by UK administration

Administration	Peat area (ha)	Reference
Scotland	1,947,750	Evans *et al.*, 2017
England	Deep: 495,828 Wasted:86,372	Natural England, 2010
Wales	90,050	Evans *et al.*, 2014
Northern Ireland	242,622	Cruikshank and Tomlinson 1990 Evans *et al.*, 2017
Isle of Man	475	Evans *et al.*, 2017
Falkland Islands	282,100	Aldiss and Edwards 1999 Evans *et al.*, 2017
Total (ha)	3,227,197	

The Office for National Statistics reports the range of land use changes that have occurred on peatlands in the UK from a range of sources (Trenbirth and Dutton, 2019). These figures exclude the Isle of Man and the Falkland Islands – making the area reported 2,962,626 ha. Within the UK, only 1,949,561 ha of peatland soil area has avoided conversion into another land use. The remaining peatland area has been converted into another land use for agriculture or silviculture: Cropland (194,125 ha), Forest (439,292 ha), Grassland (234,761 ha). A further use is peat extraction (144,887 ha).

Lindsay and Immirzi (1996) demonstrate that since the early 1800s the extent of lowland raised bogs in Britain has declined from 69,664 ha to 3,836 ha of near-natural remaining habitat, mostly as small remnants, and some 5,000 ha of degraded or drained bog habitat. For the near-natural habitat this represents 94% loss.

The most famous example of peatland conversion to alternative uses, embracing both lowland bog and fen, is found in the East Anglian Fens.

Following land conversion efforts mostly since the mid-1700s, less than 1% of the original 4,000 km^2 of the East Anglian Fens remains as wetland habitat, (Sheail and Wells, 1983; Rotherham, 2013). The sole aim of this land conversion was to increase 'productive' land, according to the Institute of Civil Engineers. At least two major periods of drainage infrastructure development took place in the East Anglian Fens during 1829-1845 and 1964-1974 (ICE, 2023). Such dramatic drainage has undoubtedly led to wide ranging impacts on the peat-soil resource, landscape hydrology, and regional climate, several aspects of which will be explored in the next section.

1.10 The consequences of agricultural effects on wetland landscapes

Appropriate hydrological regimes are required for wetland ecosystem functions and processes to operate in ways that benefit humanity in sustainable ways. Despite this, anthropogenic land use change on wetland ecosystems has been dominated by land drainage as a management strategy. Drainage as a management tool has allowed humanity to increase the provisioning ES that provide raw materials and food production, but such land use change has resulted in a very substantial reduction in the number of positive ES provided by wetlands, as well as giving rise to a variety of significant negative impacts.

Drainage of peat soils seeks to lower the water table and reduce waterlogging in the surface soil layers. However, the drainage of peat soils results in fundamental changes to physical and chemical processes in peatlands. Drainage of peatlands causes oxygen to enter the peat soil, with subsequent decomposition of stored organic material, followed by an increase in CO_2 and N_2O emissions, while CH_4 emissions decrease (Kasimir-Klemedtsson *et al.*, 1997).

Drainage can be broadly divided into three critical phases: primary consolidation, secondary compression, and oxidation. Primary consolidation is a process where peat layers above the water table slump as interstitial or free water is lost (Eggelsmann, 1972). Secondary compression occurs as the drained peat layer at the surface acts as a weight on the wet peat below it. This additional load drives further loss of water from the underlying peat layers (Eggelsmann *et al.*, 1993; Heathwaite, 1993).

Primary consolidation and secondary compression are physical and mechanical processes that result in peat shrinkage and a denser peat store but do not themselves result in loss of peat (Lindsay, 2010). Change is limited to increased peat bulk density and reduced soil pore space in peat soils under drainage conditions compared to undrained sites (Leifeld, Müller and Fuhrer, 2011). These impacts lead to altered hydraulic properties including poorer water retention, greater fluctuations in the water table and reduced hydraulic conductivity due to the reduction of peat pore space (Price and Schlotzhauer, 1999).

Biochemical changes also occur within peat soils following drainage. Oxidation occurs as the formerly saturated organic matter is aerated. This promotes aerobic decomposition of peat material to CO_2 via microbial action (Clymo, 1983; Waddington and McNeil, 2002; Van Den Akker and Hendriks, 2017). Oxidation results in the finite loss of stored peat carbon.

Subsidence is a key ongoing impact following peat soil drainage. Subsidence is defined as the sudden sinking, or gradual downward settling of the grounds surface (Galloway *et al.*, 2016). Subsidence is easily quantified if surface levels are compared over time, and the proportion of subsidence attributed to each of the three critical drainage phases varies with time (Van der Molen, 1972; Grønlund *et al.*, 2008; Couwenberg and Fritz, 2012). Primary consolidation causes substantial changes in peat depth in the short term while secondary compression is a long-term process. Peat oxidation occurs at all stages but, like secondary compression, results in small changes to peat depth over long periods of time (Schothorst, 1977). Subsidence rates on peat soils can vary with several factors: peatland type, site condition pre-drainage, water table decrease, length of time since drainage and the degree of peat humification (Leifeld, Müller and Fuhrer, 2011; Pronger *et al.*, 2014; Regan *et al.*, 2019).

Importantly, subsidence occurs not only within the peat layers above the water table, but also impacts the waterlogged peat layers beneath. Impacts can be observed down the entire depth of the peat soil column (Egglesmann, 1975; Price and Schlotzhauer, 1999). This has implications for the nature of stored carbon even in waterlogged layers at depth in the peat column, for example in terms of bulk density and thus the quantity of carbon stored at depth, while also increasing risks beyond the peatland ecosystem by, for example, increasing the flood risk in downstream lowland areas (Ikkala *et al.*, 2021).

1.10.1 Loss of carbon store and sequestration

Carbon sequestration as a wetland ES function, is only achieved when a wetland is waterlogged and has a peat-forming vegetation cover. Drainage leads to a reduction in waterlogging and widespread vegetation change within peatlands (Regan *et al.*, 2019; Temmink *et al.*, 2022). The widespread drainage of peatlands reduces the potential for peat formation significantly and results in the release of stored carbon within peat soils. Therefore, drainage as a management strategy will inevitably result in the loss of a wetland carbon sequestration ES and shift the ecosystem from a carbon sink to a carbon source. In pure carbon terms, any land use that results in drainage on wetlands is not a sustainable long-term option. This section will show that drainage and land use change cause a decline in broader ES.

The large carbon store provided by peatlands is threatened by drainage and land conversion. Drained peatlands cover 0.4% of the global land area but are responsible for 5% of all anthropogenic GHG emissions (Joosten, Tapio-

Biström and Tol, 2012). The rate of annual wetland habitat loss is estimated to be 1% of the total area per year (Temmink *et al.*, 2022), therefore action needs to be taken to protect this valuable carbon store.

1.10.2 Loss of soil resource

Unabated drainage will ultimately lead to the total loss of peat soil through primary consolidation, secondary compression, and oxidation given enough time. Peat soil losses typically vary due to climate, location, and rate of drainage. A summary of peat subsidence studies is found in Hooijer *et al.*, (2012) where a subsidence loss-range is given as 200 to 600 cm lost over 40 to 130 years in non-tropical peatlands.

One example given is cultivated peat soils in the Zennare Basin, Italy, where c. 1.9m of soil has been lost due to subsidence since the 1930's, leaving c.1m peat remaining (Zanello *et al.*, 2011). Within the Zennare basin, field-based data collection over four years combined with modelling revealed variable mean annual subsidence rates of 3 – 15 mm yr^{-1}, with the greatest subsidence rate observed at 30mm yr^{-1} during a single heatwave in 2003. Based on the average rates this suggests that this area of cultivated peat soil has c. 66 to 333 years of production remaining. However, if the highest rate observed during the 2003 heatwave is considered, 33 years of soil resource remains. Higher subsidence rates due to heatwaves is an outcome that is likely as heatwaves are predicted to increase in frequency due to climate change (Lhotka, Kyselý and Farda, 2018).

In the UK, historic peat loss due to drainage of the Fens is displayed dramatically by a fixed datum known as the Holme Fen post. This cast iron post was sunk into the clay layer underlying the peat at Holme Fen (confusingly, in fact a lowland raised bog) until the top of the post was level with the peat surface prior to the drainage of neighbouring Whittlesea Mere in 1848 (Holman, 2009). The ground level today can be compared to level of the iron cap of this post now standing more than 4 m higher than current ground level. Peat wastage rates are reported as 92mm yr^{-1} for the first 27 years post drainage. The remaining losses up to 1978 showed annual losses of 11-18 mm yr^{-1}. The total loss of peat from drainage in 1848 to 1978 was recorded as 3.91 m (Hutchinson, 1980) and subsidence has continued to the present day.

The mean rate of loss reported in Hutchinson, (1980) taken over the whole period, 1848 – 1978 equates to 30.5 mm yr^{-1}. Current estimates of peat losses in the East Anglian Fens range from 9 – 19 mm yr^{-1} (Dawson *et al.*, 2010), so this remains a very current issue, especially with large areas of 'wasted' peat with < 40 cm peat remaining (Evans *et al.*, 2017).

In tropical peatlands, subsidence rates can be greater still. Subsidence on newly drained Acacia and palm oil plantations were monitored over 18 years (Hooijer *et al.*, 2012), where 75 cm of peat surface loss occurred in the first year alone (due to primary consolidation). Over 5 years the loss equated to 1.42 m and long term subsidence was determined as 50 mm yr^{-1}. A similar result was observed in Western Johore, Malaysia. The study determined the subsidence rate 14 years post-drainage at 46 mm yr^{-1}, with a long-term rate of 20 mm yr^{-1} 28 years post-drainage (Wösten, Ismail and Van Wijk, 1997). Data collection from the start of drainage was not available in the Malaysia study, though large subsidence values are likely to have occurred there.

Clearly subsidence losses to drainage are long-term processes with very real implications for the future viability of agriculture on peat. Without action, there may be the complete and irreversible loss of soil in which to use as the basis for conventional agriculture on peat soils.

10.1.3 Long term economic viability of long-term drainage

The ongoing requirement and cost implications of drainage are a further consequence of peat soil subsidence. Gravity-based drainage is insufficient to maintain low water table in most areas of drained peatland over long time scales. Where a drainage level is kept constant, peat is lost to subsidence and oxidation until a new equilibrium with the lowered water table is found. This results in further drainage and the associated drainage effects. The long-term outcome of drainage is that many areas of drained peat soils are now close to or below sea level (Hoogland, van den Akker and Brus, 2012). Gravity cannot move water uphill, therefore mechanical drainage infrastructure is an inevitable requirement to ensure dryland use on peat soils (Hoogland, van den Akker and Brus, 2012; Querner *et al.*, 2012). This comes at great cost.

The effects of land lying close to or below sea level are compounded by sea level rise due to climate change. For example, the west coast of the Netherlands contains large areas of peat soils. The drainage of peat soils for agricultural use currently causes peat subsidence of up to 8 mm yr^{-1} and has resulted in one third of the land area lying below sea level (Hoogland, van den Akker and Brus, 2012). This land is managed through extensive flood management measures, known as the polder scheme. The costs of such an approach are estimated to reach €1 billion per year by 2100, which equates to 0.1% of Dutch GDP (Stijnen *et al.*, 2014). Due to the economic and cultural importance of the polder system within the Netherlands, it is likely to remain technical and financially viable, despite the associated environmental degradation. However controlled flooding is being explored as means of managing water overflows and benefiting nature (van Staveren *et al.*, 2014).

The level of control and investment supplied by the Dutch government to maintain water levels in productive areas such as the Polderlands may not be suitable everywhere. Within the UK, the Environment Agency (EA) is revaluating the cost-benefit ratio of flood protection areas in NW England and is seeking to pass ownership of flood defences to local stakeholders. One area is the Lyth Valley, an agricultural area formerly on peat that undergoes heavy drainage. The proposal to reduce pumping by the EA (RSPB, 2017) met resistance from farmers who would bear significant cost for establishing a farmer led Internal Drainage Board (IDB). Discussions regarding a solution are still ongoing (ADA, 2017).

1.11 Societal and policy responses to wetland/peatland ecosystem loss

1.11.1 Peat in horticulture and the Peat Campaign

The UK demand for commercially available growing media began in the 1930's as shown in Figure *3*. The release of the first commercial 'John Innes' mixes was a move away from the traditional homemade compost used by professional and amateur growers (Alexander *et al.*, 2008). These mixes constituted a blend of loam, peat and sand. Peat was typically hand-dug at a small scale and loam was made from composted grass turves (Alexander *et al.*, 2008). The loam component was problematic; firstly it was difficult to source loam of sufficient quality and quantity and secondly loam-based mixes were heavy (Alexander *et al.*, 2008). The resulting heavy weight of the growing media mixes led to high transport costs and reduced profitability. To solve this, the horticultural industry investigated alternatives to loam. The solution involved steadily increasing the proportion of peat within the growing media blends, as it reduced the overall weight (Waller, 2012).

The most significant transition toward a greater percentage of peat within growing media mixes in the UK occurred in the 1950's through to the 1970's. This was driven by technological innovation and mechanised harvesting methods (Rotherham, 2011), innovations such as sausage cutting, baulk & hollow, and surface milling enabling greater volumes of peat to be harvested. This solved both the supply volume and cost issues associated with loam

(Bragg, 1998). The success of peat-based growing media was so great, that in the 1970’s 100% peat mixes dominated the growing media market (Waller, 2012).

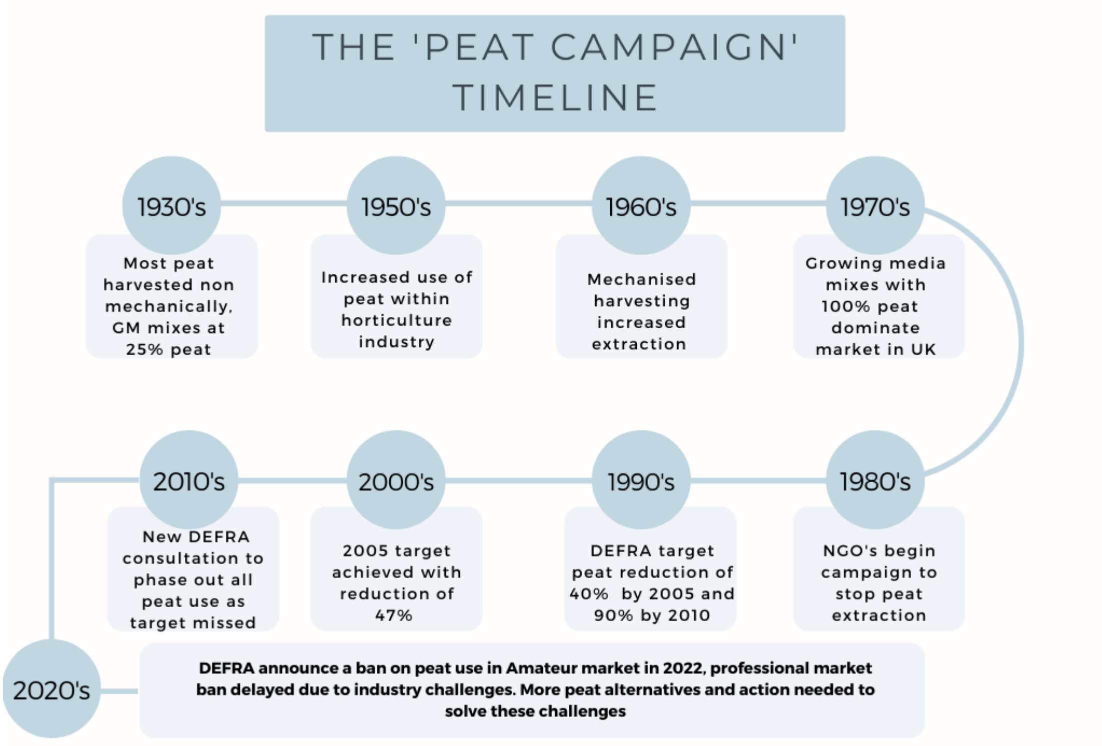


Figure 3: The Peat campaign timeline, created using data in (Alexander *et al.*, 2008; Waller, 2012; DEFRA, 2021a, 2022b).

The use of new machinery to extract peat commercially demanded a step change in extraction site management as the sites needed to be drier to permit machinery access and narrow-gauge railway construction to remove harvested peat. This involved large pumps, drainage channels and wholescale removal of peatland vegetation (Rotherham, 2011). The speed and scale of this new management for extraction led to significant and rapid conversion of lowland raised bogs into peat extraction sites (Rotherham, 2011; Karofeld *et al.*, 2017).

Thorne and Hatfield Moors in the Humberhead Levels became iconic examples of these changes. At the start of the 1990's >50% and c. 80% of the two peatlands were stripped of vegetation, drained and prepared for commercial peat extraction (Eversham, 1991; Bain and Eversham, 1992).

This huge extraction and consumption of peat in the UK attracted the attention of Conservation NGOs, industry, and government bodies (Alexander *et al.*, 2008). An early report by the Nature Conservancy Council (NCC, 1981) in the 1980s highlighted the loss of peat habitat (Bragg, Lindsay and Robertson, 1984). This report highlighted the scale of change which had occurred between 1840 and 1978 on lowland raised bogs in four major peatland areas (Lancashire, South Cumbria, Solway and the Forth Valley), revealing that 96% of the original natural habitat had been lost from these areas with only 4% remaining in good enough condition to permit peat formation. The report estimated that less than 10,000 ha of lowland raised bog habitat remained across England and Wales, and that if losses occurred at the same rate, the entire habitat would be lost within 10 to 20 years (RSNC, 1990; Lindsay, 1993). As the habitat was below 10,000 ha this triggered a critical threshold within the NCC that stated if a habitat fell below 10,000 ha, it should be protected in its entirety.

The increased awareness of the serious loss of peatland habitat demanded action. This gathered momentum and paved the way for development and launch of what was termed the Peat Campaign in the 1990's (Rawcliffe, 1998).

The Peat Campaign was launched In March 1990 by the Peatland Campaign Consortium (PCC) – a group of ten environmental NGOs. The campaign gathered evidence of 'the value, importance, damage, destruction and decline of peatlands in the UK'. The consortium recognised that peatlands offered direct wildlife interest in terms of biodiversity. However, they also argued for wider benefits such as biological indicators, genetic resource, refuges for rare species, carbon sinks and international heritage (Alexander *et al.*, 2008) – things that would be grouped under ES today (Kumar, 2012; Costanza *et al.*, 2014). This evidence, like the NCC report before it, highlighted the impact on lowland raised bogs, typically threatened by horticultural peat extraction'. The key to the campaign was the simple message: ban the use of peat to reduce the negative impacts (Rawcliffe, 1998).

The PCC produced a wide range of literature for the public to spread the ban-peat message. Friends of the Earth produced two guides for the amateur gardener 'the peat alternative manual' and 'gardening without peat'. Friends of the earth went beyond emphasising habitat loss and shifted focus towards the Green House Gas (GHG) impact of peatland extraction by publishing books such as: 'Do not disturb! Peatbogs and the Greenhouse gas effect' (Maltby, Immirzi and McLaren, 1992) and 'The global status of peatlands and their role in Carbon cycling' (Immirzi, Maltby and Clymo, 1992).

Continued cooperation between the NGO groups is evidenced through joint publications for example, the book 'Out of the mire, a future for lowland peat bogs' was produced by Plantlife and the RSPB on behalf of the peatland campaign. This publication highlighted the problems around peat extraction, placed them in their wider context and developed action points to overcome them. Key messages included the cessation of peat extraction on protected peatland sites such as SSSI's, the need to develop national monitoring and restoration techniques, and move to peat-free products to reduce pressure on peatlands (Bain and Smart 1993).

Journalists increasingly communicated these messages as interest picked up. A major piece detailing the history and expanded peat extraction by Fisons on the Thorne and Hatfield moors was published in the New Yorker magazine (Caufield, 1991), while media uptake in the UK was high. With publications in the New Scientist (Moss, 1991), Farmers Weekly (Gates, 1991), specialist interest magazines such as the Tree council magazine (Anderson, 1991) and the leisure manager magazine (Joyce, 1991) also brought the debate to wider audiences.

Many companies linked to the gardening or horticultural industries reacted to the perceived negative PR of peat use by advertising their peat-free credentials. One advertisement for B&Q in the Daily Mirror dated 18/03/1992 highlighted the problem of peat extraction in relation to SSSI's. In this advert B&Q stated that they would cease to purchase peat from any SSSI sites (B&Q, 1993). This was significant, as at the time B&Q was the third largest home and garden retailer in the world, and 60 – 80% of the peat sold via B&Q was taken from SSSI's or their national equivalents (Alexander *et al.*, 2008). The advertisement also promoted B&Q peat-free products, which is evidence for market demand shifting from peat products to peat-free alternatives.

In response to growing awareness, the Peatlands Working Group (PWG) was established by the Department of the Environment in 1992. Its purpose was to balance environmental, conservation and extraction interests on peatlands and issue policy guidance. The group initially consisted of statutory conservation agencies and representatives from the commercial extraction companies. The working group response was to suggest consultation with the NGO''s in 1993. In 1994 the PWG recommended conservation of the 'critical natural capital' of peat bogs. It proposed that the government should 'conserve examples of all peatland habitat types; establish a land use planning framework to constrain UK peat extraction to the level of horticultural demand; encourage the development and use of suitable alternatives to peat; and provide a framework for updating the conditions on existing peat planning permissions', marking a key turning point in attitudes to peat.

A House of Commons seminar took place in April 1993 between representatives of the peat campaign, the Peat Producers Association (PPA) and members of the House of Commons and House of Lords. Minutes from this meeting largely suggest that the PPA sought to dissuade a move towards peat-free alternatives, which exposed producers reluctance to change (Immirzi, 1993). The main argument put forward by the PPA was that peat accumulated faster in Europe than it was extracted, making the resource sustainable. This is an argument that makes little sense today with broad societal understanding of local, regional and global ES (Maltby and Acreman, 2011; Davidson *et al.*, 2019).

Planning consents issued before peat extraction were a problem as they were governed by planning law and were difficult to change. Consequently, even for government agencies, the only real option was to purchase or compensate extraction companies to remove the sites from extraction.

High profile peatland cases included: the Fisons company extracting peat on Thorne and Hatfield Moors and the Croxdens company extracting peat on Fenns, Betsfield and Whixall's Mosses. On October 31 1990 the NCC purchased and leased 75% of the site (1500 acres) leaving 500 acres under extraction. The purchase price was £1.6 million, making this the largest purchase in the history of the NCC. Such was the public outcry and concern for the company's image, that Croxdens accepted a realistic price with acceptable terms to the NCC (Jones, 1991).

1.11.2 Previous policy moves to reduce peat use in the UK

To combat the continued use of peat in the UK, there have been policy moves designed to reduce and ultimately ban peat use in the horticultural sector. The Minerals Planning Guidance on Peat MPG13 (Department of the Environment, 1995) introduced in England in 1995 set out the first peat-reduction target, whereby 40% of the materials used in growing media and soil improvers should be peat alternatives by 2005 and set a cap on the size of new peat extraction permissions of 1000 ha (Whitfield *et al.*, 2011). As the area of near natural lowland raised bog in the UK was estimated to have fallen by 94% from an original c.95,000 ha to c.6,000 ha by 1999, increased conservation measures in the 1999 UK Biodiversity Action Plan for lowland raised bogs, with a target to 'Undertake and promote research and development of sustainable alternatives to peat to speed up reduction of peat used in both amateur and professional markets'. It set an aim for a minimum of 40% of total market requirements to be peat-free by 2005 and 90% by 2010 (Lambert, 1999).

These voluntary peat reduction targets of 40% by 2005 and 90% by 2010 across both the amateur and professional markets brought partial success. The 2005 target was achieved, but the 90% reduction by 2010 was missed. To address this missed target, a new DEFRA consultation was launched on the reduction of peat in December 2021 (DEFRA, 2021a).

1.11.3 Peat extraction rates in the UK

Amongst this backdrop of awareness and policy, Peat extraction within the UK has generally declined. Data for annual peat extraction volumes in the UK are available from the Office for National Statistics (ONS). The available data ranges from 1997 to 2015. Peat extraction peaked in 2003 at just over 2 million cubic metres, with declining annual volumes post 2003. The total income generated from the peat extraction has a similar declining trend, as total sales declined from £119 million in 1997 to £36.2 million in 2015, based on 2017 prices (Trenbirth and Dutton, 2019).

However, the overall growing media sales volume data is available from 2011 to 2021 in Table 6. Although this does not cover the same period as the ONS statistics, it illustrates the large volumes of GM required for the total market, ranging from 3.56 to 4.93 million m3, of the growing media sales, the vast majority 77% - 91%. has a peat component, so even with recent increases in the peat-free GM volumes, more work is needed to decrease peat use.

Table 6: UK Growing media market sales by volume (million cubic metres) adapted from statistical data published by the Horticultural Trade Association (Waller, 2022)

UK Growing media volumes in millions of cubic metres (2011 - 2021)									
	2011	2012	2013	2014	2015	2018	2019	2020	2021
Amateur peat-free GM	0.19	0.25	0.23	0.23	0.24	0.12	0.12	0.54	0.67
Amateur peat-based GM	2.88	2.34	2.26	2.42	2.38	2.45	2.45	3.28	2.49
Professional peat-free GM	0.23	0.18	0.17	0.18	0.19	0.18	0.19	0.25	0.31
Professional peat-based GM	0.91	0.87	0.81	0.82	0.82	0.83	0.88	0.87	0.86
Total Amateur GM volume	3.06	2.59	2.49	2.64	2.62	2.57	2.58	3.81	3.16
Total Professional GM volume	1.14	1.04	0.98	0.99	1.01	1.01	1.08	1.12	1.17
Total GM market volume	4.20	3.64	3.46	3.63	3.63	3.59	3.65	4.93	4.34
Percentage of peat-based GM	90.2	88.2	88.3	89.0	87.9	91.5	91.4	84.1	77.3
Percentage of peat-free GM	9.8	11.8	11.7	11.0	12.1	8.5	8.6	15.9	22.7

This data on domestic peat production figures in the UK indicates that the problem of peat use is reducing gradually. However, despite declining peat extraction in the UK, peat demand and use has increased. This is largely due to increasing imports of peat from Europe as UK-produced peat accounted for just 22% of the total UK peat usage in 2014.

According to the Observatory for Economic Complexity, the UK was the 7th largest importer of peat in the world, importing 63.9 million US dollars' worth of the product in 2022. The largest importer source is Ireland ($51.3 million), Netherlands ($5.2 million), Belgium ($2.65 million), Latvia ($1.83 million), and Estonia ($852 thousand) (OEC, 2022). In fact, DEFRA states that 'Two thirds of the peat sold in the UK is imported from the rest of Europe' (DEFRA, 2021a). Therefore, the evidence indicates that the UK has simply exported the many problems associated with peat extraction to the nations from which it imports the peat products. Within this wider context, the problem is still not being addressed.

Since 2017, the use of peat in the UK has fluctuated (HTA, 2022). Between 2018-19 peat consumption in the UK declined by 2.3%, before rapidly increasing by 9% as Covid-19 lockdowns increased consumption of growing media by amateur users. During the same time, professional users reduced their peat consumption by 5% (HTA, 2022). The rapid change in demand during 2020 (shown in Table 6) was reduced in 2021 but shows how quickly the retail market response can have an impact on peat consumption. Limiting the amateur market to peat-free options could reduce this impact considerably. The professional horticultural sector in the UK has a fairly consistent peat use, averaging 0.85 million m^3, which may reflect difficulties in transitioning to peat-free alternatives (Waller, 2022).

1.11.4 Current policy moves to reduce peat use in the UK

There are promising policy developments: a consultation on the future of peat use in England was undertaken in 2020. Wildlife NGOs once again sought to put information into the public realm. For example, the Wildlife Trust reported that since 1990 'policy failure to stop peat extraction has caused up to 31 million tonnes of CO_2 to be released since 1990' (Doar, 2022). This is a figure based on 81 million m^3 of peat being extracted and the upper range for the kg of CO_2 contained in a standard cubic metre of peat given as 47 -104 kg of carbon in

Lindsay, (2010) being oxidised completely. The total oxidation of the standard cubic metre of peat has the potential to release 385 kg of CO_2 per cubic metre. This figure only accounts for the carbon losses from extracted peat, not the ongoing carbon losses from peatland degradation which will be substantially higher at 10 - 25 of tonnes CO_2e per ha per year until either the peat is wasted, or re-wetted (Evans *et al.*, 2017, 2021).

In 2022 DEFRA announced their intent to ban on the sale of peat for use within the amateur gardening sector, following a public consultation. The legislation will come into force by 2024 (DEFRA, 2022b). This is a crucial step in limiting the direct damage and drainage of peat bogs for growing media. The reach of the ban is important, as the amateur sector is responsible for c.70% of all peat sold within England (Trenbirth and Dutton, 2019). The ban doesn't extend to the professional horticultural industry at present, as DEFRA recognises that the professional horticulture sector faces additional technical barriers that will take longer to overcome. Prior to a ban for the professional horticulture sector, the government has pledged to 'work closely with the professional horticulture sector on speeding up their transition to peat-free alternatives ahead of a ban' (DEFRA, 2022b). Therefore, even in 2022, a satisfactory peat-free growing media solution remains elusive for the professional sector (Wallace *et al.*, 2010; AHDB, 2018; Mulholland *et al.*, 2020).

The industry has found it difficult to develop a peat-free growing media to serve as a direct replacement for fossil peat (Bustamante *et al.*, 2008). Peat has been in such high demand due to its unique set of characteristics. Desirable attributes include physical attributes such as a high water-holding capacity, chemical attributes such as low Ph and nutrient composition, and aesthetic attributes such as an attractive brown colour and low odour (Pryce, 1991). Furthermore, peat is cheap to produce and has received long term focussed scientific development as a material (Ogg, 1937; Robertson, 1962; Schmilewski, 2008; Turunen *et al.*, 2019). Due to a long history of development and its many positive attributes, peat remains the dominant growing media component in professional horticulture and has been regarded as indispensable by the horticultural industry (Michel, 2010).

The reduction of peat has also faced opposition from the growing media industry, who believe the total replacement of peat within GM to be both unrealistic and unfair (Waller, 2012). The International Peat Society (an association of GM producers) claim that as less than 0.1% of Britain's bogs are used for peat extraction, and contribute to 0.07% of Britain's annual CO_2 emissions, there is no substantial problem (Rieley, 2012).

However, peat extraction is evidently unsustainable from a material production view alone. Peat extraction requires the total removal of the vegetated acrotelm layer, which prevents new peat formation (Lindsay, Birnie and Clough, 2014). Extraction removes up to 25 cm of peat annually, and commercial extraction sites are typically exhausted of peat in 30 years (Waddington *et al.*, 2009). It is very likely that the lack of long-term material sustainability, changing policy environment, and the focus on enhancing peatland ES will ultimately ensure that peat extraction is phased out in the UK.

Nevertheless, future policy efforts to reduce the sale of peat are complex. This is in part because the sale of peat is an issue overseen by each of the devolved UK governments (Trenbirth and Dutton, 2019). For example, Wales is expected to enact a similar peat ban to England by May 2024, but in Northern Ireland proposals to phase out peat compost sales by 2025 were dropped from its most recent Peatland Strategy that covers 2022-2040. In Scotland, the SNP's 2021 election manifesto pledged to end the sale of 'peat-related gardening products', but no date for this has been set (SNP, 2021). A UK-wide unified approach is needed but may be difficult to enact, especially as Ireland have historically been a large peat exporter (OEC, 2022). However, it is clear that the voluntary approach to ceasing the use of peat in horticulture has been ineffective (Doar, 2022) and a legislative solution across both the retail and professional sectors is required.

1.11.5 Peat-free alternatives

Peat-Free Alternatives that have been developed and are now being used include materials such as coir, bark mulch and composted green bin waste, all of which offer challenges for the growing media industry (Alexander *et al.,* 2008). A key issue surrounding the uptake of alternative growing media components is viable supply over the appropriate timescales. Best estimates show the supply of peat alternatives falling short of demand up to 2025 (DEFRA, 2009). Cost is also an issue for many GM producers. But ultimately many growers feel that alternative GM components offer inferior performance to peat, and that peat in a diluted form will remain a key component necessary to balance the inferior GM alternatives within a GM blend (Schmilewsk, 2008).

1.12 Broader Peatland policy

1.12.1 Global policy

The importance of Peatlands is recognised internationally through many regulatory frameworks and multilateral agreements. These regulatory features provide incentives and pathways to peatland protection and restoration.

RAMSAR & Peatland Wise use concept

The carbon storage and sequestration potential of peatland is directly relevant to the United Nations Framework Convention on Climate Change (UNFCCC). The UNFCCC was proposed in 1992 and became effective in 1994 (UNFCCC, 1992). As of 2020, The UNFCCC has over 197 signatories, and acts as an international framework to develop collaborative action to limit the effects of anthropogenic climate change (UNFCCC, 2023). The UNFCCC has fostered progress since its inception. Through the framework, The Kyoto protocol was developed in 1997 (Protocol, 1997; Böhringer, 2003). Following ratification, it became effective in 2005, with 192 signatories. The Kyoto protocol was superseded by the 2015 amendment known as the Paris Agreement. The Paris Agreement commits the 196 signatories of the UNFCCC to limit global warming to less than 2°C, with the aim of 1.5˚C or less (Schleussner *et al.*, 2016). Peatlands will play a key role in limiting global warming impacts if they are protected, restored and remain functional carbon stores and sinks (Humpenöder *et al.*, 2020).

Convention on Biological Diversity

Peatlands are havens for a wide range of biodiversity. The Convention on biological diversity (CBD) was proposed in 1992 and ratified in 1993. The CBD was designed to commit signatories to restoring ecosystems and preserving biodiversity for their intrinsic value, and their benefit to humanity. The AICHI targets developed within the CBD provided strategic goals for biodiversity improvement for the period of 2011 – 2020. Of interest for peatlands was the AICHI target 15, which committed signatories to the restoration of 15% of their degraded ecosystems (Navarro *et al.*, 2017).

Sustainable Development Goals

The Sustainable Development Goals (SDGs) are a series of 17 goals developed by the United Nations designed to promote sustainable development (Jaramillo *et al.*, 2019). The SDGs were developed from the UN's previous achievements surrounding the Agenda 21 plan at the Rio earth summit in 1992, and the Millennium Development Goals in 2000 (UN, 2023). The SDGs form a key focus for global development that can support human life while protecting the environment. In 2015 the 17 SDG's were embedded within the latest global initiative 'Transforming our world: the 2030 Agenda for Sustainable Development' (United Nations General Assembly, 2015). Key targets for peatlands within the SDGs include goals 6 'Clean Water and Sanitation', 15 'Life on Land', 14 'Life below Water' and 13 'Climate Action'. The sustainable management of wetlands will be essential to ensuring SDGs are met (Jaramillo *et al.*, 2019; Seifollahi-Aghmiuni, Nockrach and Kalantari, 2019).

1.12.2 EU policy

Global initiatives must filter down to regional and local levels to ensure effective uptake. The EU has implemented several directives to meet these global agreements. There are three key EU level policies that are directly and indirectly relevant to peatlands.

EU Habitats and Birds directives

Concern for wetlands initially focused on their importance as biodiversity corridors for migratory birds. This was explicitly acknowledged through the EU Birds directive adopted in 1979. The birds directive resulted in the designation of Special Protection Areas (SPAs) that protect wetlands of international importance for birds. The Habitats directive facilitates the designation of Special Areas of Conservation (SACs) and Sites of Community Interest (SCIs), (McLeod *et al.*, 2005; Born *et al.*, 2015). Both SPAs, SACs and SCIs are connected through the Natura 2000 framework, the EU's ecological network of protected areas. These approaches have been widely adopted in member states, with 27,000 designated sites covering 18% of the EU land area, and 6% of the EU marine area held within the Natura 2000 network (Jantke, Schleupner and Schneider, 2011; European Commission, 2023a). Wetlands form a high proportion of EU designated sites within the Natura 2000 network (Evans, 2012).

EU water Framework directive

The EU Water Framework Directive, (WFD) facilitates the restoration and sustainable management of water bodies within member states. The directive supports these actions by providing a methodology and legal framework to set and monitor water quality standards. The WFD does not recognise peatlands as explicit water bodies, but recognises that they can act as buffer habitats for other water bodies, thus implicitly encourages the restoration of wetlands (Peters and Unger, 2017).

Common Agricultural Policy

The Common Agricultural policy (CAP) was developed in 1962 and is the longest serving policy adhered to by EU member states. The CAP aims to provide affordable, safe food for EU citizens, ensure a fair standard of living for farmers and preserve natural resources and respect the environment (European Commission, 2023b). The CAP achieves its aims through direct support, market measures and rural development. The CAP is a dynamic policy, that has undergone several reforms since its inception (Van Zanten *et al.*, 2014; BIOGEA, 2019). Following reform in 2003, the two pillar system was developed. Pillar one payments provided direct income support, and pillar two payments encouraged rural development actions. Critics of the CAP argue that the policy provided limited opportunities for the uptake of environmental best practice (Schmid, Sinabell and Hofreither, 2007; Scown, Brady and Nicholas, 2020). Environmentally positive actions such as adaptation to climate change or improving carbon sequestration and storage were not a component of the CAP. Prior to the reforms of 2003, the CAP incentivised overproduction at the expense of environmental degradation (Jack, 2020). The CAP reforms of 2003 disincentivised such overproduction by decoupling farm income from production and shifted the CAP focus to area-based payments.

The CAP reforms of 2013 introduced additional greening measures. The 2013 reforms aimed to restore, preserve and enhance ecosystems related to agriculture and forestry. Cap reforms for 2021 – 2027 are being formulated. These are expected to increase the focus on payments for ES and public goods (Plieninger *et al.*, 2012; Pe'er *et al.*, 2020). This represents an opportunity for peatlands (Dupraz and Guyomard, 2019; Tanneberger *et al.*, 2021). A joint letter from key peatland conservation institutions argues for three key areas of

CAP reform (Greifswald Mire Centre, National University of Ireland, and Wetlands International European Association, 2020; Wetlands International, 2020). The three key areas are:

1. Guaranteed eligibility of paludiculture systems for CAP payments, to provide financial incentives;
2. Phasing out CAP payments for areas of agriculture operating on drained peatlands, to reduce further peat soil losses from agriculturalised peat soils;
3. Payment through the future CAP should follow a results-based agricultural payment approach that rewards ecosystem service provision, especially actions that reduce greenhouse gas emissions from peatlands.

Within the European Cap framework, policy amendments such as those referred to above can improve the management of peatlands.

1.12.3 UK specific policy

At the UK level, all devolved country regions are full signatories to the international agreements referred to earlier. From 1973 to 2020, UK policy was also aligned with that of the EU. On the 1st January 2021, the UK formally withdrew from the European Union, in an event colloquially referred to as Brexit (Arnorsson and Zoega, 2018). The long-term impacts from Brexit are unknown and many EU roles must now be redesigned into UK-only policy (Reid, 2016).

Irrespective of Brexit policy changes, environmental protection is supported by the UK government. In 2018 £ 14.5 billion was spent on environmental protection, equating to 1.7% of all UK government expenditure (Harris and Tam, 2022). Within the context of this funding environment, the UK 2020 budget announced a new £640m 'Nature for Climate Fund' targeted at tree planting and peatland restoration to reduce GHG emissions (DEFRA, 2021b).

The climate change act of 2008 committed the UK to reducing GHG emissions by 80% compared to pre-1990 levels. In 2019 the UK achieved a reduction of 40% compared to pre-1990 levels (Climate Change Committee, 2023). The UK government then raised its in ambition in 2019 by passing the Climate Change Act 2008 - 2050 Target Amendment Order. This amendment ensures that the UK is legally committed to achieving net zero emissions by 2050 (Priestly, Hirst and Bolton, 2019). This action commits the UK to reducing its contribution to global warming in any capacity to zero by 2050. This is a hugely ambitious target that will require effective and coordinated action across the UK to be met. This will be a powerful driver for improved peatland management, as damaged peatlands emit c. 23 million tonnes of CO_2e yr^{-1} (Evans *et al.*, 2017). For perspective, this is 5% of the total UK territorial greenhouse gas emissions in 2021, which totalled 424.5 million tonnes CO_2 (O'Sullivan, 2021). As part of the net zero ambitions the UK Committee for Climate Change (CCC) provides strategic policy recommendations to the government to help reach Climate Change targets. The CCC advised that to reach net zero, a necessary reduction of 35-80% in agricultural GHG emissions is needed (Climate Change Committee, 2018). The CCC also states that existing agricultural land would need to transition to alternative land uses including afforestation and peatland restoration.

More recently, a post-Brexit policy change came about in the form of the 25-year Environment Plan (DEFRA, 2018). The plan set out ambitious aims for a shift towards sustainable management of habitats across the UK. Of interest for peatlands is the recognition that 70% or more of the UK's peatland habitat is damaged, and some areas such as the East Anglian fens have 30 to 60 years of soil resource remaining without action. The plan made a firm commitment to restore and protect peatlands, phase out horticultural peat and move away from conventional agriculture on peat as it is unsustainable in terms of GHG emissions and soil health.

To assist in the 25 year plans aims, the agricultural transition plan was developed as a means of transitioning UK agriculture away from previous EU land subsidy schemes and states that it will introduce new methods to reward UK farmers for producing public goods (DEFRA, 2020) in a payment mechanism known as Environmental Land Management schemes (ELM’s) (DEFRA, 2022a) which is still being designed (Agri-Tech, 2023).

1.12.4 Peatland policy context

Recent policy developments and recommendations for peatlands have been advocating for a new approach to peatland management.

The UK has committed to developing and maintaining the natural capital of peatlands. This was recognised formally in a letter signed by the four devolved UK governments in 2013. In this letter the four Environment ministers, outlined their agreement to deliver policy coordination, partnerships, and a joint framework for peatland actions. This coordinated action acknowledges that past land use on peatlands had degraded peatlands and provides the context for renewed political action for peatland policies in the UK. In 2017 the English government provided £10 million as a fund to restore degraded peatland habitats. This funding aimed to restore 6000 ha of peatland (DEFRA, 2017).

The UK peatland strategy developed by the International Union for Conservation of Nature UK Peatland Programme (IUCN UK PP) launched in 2018 was designed to highlight key issues with peatland management in the UK. The strategy has an ambitious target which is ensuring two million ha of UK peatland are in good condition, under restoration, or being sustainably managed by 2040 (IUCN UK PP, 2018). The IUCN UK PP promotes 6 methods achieving this target as seen in Figure 4 below.

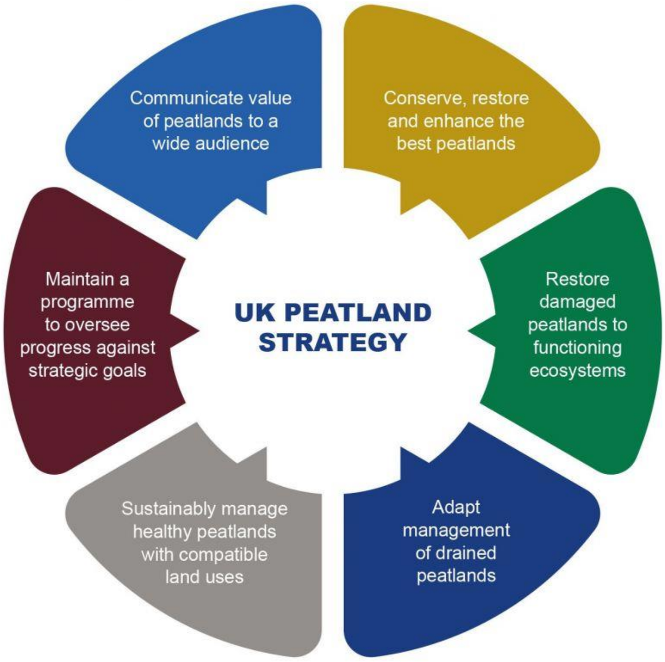


Figure 4: The 6 key components of the UK Peatland strategy as proposed by the IUCN UK PP in 2018.

The overall UK strategy is intended to provide a policy steer for England and the devolved governments as they develop their individual country level strategies. Country level plans must drive policy development and achieve innovative financing to achieve the target of the UK peatland strategy by 2040.

The England Lowland Peat strategy has been developed to further sustainable management of England's peat resource and deliver many aspects of the UK peatland strategy. In a keynote address to the IUCN UK PP conference 2020, Rebecca Pow the DEFRA minister for England gave key highlights of the strategy as phasing out peat use and demand for horticulture, driving restoration of 35,000 ha of peatland and developing commercially viable paludiculture (IUCN UK PP, 2020a)

Evidence gaps impacting policy

To date no systematic soils directive exists in the UK following the withdrawal from the 2014 EU Soil Framework Directive. Key policy decisions are also impaired by lack of complete and reliable soil condition monitoring within the UK. The Sustainable Soils Alliance published the results from a Freedom of Information request (FOI) from DEFRA in 2020. This revealed that the UK spends very little on soil quality monitoring: the UK spent £60.5m on water quality monitoring, £7.65m on air quality monitoring and £0.28 million on soil quality monitoring in the year 2017/2018 (SSA, 2023). Due to this disproportionally low funding, there are limited high resolution soil data available in the UK. The lack of a detailed evidence baseline makes it challenging to make informed policy decisions and evaluate the impacts of new policy actions on soil quality, especially regarding soil carbon.

This evidence gap must be closed. Avoiding losses and protecting current carbon stocks are a key policy recommendation set out by the UK Natural Capital Committee (NCC, 2020), who argue that soil asset statistics are not routinely incorporated into national statistics which must change as the NCC states: “The maintenance of biocarbon stocks held within natural assets such as soils is as, if not more, important than creating new stocks of biocarbon”. As a result, the Government should devise ways of monitoring and improving these metrics.

This is a problem for peatlands. In an assessment of the UK’s peatlands in 2011, the JNCC reported that a coordinated and consistent UK wide approach to peatland monitoring was required to enable accurate wetland assessment in future (JNCC, 2011) - something that still does not exist.

1.12.5 Societal campaigns and policy steps – conclusions

Overall, within the current UK policy environment, great strides will be required to achieve the necessary GHG reductions for net zero by 2050. The agricultural sector will remain one of the largest emitting sectors in the UK by 2050 (Climate Change Committee, 2018, 2020) even under the 'ambitious' reduction scenarios proposed by the CCC, which, if achieved, would deliver a reduction in emissions from the agriculture, land use and peatlands sectors by just 64% to 21 Mt CO_2 e by 2050. As peatlands are currently responsible for c.5% of the UK's total emissions (Evans *et al.*, 2017; O'Sullivan, 2021), there would still be a substantial percentage of residual emissions to account for.

Despite more than 30 years of the Peat Campaign to replace peat in horticulture, with the message of 'don't use peat' being repeated again and again through TV gardening programmes, gardening organisations such as the Royal Horticultural Society, and through government policy announcements, the fact remains that peat has continued to be used widely for gardening and horticultural purposes throughout this time. Even the latest Government announcement about the halting of peat use for the retail market, while a step in the right direction, still accepts the argument that no adequate replacement for peat yet exists for the professional horticultural sector.

It is this lack of an adequate universal substrate which has always frustrated, and continues to hold back, the movement to halt the use of peat entirely as a growing medium. It is also, however, the specific driving force behind the research described in this thesis. It is a driving force which arises from, and has been given powerful impetus by, the increasing recognition that conventional agriculture on peat soils cannot continue with 'business as usual' because of the high carbon emissions associated with such practices. This has resulted in a willingness to consider novel alternative approaches to agricultural production on peat soils.

1.13 Climate mitigation on agricultural peat soils

The mitigation of current GHG emissions from peat can be delivered in two ways. The preferred first method is preventing new drainage of peat soils. The second is reducing emissions from currently drained areas (Regina *et al.*, 2016). Additionally, the large volume of agricultural output and economic activity occurring on peat soils may make the complete restoration of the degraded area impossible (FAO, 2021). By adapting current land use, wider ecosystem service benefits may be reinstated. Methods for adaptation include cultivation of perennial crops, the selection of crops suited to higher water tables, reduced tillage and raising the groundwater table (FAO, 2012).

One proposed method is maintaining conventional agriculture on peat soils with a raised water table. The effect of water table on crop growth is dependent on plant species and the waterlogging tolerance of their rooting systems (Wen et al, in press). The raising of water tables on agricultural soils also need to balance peat oxidation, GHG emissions (CO_2, Ch_4, N_2O) and crop production (Renger *et al.*, 2002). Studies have suggested that higher agricultural water tables may cause a reduction in GHG emission (Renger *et al.*, 2002; Musarika *et al.*, 2017), few studies have observed increased GHG emissions with raised water table (Berglund and Berglund, 2011).

Radish crop biomass yields were shown to respond favourably to raising of water table from -50 cm to -30 cm on fen peat soil taken from Norfolk in a laboratory study (Musarika *et al.*, 2017). However, conventional cultivars of commercial crops are rarely selected for tolerance to high water tables, suggesting these positive results are unlikely to be replicated across other common crops.

Another study investigated raised water table with two typical UK winter cover crops, Rye and Vetch, which were assessed in an outdoor, peat monolith mesocosm experiment. When the water table was raised from -50 cm to -30 cm daytime ecosystem respiration (reco) was reduced by two thirds compared to an unvegetated control (Wen *et al.*, 2019). However, no significant difference was found between Net Ecosystem Exchange (NEE) or Gross Primary Productivity (GPP) across differing water table treatments. Nonetheless, raised water tables on agriculturalised peat, during the non-cropping period shows clear promise for the reduction of CO_2 losses. Additional field scale trials of a wide range of conventional crops will be needed to assess the real-world benefits of raised water tables.

The concept of carbon farming may also offer a novel mitigation option on wetlands, particularly agricultural peat soils. Carbon farming is a new activity, designed to feed into carbon offsetting and payment for ES. The process involves restoring a damaged peatland to a high level of ecosystem function, with a functioning vegetation and hydrology. The critical difference to paludiculture is that the vegetation is not harvested, and is left to accumulate as peat, the ‘crop’ is the accumulated carbon and avoided carbon losses, which can be sold by the landowner as carbon credits. This method could ensure long term carbon sequestration. If made reality, carbon farming as an activity would offer an additional pathway for achieving GHG policy targets (Tanneberger *et al.*, 2020). Currently one carbon farm is under development at Winmarleigh moss and is run by the Lancashire Wildlife Trust.

Amongst the complex, dynamic history of land use change and degradation on wetlands, many key issues still exist. Most conflicts revolve around the balance of different ES provided by peatlands. By establishing a land use that maintains positive ES for both people and the environment, these conflicts could be avoided. To be successful, this new land use would need to drastically reduce carbon emissions and achieve environmental policy outcomes, whilst ensuring a livelihood for stakeholders operating on peat soils. This land use is developing now, and it is called paludiculture.

Chapter 2. Paludiculture as a new paradigm

2.1 What is paludiculture?

Paludiculture first appeared as a concept within English scientific literature in 2007. Originally defined in Germany as the sustainable production of biomass on re-wetted peat soils (Wichtmann and Joosten, 2007). The concept of paludiculture has since developed into an innovative, rapidly developing approach for sustainable agricultural production on peat soils (Tanneberger *et al.*, 2020). The paludiculture definition has broadened to include extensive grazing for pastoral agriculture (Sweers, Möhring and Müller, 2014) and has the potential to include additional areas of land pre-disposed to high water tables beyond those areas which support peat soils - i.e. non-peat wetlands (Mulholland *et al.*, 2020). The overall concept aims to ensure truly sustainable economic output from peat soils while delivering ES benefits (see Figure 5).

Paludiculture, encourages a shift from conventional, drainage-based agriculture on peat soils. Drainage-based agricultural use of peat is inherently unsustainable (Chapter 1). Peat subsidence, environmental degradation and a reduction in positive ecosystem services (ES), are all consequences of conventional agriculture on peat soils (Wichtmann and Joosten, 2007; Wichtmann, Joosten and Schröder, 2016).

The primary focuses of paludiculture are carbon management and crop production. The first of these is undoubtedly the primary policy driver for paludiculture because governments around the world are increasingly committed, indeed are increasingly required by their obligations under the UN Framework Convention on Climate Change, to reduce national carbon emissions, and conventional agriculture on peat soils is now acknowledged as one of the largest sources of land-use carbon emissions. The second objective is, however, equally important because it emphasises the potential for a just transition to new forms of financially viable agriculture based on commercially successful wetland products.

The fundamental difference between conventional agriculture and paludiculture is that the former seeks to lower soil water tables in order to create dryland conditions whereas the latter seeks to increase soil moisture to levels that provide wetland crops with a competitive advantage over more typically dryland species which might otherwise compete with the wetland crop. Most, though not all, wetland plant species are capable of growing in non-wetland conditions, but the reverse is not true. Paludiculture therefore seeks to provide soil-moisture conditions that provide wetland plants with a competitive advantage over species which lack adaptations necessary for survival in wetland conditions.

Carbon dioxide emissions are generally reduced via peatland rewetting (Joosten, Tapio-Biström and Tol, 2012; Wichtmann, Joosten and Schröder, 2016) and thus paludiculture provides an economically viable means of doing so, but rewetting also involves a delicate balance because high water tables also tend to result in greater emissions of methane (CH_4) which is 28x more powerful as a greenhouse gas than CO_2, although it has a very short lifespan within the atmosphere compared to CO_2. If a sole focus on greenhouse gas balance is used to justify adoption of paludiculture, it is important to take into account the timescales for beneficial climate effects together with the likely quantities of CH_4 emitted versus the quantity of CO_2 emissions avoided during such timescales. From this perspective, water levels must therefore be managed to ensure that the cessation of CO_2 emissions is not achieved at the expense of increased CH_4 emissions (Couwenberg and Fritz, 2012; Kandel *et al.*, 2020; Ojanen and Minkkinen, 2020). Nevertheless, though currently a major policy driver of paludiculture, greenhouse-gas balance is not the only benefit of, nor justification for, paludiculture.

If undertaken appropriately, paludiculture should also result in the conservation of the peat body. This is achieved by encouraging and maintaining high moisture levels within the peat soil. Rewetting encourages the conservation of the peat body by halting and even reversing subsidence and oxidation losses (Galloway *et al.*, 2016). This ensures long term productive use rather than continued physical soil losses. The ideal water table height will vary by crop but largely it is expected that near surface level (WT > -40 cm) will be essential to maintain high levels of ES within rewetted peatlands (Geurts *et al.*, 2019).

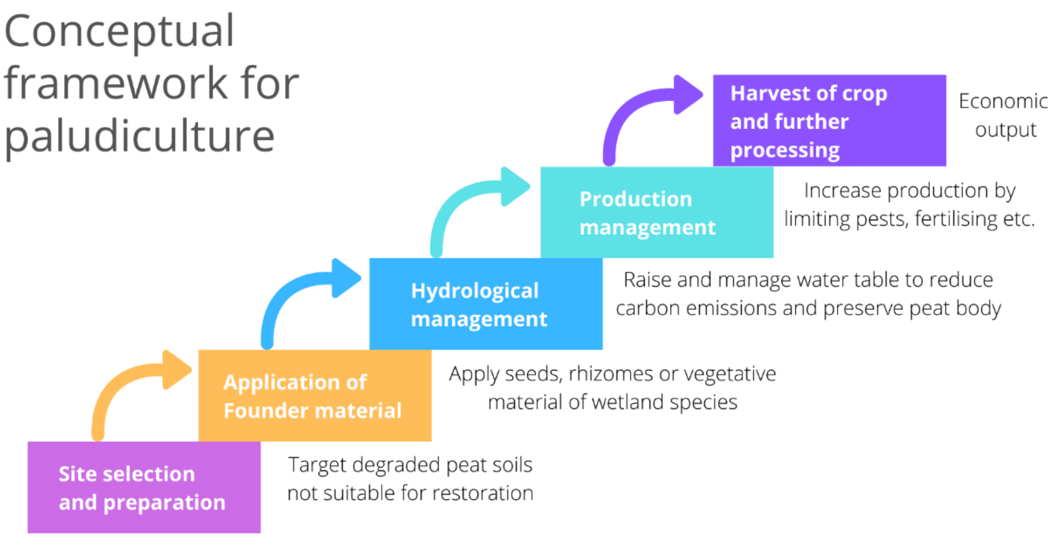


Figure 5: The Paludiculture conceptual framework.

This preservation of peat soil is important for agricultural production. For example, a major part of what creates Grade 1 agricultural land in the East Anglian Fens, which are described as the breadbasket of England, is the presence of the peat soil. Beneath this are alluvial and marine sediments which are much poorer in agricultural terms. Loss of the peat soil can be reduced or stopped using paludiculture, this means that productive agriculture can continue indefinitely, though this requires the introduction and development of species adapted to wetter soils (Wichtmann, Joosten and Schröder, 2016). Doing so, however, removes the need for deep drainage and increases the provision of positive ES that are closer to those found on undamaged wetlands (Joosten, Tapio-Biström and Tol, 2012). Crop selection is guided by three founding principles. Plant species for paludiculture systems should: thrive under wet conditions, produce biomass of high quantity and quality and contribute to peat formation (Wichtmann and Joosten, 2007).

The paludiculture concept developed following research into peatland restoration efforts in the temperate conditions of Europe and North America (Wichtmann and Joosten, 2007). The land use concept has become an international research focus with studies in Canada, Germany, Holland, Belarus and Asia (Indonesia, Sumatra, Malaysia) (FAO, 2012; Wichtmann, Joosten and Schröder, 2016; Prastyaningsih, Hardiwinoto and Agus, 2019). Crops vary by location, but the overriding focus of research is on wetland biomass production for food, biogas, building materials, growing media and raw material for additional uses (Abel and Joosten, 2012; Abel *et al.*, 2013).

Paludiculture ensures long term sustainable management of peat soils. Without rewetting there will be a loss of productive agricultural land on peatland soils due to subsidence and oxidation of the peat soil and the subsequent reduction in yields as explored in chapter one.

2.2 Distinction between paludiculture and peatland restoration

There is a real danger that paludiculture is perceived as peatland restoration under another name and is therefore regarded as an activity only of interest to, and driven by, environmental conservation interests. It is therefore vital to clarify that paludiculture is an exclusively agricultural activity having no direct connection with nature conservation. It may be that paludiculture crops will attract wetland wildlife in the same way that wheat fields attract seed-eating bird populations but attracting and restoring wildlife is not the purpose of either wheat fields or paludiculture crop stands – their purpose is to generate commercially profitable products.

As a new form of land use, paludiculture may suffer from misconceptions. Many paludiculture systems may appear visually similar to habitat restoration, this has the potential to influence stakeholder perceptions and expectations. It is therefore extremely important to make a clear distinction between habitat restoration and paludiculture operations, and also provide guidance setting out the baseline conditions under which habitat restoration may be deemed the appropriate course of action and when paludiculture would be an appropriate choice.

Restoration efforts have been taking place to preserve the remaining areas of peatland for their biological diversity and their ES. The IUCN UK peatland programme reported that In the 1980's to 1990's most restoration projects were small but more recently are scaling up to landscape scale restoration projects (Cris *et al.*, 2011). In the same IUCN report the restoration case studies presented cover a total area c.56,945 ha, with an ambition for 1 million hectares under restoration.

However, restoration can have conflicting aims with agriculture, horticulture or silviculture (tree cultivation) on peat soils. These land uses achieve high value crops and markets – so must be displaced by new crops or other economic activity of a similar value. Alternative economic replacements could be achieved through direct crop value or subsidy. For example the East Anglian Fens in England have a large crop area of 133,000 ha (Graves and Morris, 2013) and are very productive. Land under cultivation in the Fens represents 4% of England's agricultural area, but accounts for 7% of agricultural production, with an estimated net value of £1.23 billon (NFU, 2019). The displacement of this profitable but damaging land use will require careful planning and implementation.

The focus of paludiculture is, however, fundamentally different to restoration. Paludiculture aims to produce economically valuable products in addition to restoring regulatory or ES. To achieve this paludiculture has several additional stages beyond that of restoration, these include production management, crop harvesting and processing (see Figure 5). This contrasts with traditional restoration that aims to restore the broad range of ES seen in Table 3 but excluding a focus on raw material or food production (Barbier, 2011; Russi *et al.*, 2013).

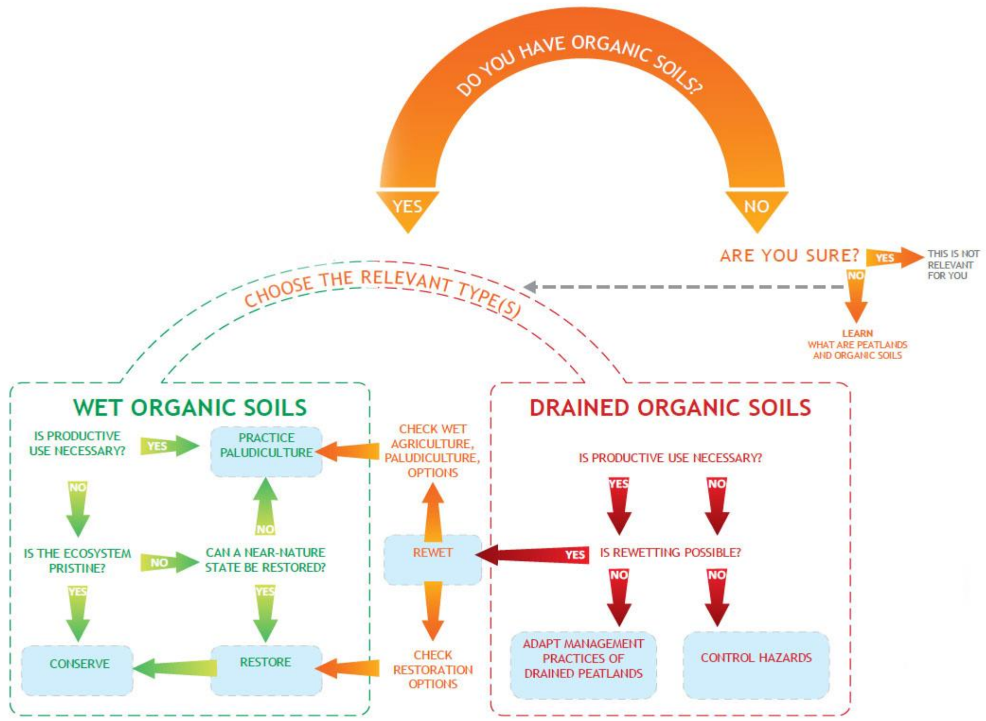


Figure 6: The Decision support framework for the management of peat soils produced by the FAO, highlighting the various pathways for conservation, restoration, paludiculture, adaptive management and hazard control (Source: FAO, 2012).

An early decision support framework for differentiating between paludiculture or restoration choices on degraded peat soils has been developed by the FAO, shown in Figure 6 shown above.

The FAO decision support framework for peat soils provides clear pathways to achieving the sustainable management of peat soils. It demonstrates that paludiculture should only be practiced where a peatland ecosystem is damaged and near natural states cannot be restored. Paludiculture offers an attractive land use where productive use is necessary, rewetting is technically possible, and restoration is not an option.

2.2 History of paludiculture

Paludiculture is a new term, but wetlands have been utilised by humanity for millennia. During the Mesolithic era there is evidence from Europe that many populations used wetlands for subsistence purposes, such as hunter-gathering, but settled on drier areas (Nicholas, 1998, 2007). From the late Mesolithic – early Neolithic, evidence of sacrifice and offerings suggest a more intense cultural relationship with wetlands and greater numbers of settlements on wetlands are observed from the Neolithic era (Larsson, 2007, 2011)

There is evidence of Neolithic wetland agriculture in the Netherlands (Cappers and Raemaekers, 2008), where settlements appear to have adopted barley cultivation close to the Swifterbant river. However most inhabited areas in the Netherlands appeared to be located on dryland, surrounded by wetlands (Out, 2008). Field experiments in the 80's suggested that unprotected salt marsh wetlands could technically have been used for cereal cultivation but were very vulnerable to flood and saltwater incursion (van Zeist *et al.*, 1976). However, it is impossible to provide a quantified, or estimated scale for the extent of early wetland agriculture.

The fenland area of the UK provides many examples of changing human interactions with wetlands. Evidence has been found for permanent site occupation and field systems from the Bronze Age (Pryor, 2005) while Iron Age evidence suggests continued use of 'fen edge' habitats. However, during the

Iron Age, as agriculture gained in importance, intensification of land use occurred in wetland habitats (Menotti and O'Sullivan, 2012).

During the medieval period many monasteries were founded on the mineral 'fen islands' scattered through the waterlogged landscape (Darby *et al.*, 1979). These monasteries as landowners showed little or no enthusiasm for drainage. The wildness of the fens was valued for its own sake, with one monk writing in 1150: "the water, standing on unlevel ground, makes a deep marsh and so renders the land uninhabitable, save on some raised spots of ground, which I think that God set up for the special purpose that they should be the habitations of His servants who have chosen to live there" Hugonis Candidi quoted in (Page, Proby and Ladds, 1936). The wetland habitat offered many useful resources pre-drainage (Williams, 1991). The large lakes or meres provided food in the form of fish and waterfowl. Seasonally dry areas provided pasture for animals. Reed and sedge were cut for thatching and turf cutting provided fuel.

The fenland economy was unique in England. The abundance of eels caught by fishermen gave them a currency like status. Eels were traded for debts, rental payments and tithes (Darby, 1932). Thatch materials were controlled via strict regulations such as cutting at specified times of the year, and sales were permitted within the medieval manors only. Lawbreakers often found themselves imprisoned or fined (Page, Proby and Ladds, 1936). Summer grazing on fen pasture was also hugely economically important, with grazing on common land allowed from 1140. Such was the importance of grazing that steps were taken to maintain summer grazing by preventing access during rogation days, which were festivals devoted to crops. Management ensured that fen vegetation could grow and recover before grazing activity took place again in the summer (Darby, 1932).

Some wetland crops have been harvested as far back as records go, with berries an obvious food source, while nettles are known to have been harvested to make clothing at least as far back as the Iron Age. Such crops fall under the provisioning ES of wetlands, and wild berries continue to be of particular importance, most notably bilberry (*Vaccinium-myrtillus*) crowberries (*Empetrum hermaphroditum*), cranberries (*Vaccinium oxycoccos*), cloudberries (*Rubus chamaemorus*) lingonberries (*Vaccinium vitis-idaea*) and raspberries (*Rubus idaeus*). The Finnish domestic berry harvest in 2011 for example yielded 26.5 million kg, with a commercial harvest of 8.4 million kg (Vaara, Saastamoinen and Turtiainen, 2013).

The American cranberry (*Vaccinium macrocarpon*) currently fuels a multi-million dollar global industry, whereas in Western Europe the bilberry remains an ecologically and economically important wetland species. It provides food for iconic species such as bears, deer and grouse (Hertel *et al.*, 2016) . The gathering of wild bilberries contributes to income for underemployed households (Barszcz, 2006). However, in Nordic countries, the abundance of bilberry as a modern crop has been impacted by the increase in commercial forestry on peat soils (Hedwall *et al.*, 2013) and the intensification of berry harvest (Hamunen *et al.*, 2019). Therefore, careful habitat management (Lõhmus and Remm, 2017) and sustainable harvesting practices will be required to prevent further losses and ensure the future viability of bilberry crops. These are sensible recommendations that could be applied to most wild berry crops found on peat soils.

Other food crops associated with wetlands have a history of use but have long been forgotten. Sweet manna grass (*Glyceria fluitans*) is one such species. *G. fluitans* is a cosmopolitan plant that can tolerate soils ranging from weakly acid to base rich conditions, (Hill *et al.*, 2000). *G. fluitans* was therefore found in a variety of wetland areas. The seeds of *G. fluitans* were gathered for their high-quality food value until the start of the 20th Century, especially in the Baltic states, and were used to pay church tithes (Łuczaj *et al.*, 2012) although it was only ever gathered from the wild using labour intensive methods. There is no clear evidence in English literature that *G. fluitans* was ever domesticated and cultivated on any large scale. *G. fluitans* was regarded as nutritious and was recommended by pharmacopoeias to patients with debilitating diseases and malnourished children (Hozyasz, 2020). In the modern world, there is new interest in investigating the gluten-free status of *G.fluitans* (Moreno *et al.*, 2014)

2.3 Abandonment of paludiculture and wild harvesting

Page Proby and Ladds, (1936), provide a thorough account of changing attitudes to wetland use in the East Anglian Fens. In the Middle Ages, the natural course of six major rivers ran through southern fenland into the sea via the Wisbech estuary. By 1292, this estuary had become infilled with silt and sediment. This caused almost all freshwater routes across the fenland region to alter their original courses. A royal commission encouraged attempts to restore these altered water courses, but all attempts failed. The new river courses left the fenland area more vulnerable to flooding. Local monasteries acted as landowners and managers and acted to secure vulnerable drier arable land using embankments for protection, whilst leaving pastureland unprotected to benefit from the nutrient enrichment brought by flooding. However, frequent winter and occasional summer floods regularly caused damage to both areas as evidenced by frequent references in the 1300's to flood damage within church records.

In the fenland example, following the dissolution of the monasteries, land ownership became fragmented. The traditional management roles governed by the monasteries were lost and much of the infrastructure for water management fell into disrepair. Consequently, political attitudes towards the fen changed. Although most of the fen area was common land, new landowners desired to increase the productivity and profits from the fenland area. The act of Sewers passed in 1531 created permanent commissioners and courts that were responsible for the management of coastal areas and flood prone agricultural land (Page, Proby and Ladds, 1936). Under the act of Sewers, commissioners could force the construction of drainage infrastructure. The first efforts at scale to drain the fens were made by a Mr George Carleton, in 1580, who drained 1000 ha in the South Holland district of Lincolnshire (Kennedy, 1983).

Wider drainage occurred with the passing of the 1600 General drainage act. This encouraged drainage as it allowed landowners to secure the services of external parties ‘for the draining and keeping dry perpetually’ of the fen, in exchange the external parties received an area of reclaimed land. This drove a boom in fen drainage to convert it from waste ground to productive land (Grove, 1981). Large numbers of venture capitalists invested in these reclamation projects and were known as Gentleman adventurers. The Earl of Bedford, was one of the most prolific adventurers, forming the Bedford Level Corporation in 1663 that drained c.38,400 ha of the fens. This was achieved using Dutch engines (windpumps), drains, and sluices (Merchant, 1983).

Technological advances in the 18th and 19th centuries enabled wider drainage. Coal and steam engines such as the Stretham Old engine developed in the 1820’s allowed greater volumes of water to be pumped (Glynn, 1836). These were replaced by diesel engines, and subsequently today’s modern electric pumping stations. A network of sluices and locks are maintained to control flooding as land subsidence continues. As discussed in Chapter 1, continued pumping is essential to maintain the agricultural viability on drained peat soils. Such pumping will be required whilst drainage continues.

The traditional pastoral farming and peasant system maintained a balance between human, environment and soils for hundreds of years (Merchant, 1983). This balance was gradually lost from the 1500’s as a move towards intensive modern farming and increased production led to increased drainage. This improved water management provided the opportunity for intensive agriculture on reclaimed peat soils. Reclaimed land provided more space and sluices reduced the risk of flooding. This allowed great expansion of arable farming in the fens. Wheat, flax, sugar beet, oats and rye became common crops. At the time of the Great agricultural depression of 1873-1896, fenland farmers were almost entirely dependent on the sale of these arable crops, and few farms kept livestock (Fletcher, 1961).

Technological advances drove the loss of traditional farming methods as horse drawn ploughs were replaced with tractors (Honnor and Lane, 2002). Industrialisation encouraged the use, purchase and trade of farming resources from elsewhere rather than the use of locally sourced materials (Grigg, 1987). Globalisation also opened farming to external markets.

The outbreak of WW1 and WW2 encouraged greater agricultural expansion and production on peat soils. By the 1950's arable cash crops had become the dominant agricultural output on peat soils in England (Merchant, 1983).

Changing National and International policy also played a role. For example the CAP provided grants to reclaim wetland (Baldock, 1984; Field, 1991). Following such agricultural intensification, less than 1% of the natural fen area present in the Middle Ages remain (Ratcliffe, 1984). The impact of this was much abandonment of paludiculture in Britain.

2.4 Current paludiculture

Paludiculture is technically possible wherever there is degraded or damaged peat soil not being used for restoration as shown in Figure 6. Therefore, paludiculture is suitable across the global peatland area from tropical (Tan, Lupascu and Wijedasa, 2021) to temperate peatlands (Wichtmann, Joosten and Schröder, 2016).

Globally there is a diverse range of wetland species with a history of human use that have potential as modern paludiculture crops (Williams, 1991; Oehmke and Abel, 2016). An early effort to create a global database of potential paludiculture plants (DPPP) was developed, with 812 species identified (Abel *et al.*, 2013). Species were grouped into four categories for paludiculture potential. Potential is scored on the predicted level of suitability, market demand, peat conservation and previous evidence of successful cultivation. Wetland species were also classified into six key main human use categories: agricultural conditioners, energy, food, medicine, ornamental and raw material, as detailed in Abel and Joosten, (2012).

Globally, 333 species across all the six human-use categories are scored in the highest paludiculture potential category (Abel, pers. comm.). At a UK level, 75 native species have been identified as being very promising, highlighted in Mulholland *et al.* (2020). These species lists are not definitive as species are continually being assessed for potential use and more species will undoubtedly be added over time.

Based on the DPPP (Abel and Thiel, 2016) tropical peatlands have a broader array of crops suitable for paludiculture than temperate or boreal areas. Tropical peat swamp timber trees such as Jelutung (*Dyera latex*), Gemor (*Alseodaphne coriacea*), Belangiran (*Shorea balangeran*) or Ramin (*Gonystylus bancanus*) are suitable for paludiculture (FAO, 2012). Most tropical peatlands are naturally forested, so a system of paludiculture based agro-forestry system is likely to represent the most sustainable option for the ecosystem (Tan, Lupascu and Wijedasa, 2021). Over 1,300 species with paludiculture potential have been found on tropical peat swamp forests (PSF) (Giesen, 2015), and more than 80% of these species have the potential to be utilised for Non-Timber Forest Products (NTFPs).

NFTPs offer many additional paludiculture choices. Species may include sago, illipe nut, banana, pineapple and edible spinach (Uda, Hein and Adventa, 2020). It is important that species selected for paludiculture in peat swamp forests are truly native species, with the necessary adaptations to thrive on the higher water tables required (Chitra, Wicaksono and Sari, 2018).

Few fully developed paludiculture projects exist in the tropics and mistakes such as incorrect species selection, or limited re-wetting, have occurred (Budiman *et al.*, 2020). Greater knowledge transfer surrounding paludiculture and adherence to the principles of paludiculture will lead to increased success of paludiculture projects in the tropics. Finally, the evaluation of research gaps and the development of tropical paludiculture frameworks will improve the uptake of paludiculture in tropical regions (FAO, 2012; Tan, Lupascu and Wijedasa, 2021; Harris and Tam, 2022).

In the temperate peatlands of the Northern hemisphere paludiculture research is more widespread and more developed as a concept. Biomass and bioenergy crops dominate the research, with key focuses on common reed: *Phragmites australis* (Wichtmann and Schäfer, 2007; Köbbing, Thevs and Zerbe, 2013; Wichtmann and Couwenberg, 2013; Wichmann, 2017) and common reedmace: *Typha latifolia* (Rebaque *et al.*, 2017; Geurts and Fritz, 2018; Vroom *et al.*, 2018). Research primarily focuses on these crops as bioenergy feedstocks or as raw material for traditional & alternative uses.

As in tropical peatlands, timber and forest products have paludiculture value in temperate northern peatlands. Black alder (*Alnus glutinosa*) is one suitable species. Alder is used for timber production and has been shown to offer net positive GHG flux change following rewetting (Bereswill *et al.*, 2017; Huth *et al.*, 2018). Natural *A. glutinosa* swamp forests are in decline within Europe (Natlandsmyr and Hjelle, 2016), therefore the cultivation of *A. glutinosa* is likely to increase the abundance of this threatened species and offer additional refugia for species inhabiting natural alder forests (Claessens *et al.*, 2010).

Paludiculture research has expanded beyond novel crop identification, as technological and methodological adaptations for the harvest of crops on rewetted peat soils have been conducted, proving that harvesting under peat conserving condition is possible (Schröder *et al.*, 2015). Studies have debated the political and social developments required for widespread adoption of paludiculture. Potential land conflicts and methods to resolve these have been proposed (Tanneberger *et al.*, 2018). Furthermore, the identification of paludiculture as a future policy objective and its role as a nature based solution to climate change has been investigated (Tanneberger *et al.*, 2021)

2.4.1 Scale

The geographical potential for paludiculture is broad. Globally there are over 80 million ha of drained peatland soils (Wichtmann and Joosten, 2007). These areas require restoration, adaptation or mitigation to reduce the negative effects of peatland drainage and degradation (as described in chapter 1). In a global review, the areas with the highest potential for paludiculture were within Europe and East Asia. Europe contains c.220 000 km² of suitable peatland area. Key countries include Russia, Belarus, Finland, Germany, Sweden and Poland. Key areas for paludiculture in East Asia accounted for 200 000 km² of potential land. Key countries identified with paludiculture potential are Indonesia, China, Malaysia and Mongolia (Barthelmes *et al.*, 2014; Renou-Wilson *et al.*, 2016).

The potential for paludiculture in Europe is also great. Permanent grassland on peat soils represents one of the largest agricultural categories for peatlands within Europe. Rewetting of organic soils used as grassland has been shown to restore the sequestration of carbon, following the removal of grazing animals (Renou-Wilson *et al.*, 2016).

Paludiculture research in Europe has largely been pioneered by the Greifswald Mire Centre (GMC), based in Germany. Within NE Germany, there exists 291,361 ha of peatland. These peatlands are highly agriculturalised. 50% of the peatland area is permanent grassland, with 7% used as arable land. The GMC has developed a spatial planning tool, that suggests the entirety of this agriculturalised peat could be converted to paludiculture (Tanneberger *et al.*, 2020; Tanneberger *et al.*, 2021). Paludiculture use is split into two modes: cropping systems and permanent grassland systems. 52-82% of the agriculturalised peat area in NE Germany may be suitable for either type. The remaining 17% of the peatland area may be limited to permanent grassland systems only, due to regional nature conservation restrictions.

Numerous European countries are the focus for paludiculture projects. Most recent paludiculture research projects have operated at pilot or field scale. However, further research is required to evaluate the potential areas for paludiculture in greater details at international, national and regional level (Geurts and Fritz, 2018; Geurts *et al.*, 2019).

The most developed paludiculture crops are shown in Table *7* on the next page, which shows that as of 2018, c.29,000 ha of paludiculture pilot projects exist. This area rises to 1.3 million ha if paper production utilising *P. australis* in China is included. Key areas to evaluate for paludiculture potential include areas such as Belarus, Canada, Germany, Romania, Ukraine and the USA.

Table 7: Overview paludiculture crops, range in water levels, important production areas, potential for carbon and blue credits based on suitability for ES such as water purification (P) and water storage (S). Table taken from (Geurts *et al.*, 2019).

Crop	Water level (cm +/- soil surface)	Product	Potential for carbon credits	Potential for blue credits	Important production areas including pilots (in ha) and potential areas (in italics)
Cattail *(Typha sp.)*	0 to +20	insulation and building material	+	P + S +	Kamp (D) 30 Zuiderveen (NL) 4 Peel (NL) 1 Bûtefjild (NL) 0.1 *Danube delta (RO)*
		bedding material	+	P + S +	Peel (NL) 1 Zegveld (NL) 0.4
		extraction of protein, fibres, cellulose	0/+	P ++ S +	*Canada*
		feed for pest-controlling predatory mites	0/+	P ++ S +	Zegveld (NL) 0.4
		fodder	-/+	P ++ S +	Peel (NL) 1 ha Zegveld (NL) 0.4
		combustion	-/+	P + S +	Canada > 500
Reed *(Phragmites australis)*	-20 to +20	thatching, insulation and building material	++	P + S ++	UK 6,500 Netherlands 4,500 Mecklenburg-Vorpommern (D) 550 Poland 8,000 Hungary 7,500 Austria 1,500 Denmark, China *Romania 190,000* *Ukraine >100,000*
		paper	++/+	P + S ++	China > 1 million
		extraction of protein, fibres, cellulose	0/+	P +/++ S ++	*Germany*
		combustion/ biogas	-/+	P +/++ S ++	Italy 0.75 Germany *Belarus & Ukraine: large potential areas*
Peat moss *(Sphagnum sp.)*	-15 to -5	high quality substrate in horticulture	++	P + S 0/+	Hankhausen (D) 14 Twist (D) 10 Ilperveld (NL) 8 Canada 8 Finland, Chile
Grasses like reed canary grass *(Phalaris arundinacea)*	-30 to +10	combustion/ biogas	-/+	P 0 S +	Malchin (D) 200 Denmark, Estonia, Belarus
		fodder	0/+	P 0/+ S +	Mecklenburg-Vorpommern (D)
Alder *(Alnus sp.)*	-40 to +5	wood/timber	++	P 0/+ S ++	Mecklenburg-Vorpommern (D) USA

2.4.2 Monetisation of ecosystem services

There are efforts to develop carbon and blue credits for wetlands and paludiculture (Bonn *et al.*, 2014; Geurts *et al.*, 2019).

Carbon credits translate carbon storage and sequestration from restoration or paludiculture into quantified tradable permits. These can be bought or sold to allow the emitting of 1 ton of CO_2 equivalent. Blue credits are similar and translate water purification, water storage, water retention and depuration, services to tradable permits. These are financial units which can be traded. Quantifying, verifying, and enabling the trading of these credits is likely to play a key role in ensuring the economic success of paludiculture.

2.5 *Sphagnum* as a multi-purpose paludiculture crop

Sphagnum moss is a key potential paludiculture crop seen in Table 7. It has been suggested that live *Sphagnum* can be used as a component of growing media (Pouliot, Hugron and Rochefort, 2015; Gaudig *et al.*, 2017; Kämäräinen *et al.*, 2018; Kämäräinen, Jokinen and Lindén, 2020). As a result, *Sphagnum* Farming (SF) is an emerging method of cultivating *Sphagnum* on a large scale (Gahlert *et al.*, 2012; Gaudig *et al.*, 2014, 2017). Successful cultivation of this bryophyte provides an opportunity to create a sustainable source of live *Sphagnum.* Such a supply would reduce the requirement for peat extraction significantly, as peat is largely made up of dead, semi decomposed *Sphagnum*.

Fresh, undecomposed cultivated *Sphagnum* could provide a much-needed direct replacement for 'White Peat', which is the lightly humified *Sphagnum* peat found in the upper peat layer on ombrotrophic bogs. This is a highly sought after growing media component due to its structural stability, low bulk density, high porosity, and low pH, nutrient and nitrogen immobilisation levels. As a result of demand, the stocks of 'White Peat' in Europe are depleting and demand Is met via imports from the Baltic states and Canada, which simply offshores this problem (Joosten, 2012, Gaudig *et al.*, 2014).

Sphagnum farming also has many additional uses and co- benefits beyond growing media. *Sphagnum* moss has a long history of medicinal use. *Sphagnum* moss has been used for wound dressings and the inhibition of microbial activity (Boateng *et al.*, 2017). Additional value is created through habitat provision. *Sphagnum* farms enhance biodiversity as they provide refugia for threatened wetland species (Muster *et al.*, 2015; Muster, Krebs and Joosten, 2020). The establishment of *Sphagnum* farms provides additional habitat that benefits wetland species of medicinal value, such as *Drosera sp.* (Egan and van der Kooy, 2013).

Cultivated *Sphagnum* biomass could also be used as founder material for the restoration of degraded peatlands. For example, the Yorkshire Peat Partnership and the Moors for the future initiatives have applied millions of *Sphagnum* plug plants and beads which were cultivated in the greenhouse via micropropagation techniques or from hand gathered sources (Wittram *et al.*, 2015). Demand for *Sphagnum* founder material will by necessity remain high if UK peatland restoration targets are to be met.

Sustainable production on peat soils coupled with enhanced peatland ES are key benefits of paludiculture (Tanneberger *et al.*, 2020). The multiple benefits of *Sphagnum* farming suggest *Sphagnum* moss will be a key crop within paludiculture systems. As a result, *Sphagnum* cultivation presents a fascinating, novel area of research. This will be explored in the next chapter.

Chapter 3. The *Sphagnum* plant

3.1 Sphagnum morphology and biology

Clymo and Hayward (1982) provide a comprehensive overview of *Sphagnum* morphology. *Sphagnum* mosses consist of a rather complex structure for a moss (

Figure 7). At the top of each *Sphagnum* plant is the capitulum, forming a compact hemispherical head consisting of multiple partially-developed buds from which new growth occurs. The stem beneath the capitulum has a collection of stem leaves distributed along its length. A series of branches also connects to the stem. These branches are of two distinct forms – pendant branches, which hang downwards and clasp the stem, and spreading branches which stand out at right-angles from the stem. Pendant branches act as a wick to draw water up towards the growing capitulum because, as with all mosses, *Sphagnum* has no water-conducting organs in its stem. The spreading branches act to keep individual plants spaced at regular intervals from each other because, importantly, *Sphagnum* never grows as individual plants but always as a mat, carpet or hummock consisting of many individual plants (Clymo and Hayward, 1982). The action of the spreading branches thus creates large pore spaces between interleaved branches and plants. These pore spaces permit water to flow both vertically and horizontally.

Sphagnum grows from the capitulum but has axillary buds at the angles between branch and stem. Regrowth of a capitulum from the axillary buds can occur if the original capitulum is damaged or removed (Clymo and Hayward, 1982; Diaz and Silva, 2012; Krebs *et al.*, 2018).

Sphagnum mosses are poikilohydric (Titus, Wagner and Stephens, 1983), lacking a leaf cuticle, stomata or a root system and thus lack any physiologically active means of controlling water movement in and out of the plant. (Goetz and Price, 2015). The physical morphology of *Sphagnum* has nevertheless evolved to retain large volumes of external water. Up to 90% of total *Sphagnum* water content is trapped between the dense network of branches and leaves (Clymo and Hayward, 1982). Water uptake is controlled by capillary action, transporting water largely via the pendant branches to the capitulum, where most primary production occurs (Weston *et al.*, 2015). This unusual system of water management has implications for the water provisioning strategies required for *Sphagnum* cultivation.

Microscopic features within *Sphagnum* leaves provide additional advantages that *Sphagnum* has evolved. *Sphagnum* leaves are one cell thick (Clymo and Hayward, 1982), promoting efficient water and nutrient uptake. These cells can be divided into two distinct types:

- hyaline cells are specialised cells lacking membranes or organelles and function almost exclusively for structural support and water storage (Lewis, 1988). Open pores in the hyaline cells facilitate ingress and egress of water (Lewis, 1988) which then enables cation exchange to occur between stored water and the cell wall (Clymo, 1963; Clymo and Hayward, 1982). Cell structure is maintained by spiral thickening or fibrils, which prevent cell collapse in times of drought (Rice, 2009).

- narrow chlorophyllose cells, where photosynthesis takes place, are squeezed between the much larger hyaline cells. Chlorophyllose cells are almost the only living part of a *Sphagnum* plant, which in part explains the ability of *Sphagnum* to survive and grow under conditions of extreme nutrient poverty.

Around 10% of total *Sphagnum* water content is retained internally within the hyaline cells, which is essential for photosynthesis (Proctor, 1955; Thompson and Waddington, 2008). Photosynthetic capacity is drastically reduced if internal water content is lowered through air-seeding. Air seeding is a phenomenon which occurs under drought conditions, whereby capillary pressure forces the meniscus of water across a hyaline cell pore inwards into the centre of the hyaline cell, collapsing the meniscus and allowing air to force the remaining water out of the hyaline cell (Thompson and Waddington, 2008) and can be irreversibly damaged following long droughts (Clymo, 1973; Gerdol *et al.*, 1998).

Finally, the cell walls of *Sphagnum* are made of lignin which is physically very decay resistant, but they also contain various phenolic compounds, waxes and polymerized lipids, which inhibit microbial activity under anaerobic conditions making them very decay resistant (Verhoeven and Liefveld, 1997).

These cellular adaptations enable *Sphagnum* to store water, resist desiccation, and increase biomass productivity by reducing the impacts of drought and decomposition.

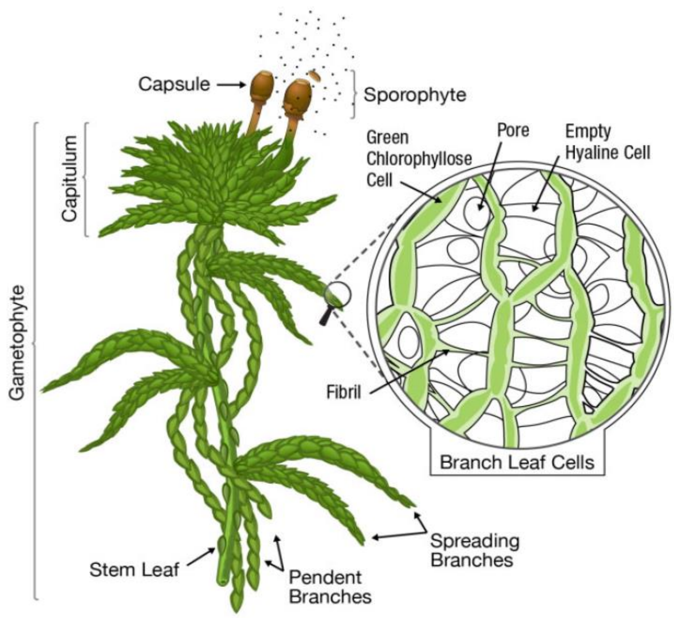


Figure 7: Schematic of *Sphagnum* moss plant. The key macro features are the stem, branches, capitulum and capsules. Key microscopic features within the branch leaf cells are highlighted: hyaline cell, chloropyllous cell, fibril and pore. (Source: Weston *et al.*, 2015).

Sphagnum has two reproductive strategies. The first is reproduction via sporulation. This process involves the development of spores which are ejected from capsules. Each capsule can hold 50 – 200 thousand spores (Sundberg, 2005). Spore dispersal is positively influenced by capsule size and spore size (Sundberg, 2010). Observation of spore dispersal shows they are ejected violently, which allows spores to cover great distances (Sundberg, 2013).

The second reproductive strategy is via vegetative propagation (Clymo and Hayward, 1982). Vegetative propagation of *Sphagnum* is achieved through the regeneration of branch and stem fragments via innovations, or new shoots (Andrus, 1986). All species of *Sphagnum* appear capable of reproducing this way (Baker and Boatman, 1990). *Sphagnum* innovations are resilient, innovations have developed from *Sphagnum* fragments within peat cores taken at a depth of 30 cm, with the estimated age of the vegetative fragments producing innovations being up to 60 years (Clymo and Duckett, 1986).

Reproductive traits are important in a paludiculture context. They will influence the ability to produce 'seed' plants for cultivation. Generally, *Sphagnum* spores have been found to establish well under controlled greenhouse conditions but perform poorly when seeded directly under field conditions (Gahlert *et al.*, 2012; Gaudig *et al.*, 2014). The vegetative reproduction strategy is therefore likely to be of greatest use for *Sphagnum* farming starter material.

3.2 Ecology of *Sphagnum* - range of habitat conditions and niche specialisation

3.2.1 Geographical Range of habitat

The distribution of *Sphagnum* is primarily driven by the supply of water and solute chemistry. In general, a habitat suitable for *Sphagnum* must have a reliable water supply to avoid desiccation and a low concentration of Ca^{2+} to avoid calcium toxicity which can cause *Sphagnum* death (Clymo and Hayward, 1982).

Genomic analysis suggests that the *Sphagnum* genus originated in cool – cold climatic regions and subsequently expanded its range into warmer tropical regions (Shaw, Cox and Boles, 2005; Shaw *et al.*, 2019). The *Sphagnum* genus contains five major evolutionary branches (clades) with species suited to high and low altitudes. It is thought that up to 90% of all genetic diversity within the genus originated in the Northern hemisphere (Shaw *et al.*, 2016).

Sphagnum mosses are highly adaptable, which has given rise to a wide species distribution across the globe. *Sphagnum* mosses are found in Asia, Europe, North America, South America and Australasia (Gunnarsson, 2005). Habitats suitable for *Sphagnum* range across latitudes from the northern arctic to the sub-antarctic (Whinam and Copson, 2006). The greatest variety and absolute abundance of *Sphagnum* species is nevertheless to be found in the northern latitudes (Gorham, 1991).

Range of habitat types

The temperate – boreal biome range provides many habitat opportunities for *Sphagnum*. Habitats that meet such conditions include mire types such as upland blanket bog, lowland raised bog, and poor fens which are low in Ca^{2+} (Gajewski *et al.*, 2001). Cooler boreal peatlands have additional mire types such as permafrost and palsa mires (Kuhry and Turunen, 2006).

In southern latitudes *Sphagnum* are often limited to high-altitude locations, where cooler and wetter conditions are more prevalent than at lower altitudes. *Sphagnum* can be found in the southern alps of New Zealand (Whinam *et al.*, 2003), the montane Parámo region of Colombia (Benavides, 2014) and the mountainous regions of Argentina and Chile (Diaz and Silva, 2012).

Subtropical *Sphagnum* is found in varied habitats such as dormant volcanic craters in the Philippines (Gates, 1915) and the high-ridge forests of Borneo (Glime, 2021). *Sphagnum* species in the tropics are uncommon, and they are not the dominant peat-forming species in the tropics. *Sphagnum* has also been found in unlikely habitats. For example, several species have been found surrounding Icelandic hot springs (Lange, 1973) thereby demonstrating the versatility and adaptability of *Sphagnum* as a genus.

3.2.2 Microtopography

Globally, *Sphagnum* mosses consist of a few hundred species (Michaelis, 2019). Their successful distribution across varied peatland habitats reflects the environmental tolerances of different species along electrochemical and hydrological gradients (Rydin, Jeglum and Bennett, 2013). Species occupy positions on these gradients according to their individual adaptations and niche preference (Johnson *et al.*, 2015). Features that influence positions on the two gradients include: desiccation tolerance (Hájek and Vicherová, 2014), favoured position relative to water table (Robroek *et al.*, 2007), cation exchange ability (Clymo, 1963) and nutrient requirements (Aulio, 1980).

Individual *Sphagnum* preferences in relation to these gradients and the associated ecological competition impacts the growth form and spatial distribution of *Sphagnum* on a peatland. These factors are represented physically through the concept of microtopography (Figure 8).

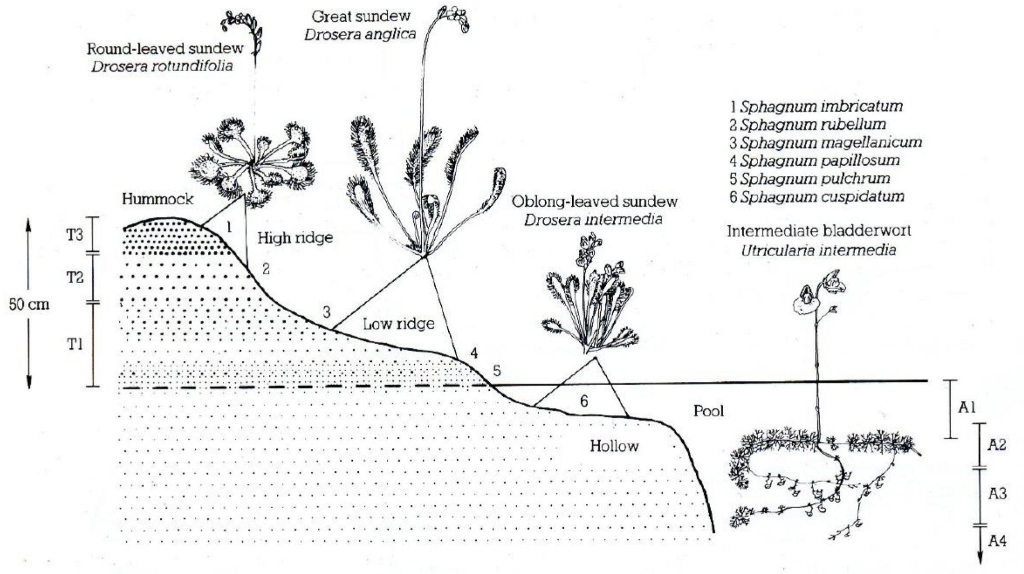


Figure 8: Microtopography within a bog ecosystem, (source: Lindsay, 2010).

Broadly, the microtopography of a peatland represents the niche that a *Sphagnum* species can occupy in the face of ecological competition and adaptation in relation to chemical and hydrological gradients. Aquatic species such as *Sphagnum cuspidatum* can thrive in very wet conditions and occupy the aquatic end of the system (A1 – A4), whereas hummock-forming species such as *Sphagnum capillifolium* can tolerate conditions further above the water table and are found in the T1 – T3 range. It is important to note that although some individual species have a competitive advantage in their micro topographical niches, if competition from other species was removed, then it is likely that individual *Sphagnum* species may occupy a wider range of microtopography zones.

3.2.2 Ecosystem engineers

Sphagnum mosses are ecological engineers. Ecosystem engineers are defined as organisms that modify, maintain, or create their own habitat through modulation of resource availability to other species, or through alteration of biotic or abiotic states (Jones, Lawton and Shachak, 1994). As detailed earlier, *Sphagnum* benefits from several physical adaptations that increase its persistence in peatland habitats. These beneficial attributes inhibit competition from other species. *Sphagnum* therefore takes an active role in managing its habitat to ensure a competitive advantage over other plant species. It achieves this by altering the hydrological and chemical state of its surroundings as well as by dominating the ground by forming extensive single-species or multi-species mats.

Hydrologically, *Sphagnum* gains a competitive advantage through its ability to retain large volumes of water in its hyaline cells. A dense network of pendant branches ensures that *Sphagnum* can resist desiccation through capillary action by drawing water up these pendant branches from deeper within the *Sphagnum* mat. During drought, the *Sphagnum* surface dries to a white colour, increasing the surface albedo and reflecting a large proportion of incident radiation, thereby substantially reducing water losses (van Breemen, 1995)

Chemically, *Sphagnum* exhibits a high cation exchange ability. The cell walls of *Sphagnum* are rich in uronic acids (Clymo, 1963) .When in solution *Sphagnum* cells exchange hydrogen ions (H+) from these acids for a base cation such as (HCO_3^-, SO_4^{2-}, Cl^-) (Vitt, 2008). This cation exchange releases H+ into the peat soil water. As a result, *Sphagnum* can both acidify its surroundings and capture the limited nutrients received via precipitation inputs (Verhoeven and Liefveld, 1997). *Sphagnum* also draws nitrogen directly from precipitation, giving it a competitive advantage over vascular plants that gain nitrogen from mineralisation (Malmer *et al.*, 2003).

3.3 Water relations of *Sphagnum* - individual plants

The morphological adaptations that *Sphagnum* has evolved with (section 3.1) influence an individual plant's ability to take up water. *Sphagnum* can only take up water via capillary action from deeper in the *Sphagnum* mat, by direct uptake through the open pores in its leaves.

Water table depth is often used as a proxy for the moisture availability within the acrotelm (Clymo and Fogg, 1984), and is undoubtedly an important component of the water story but does not provide the overall picture. However, it is argued that pore dynamics within peat soils and the living acrotelm are a more appropriate component to determine moisture availability in a peatland system (Weber, Iden and Durner, 2017a, 2017b)

Water potential (ψ) is the potential energy of water per unit volume relative to pure water. It is used to quantify the force influencing movement of water from one place to another. Movement may occur due to processes such as osmosis, gravity, mechanical pressure, and soil matrix effects such as capillary action (Hayward and Clymo, 1982; Price, 1997; Hajek and Beckett, 2007).

Pore water pressure (PWP) as a component of overall water potential relates to capillary action within a substrate and refers to the pressure of groundwater held within a soil or rock in the gaps between particles (pores). Also due to the unique structure of *Sphagnum* (see

Figure 7), pore spaces can be classified into three distinct categories:

Inter-plant pores, which are the space between individual moss plants or root channels of vascular plants (Ingram, Rycroft and Williams, 1974; Hayward and Clymo, 1982). Intra-plant space, which is the space between leaves, shoots and pendant branches (Price *et al.*, 2009; Price and Whittington, 2010). And inner-plant pores, which are found within the hyaline cells contained in *Sphagnum* leaves, branches and stems (Rydin, Jeglum and Bennett, 2013). In living *Sphagnum* moss, the three pore spaces (inter-, intra-, and inner-plant pore spaces) have been classified as having pore diameters of > 300, 300 to 30, and 30 to 10 μm, respectively (Weber, Iden and Durner, 2017a)

The experimental areas of *Sphagnum* farms essentially form a new acrotelm on the upper layers of degraded peat. In hydrological terms this means that *Sphagnum* farms sit within the unsaturated vadose zone. This is the area from the top of the ground surface to the water table below. Within the vadose zone PWP is determined by capillary action. Confusingly, within the literature, PWP is referred to as soil tension, soil suction or matric pressure. For the purposes of this thesis PWP in hPa will be the term used.

PWP is of great relevance to *Sphagnum* farming, as even small changes in PWP can have ecological impacts. Under water saturated conditions, 10 to 20% of the water is stored in pore spaces. Hayward and Clymo (1982) report that when a PWP of -100 hPa is encountered, the external capillary water available to *Sphagnum* falls by 80 – 90%. Furthermore, depending on the *Sphagnum* species concerned individual hyaline cells can completely empty at high pore water pressures within a range of 300 -1000 hPa, when this PWP threshold is reached *Sphagnum* looks 'white and papery' (Hayward and Clymo, 1982; van Breemen, 1995; Weber, Iden and Durner, 2017a).

In natural peatland systems, large volumes of water are retained within the acrotelm, and weak capillary forces are sufficient for water transport. However, in degraded systems this low capillary power can limit or even prevent an individual *Sphagnum* plant from accessing water via capillary forces. This is a challenge for individual plants when extracting water from a bare peat surface. When a bare peat surface is desiccated it can exceed the matric potential possible for *Sphagnum* to exploit, the *Sphagnum* cannot take up water, and therefore cannot photosynthesise and grow (Price, 1997).

In this context PWP relates directly to the capillary action generated by *Sphagnum*. *Sphagnum* exhibits a low capillary power and can only take on water from soils that exhibit a low PWP. This threshold is widely accepted to be

-100 mb (also represented as -100 hPa, -100 cm) (Hayward and Clymo, 1982; Price, 1997; Price and Whitehead, 2001).

3.4 Water relations of *Sphagnum* - whole peatland systems

At a peatland system scale, water table depth is often stated as the key environmental variable influencing *Sphagnum* distribution along the microtopographical gradient as shown in Figure 8. Specifically for *Sphagnum*, in hydrological terms, WT is a useful proxy for the PWP that must be overcome for *Sphagnum* to take up water via capillary action (Thompson and Waddington 2008). Generally, the lowering or raising of the water table can lead to rapid changes in PWP at the living surface. When the water table is lowered, the soil water content of the saturated peat layer is reduced, and *Sphagnum* capitula may begin to desiccate as the low capillary power generated by *Sphagnum* is not adequate to prevent the hyaline cells emptying of water (Robroek *et al.*, 2007; Rydin, Jeglum and Bennett, 2013).

Water table is the relative distance between the peatland surface and the fully saturated peat layer beneath. Natural peat characteristics such as bulk density and decomposition level influence the behaviour of the water table (Price, 1997; Berglund and Berglund, 2011). Peat soils generally have a capillary fringe that can reach the peatland surface when water table depth is around 30–40 cm. In peatlands with highly decomposed soils, this capillary zone may reach the mire surface when the water level is at 60 cm (Verry, 1997) because pore spaces are narrower and thus capillary action is able to function over greater distances. However, these narrower pore spaces also mean that the water table fluctuates across a greater vertical range and therefore tends to spend more time at critical distances from the living surface.

The ability of the capillary fringe to provide accessible water for *Sphagnum* at the surface is therefore regulated by the water table depth. Low water tables reduce the maximum height achievable by capillary rise towards the mire surface. This therefore limits the water supply and growth rate of *Sphagnum* at the surface (Rydin and McDonald, 1985).

Water table depth is also a key regulator of peatland accumulation and decomposition. A water table close to the surface encourages high productivity and low rates of decomposition, whereas a lowered water table permits greater air penetration of the peat matrix and therefore encourages faster and more extensive microbial decomposition.

At the mire surface, a very high-water table (that reaches or floods the surface) can encourage aquatic *Sphagnum* species to dominate. Aquatic species are poor peat formers, and readily decompose. As a result, peat formation is greater on the somewhat drier elevated parts of the peat surface. This difference in rate of peat formation between lower and higher parts of the peat surface is what creates the characteristic 'hummock-hollow' microtopography of a peat bog surface, with higher parts 'hummocks driving forward peat accumulation while wetter lower parts 'hollows' act as a brake on overall accumulation (Belyea and Clymo, 1998; 2001).

In near natural and pristine systems, a typical water table rarely falls below 30 cm and spends the majority of the time ('residence time') within the uppermost 5-10 cm of the ground surface (Ingram, 1983), whereas in degraded bog systems the water table may fall 1 m or more below the surface and fluctuate repeatedly across this whole vertical range while rarely approaching the peat surface.

3.5 Growth of *Sphagnum* - water, chemistry, and light

The natural productivity of individual *Sphagnum* species is varied. The global average mean dry biomass production is 260 gm^{-2} yr^{-1}, with a maximum reported value of 1450 gm^{-2} yr^{-1} (Gunnarsson, 2005), but productivity varies between individual *Sphagnum* species.

Sphagnum species can be classified by several Sections and each Section has its own characteristics in terms of potential productivity. It is important to highlight the fact that high productivity may be allied with high rates of decomposition, so high productivity does not necessarily equate to high rates of peat accumulation. Examples of sections and species are presented below.

Species from Section Recurvum *(S. angustifolium, S. Fallax, S. recurvum)* have a high natural productivity and can have a greater tolerance for eutrophic conditions than some other Sections (Gunnarsson, 2005). Eutrophic conditions are generally also associated with higher rates of decomposition, but where conditions are nutrient-poor, this highly adaptable Section can accumulate peat rapidly. Species from the Section Acutifolia (*S. capillifolium, S. fuscum, S. subitens*) and Section *Sphagnum* (*S. magellaniucm S. palustre S. papillosum*) are characterised by low rates of production but also low rates of decomposition because they are almost always associated with conditions of extreme nutrient poverty. Species from Section Cuspidata (*S. cuspidatum, S. tenellum, S. pulchrum*) have high rates of productivity but also some of the highest decomposition rates and are therefore not normally associated with rapid peat formation (Johnson and Damman, 1991).

Productivity is also influenced by individual site conditions such as water, light and nutrient status. (Rydin and McDonald, 1985; Aerts, Wallen and Malmer, 1992; Lamers, Bobbink and Roelofs, 2000).

3.5.1 Water

Competition between *Sphagnum* species for water is one of the major drivers of *Sphagnum* distribution in microtopography zones close to the water table (for example the T1 zone in Fig.6). Whereas in contrast, physiological resistance to water loss is a greater driver for competition in zones further from the water table (Pouliot, Hugron and Rochefort, 2015). An excess of water is also a problem for *Sphagnum*, as this causes elongation of *Sphagnum* stems without an increase in biomass, which therefore impacts on measures of productivity (Campeau, Rochefort and Price, 2004).

The availability of water is an important control on photosynthesis and therefore productivity. Photosynthesis declines with increasing water stress (Rydin and McDonald, 1985). Without photosynthetic inputs *Sphagnum* cannot grow, and productivity falls.

3.5.2 Desiccation tolerance

Laboratory-based experiments investigating recovery following drought report that the capitula of some *Sphagnum* species can survive and regrow following periods of several weeks at water potentials of -2 MPa at a relative humidity of 0.98, as potential water may be maintained by osmosis (Clymo, 1973). These experiments also show that no capitula can survive for very long at water potentials of - 100 MPa and a relative humidity of 0.5.

Clymo (1973) dried a range of *Sphagnum* capitula for 21 days and then tested regrowth potential at varying proportional humidities: 1, 0.998, 0.991 and 0.981. All six species tested (S. *subsecundum, S. recurvum, S. magellanicum, S. cuspidatum, S. rubellum, S. plumulosum* and *papillosum*) demonstrated that they could recover under conditions of highest humidity with 55% - 95% of capitula regenerating at 0.998 humidity. At the lower humidity of 0.981, recovery ranged between 5% and 55%. However, these high humidities are very unlikely to occur for long periods of time even under irrigation scenarios – and clearly a loss of capitula regeneration due to exceeding desiccation tolerance would be essential to avoid in a *Sphagnum* farming scenario.

Indeed, in a later paper, (Clymo and Hayward, 1982), Clymo presents the survival rates of individual 5 cm long *Sphagnum* plants subject to drought stress. These were rewetted after 3, 6, 10, 13 or 16 days of drought stress to assess recovery following desiccation (Figure 9). This is an important framework for the thesis experiments presented later.

All *Sphagnum* species recovered by 100% on rewetting after 3 days of drought, but after 6 days clear differences between species started to become apparent. Several species experienced a 50% reduction in their ability to recover after experiencing drought of 10 days, while recovery of most species was reduced to 20% or less after 13 days of drought. The characteristic hummock former, *S. imbricatum* (now *S. austinii*), showed no such reduction and survived entire after 16 days of drought. The two species most regarded as suitable for *Sphagnum* farming, namely *S. palustre* and *S. papillosum*, demonstrated little ability to recover after more than 13 days of drought and *S. palustre* recovery was reduced by 50% after only 6 days of drought. It is essential, therefore, from the perspective of *Sphagnum* farming and optimal productivity, that continuous droughts of 6 days or more are avoided to ensure good survival rates and continued growth of the *Sphagnum* crop.

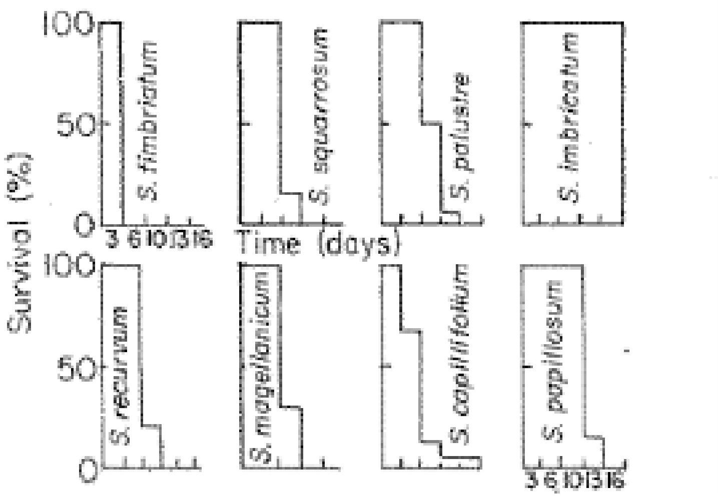


Figure 9: Survival of *Sphagnum* species. In desiccation experiments: dried *Sphagnum* species re-watered at 3, 6, 10, 13, and 16 days, with the proportion of plant capitula resuming growth recorded. (Source: Clymo 1982).

3.5.3 Water Chemistry

Sphagnum nutrient input is governed by water chemistry in combination with water availability. In ombrotrophic peatlands, *Sphagnum* receives nutrient input via rainfall, as the water table is above the influence of the underlying geology (Chapter 1).

Due to rainfall inputs, ombrotrophic bogs are typically nutrient limited (Aerts, Wallen and Malmer, 1992; Aerts *et al.*, 2001). Key limiting nutrients are nitrogen, phosphorus and potassium. However, the large surface area of a *Sphagnum* plant, coupled with a fixed negative charge across its cell walls ensure a high cation exchange capacity (CEC) (Hájek and Adamec, 2009). This high CEC ensures that *Sphagnum* can effectively take up mineral nutrients (Graham and Vitt, 2016).

This high CEC ensures that *Sphagnum* retains most of the atmospheric nitrogen it is exposed to (Li and Vitt, 1997) but this ability can also result in toxicity if nutrient loadings become sufficiently elevated. Increasing nitrogen levels through fertiliser application or increased atmospheric inputs can impact negatively on *Sphagnum* performance. Generally high N loading results in decreased *Sphagnum* productivity (Rochefort, Vitt and Bayley, 1990), (Limpens *et al.*, 2011). However, there are species dependent responses to N limitation. Negative impacts also include increased competition from vascular plants following the decline of key *Sphagnum* species such as *S. balticum* (Gunnarsson, Granberg and Nilsson, 2004), reduced photosynthetic capability *S. balticum* and *S. fuscum* (Granath *et al.*, 2009) and a reduction of physiological measurements such as shoot production (Granath, Strengbom and Rydin, 2012).

It is generally considered that the almost complete absence of *Sphagnum* from the Peak District, an upland area in the English midlands, during the last century was a response to the extremely heavy loadings of both sulphur and nitrogen oxides during the height of the Industrial Revolution. Now that levels of both atmospheric pollutants are falling, sulphur particularly so, there are significant signs of *Sphagnum* recovery throughout the area (Lee, Baxter and Emes, 1990; Caporn *et al.*, 2006; Carroll *et al.*, 2009). In particular, it appears that *Sphagnum* mosses are effective at rapid uptake of nitrogen following rainfall. The uptake rate of nitrogen is faster in N-limited sites compared to nitrogen-enriched sites (Fritz *et al.*, 2014), which suggests some *Sphagnum* species have the ability to vary nitrogen uptake which thus enables some species to reduce the risk of N toxicity at high deposition levels.

3.5.4 Light

Light can affect *Sphagnum* growth. The presence of vascular species can provide shade, which is beneficial in terms of reducing water and temperature impacts on *Sphagnum* growth (Clymo, 1973; Clymo and Hayward, 1982). However, too much shade can have deleterious consequences. Studies by Lamers, Bobbink and Roelofs (2000) and Berendse *et al.* (2001) suggest that in intact vegetation, the depression of *Sphagnum* growth under elevated N deposition conditions is explained mostly due to increased shading by vascular plants. Limpens, Berendse and Klees, (2003) conducted shading and nitrogen input experiments on *Sphagnum magellanicum* mesocosms seeded with *Betula pubescens* and *Molina caerulea* plants. The results suggested that shade positively influenced the incremental increase in *Sphagnum* height up to 53% light interception by the vascular canopy. However, beyond shading of 53% the effect of shade on *Sphagnum* productivity was found to be negative.

3.5.5 Temperature

Temperature is another environmental variable that can affect *Sphagnum* growth. In a greenhouse study, Breeuwer *et al.*, (2008) investigated the effects of four temperature scenarios on the growth of *S. fuscum* and *S. balticum* from northern Sweden and *S. magellanicum* and *S. cuspidatum* from southern Sweden. The study found that increased temperature resulted in increased *Sphagnum* height and biomass production. However, bulk densities declined with increased temperatures. Temperature increases also influenced inter-species competition. The disparity between *S. fuscum* and *S. balticum* height and biomass production decreased with increasing temperature, until no significant difference was observed at the highest temperature increase.

3.6 Reflections on the *Sphagnum* genus as a potential paludiculture crop

Sphagnum is a remarkable genus that has evolved and adapted to the cool and humid conditions found in temperate northern peatlands such as those in the UK. An extensive range and variety of degraded peatland habitats exist that offer suitable locations for *Sphagnum* farming. Meanwhile the range of *Sphagnum* species present offers an area of novel research. The development of *Sphagnum* as a crop has significant justification in the UK.

However, as this chapter sets out, *Sphagnum* mosses are highly adapted to a variety of hydrological and chemical gradients. *Sphagnum* farming should take advantage of these adaptations if it is to be a successful paludiculture option. Species selection could be tailored to the specific environmental characteristics of a selected cultivation area and to desired crop outcomes.

As a new paludiculture crop, *Sphagnum* has attracted a significant amount of interest (Gaudig *et al.*, 2014, 2017). As described in this Chapter the provision of water, nutrients and methods for reducing desiccation are key variables to control for successful production. Research into optimal *Sphagnum* cultivation is developing across the global community. However, to develop the crop potential of *Sphagnum*, several research objectives remain unanswered and must be investigated. These will be explored in the next chapter.

Chapter 4. *Sphagnum* farming as a paludiculture crop

4.1 Where *Sphagnum* farming is being attempted currently

Sphagnum farming was first designed as a concept to provide material for peatland restoration, with the suggestion of using aquatic *Sphagnum* grown in trenches for a source of *Sphagnum* material (Money, 1994). This later became regarded as a suitable means of providing biomass to displace the use of extracted peat for horticultural use (Joosten, 1998). *Sphagnum* farming as a paludiculture activity has so far typically taken place on degraded peat soils, though alternative artificial habitats have been trialled. Due to the broad range of geographical distribution and adaptations exhibited by *Sphagnum* (as shown in Chapter 3) it is helpful to differentiate *Sphagnum* farming research by geographical location. Currently, two key research areas for *Sphagnum* farming experiments are the boreal peatlands of Canada and the temperate peatlands of Europe.

4.1.1 Boreal Peatlands

Typically, these studies are focused on commercial peat extracted sites which have since been abandoned by the extraction industry for a variety of reasons. In Canada, the predominant peat extraction techniques have consisted either of the ‘block cut’ technique which results in a peat surface characterised by long trenches and raised baulks, or alternatively by the more modern approach of peat milling and vacuum extraction. Canadian *Sphagnum* farming experiments have used both block-cut and milled sites.

Canadian *Sphagnum* farming areas are often surrounded by large surviving areas of vegetated bog. This means that a ready supply of wild-harvested founder material is readily available. The extent to which experimental Canadian *Sphagnum* farming sites differ from European *Sphagnum* farming sites is debatable. Perhaps the Canadian sites experience a harsher winter climate and therefore shorter growing season (Pouliot, Hugron and Rochefort, 2015), but the actual climatic differences may not be so great at least for some of the more Central and Eastern European experimental sites. Any such differences are likely to reduce yields in Canada compared with European trials, but lessons learned are highly likely to be applicable across both climatic regions.

The main Canadian experimental site has been the Shippagan 1 site, established in New Brunswick, Canada on a former block harvested peatland. Extraction took place from 1941-1974. After this period the site was abandoned and left to revegetate naturally. Spontaneous revegetation resulted in *Sphagnum*-filled trenches and tree-covered ridges (Poulin *et al.*, 2005). Shippagan 1 was established over 11 different production cycles between 2006 and 2012 – a production cycle representing the growing seasons of late spring to early autumn because there is little or no growth during the winter season (Pouliot, Hugron and Rochefort, 2015). A second cultivation area, Shippagan 2 was established in 2012, as a new site close to Shippagan 1 (Brown, Strack and Price, 2017; Gaudig *et al.*, 2017).

4.1.2 Temperate peatlands

Germany has been the main location of *Sphagnum* farming within Europe. Two major study sites are found in Lower Saxony, Germany. The first is the Rastede site situated on former bog grassland and established in 2004. The second is the Ramsloh site, established on a commercially extracted site in 2011(Gaudig *et al.*, 2014, 2017)

Pilot studies have been conducted on 5 sites in the Kolkheti lowlands, Georgia. The sites of unknown size were situated on degraded peatland areas and adjacent mineral soil areas (Krebs, 2008; Gaudig *et al.*, 2017). Pilot studies were also developed in Latvia at the Malpils site in 2015, but no publications from those studies have yet appeared (Gaudig *et al.*, 2017). Sites of around 2 ha in Sweden were also developed in 2018, these are c. 2ha (Ludwig, 2019) and publications detailing these pilot studies and their next steps are in development.

4.1.3 UK locations

Within the UK a few *Sphagnum* farming sites have been developed. A small-scale *Sphagnum* site (0.32ha) was established at Borth Bog, Wales in June 2017, but this was non irrigated and unmanaged so remained closer to a habitat restoration scheme in both character and design (Mike Bailey – pers. comm.). The first large scale pilot site was established during August – December 2018 in coordination with this present PhD study at two sites, one on an abandoned commercially-milled peat extraction site to the west of Greater Manchester and one in an agricultural field in Leicestershire. Data from these sites form the core of this thesis.

Since this PhD research programme began in 2016, the author has also been involved through the Waterworks project with an additional paludiculture site (5 ha) that contains a *Sphagnum* farming element. The Waterworks site was established in 2020 (Great Fen, 2019). Further sites have been established through the CarePeat project in Lancashire (Lancashire Wildlife Trust, 2019), although the CarePeat project largely focuses on carbon farming rather than extractive biomass crop production.

4.1.4 Broader global locations

Researchers in Japan, Korea and Chile have some experience of *Sphagnum* farming (Landry *et al.*, 2011). *Sphagnum* farming on mineral soils has also been identified in China and SE Asia, this practice has been found to be economically viable on ex-paddy field sites (Ludwig, 2019).

4.2 What has been the approach used?

Sphagnum farming has been defined as the cultivation and harvest of *Sphagnum* moss through active management. There are several key stages to achieving active management. As such, most *Sphagnum* farming trials have employed methods originally developed for active peatland restoration, at least as the initial means of establishing the crop. In particular, methods have been based on, or at least mirrored, a standardised approach developed in Canada known as the Moss Layer Transfer Technique or MLTT (Quinty and Rochefort, 2003; Rochefort *et al.*, 2003).

The method, pioneered by Quinty and Rochefort (2003) is used to establish *Sphagnum* on bare peat sites such as those formerly subject to commercial extraction of peat. As detailed in González and Rochefort, (2014) many studies have utilised the MLTT. The technique (Quinty and Rochefort, 2003) can be summarised as follows:

- Create a level surface and/or remove the oxidised top layer of peat;
- Restore the hydrological condition of the site as far as possible by blocking drainage channels, using bunds and irrigation;
- Apply a donor *Sphagnum* material from a healthy peatland site and cover with straw mulch;
- Monitor the growth of the crop;
- Harvest once a high percentage cover of *Sphagnum* has been reached and vigorous growth is observed.

The MLTT approach, whilst widely employed and addressing some of the most challenging aspects of peatland restoration, also brings with it a number of further challenges which have a particular bearing on any attempt to establish commercially viable *Sphagnum* farming.

4.2.1 Surface preparation

Degraded peatland sites may be initially unsuitable for *Sphagnum* planting. Degradation causes the loss of the peatland acrotelm. The remaining catotelm peat is left in a highly altered state with vascular species cover, a heavily oxidised bare peat surface, which has markedly modified and highly variable surface levels. Surface preparation therefore involves rendering the remaining catotelm peat into a surface more suited to *Sphagnum* establishment. This may involve levelling the surface to remove elevated dry parts (Gaudig *et al.*, 2017) and/or removing the surface layer of highly degraded peat which may represent a particularly hostile environment for initial *Sphagnum* establishment (Hayward and Clymo, 1982; Price and Whitehead, 2001; McCarter and Price, 2014). The peat thus removed can be used to construct causeways for access, or bunds for hydrological management, so is not wasted (Wichtmann *et al.*, 2017). Levelling aims to ensure even water distribution across the cultivated area.

The levelling process has sometimes proved to be less straightforward than is the case when levelling mineral soils as a standard part of conventional agriculture. For example, prior to *Sphagnum* establishment at the Ramsloh site it was deemed necessary to remove as much as 65 cm of 'white peat' in order to provide a surface more amenable to surface levelling and hydrological control (Gaudig *et al.*, 2014). This is despite the fact that white peat is less decomposed and closer in structure to living *Sphagnum* and might therefore be thought of as a more suitable matrix into which *Sphagnum* can be established. Removal of 65 cm also represents a very substantial sacrifice of naturally accumulated peat. The white peat did not lend itself to achieving a suitably level surface because of the loose, light and fibrous nature of the material, while its large pore spaces meant that hydrological control was thought likely to be more challenging than would be the case in the denser peat beneath the layer of white peat. Unfortunately, no experimental work was undertaken to establish whether these concerns were valid.

Furthermore, obtaining a perfectly level peat surface is initially possible in denser more decomposed peat. However, it is almost impossible to maintain such a surface over time periods of more than a few weeks, particularly when working with dense relatively decomposed peat, as peat is not a uniform material. It is created by a mosaic of plant materials, some of which are more resistant to decomposition, and therefore more resistant to drying and shrinkage, than others. This mosaic of differing peat types can be seen in studies which provide longitudinal or even 3-dimensional illustrations of the peat matrix (e.g. Barber, 1981). Commercial peat extraction companies are thus obliged to re-level their 'peat fields' at least once a year and often more frequently than this in order to maintain a surface which is sufficiently level for peat milling operations to function effectively. A peat surface levelled for *Sphagnum* farming will experience similar differential subsidence throughout its area, albeit to a less marked degree than on a drained milling field because the high water table of *Sphagnum* farming will minimise, though not prevent, such effects. Consequently it is inevitable that dry spots and wet spots will develop across a *Sphagnum* farming field over time (Gaudig *et al.*, 2014; Wichtmann, Joosten and Schröder, 2016).

4.2.2 Hydrological management

A high and stable water table is one of the characteristic features of a natural *Sphagnum*-rich peat bog ecosystem. On this basis, it is natural to assume that a high and stable water table is also 'the most decisive factor for *Sphagnum* growth' within a *Sphagnum* farming context(Gaudig *et al.*, 2014; Brust *et al.*, 2018). This assumption has been supported by a number of laboratory and field studies (Gaudig, Joosten and Kammerman, 2008; Gaudig *et al.*, 2014; Brust *et al.*, 2018). Most *Sphagnum* farming studies to date have thus aimed to achieve high productivity of *Sphagnum* by substantially supplementing water inputs from direct precipitation with various approaches to raising water tables within the peat soil (Gaudig *et al.*, 2014; Brust *et al.*, 2018).

Water tables can be restored via irrigation canals, subsurface irrigation and blocking of site drains, the aim of these techniques being to re-wet the site and raise the water table closer to the peat surface to provide adequate supply for *Sphagnum*. However, the hydrological challenge for *Sphagnum* farming when using such techniques on a degraded site is that highly humified catotelm peat in terms of von post level, offers a reduced rate of hydraulic conductivity than less humified peat, so lateral water flow across the peat layer can be limited if this is not removed (Rycroft, Williams and Ingram, 1975; Price, Heathwaite and Baird, 2003). Consequently, irrigation canals may need to be closely spaced, much as drainage ditches are closely spaced to drain a peatland. This means that the production area of a *Sphagnum* farming field may be significantly reduced by the presence of multiple irrigation channels within which the crop cannot be grown.

Furthermore, during periods of prolonged intense rainfall, peat of low hydraulic conductivity will tend to cause water to pond on the surface rather than drain away through the peat to the irrigation channels. This will be particularly the case if such decomposed peat has also subsided during periods of dry weather. Excess surface water tends to inhibit *Sphagnum* growth, so overflow controls are required to prevent flooding (Gaudig *et al.*, 2014; Wichmann, Prager and Gaudig, 2017).

Hydrological control can be established with manual or automatic methods and include both passive and active management. Manual methods include pumping water into irrigation canals via diesel generators, and the use of a sluice gates for site outflow, while automatic methods can include automated solar, or wind pumps systems for water input and v notch weirs or float systems at outflow which can be controlled by telemetry. Maintaining hydrological balance is thus an essential component of any *Sphagnum* farm.

4.2.3 Supply of *Sphagnum*

A supply of *Sphagnum* biomass is required as founder material, as *Sphagnum* grows from vegetative fragments more readily than spores (Clymo and Hayward, 1982; Sundberg, 2013) as detailed in Chapter 3. So far, experimental sites in Canada and continental Europe have typically used donor sites from which founder material is wild harvested (Quinty and Rochefort, 2003).

The Shippagan site in Canada, was established using donor material taken from a more natural part of the surrounding peatland area. This was achieved by removing established *Sphagnum* moss at depth of 10 cm from a donor area. The donor material was applied on the new *Sphagnum* farm area at a ratio of 1:10 so to create a 10 m^2 area of *Sphagnum* farm, a donor area of 1 m^2 at 10 cm depth was required (Pouliot, Hugron and Rochefort, 2015).

Such an approach raises questions about the sustainability of using wild-harvested founder material, particularly when the extent of remaining near-natural peatland habitat in the area may already be extremely limited., as is the case for bog habitat in many parts of Europe and even in the more southerly provinces of Canada.

A key weakness of the MLTT approach used in most *Sphagnum* studies to date, is that it relies on donor *Sphagnum* material being readily available. Donor material is required at a ratio of 1:10, and the required volume may simply be unavailable in sufficient quantity where a suitable local site does not exist (Gaudig *et al.*, 2014). Sourcing *Sphagnum* from a non-local site could pose cost, transport, and logistical issues (Wichmann *et al.*, 2020). Removal of donor material may also be prohibited by local, regional or national legislation (Gaudig *et al.*, 2017). There are also concerns that wild harvesting may damage a donor peatland site due to potentially long recovery times and the area required for founder material (Silvan *et al.*, 2017). Perhaps more significantly, a concern is that donor material could introduce undesirable species such as vascular plants. These species could outcompete the *Sphagnum* crop and reduce the quality of the harvested *Sphagnum* biomass, especially for growing media purposes where low contamination of growing media is desired (Mulholland *et al.*, 2020).

Studies investigating *Sphagnum* recovery from natural sites following wild harvest of *Sphagnum* reveal a range of recovery rates. Elling and Knighton, (1984) showed that harvesting *Sphagnum* from a donor site for restoration purposes could result in a recovery time of 5 to 20 years based on total biomass removed. The authors suggested that a rotation of 20 years was necessary to provide maximum yields for harvested material.

Silvan *et al.* (2017) proposed harvesting *Sphagnum* at a depth of 30 cm from natural sites in Finland on a 30-year rotation to provide biomass for horticulture at yields of 1.5 t per ha per year. The approach by Silvan *et al.* (2017) suggests that medium-term recovery is possible following harvest. Diaz and Silva (2012) propose that 12 cm is the maximum depth for harvesting moss in Chile, as there is a 90% chance of moss recovery at this depth. The authors found that if harvesting deeper than 12 cm there is a lower chance of recovery. The appropriate depth is likely limited by the depth at which the axillary buds of the *Sphagnum* plant are viable as described in Chapter 3.

In New Zealand, wild harvesting of *Sphagnum* continues to be a widespread practice. Some voluntary recommendations exist, but none is enshrined in law (Whinam and Buxton, 1997). The recommendations suggest that no machinery be used and only lowland sites be harvested because of the long recovery times required for upland sites due to altitude. Finally, the recommendations suggest that drains are not used. However, opportunist acts have led to whole bogs being harvested on a one-off basis, leading to the loss of entire peat forming systems (Whinam and Buxton, 1997; Whinam *et al.*, 2003). These actions suggest that encouraging, or at least not adequately controlling, wild harvesting as a profit-making venture can lead to damaging behaviours.

Guêné-Nanchen, Hugron and Rochefort (2018) have suggested that harvesting from donor sites causes minimal damage, stating that donor sites should regenerate with *Sphagnum* levels comparable to natural sites in Canada within 10 years. However, they acknowledge that appropriate harvesting methods are essential to prevent ruts forming on the donor site. Ruts alter the distribution of surface water and can impact on *Sphagnum* recovery if the area floods. The authors recommend harvesting donor stock while the sites are frozen to minimize impacts. They also acknowledge that microtopography and species composition can be altered on donor sites. These changes could have immediate and long-term impacts on species diversity, as *Sphagnum* biodiversity increases with a greater range of microtopography and it also acts as an ecosystem engineer (Rochefort, 2000; Johnson *et al.*, 2015).

Some small pilot studies on undesignated land in Derbyshire, UK, have been conducted where *Sphagnum palustre* was harvested by hand at a rate of 10 handfuls of *Sphagnum* per 1m^2 quadrat, n=25. The aim of this exercise was to provide material for translocation and restoration. The results of the Derbyshire trials suggest a 57% recovery after 3 years, with 100% recovery estimated at 5 years (Benson *et al.*, 2019). However, the study yielded a low volume of harvested *Sphagnum* material - 250 handfuls. Unfortunately, a comparative unit for a handful was not quantified in the study. If a handful is assumed to be 100g, this would yield 25kg of donor material. Such a low yield would certainly limit this method's application as founder material in a commercial, large-scale *Sphagnum* farming scenario.

A key issue in the UK context is that many sites suitable for *Sphagnum* farming do not have suitable donor sites close by. For example, two NGOs: the Yorkshire peat partnership and Moors for the future have projects in the UK.

These projects are attempting to re-establish *Sphagnum* cover in the Yorkshire Dales, North York Moors and the South Pennines. These areas have been heavily damaged by industrial pollution, overgrazing and prescribed burning (Noble *et al.*, 2018). Such damage has reduced the prevalence of *Sphagnum* locally. As a result, restorative *Sphagnum* planting is enabled via the planting of externally sourced Micropropagated *Sphagnum* (Caporn *et al.*, 2017). Micropropagation techniques are advantageous for restoration as they require a very small amount of donor *Sphagnum* material to be gathered from a donor site, which reduces further negative impacts on areas with low *Sphagnum* prevalence. the disadvantage is the high cost of their use as they are produced in the glasshouse. *Sphagnum* farming may help to provide additional founder material for peatland restoration and reduce these costs if grown at scale.

The UK's temperate climate prevents complete freezing of sites, which may prevent use of the donor material harvesting method used in Canada where sites are accessed in winter when the ground is frozen. Taking Scotland as an example, poor weather including snow and ground frost prevents rather than encourages site access for peatland restoration (Artz *et al.*, 2019; Novo *et al.*, 2021). The remoteness of sites is also a problem, the average time to access and egress a site with machinery in Scotland is listed as 12 hours. Access to harvest donor *Sphagnum* is potentially a very real problem, assuming these logistical constraints for donor site access across the UK as whole, the logistical difficulties and associated costs of donor harvest are clear. Fortunately, access to *Sphagnum* farming sites located on peatlands following peat extraction, or on agricultural land is likely to be logistically easier than donor harvest.

Finally, environmental designations may prevent the removal of material from near-natural sites. *Sphagnum*-rich locations are usually located within conservation areas and many of these are regulated under the EU Habitats Directive (Caporn *et al.*, 2017). This potentially closes off these locations as sources of donor *Sphagnum* material within the UK.

When considering these various reports of recovery from wild harvesting, it is important to be clear that simple recovery of a moss carpet is not the same as ecosystem recovery. In some cases, the claimed 'recovered' moss carpet may not even be *Sphagnum* (to commercial operators one moss looks very much like another), while development of a post-harvest single-species flat sward of *Sphagnum* is no replacement for a natural *Sphagnum*-rich community with its characteristic microtopography. Clear and accurate definitions of 'recovery' are thus vitally important when reviewing descriptions and assessments of wild harvesting.

The Ramsloh site in Germany sourced its initial founder material from donor sites. However, a crucial step for subsequent German sites was that harvested material from the Ramsloh site was then used as donor material for the second site at Rastede (Gaudig *et al.*, 2014, 2017; Gaudig and Krebs, 2016). This highlights the potential for existing *Sphagnum* farms to act as donor sites for future *Sphagnum* farms or restoration areas and thus reduce or dispense with the need for wild harvesting of founder material.

The use of *Sphagnum* spores as an alternative and more environmentally sustainable founder material unfortunately has limited potential. This is largely because dioecious *Sphagnum* species rarely sporulate (Sundberg, 2000). If capsules are present, they cannot be harvested mechanically, so must be harvested by hand, which increases labour. In terms of knowledge, the factors inducing sporulation and germination are poorly understood (Sundberg, 2000; Gahlert *et al.*, 2012). Generally, spores have been found to establish well under controlled greenhouse conditions have then performed poorly in *Sphagnum* farming scenarios when seeded directly under field conditions. As a result of the above factors, use of spores as a founder material is more problematic than using *Sphagnum* fragments for vegetative propagation (Gaudig *et al.*, 2014).

4.2.4 Application of mulch covers

The MLTT approach employs a mulch cover during the establishment of *Sphagnum* material. The aim of the mulch cover is to provide a suitable microclimate for *Sphagnum* growth. The mulch cover also provides protection to young establishing mosses which are particularly susceptible to desiccation during establishment. The typical mulch cover used in the MLTT is straw (Quinty and Rochefort, 2003). The inclusion of a straw mulch has been shown to increase surface soil moisture on bare peat by 15% (Price, 1997) and as a result of work on Canadian peatlands, a recommended rate of straw application has been determined at 1,500kg per ha (Rochefort, 2000). The use of straw mulch cover provides a range of other benefits, including a decrease in the temperature variation between day and night, decreased evaporation from the underlying peat soil, a sustained higher water table closer to the peat surface, reduced PWP and increase relative humidity at the surface (Price, Rochefort and Quinty, 1998). A mulch cover offers many clear benefits for *Sphagnum* establishment.

Potential disadvantages of utilising a straw-based mulch include the introduction of vascular plants through seed fall (Gaudig *et al.*, 2017) , increased fertiliser effect due to leaching and released carbon emissions as the straw decomposes (Brown, Strack and Price, 2017). While the reduction in light due to the mulch cover may also have an impact on *Sphagnum* growth (Lamers, Bobbink and Roelofs, 2000; Limpens and Heijmans, 2008). The use of other mulch covers remains under-researched in a *Sphagnum* farming context, although recently, (Grobe, Tiemeyer and Graf, 2021) compared geotextile mulch covers with straw and found that geotextile covers resulted in lower *Sphagnum* productivity on a shallow peat site.

4.2.5 Monitoring

Studies investigating the establishment of *Sphagnum* on bare peat surfaces either for restoration or for *Sphagnum* farming are varied in their location and accessibility, which can influence their experimental design and methods. However, a common theme throughout these studies is the need to monitor the vegetation change over time and attempt to quantify this growth against other influencing covariates (Gaudig *et al.*, 2014; Pouliot, Hugron and Rochefort, 2015).

Key variables monitored include the percentage cover of *Sphagnum* over time, the depth of *Sphagnum* carpet, the bulk density of the growing *Sphagnum* and the relative proportions of vascular species found within the *Sphagnum* crop (Gaudig *et al.*, 2014).

Environmental variables are also monitored. These can include the use of a weather station to assess precipitation, temperature, humidity, wind speed, wind direction and global radiation (Brust *et al.*, 2018). Water table, water input and water outflow are useful measures that can be incorporated into water balance models (Gaudig *et al.*, 2014; Glatzel and Rochefort, 2017; Brust *et al.*, 2018) (Brust et al., 2017). Pore water pressure (PWP) is also a key variable (Price, Rochefort and Quinty, 1998; Thompson and Waddington, 2008), however this is often not observed in *Sphagnum* farming studies.

4.2.6 Harvesting

Harvesting paludiculture crops poses a challenge. This is because of high water tables following rewetting. High water tables can prevent conventional farming machinery accessing sites, due to the low bearing pressure of saturated peat and the high pressure exerted by heavy machinery (Schröder *et al.*, 2015). Specially adapted machinery or methods are required to overcome this issue. Two clear approaches have emerged. One approach is to physically access the *Sphagnum* growing area (Gaudig *et al.*, 2014). This necessitates the use of tracked or double wheeled vehicles that exert a low ground pressure (Schröder *et al.*, 2015). The second method uses machinery with a long reach that can efficiently harvest from the *Sphagnum* cultivation areas without physically accessing the area (Gaudig *et al.*, 2014).

There have been few actual attempts to harvest farmed *Sphagnum* at this time. Indeed the only large-scale mechanical harvest to date took place on the Rastede *Sphagnum* farming site, Germany in June 2016, five years after establishment. The harvesting method made use of a long-reach excavator equipped with a long arm and mowing bucket which was positioned on a causeway. This enabled the harvest of *Sphagnum* to be gathered without exerting any pressure on the growing area. Harvested *Sphagnum* material was loaded directly into a tractor-pulled dumper truck on the causeway, this allowed for transport off the site (Wichmann *et al.*, 2020).

Three factors have been identified as being critical for improved harvesting (Schröder *et al.*, 2015):

1. Firstly, the appropriate selection of machinery to reduce the machine weight and reduce the ground pressure exerted;
2. Secondly, logistical considerations to reduce impacts include keeping the number of vehicle crossings low to reduce pressure and ensure adequate turning circles to reduce shearing forces, particularly if a tracked vehicle is used;
3. Finally, prior infrastructure planning should be used to reduce bottlenecks, reinforce biomass removal areas and establish suitable locations for wider transport offsite.

4.3 What results have been achieved so far?

4.3.1 Mire type suitability

Two main peatland types have been identified as suitable habitats for *Sphagnum* farming: former commercially extracted raised bogs (Pouliot, Hugron and Rochefort, 2015; Gaudig *et al.*, 2017; Graf *et al.*, 2017), and agriculturalised peatlands mostly concentrated on former bog grassland (Krebs *et al.*, 2012) and (Gaudig *et al.*, 2014). These areas represent large areas of degraded peat, which are subject to commercial drainage-based activities, as such, any change in land use needs to allow for economic output, making paludiculture the only real sustainable option on these land use types (Gaudig *et al.*, 2014). Although no publications have assessed upland peatland use for *Sphagnum* farming, it is likely that the best land use in these areas is peatland restoration rather than paludiculture, due to their relatively remote and undulating terrain which would serve to make infrastructure development, harvest, and transport costs very high.

Artificial habitats have also been investigated as potential locations for *Sphagnum* production. The main artificial habitat investigated being floating mats on open water at mining sites, (Blievernicht *et al.*, 2011, 2012) and (Gaudig *et al.*, 2014). Hydroponic cultivation on floating mats was considered to have the potential to increase the available space for *Sphagnum* farming, with the potential for lower infrastructure investments compared with land-based techniques because water tables in relation to the crop would remain high and steady. However, floating mats were not without their own challenges, Wichmann, Prager and Gaudig, (2017) reported further on the procedure, economics and area potential of various *Sphagnum* farming scenarios and identified that floating mats for water-based cultivation resulted in the highest cost estimates, almost double that of land-based techniques €17.34 m^{-2} to €21.43 m^{-2} vs €8.35 m^{-2} to €12.80 m^{-2} respectively. The floating mat costs were substantially higher as they required additional pre-cultivation in a greenhouse and transport prior to installation on site.

4.3.2 Promising species identified

In Europe, the most promising species identified have been *S. palustre* at the Ramsloh site and *S. papillosum* at the Rastede site (Gaudig *et al.*, 2014). In Canada species trialled were *S. fuscum*, *S. rubellum* and *S. magellanicum* as part of a mixed species founder material taken from a donor site (Pouliot, Hugron and Rochefort, 2015). However, both studies acknowledged that the authors only explored a limited number of species and recommended that further species of varying provenance and productivity should be investigated. There are many species to trial, and the location of an individual *Sphagnum* farm may make certain species more suitable based on climate, rainfall, nutrient levels and other factors (Gunnarsson, 2005).

4.3.3 Hydrological management results

Most *Sphagnum* farming studies to date have aimed to achieve high productivity of *Sphagnum* by utilising water inputs from natural rainfall from above, supplemented with raising the water table from below the peatland surface (Gaudig *et al.*, 2017). A hydrological study of the Rastede site has confirmed that distributing irrigation canals every 10m, fed with stream water, is an effective form of maintaining water levels within *Sphagnum* farming areas (Brust *et al.*, 2018). The water table was monitored via manual dipwells and automatic level-loggers with barometric compensation.

Beyer and Höper, (2015) reported water table results from a *Sphagnum* farming site in Germany referred to as 'Westermoor'. The site involved active management of the water table in the cultivation area. The mean annual water table was recorded in 2010 and 2011 as 6.1 cm and 9.2 cm below the mire surface respectively, although other important metrics recorded were the water table range and the maximum duration for single periods of low water table.

At Rastede, another Germany *Sphagnum* farm site a water balance was calculated from September 2012, through to October 2013. The Rastede site was equipped with surface irrigation canals fed by automated pumps from a stream water source. The system automatically pumps water into irrigation channels if the water level falls below 8 cm depth. The aim of this system is to keep the water table within 3 cm of the *Sphagnum* surface (Brust *et al.*, 2018). Water losses were attributed to seepage and evapotranspiration, these were compensated via the irrigation channels. Given the target 3 cm tolerances, it is difficult to see how this could be achieved when differential shrinkage and subsidence of the peat surface, as discussed earlier, will undoubtedly have resulted in height differences across the peat surface of at least a few centimetres.

Pouliot, Hugron and Rochefort, (2015) assessed the performance of cultivated *Sphagnum* grown on block cut peatland in Canada. *Sphagnum* production basins sized between 800-1,500 m^2 were assessed over 6 yearly production cycles. The study achieved this by observing the vegetation cover and dry weight biomass production of *Sphagnum* grown in trenches, whilst comparing these with hydrological and meteorological data gathered during the same period. The study found that the observed water table varied throughout production years and confirmed the widely recognised link between a high-water table and *Sphagnum* growth. The paper considered that the variation in production cycles (i.e. growing seasons) was most likely due to variation in water table and plant-water interactions. (Gaudig *et al.*, 2017)

Raising and maintaining a high-water table on *Sphagnum* farming sites is a key factor for success (Gaudig *et al.*, 2014). This requires knowledge, monitoring and the technical infrastructure to maintain appropriate water levels.

Ensuring adequate water provision is a challenge. For example, the volume of irrigation water applied over the hydrological year at the Rastede site in Germany totalled 6,590 m^3 ha-1 (Brust *et al.*, 2018). This is equivalent to a daily average input of 1.8mm water per m^2. The study acknowledged that this was an unnecessarily high total as the irrigation water input, and the outflow control systems were activated simultaneously due to equipment failure. The calculated irrigation requirement to overcome an expected 6-month summer water deficit at the Rastede site for the equivalent period was 3,588 m³ ha^{-1} equivalent to a daily irrigation depth of 0.98mm per m^2. This highlights the need for good water supply and delivery mechanisms.

Environmental factors inevitably play a role in calculating necessary irrigation requirements. At the Shippagan site in Canada, irrigation was supplied at a rate of approximately 30 mm per month during summer water deficits in 2015 (Brown, Strack and Price, 2017). The cooler and wetter climate in Canada delivered 50% more precipitation and 50% less evapotranspiration over the study period compared to that of the Rastede site in 2012 – 2013. Consequently, the irrigation volume required at the Shippagan site was half that of the Rastede site in 2012-2013 (Brust *et al.*, 2018). This highlights the fact that irrigation supply cannot be met by a one-size fits all solution, and a detailed understanding of site conditions and locale are required.

4.3.4 Potential yield of harvested *Sphagnum*

Research has shown that achieving a 'canopy closure' quickly is essential for achieving high yields, with high productivity and efficient regeneration featuring as recurring themes across many *Sphagnum* farming studies (Landry *et al.*, 2011; Pouliot, Hugron and Rochefort, 2015; Wichmann, Prager and Gaudig, 2017). At the Rastede site, the greatest productivities of *Sphagnum papillosum* were observed at locations with continuously high-water levels, where productivity of 6.9 t dry matter (DM) ha -1 vs the mean value of 3.6 t dry matter (DM) ha^{-1} was recorded (Gaudig *et al.*, 2014).

(Wichmann *et al.*, 2020) analysed *Sphagnum* yield following 5 years of growth at the Rastede site. At harvest it was not possible to measure the yield directly but an estimated yield was obtained based on the difference between dry weight biomass of *Sphagnum* samples taken before harvest and the dry weight biomass remaining in samples taken post-harvest. Dry weight biomass of remaining *Sphagnum* post-harvest. This was achieved by removing biomass from 30 quadrats of 15 cm x 15 cm prior to harvest, then taking repeat samples after harvest. Dry biomass weights were converted into volumes by calculating a conversion factor using the European standard DIN EN 12580 which is designed to provide a dry bulk density value (DBD).

The various assumptions involved within the study meant that a range of estimated yields was obtained. The mean harvested dry mass yield value was calculated at 16 t ha^{-1}, the estimate range was 10 t ha^{-1} to 22 t ha^{-1}. The study acknowledges that a key research gap is uncertainty around the conversion factor. Future studies that should aim to determine a standard range of conversion factors for individual *Sphagnum* species. Such an approach could increase confidence surrounding potential profits for *Sphagnum* farming and increase future production of *Sphagnum*.

4.3.4 Quantification of Ecosystem service Benefits

As *Sphagnum* farming is in its infancy, a few studies provide early quantification of the Ecosystem services (ES) provided by *Sphagnum* farming. There is evidence of reduced GHG emissions (Beyer and Höper, 2015; Günther *et al.*, 2017), enhanced nutrient cycling (Temmink *et al.*, 2017) and increased biodiversity (Muster *et al.*, 2015; Muster, Krebs and Joosten, 2020) which will be explored below.

Green house gas results

An additional area that must be considered when *Sphagnum* farming is that of the carbon balance. The GHG balance of the Rastede *Sphagnum* farming site is reported in (Günther *et al.*, 2017). The study was balanced in its approach as it measured the GHG flux on both the *Sphagnum papillosum* covered areas which have the potential to sequester carbon, and the open water irrigation ditches which have the potential to emit methane. Measurements took place every 4 weeks from September 2009 to December 2011, so were not limited to a short campaign approach.

Gas flux was observed via box chambers with a rubber base on production areas and via floating circular chambers with a polystyrene base for the irrigation ditches. The headline figure for this study was that *Sphagnum* production strips were net carbon sinks of 5 – 9 t CO_2 e ha^{-1} yr^{-1} while the irrigation ditches resulted in a net GHG emission of 11 t CO_2 e ha^{-1} yr^{-1}. Evidence from this study shows that reducing the area of irrigation ditches is desirable to reduce the negative greenhouse gas emissions, and that other methods of supplying water to the *Sphagnum* crop could provide greater greenhouse gas emission benefits.

Greenhouse gas monitoring at the Shippagan site in Canada was conducted in summer campaigns during 2014 and 2015. This is a limitation of the location, as the *Sphagnum* would be under snow in the winter. The decomposition of straw mulch was found to account for almost 50% of the seasonal summer ecosystem respiration. When this was removed from calculated Net Ecosystem Exchange (NEE) values, the *Sphagnum* farming areas were found to be sinks (Brown, Strack and Price, 2017). This highlights that awareness of the mulch decomposition dynamics is needed when assessing GHG flux on establishing *Sphagnum* farming areas. The study found that there was no significant difference in the CO_2 uptake of *Sphagnum* between production basins with water table targets of -10 and -20 cm.

Nutrient balances

A common concern surrounding *Sphagnum* cultivation on bog grassland or agriculturalised peatland sites is one of altered nutrient availability. *Sphagnum* dominates in its natural habitat due to a high nutrient retention capability, giving a competitive advantage over other species (Chapter 2). However it was unclear how *Sphagnum* would fare in nutrient rich conditions. Temmink *et al.*, (2017) reported results from a study investigating the nutrient balance of the Rastede *Sphagnum* farming site in Germany. The study site was subjected to highly enriched irrigation water from the surrounding land and received substantial nitrogen (N) inputs from airborne sources. The study concluded that if moisture supply is sufficient, and large graminoids (weeds) are supressed then high biomass yields can be attained. *Sphagnum* growth is enhanced if irrigation water contains high concentrations of phosphorus (P) and potassium (K) as low concentrations of these nutrients could be growth limiting in a high N scenario. The authors advocate removal of enriched topsoil prior to *Sphagnum* planting, frequent mowing of graminoids, high water availability and maintain a low soil pH. *Sphagnum* within this study was also shown to have a nutrient sink ecosystem function achieving nutrient sinks of 34 kg N, 17 kg P and 4 kg K ha−1yr−1.

Biodiversity benefits

Sphagnum farms have the potential to enhance habitat diversity at the landscape scale. They can provide refuge for specialist species threatened by ephemeral habitat loss, and offer habitat connectivity for rare wetland species (Muster *et al.*, 2015).

Spider communities (Araneae) have been suggested as bioindicators at the Rastede site (Muster *et al.*, 2015). This was the first invertebrate study on a *Sphagnum* farm within the literature. Spider communities were selected as bioindicators as they have a high dispersal ability and are top predators amongst invertebrates. Spider communities could therefore reflect the various abiotic and biotic changes following the establishment of *Sphagnum* farms. Certain spider species are also closely associated with peatland habitats and have been used as ecological indicators in the past. The Muster *et al.,* (2015) study appears well thought out and consisted of 6 test plots within the Rastede site, and 5 reference plots located on nearby peatland habitats. Reference plots were situated on bog grassland, Juncus fallow areas, degraded peat bog, sedge reed, and myrtle bush rich areas to provide adequate controls.

The study found that the proportion of peatland specialist spider species across all areas were low <10%. Overall abundance and species richness of spiders on the *Sphagnum* farming plots were typically lower than reference areas. This is most likely due to a lag time between the habitat creation and spider succession and reduced structural diversity within the *Sphagnum* cultivation areas. Generally, there was a rapid change in the spider communities. The community structure developed from disturbance specialists (pioneer species) towards generalised peatland species. However, the *Sphagnum* farming plots provided habitat for coastal specialist species such as *Robertus heydemanni* and *Pardosa agrestispurbeckensis*. Finally In the last 2 years of monitoring, a German red data book (a list of endangered species) species was recorded: the pirate wolf spider *Pirata piscatorius* was found on the *Sphagnum* farm sites. *P.piscatorius* is a wetland specialist, and became the most abundant spider found on the *Sphagnum* farm areas (Muster *et al.*, 2015).

A follow up paper identified *Sphagnum* percentage cover as the largest environmental driver of spider community assemblage and revealed a high turnover of species dominance on the *Sphagnum* farming areas (Muster, Krebs and Joosten, 2020). These early results suggest that *Sphagnum* farms may never act as direct replacements for natural wetland habitats. However, it is likely that *Sphagnum* farms may provide a viable surrogate habitat for at least some endangered species. The rapid contribution of *Sphagnum* farming to enhancing biodiversity is encouraging, as 52% of wetlands may have a natural biodiversity recovery rate of >1000 years (Pezzati *et al.*, 2018). Future research is needed to identify the impacts of *Sphagnum* biomass harvesting on invertebrate assemblages and wider biodiversity. It is likely that a mosaic or rotational approach to harvesting will maximise biodiversity benefit.

4.4 Alternatives to the MLTT approach to *Sphagnum* farming

4.4.1 On-site challenges for the MLTT approach

Although the MLTT approach to *Sphagnum* farming, together with variants developed in continental Europe, have shown collective promise, several factors have been highlighted above which may pose lesser or greater challenges to any proposed *Sphagnum* farming scheme. The list of challenges includes:

- Selecting a site which can be levelled to the necessary degree of accuracy with the resources available;
- Designing an irrigation system that maintains the maximum possible area of peat with the ideal water table for the entire growing cycle while minimising the cropping area lost to that irrigation system;
- Maintaining the surface level over time in order to ensure that the maximum possible area of crop is subject to the ideal water table for the whole growing cycle;
- Ensuring that sufficient water of the right quality is available at all times of need and that it can be fed efficiently and effectively to all parts of the growing area;
- Availability of suitable donor material obtainable without adversely affecting the existing natural capital of the *Sphagnum* ecosystem resource;
- Freedom from non-crop species;
- Availability and affordability of suitable mulch materials in order to protect the crop from sun- and wind-driven losses and thereby maintain high humidity within the crop;
- Accessibility for maintenance and harvesting machinery.

4.4.2 Off-site socio-economic challenges for the MLTT approach

In addition to the on-site challenges listed above, particularly within a landscape of lowland Britain where conventional farming is the predominant land-use and therefore intensive land drainage is the norm, there are certain major socio-economic and even cultural challenges which must be addressed. The MLTT approach is likely to experience considerable local and even regional resistance to the concept of raising water tables within a landscape where the infrastructure is designed and managed, usually by Internal Drainage Boards, almost wholly for the purposes of drainage.

Local communities may also be alarmed by the idea of water tables being raised, thereby (in their eyes) increasing flood risk. Farmers who have been trained all their working lives to ensure that their land is in ‘good condition’, which is usually considered synonymous with ‘well drained’, risk being labelled within their community as being poor managers of their farm and even being ostracised by their neighbours (Reed *et al.,* 2020). Indeed, there is a real danger that raised water tables within a *Sphagnum* farming field will have genuine consequences for the water tables of adjoining fields, meaning that the *Sphagnum* farmer is truly a ‘bad neighbour’.

4.4.3 The micropropagated irrigation-from-above (MIFA) approach

The present thesis describes an alternative to that of the MLTT, devised in partnership with Micropropagations Services Ltd. The approach differs from the MLTT method of *Sphagnum* farming in two important ways:

- Founder material is provided in the form of micropropagated *Sphagnum* plants rather than wild-harvested material; and
- Irrigation is provided in the form of surface drip-feed and/or overhead spray rather than irrigation from below in the form of raised water tables.

Rationale for the use of micropropagated *Sphagnum*

There are three key benefits of using micropropagated *Sphagnum*. Firstly, it requires only a single *Sphagnum* plant to be taken from the wild for this then to be micropropagated into founder material. The impact on the local or regional natural capital of *Sphagnum* habitat is therefore negligible. Secondly, the micropropagated material can be created from the nearest source, meaning that as far as possible, local genetic variation is maintained. Thirdly, the micropropagated material is free from other plant or fungal species, meaning that if the *Sphagnum* field can be suitably prepared to remove any seed bank and the material is then covered to prevent, or at least minimise, ingress of weed seeds, the weed load within the eventual *Sphagnum* crop can be minimised.

Rationale for irrigation from above

The predominant direction of water movement in a natural *Sphagnum*-rich peat bog is either downward from the atmosphere or laterally through the acrotelm layer. While capillary action via the *Sphagnum* pendant branches can provide a limited degree of upward water movement, the main source of water for an ombrotrophic peat bog is, by definition, from above. Rising and falling groundwater tables are more characteristic of fen peatlands, often with degrees of rise and fall that significantly exceed the water table movements typically observed in a natural peat bog system.

Both drip irrigation and overhead spray closely mirror the predominant direction of water input into a peat bog system, and are also both methods of irrigation that are familiar to the farming community. With surface irrigation it is not necessary to ensure that the peat surface is completely level, so if there is differential shrinkage and subsidence across a *Sphagnum* field this is not as serious an issue as when relying on fine tolerances of water table maintenance.

Surface and overhead irrigation do not require water tables to be raised and therefore do not risk antagonising neighbouring farm holdings or causing concern about flood risk within the local community. This is particularly the case because both drip irrigation and overhead spray are irrigation methods which are familiar sights within the conventional agricultural landscape.

Surface irrigation or irrigation from above also needs no infrastructure of canals or other open-water bodies which create barriers and hazards to agricultural machinery. If any form of site management is required, the irrigation system can be instantly turned off for the duration of the work, then just as instantly turned back on again, whereas a cabal irrigation system designed to raise water tables requires much more time and effort to drain down and then re-wet a field if site management requires it.

Top-down irrigation utilises a variety of trickle, drip, spray, or mist type systems. These have been observed to increase the water use efficiency of farms in England (Gadanakis *et al.*, 2015) . The use of top-down irrigation over surface irrigation channels may be more appealing to farmers. The use of top-down irrigation is advantageous as it does not lock a farmer into a land use system. Top-down irrigation could reduce the infrastructure adaptation required at a farm level, although the economic viability of this system for *Sphagnum* farming purposes still needs assessment.

The MIFA approach has a clear rationale and could offer a potential new option for *Sphagnum* farming. However, the approach has never been taken before and many questions must be answered to develop the concept further. The next chapter sets out the specific aims and objectives of the thesis to advance the MIFA approach.

Chapter 5. Aims and objectives

5.1 Overall aim

The overall aim of this thesis was to perform exploratory research into a new *Sphagnum* farming method: The Micropropogated with irrigation from above method (MIFA). The research was highly novel, this being the first time micropropogated *Sphagnum* had been used as a founder material, and to our knowledge the first-time irrigation from above had been used in the absence of active water table management for *Sphagnum* farming (SF) on peat soil.

As discussed in Chapter 4, if proved successful, The MIFA approach may be a desirable option for SF as it does not use active water table management. It enables paludiculture options in areas where complete raising of the water table is technically or socially difficult, or in areas where no large supply of donor material exists.

5.2 Challenges and questions about the MIFA approach

While it was possible to identify a number of potential advantages to be gained by adopting the MIFA approach, it is important to recognise that this form of SF is likely to come with its own significant challenges and uncertainties, particularly as the approach has even fewer experimental studies to draw on than studies using the MLTT with raised water table-based approach.

As discussed in Chapter 4, based on literature for current SF, the critical factors for successful *Sphagnum* cultivation are: the provision of founder material, hydrological management, and the use of protective mulch during establishment as mentioned in the MLTT approach (Quinty and Rochefort, 2003) and in key review papers (Gaudig *et al.*, 2014; Gaudig *et al.*, 2017). The result from achieving these critical factors should be a high yield of *Sphagnum* biomass, typically presented as the tons of dry mass per ha per year (Wichmann *et al.*, 2020).

Additionally, any new *Sphagnum* farming production method should strive to achieve more than just biomass production, achieving the ES benefits of paludiculture particularly in terms of GHG balance, and be technically and economically viable to ensure wider uptake of the method.

Based on the existing framework of critical factors for success within the literature, and expert discussion during the bidding process of the *Sphagnum* farming Innovate UK project that funded the main study experimental sites, several key challenges and questions surrounding the MIFA approach were identified:

Founder material

- Which forms of micropropagated material respond best to the MIFA approach? BeadaMoss offer multiple micropropagated *Sphagnum* products, each with their own application technique and cost implications.
- Do some species of *Sphagnum* respond better to the MIFA approach than others? *Sphagnum palustre* or *Sphagnum papillosum* are established options, but other species could be trialled.

Hydrological management

- As the MIFA approach does not rely on WT manipulation, what exactly is the critical hydrologic metric to be measured when using the MIFA approach? Almost all *Sphagnum* farming studies have used WT depth as their key metric, however pore water pressure (PWP) may be more appropriate as it can provide information at the capitula level.

- Can irrigation from above provide sufficient water to maintain a *Sphagnum* crop and enable it to thrive? A simple proposition: can *Sphagnum* survive under the MIFA approach?
- Which of the possible top-down irrigation methods – trickle, drip, spray or mist – provides optimal conditions for a healthy and vigorous *Sphagnum* crop? There are multiple irrigation options available to farmers, and none have been tried before, so trialling several seems prudent.

Protective Mulch covers

- Does the MIFA approach still require a protective mulch and is straw still the best medium? It is possible that the MIFA approach may reduce the need for any mulch cover by applying water directly at the *Sphagnum* surface.

Yield of *Sphagnum*

- Can successful *Sphagnum* production be achieved if water is supplied from above even though the water table is lower than typically employed in the MLTT approach?
- How best to measure growth and productivity of differing forms of micropropagated *Sphagnum*? Can a non destructive method be used?

Economics

- Irrigation from above requires power, but how much power compared with that required for automated water-level control with the MLTT approach? This could have an implication on the running costs.
- What are the infrastructure costs for the MIFA approach?

ES benefits

- What would the GHG balance differences between the MLTT with raised water table and the MIFA approach be? As raising water tables under the conventional approach lowers GHG emissions, an alternative method should also seek to reduce these emissions.

- What are the GHG Life Cycle Analysis implications of the MIFA approach? An LCA encompassing all the potential sources and sinks of GHG would be required to fully support widespread adoption of the MIFA approach.

Most of these questions cannot currently be answered entirely but this thesis describes a research programme with the objective of providing initial answers to some of these key questions. It seeks to provide a foundation for what is hoped will be a whole new field of research. This will involve experimentation, agricultural testing and trialling, product development and market penetration for farmed *Sphagnum*. The results of this new area of research which will not be limited to the world of horticulture but applicable across a wide range of other potential sectors as well.

5.3 Thesis Objectives

As the MIFA approach is highly novel, the work has necessarily been exploratory in nature. Boundaries had to be applied as all the above key questions cannot be answered. Specifically, the thesis objectives centred around the following questions:

Hydrology

- Is PWP suitable as the critical hydrology metric to be measured when using the MIFA approach?
- Can irrigation from above provide sufficient water to maintain a *Sphagnum* crop and enable it to thrive?
- Do drip or spray irrigation systems provide optimal conditions for a healthy and vigorous *Sphagnum* crop?

To achieve the objective of answering these hydrology questions, pilot studies and a main study were developed. Each study had its own hypotheses to answer, which are introduced in their relevant chapters. The primary data collected to answer these, relied on PWP measurements obtained through tensiometers, which provide a more ecologically meaningful measurement of water availability to the *Sphagnum* capitula (Clymo and Hayward 1982, Price, Thompson and Waddington 2008, McCarter and Price 2014, Weber *et al.*, 2017a). These papers provided a well-defined framework for PWP thresholds, whereby *Sphagnum* hyaline cells loose water causing desiccation, which reduces photosynthesis at PWP between 100-600 hPa. This also enabled a framework for *Sphagnum* recovery following PWP desiccation events for the moss species used in the experiments *S. palustre*. Whereby full recovery is possible after 6 days for *S. palustre* but declines to 50% recovery at 10 days and 0% recovery at 16 days.

Growth and impact of protective mulch covers

- Can successful *Sphagnum* production be achieved if water is supplied from above without any effort to maintain water tables?
- Does the MIFA approach require a protective mulch?
- Which forms of micropropagated material respond best to the MIFA approach Plugs or Gel?

To achieve the objective of answering these growth questions pilot studies and a main study were conducted. Each study had its own hypotheses to answer, which are introduced in their relevant chapters. The primary data collected to answer these relied on growth metrics including percentage cover, *Sphagnum* carpet depth and literature values for the dry bulk density (DBD) of both natural and farmed *Sphagnum*. S. palustre was used for its versatility and commercially available mesh and plastic mulches were used in addition to straw to investigate the potential of alternative mulch covers.

Summary

To achieve these objectives several small-scale pilot studies were undertaken to investigate hydrological aspects of growing *Sphagnum* in the absence of a water table which are presented in Chapter 6. The results of these pilot studies were further explored through a larger main study which applied the MIFA approach at field scale at two contrasting peatland sites which had water tables, using two irrigation methods and several protective covers. The experimental layout for the main study is described in Chapter 7, the hydrological investigations in Chapter 8, and the growth investigations in Chapter 9. Finally, the overall discussion and conclusions are discussed in Chapters 10 – 11.

Chapter 6. Initial pilot studies

6.1 Introduction

At the beginning of this PhD study, there was a pressing need for a demonstration *Sphagnum* farm in the UK. The UK government ambition was clear in terms of its intention to phase out peat in horticulture (Chapter 2) but several key research gaps (Chapters 4 and 5) still needed to be addressed if *Sphagnum* farming was to be taken forward in a UK context.

Micropropagation Services Ltd. (MPS) was the commercial partner for this study. Their role was to produce micropropagated *Sphagnum* under greenhouse conditions at their facility in Leicestershire. The *Sphagnum* production system at their facility is managed through temperature and irrigation controls and is monitored regularly. The conditions for *Sphagnum* growth are regarded as near optimal in this system. As a result, large quantities of *Sphagnum* moss are successfully produced for sale around the UK annually, mostly to provide material for peatland habitat restoration. Some brief detail is provided about the MPS system here but full details will not be included in order to protect the trademarked products and process.

As described in Chapter 4, *Sphagnum* founder material is needed to start a *Sphagnum* farm. Typically, donor material used in *Sphagnum* farming studies is harvested from near natural sites and applied at a ratio of 1:10 using the MLTT method (Quinty and Rochefort, 2003). The MPS method requires a very small amount of initial *Sphagnum* donor material which is then multiplied-up in a lab using micropropagation techniques to generate a large volume of *Sphagnum* material (Caporn *et al.*, 2017) . The advantages of this approach are that there is minimal impact on the donor site, a weed free founder material is generated, and options exist to select a multi or single moss species mix. In the greenhouse, *Sphagnum* is cultivated in complete segregation from a water table. Water is applied via a top-down irrigation system, designed to apply water directly to the capitula of growing *Sphagnum* mosses.

As part of the present PhD research programme, three pilot projects were conducted at the MPS business premises prior to undertaking the main study. *Sphagnum* is cultivated in the absence of a water table at the MPS premises, therefore water table impact on the *Sphagnum* is non-existent in this setting. This suggests that *Sphagnum* could be farmed without any attempt to control the water table dynamics if adequate water is provided via irrigation. The pilot studies were conducted in order to evaluate pore water pressure (PWP) dynamics of cultivated *Sphagnum* in interior (glasshouse) and exterior (outside) settings. PWP is the key dynamic affecting *Sphagnum* growth as at certain thresholds *Sphagnum* desiccation and death occur.

This pilot approach ensured that experience in utilising new equipment could be gained, sample data could be obtained, and an initial analysis of pore water dynamics in micropropagated *Sphagnum* could be undertaken with a view to establishing some context for the MPS *Sphagnum* cultivation system in relation to key ecohydrological threshold for *Sphagnum* PWP of 100 hPa as described in Clymo and Hayward, (1982) and Price and Whitehead, (2001).

6.1.1 Methods common to all the pilot studies

PWP within each pilot experiment was monitored with eight mini electronic tensiometers, connected to an 8-channel DataHog datalogger produced by Skye industries Ltd. This methodological approach was based on a 10 cm length of tensiometer tube, designed for low PWP readings in shallow substrates, an appropriate data resolution within the logger (logging every 15 minutes) and weatherproof specification for the equipment (weatherproof and suitable for inside and outside applications with an ingress protection rating of IP65).

Tensiometers consist of a porous ceramic tip and a clear acrylic plastic shaft of varying lengths according to the required depth of measurement in the soil, which is usually the rooting depth of the crop. The electronic tensiometers each have a pressure transducer fitted and are fully automatic in terms of data logging.

The water within the tensiometer is able to flow through the porous ceramic tip in either direction. When the ceramic is in contact with a dry soil, water flows out of the tensiometer leaving a vacuum behind. This vacuum inside the tensiometer equalises to the PWP surrounding the ceramic tip in the soil or growing medium. This is then directly measured using the electronic pressure transducer in in Hectopascals (hPa).

If the soil (or growing media) is irrigated or it rains, the PWP reduces and moves closer to 0 hPa. If the soil becomes saturated, more water will enter the tensiometer until the vacuum is filled. The pressure transducer will now read zero as there is no soil suction at soil saturation. When the soil dries and is rewetted by irrigation or rainfall, tensiometers will re-equalise and read the soil suction directly. Generally, Low PWP readings nearer zero mean wetter soils while higher PWP readings mean drier soils.

Problems can arise in tensiometers when: there is poor contact between the ceramic tip and the soil, there are dry conditions for a long duration, where air bubbles develop within the acrylic tubes, and when tensiometers freeze; all of which can result in unreliable PWP readings. Though these were deemed unlikely to occur in the irrigated pilot projects.

6.2 Pilot Study 1: *Sphagnum* pore water pressure within a greenhouse environment irrigated by overhead spray.

6.2.1 Aims

Before the pilot study, no primary data had been obtained evaluating the irrigation systems used by MPS in relation to the ecohydrological thresholds of *Sphagnum*. Unlike most studies investigating *Sphagnum*, at the MPS site the moss is grown in the absence of a water table. Therefore, the aim of this first pilot study was to assess whether tensiometers were a suitable method for monitoring PWP, and whether key pore water thresholds were avoided (PWP not exceeding 100 hPa) under the MPS top-down irrigation regime in a greenhouse environment. Two hypothesis were proposed:

Hypothesis 1: Mini electronic tensiometers can detect pore water availability when inserted directly into a *Sphagnum* moss carpet rather than a soil substrate.

Hypothesis 2: PWP will be kept below the critical 100 hPa threshold for *Sphagnum* pore water availability where top-down automated irrigation is used in a greenhouse setting.

6.2.2 Method

A pre-established *Sphagnum palustre* carpet was provided by MPS as the mesocosm for this study. This carpet was created through application of micropropagated BeadaGel^{TM} founder material to a thin layer of peat substrate <4 cm depth. *Sphagnum* carpets were created directly on the greenhouse floor at the MPS facility. The greenhouse floor comprised a concrete foundation, covered with a depth of sand c.15 cm depth which formed the sub layer. The sublayer was topped with a permeable woven geotextile which acted as the greenhouse floor. The shallow peat layer and gel founder material was applied directly to the floor for cultivation.

The tensiometers were distributed as two line transects of four tensiometers installed directly on the upper surface of the *S. palustre* carpet (Figure 10). This was achieved by creating gaps in the *Sphagnum* carpet using a knife to gently tease apart the dense surface of entwined *Sphagnum*. Tensiometers were inserted into these gaps so that the 2 cm long ceramic bulbs long, were fully covered to a depth of 2 cm within the carpet a necessary feature of their use as ceramic bulbs not fully immersed in the *Sphagnum* carpet would lead to erroneous results. This allowed the PWP within the macropores as close to the *Sphagnum* capitula as possible to be monitored. Finally, the *Sphagnum* carpet was gently pressed against the tensiometer tube. This ensured good contact with the *Sphagnum* carpet and the ceramic bulb of the tensiometer. The protruding tensiometer tube was fixed in place by attachment to a steel peg anchored in the sand below the *Sphagnum* carpet.

Top-down irrigation was applied from above via mist nozzles suspended from the greenhouse ceiling above the *Sphagnum* carpet. The spatial distribution of the irrigation points was pre-determined by MPS and mirrored the irrigation distribution across the rest of the facility which had been designed to provide an even coverage of water supply.

Irrigation application was applied as normal by MPS. The first method used a manually programmed timer which supplied 3 minutes of mist irrigation in the evening. The second irrigation application employed an evaposensor control that triggered additional irrigation from the mist system if evaporation exceeded pre-set limits defined by MPS.

The PWP of a *Sphagnum* carpet in the greenhouse was monitored for a period of 55 days during the summer months, from 16th May to 9th July 2017.

Figure 10: Pilot one experimental layout within MPS greenhouse, tensiometers are distributed across 2 transects, spaced approximately 30 cm apart, with a 50 cm gap left on all sides to reduce the likelihood of edge effects influencing the results.

6.2.3 Results

PWP across all individual tensiometers ranged from 20 – 93 hPa during the study period of 55 days. The key result shown in *Figure 11* below is the fact that PWP for every tensiometer remained below the 100 hPa threshold (shown as a grey dashed line) for 100% of the time.

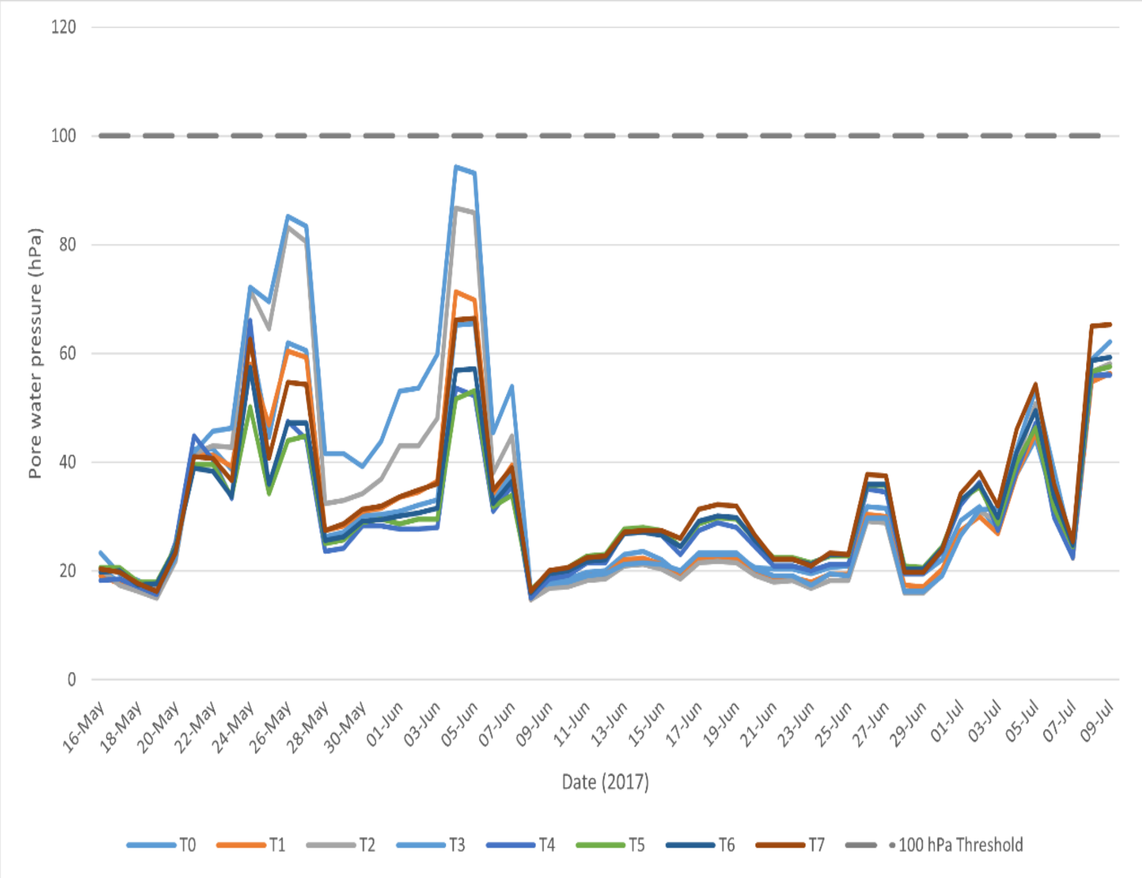


Figure 11: Maximum daily pore water pressure recorded within pilot study one.

A rise in PWP was detected for all tensiometers during the period 23rd May – 8th June, particularly for tensiometers at locations T1, T2, T3 and T7. This suggested a problem with the overhead irrigation system. Prior to the data download, MPS confirmed an issue related to the irrigation system was identified and corrected, resulting in lower PWP readings for the rest of the study. This demonstrates that the tensiometers could detect small-scale changes in PWP within the *Sphagnum* carpet.

One unanticipated finding was the variation in *Sphagnum* growth across the established *Sphagnum* carpet. At the beginning of the study, tensiometers were inserted at the same depth across the carpet. At the conclusion of the study period, a variable depth of moss had developed, evidenced by the differing length of tensiometer tube remaining visible at the surface. In all instances the *Sphagnum* carpet had grown to reach the black pressure transducer cap of the tensiometer, representing a minimum of 8 cm of growth. In some instances, the moss had almost totally overgrown the tensiometers after just 55 days.

6.2.4 Discussion

Firstly, the study met the primary objective of assessing Hypothesis 1. The mini electronic tensiometers supplied by Skye Industries had the appropriate PWP resolution to detect pore water responses to irrigation when installed directly in a *Sphagnum* carpet as Figure 11 shows that there are PWP responses, rising and falling with irrigation events in the greenhouse.

The results also confirm Hypothesis 2. PWP was prevented from reaching the critical 100 hPa threshold for *Sphagnum* under top-down irrigation. As expected, this method provided sufficient water provision in an internal, controlled environment. Water uptake by *Sphagnum palustre* was not a limiting factor for *Sphagnum* growth within the controlled greenhouse environment at the MPS facility.

Future research could focus on automating the process of reducing PWP by incorporating tensiometers into the irrigation control system at MPS. This would involve diverting control of the automated evaposensor irrigation system to that of a tensiometer one. This would trigger irrigation events when PWP thresholds were reached. Maintaining steady PWP levels within the *Sphagnum* carpet could potentially increase productivity.

This dataset is novel and provides the first objective tensiometer data within a micropropagated *Sphagnum* carpet grown and maintained in a greenhouse setting.

6.3 Pilot Study 2: *Sphagnum* pore water pressure in an external environment irrigated by drip-feed.

6.3.1 Aims

In Pilot Study 1, it was shown that key *Sphagnum* pore water thresholds were not exceeded in a controlled, internal environment using mist irrigation. This led to a similar spray irrigation system being proposed for the main field-trial study. However, it was anticipated that a second, or perhaps back-up, irrigation option might also be useful in the main study. This gave rise to a further research question, namely could a drip irrigation system prevent key ecohydrological thresholds from being exceeded in an external environment.

Dripline irrigation was proposed as one of the irrigation options for the main field trials study as it is commercially available, familiar to farmers and it offers efficient water use (Camp 1998) and (Wang *et al.*, 2022). There was no evidence within the *Sphagnum* farming literature of this being employed on a *Sphagnum* crop. The opportunity therefore existed for such an irrigation approach to be tested as a suitable water supply and distribution and compare results with those obtained from the mist irrigation used in Pilot Study 1. Rather than mimicking precipitation, driplines lie in direct contact with the *Sphagnum*, delivering water in the form of lateral seepage in much the same was as water seeps laterally through the acrotelm of a natural peat bog, permitting water to be captured and stored in the hyaline cells of the leaves as it passes through the *Sphagnum* carpet (Clymo and Hayward, 1982).

In addition to the above aims, Pilot Study 2 was also carried out in an exterior setting in order to begin more closely mirroring field conditions of *Sphagnum* farming, albeit at a small scale.

Three hypotheses were proposed:

Hypothesis 1: PWP across all tensiometers would not exceed 100 hPa.

Hypothesis 2: Mean PWP would increase with distance from dripline irrigation input.

Hypothesis 3: Tensiometers spaced 30 cm away from dripline input, would be most suitable for *Sphagnum* irrigation in an external setting as this standard spacing is suggested for most agricultural use.

6.3.2 Method

A *Sphagnum palustre* carpet measuring approximately 220 cm long by 40 cm wide was placed in an external growing area at the MPS facilities in Leicestershire. The micropropagated *S. palustre* carpet was created using the same methods as the first pilot, with micropropagated *Sphagnum* material being grown on a shallow layer of peat <4 cm. The external growing area consisted of a topsoil base with c.15 cm depth of sand layered over this and topped with a permeable woven geotextile.

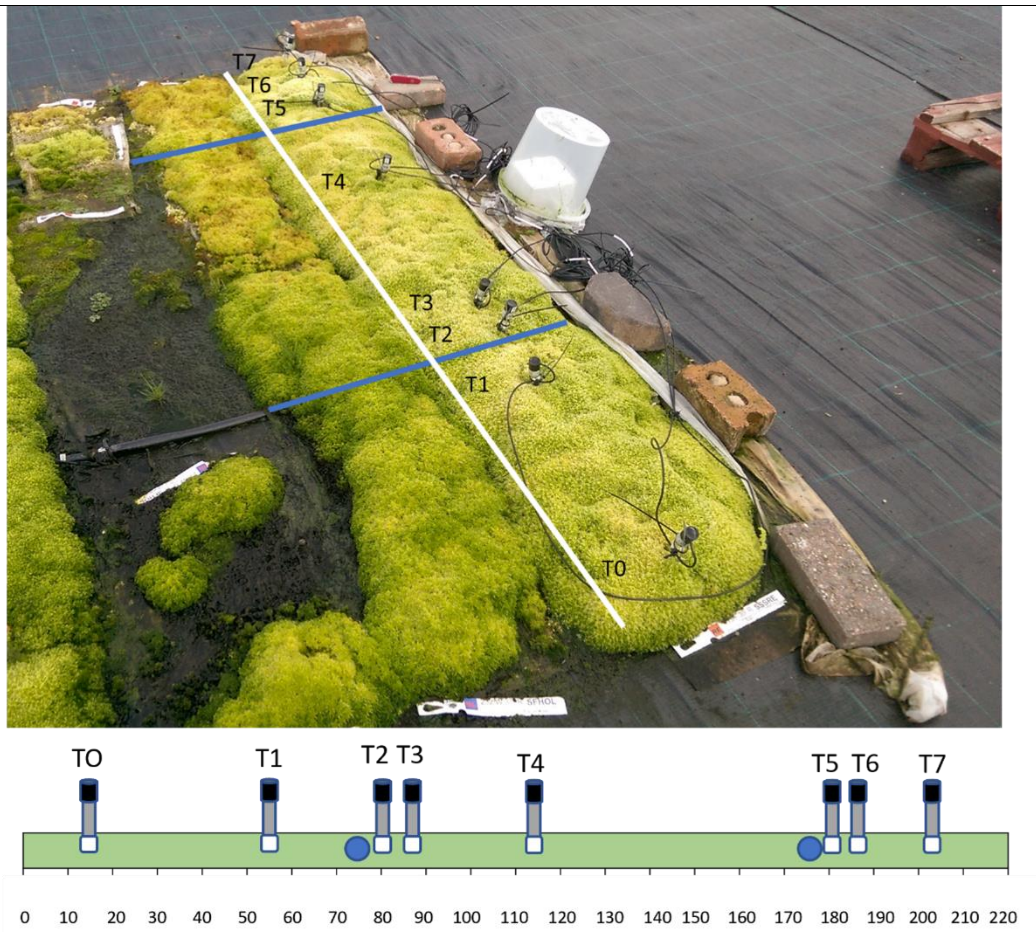


Figure 12: pilot study 2 experimental layout. Transect with cm markings (bottom) shows dripline inputs as blue circles, and tensiometer positions. Photo (above) shows tensiometers at collection, with Sphagnum Growth engulfing tensiometer tubes. A cover was placed on top of the Datahog data logger for extra weather protection.

The *Sphagnum* carpet was rectangular, with a dimension of 220 cm for its longer axis. Irrigation was supplied to the *Sphagnum* carpet via two driplines which lay within the *Sphagnum* carpet oriented along its shorter axis. Rather than using the commonly employed spacing for driplines of 30 cm, it was decided that the experiment would test a substantially wider spacing which might then reduce infrastructure costs at real-world field scale. Spacing between driplines was therefore set at 65 cm and 175 cm positions along the long axis of the carpet shown in Figure 12.

The mini electronic tensiometers were positioned along a transect oriented parallel with the long axis of the carpet and were grouped at different distances from, and at right angles to, the driplines. The tensiometer positions were categorised for the purposes of assessment as follows:

- Next to: 5 cm from dripline (Loggers 2 and 5)
- Close: 10 cm from dripline (Logger 6 and 3)
- Near: 20 cm from dripline (Logger 1)
- Far: 30 cm from dripline (logger 7)
- Furthest: 60 cm from dripline (Logger 0 and 4).

Drip irrigation supply was controlled by an evaposensor, a sensor that mimics a transpiring leaf and gives a continuous electrical output proportional to the rate of evaporation, pre-programed to activate irrigation using the same settings as the greenhouse settings in Pilot Study 1. Data from a nearby weather station were used to calculate water input from rainfall across the experiment period. The weather station comprised a Davis Vantage Pro2 utilising an anemometer, a rain gauge and a thermo-hydro sensor and was situated c.3km away from the site. The pilot ran from the 17th of August to 21st October 2017

6.3.3 Results

Generally, the distance from the dripline did not greatly affect the PWP readings across the tensiometers. The maximum daily recorded PWP and means are shown in Table 8.

Table 8: Maximum Tensiometer readings and mean results grouped by distance from dripline irrigation source. *For distance groups with replication 5 cm (n =2) and 60 cm (n = 2) the mean results across all replicates are presented.

Distance from dripline irrigation source (cm)	Tensiometers used	Range of maximum daily pore water pressure (hPa)	Mean daily pore water pressure across tensiometers (hPa)
5	2, 5	5.90 - 24.18	9.61*
10	3,6	7.67 - 20.94	9.94
20	1	6.48 - 22.10	9.64
30	7	9.16 - 22.45	11.22
60	0, 4	5.31 - 20.65	8.13*

Relatively little difference was observed between the maximum pore water pressures recorded on any given day by the tensiometers at different distance groupings from the dripline locations, values ranging from 20.65 to 24.18 hPa.

Table 9: Weather data summary from publicly available weather station Source: (East Leake Weather, 2018)

Year	Rainfall observed 17th August to 21st October (mm)	Annual rainfall (mm)
2014	121.1	951
2015	127.8	758
2016	116.1	812
2017	178.5	854
	Annual mean (2014 – 2017)	844

The rainfall volume over the monitoring period 17th August to 21st October 2017, compared with the same period over the previous 3 years, are presented in Table 9. Weather data was available as summary data, and not at the same resolution as the tensiometer loggers, and was available from 2014 onwards (East Leake Weather, 2018).

6.3.4 Discussion

The results of the outdoor irrigation assessment showed that the established *Sphagnum* carpet consistently experienced a relatively narrow range of PWP over the monitoring period. Table 8 demonstrates that across all distance groupings, maximum PWP range was 5.31 to 24.18 hPa. This is well below the 100 hPa threshold which limits water uptake by *Sphagnum* plants. The mean of daily maximum PWP across all treatments was 15.51 hPa with a standard deviation of 3.48 hPa. This is lower than in pilot study one, perhaps reflecting the denser growth form of *Sphagnum* in an external environment compared to in the greenhouse, though this was not quantified as part of the pilot studies and is therefore anecdotal.

Hypothesis 1 was thus found to be true: dripline irrigation prevented the 100 hPa pore water threshold being reached on a *Sphagnum* carpet in an external setting.

Hypothesis 2 and Hypothesis 3 sought to evaluate whether proximity to dripline irrigation input influenced PWP in an external setting. The evidence presented here suggests that at a maximum distance of 110 cm between driplines there are no ecologically significant differences between distance groupings in an established *Sphagnum* carpet.

Hypothesis 2 has been disproven as the tensiometers at 60 cm distance from drip line inputs generally experienced maximum PWP ranges lower than the groupings with closer proximity to dripline input.

Hypothesis 3 has also been disproven, the 30 cm distance grouping of tensiometers offering no hydrologically meaningful benefit over any other distance grouping used in this experiment.

The PWP range observed across all distance groupings during the study was relatively narrow. This narrow PWP range could be due to multiple water inputs from the drip irrigation and natural rainfall. Table 9 shows that the rainfall during the study period in 2017 was greater than the same period over the previous three years. Consequently, higher rainfall than normal could have regularly saturated the *Sphagnum,* thereby lowering the PWP values. Pilot Study 2 took place towards the end of summer/early autumn, so the evapotranspiration demand may have been lower than if the trial had taken place earlier in the summer, though assessing evaporative demand was beyond the scope of the pilot studies.

Another interpretation of these results is that the pre-established *Sphagnum* carpet was able to maintain a high level of water availability across the entire moss area due to the high density of *Sphagnum* individuals within the carpet. Due to the architecture of *Sphagnum*, which permits water to be transported via capillary action (explained in detail in Chapter 3), water moves both up and down its stem (via the pendant branches) but also laterally through the interconnections made between the spreading branches of adjacent plants. This potentially allows for water to be transported across the entire *Sphagnum* carpet – at least over the distances involved in the present trial. This capillary transport of water may thus be more influential than dripline spacing in an established *Sphagnum* carpet. However, Pilot Study 2 provides no information about whether drip line supply could be used to support *Sphagnum* development on an initially bare peat surface. Further research is required to assess this.

The data reveal that supplementary dripline irrigation in combination with natural rainfall is likely to be capable of maintaining the PWP below the 100 hPa threshold within an established *Sphagnum* carpet, at least using dripline spacings of up to 1.1 m. These data are site specific, to a site close to East Leake, with an average rainfall of 844 mm. However, the mean UK annual rainfall has ranged from 1020mm to 1889 mm since 2001 (Statista, 2022) suggesting that most UK locations could support *Sphagnum* farming under this hydrological pattern. However, further study is needed across a wider range of climatic conditions in the UK, especially in regions with lower annual rainfall, for a clearer picture to be developed of the national potential for *Sphagnum* farming based on dripline irrigation. This complements the data in pilot study one, that shows mist irrigation can support *Sphagnum* growth in a glasshouse environment.

Pilot Study 2 at least suggests that drip irrigation has potential as an additional irrigation method for future *Sphagnum* crops and merits testing at scale on peat soils. The system may offer cost advantages over site wide infrastructure works due to the commercial availability of the components and ability to be installed quickly with little training. Hydrological opportunities of dripline include the potential reduction in evaporative losses compared to spray or boom irrigation methods. Further research into these areas is necessary as economic viability and water demand are likely barriers to expansion of *Sphagnum* farming.

6.4 Pilot Study 3: Mulch cover and pore water pressure in an external environment.

6.4.1 Aims

It is widely considered within *Sphagnum* farming literature (Pouliot, Hugron and Rochefort, 2015; Wichmann, Prager and Gaudig, 2017) that a mulch cover is required to enhance establishment and growth of the crop. Typically, a straw mulch cover is used during the MLTT approach, as discussed earlier (Quinty and Rochefort, 2003). Straw mulch has proven benefits, but, because it tends to decompose relatively rapidly it does not remain in place and is not present for the whole duration of a *Sphagnum* cultivation cropping cycle. Therefore, the protective effect of straw cover declines rapidly as the straw degrades (Gaudig *et al.*, 2017). Alternative mulches are available and could be employed if they conferred a longer protective period and additional competitive advantage for *Sphagnum* over non-target species during establishment and beyond. This pilot aimed to assess, at least on a small scale, whether any differences in PWP could be observed under different mulch cover treatments.

Hypothesis 1: There will be variation in PWP across mulch cover treatments.

Hypothesis 2: Bare peat areas will experience the highest PWP.

6.4.2 Method

A set of three experimental test surfaces onto which a variety of mulch types was applied formed the basis for this experiment, located at the external facility on the MPS premises. The entire experimental area consisted of a layer of sand which formed the base substrate, overlain with a permeable woven geotextile. The ground was levelled for uniformity and three shallow peat strips with dimensions 8 m (length) x 2 m (width) x 0.04 m (depth) were then constructed over this geotextile base. This ground preparation system was chosen by MPS to reflect the internal set up within their facility, though in this case applied to an exterior setting.

Gel founder material was applied evenly across all three peat strips. A total of nine different mulch cover treatments were distributed across the newly applied gel material, leaving three areas uncovered as bare control plots. The mulches chosen aimed to be representative of the commercially available options, there were no prior expectations about their performance, only that they might retain water and aid *Sphagnum* establishment. Mulch treatments were not replicated or standardised in terms of area because the size of the experimental strips did not provide enough surface area to allow for such accommodations.

The experimental layout is shown in Figure 13. Strip 1, which included one of the control plots, was constructed first and monitored from 16/04/18 - 7/06/18. Strips 2 and 3 were constructed later and monitored from 07/06/18 - 17/08/18. Strip 2 included two control plots.

	MPS buildings									North				
Strip Three	low density bark			clear plastic	woven fabric					very perforated plastic				
Plot length (cm)	0-50	50-100	100-150	150-200	200-250	300-350	350-400	400-450	450-500	500-550	550-600	600-700	700-750	750-800
Strip Two	Straw (low density)	Bare peat control	spun fleece						bare peat control					
Plot length (cm)	0-50	50-100	100-150	150-200	200-250	300-350	350-400	400-450	450-500	500-550	550-600	600-700	700-750	750-800
Strip One	Straw (normal density)	Bare peat control	woven plastic						biodegradable plastic					
Plot length (cm)	0-50	50-100	100-150	150-200	200-250	300-350	350-400	400-450	450-500	500-550	550-600	600-700	700-750	750-800
	Road													

Figure 13: Experimental layout of mulch cover trials: plastic (clear/ perforated), woven fabric, spun fleece, straw (low and normal density), biodegradable plastic conducted at MPS external facility, irrigation provided by spray irrigation system. Mulch covers are on top of a *Sphagnum* gel founder material, applied to a c. 4 cm layer of peat substrate.

Irrigation was provided through a commercially available spray irrigation system. The spray system surrounded the boundary of the three irrigation areas, and multiple spray nozzles provided an even application of water across the experimental area. Irrigation was applied manually when judged to be required, or via a timer. Unfortunately, no data were obtained regarding the applied water volume or frequency of application, although an estimated input by MPS was given as 1 to 2 ml per day per m^2.

As in Pilot Study 2, the same local weather station was used to assess weather conditions during the monitoring period.

Tensiometers were installed at a density of two per mulch treatment within the peat substrate. A minimum level of replication was used in case of equipment failure during the experiment, but replicates were limited due to equipment constraints. The tensiometers were attached to a wooden baton suspended over the treatment areas to prevent compression of the peat surface below. Suspension of the wooden baton prevented experimental error from water pooling around the which could have artificially lowered tensiometer readings.

To enable accurate comparisons between strips, all strips were constructed using the same materials and methods, over a small area. Due to time constraints and overlap with the first pilot study, It was deemed unnecessary to assess the strips as bare peat surfaces prior to application of the *Sphagnum* treatment or mulch covers. Large PWP differences would likely only be experienced in this experiment if the depth of peat substrate varied wildly, which was minimised via the same construction techniques.

6.4.3 Experimental constraints

As shown in Figure 13 not all areas covered by mulch were equal in size, which may have influenced the PWP due to a reduction in water storage capacity within the peat area under each mulch treatment. However, all mulch treatment areas were within range of the spray irrigation system, so variation was expected to be minimal.

Replication with tensiometers was limited to two due to equipment restrictions. Ideally replicates would be closer to 5 tensiometers per treatment for statistical validity. However, given that this pilot study was a basis for ‘proof of concept’ and equipment availability was a constraining factor, two tensiometers per treatment gave a degree of redundancy for breakages or other logger issues.

Due to equipment limitations strip one was monitored early in the year, during this monitoring period, temperatures were cooler (maximum temperature 27.6°C), and precipitation higher (96.4 mm) than the monitoring for Strip two and three. Strips two and three were monitored later in the year, and experienced warmer temperatures (maximum temperature 33°C) and lower precipitation (52.5 mm). Due to temporal differences in monitoring it could be argued that comparing the impact of mulch covers between the two monitoring periods was invalid. However, the use of a bare peat control area in strip one, and a bare peat control in strip two, that could be used as a proxy for strip three gave a control over both monitoring periods and helped to overcome this. Nonetheless, further work should aim to monitor all treatments on the same temporal scale.

6.4.4 Results

As reported earlier, the recovery of *S. palustre* in the face of drought is sub optimal when PWP exceeds 100 hPa for 6 days or more presented in Figure 9 (Clymo and Hayward, 1982). Taking a conservative approach, drought stress was defined as a day where daily maximum PWP > 100 hPa, which is shown in Table 10, Table 11, and Table 12. The combined effect of spray irrigation together with the mulch cover treatments prevented critical thresholds being exceeded in 8 out of 9 mulch treatments. Although several instances occurred of PWP exceeding the critical threshold of 100 hPa for relatively short periods of time, the duration of such events only exceeded the critical duration of 6 days or more (12 days) in the case of low density bark mulch in Strip 3, while even bare peat only just reached that time threshold.

Table 10: Pore water pressure results for Strip 1 – monitored for 53 days (16/04/18 to 7/06/18)

Mulch trial: Strip 1.	Straw normal density	Straw normal density	Bare peat	Bare peat	Woven plastic	Woven plastic	Bio-degradable plastic	Bio-degradable plastic
Tensiometer number:	0	1	2	3	4	5	6	7
Days in trial where max reading >100 hPa	0	0	0	0	2	0	0	0
Length of longest period >100 hPa (days)	0	0	0	0	2	0	0	0
Average hPa	21.60	19.66	19.00	16.61	32.84	23.33	21.34	19.57

Table 11: Pore water pressure results for Strip 2 – monitored for 72 days (07/06/18- 17/08/18)

Mulch trial: Strip 2	Straw low density	Straw low density	Bare peat	Bare peat	Spun fleece	Spun fleece	Bare peat	Bare peat
Tensiometer number:	0	1	2	3	4	5	6	7
Days in trial where max reading >100 hPa	3	3	6	1	1	4	0	0
Length of longest period >100 hPa (days)	3	3	5	1	1	2	0	0
Average hPa	36.26	30.17	54.25	31.51	30.09	49.25	27.45	31.24

Table 12: Pore water pressure results for Strip 3 – monitored for 72 days (07/06/18- 17/08/18)

Mulch trial: Strip 3	Bark low density	Bark low density	Clear plastic	Clear plastic	Woven fabric	Woven fabric	Very perforated plastic	Very perforated plastic
Tensiometer number:	0	1	2	3	4	5	6	7
Days in trial where max reading >100 hPa	12	5	0	5	4	3	2	0
Length of longest period >100 hPa (days)	12	3	0	3	2	2	2	0
Average hPa	183.25	63.53	11.69	50.40	41.91	38.97	56.17	27.60

6.4.5 Discussion

The only ecologically critical result in terms of PWP across the three mulch trial strips was recorded in the by T0 in the low-density bark mulch in strip three Table 12. However, it is possible that the observed readings were caused by another factor such as logger disturbance, given that even the bare peat controls only just approached the critical duration of 6 days in only one of the three bare peat control plots Table 11.

In an optimised *Sphagnum* farming system, decisions would need to be made about whether to employ a mulch, and if the decision is made to use mulch, then which choice of mulch treatment would contribute best to the prevention of pore water thresholds being reached. An optimum mulch treatment to limit the period where thresholds are exceeded to a duration of less than 6 days may be appropriate for *Sphagnum palustre,* but this threshold is likely to be species specific, so particular mulches may be suitable for particular species.

Based on this study, it might be concluded that the MLTT approach of using straw mulch as standard may be un-necessary if top-down irrigation is provided because two out of three bare peat plots performed no worse than those covered with normal density straw as shown in Table 10 and Table 11. Overall, bare peat controls performed markedly better than low-density straw mulch in terms of reducing drought duration. Perhaps even more surprising is the fact that none of the other mulches performed better than bare peat under this irrigation regime. This might lead one to conclude that costs could be kept low and site-management effort reduced by dispensing with the use of a mulch. Although such a conclusion may be supported by the hydrological evidence, subsequent experience of the proposed MIFA approach at the field scale has revealed other arguments for employing some form of mulch cover, such as protection from animal disturbance, which will be explored later in the present thesis.

6.5 Conclusions from pilot studies

In terms of developing a working knowledge of tensiometer use, the pilot studies have shown that tensiometers and their dataloggers are robust and capable of detecting rapid changes in PWP under both interior and exterior conditions, indicating that their use should also be suitable for use in the main field-scale study areas.

The pilot studies have shown that the tensiometers are capable of recording PWP changes in both *Sphagnum* carpets and in peat substrate. The data can be used to determine when PWP thresholds >100 hPa are reached, and to calculate the number of days affecting desiccation tolerance during the pilot experiments.

The pilot studies have shown that the MIFA approach of using top-down irrigation with micropropagated founder material is effective, at least on a small scale, and choice of mulch cover may be important. A critical point about these pilot studies, and a fundamental break from the MLTT approach, is that the *Sphagnum* crop was cultivated in the absence of a water table, all moisture being supplied from above rather than the crop depending on supply from below. This represents a radical change from all previous work on *Sphagnum* farming and opens up many new directions for the future of *Sphagnum* farming, but the critical question, therefore, is whether the MIFA approach can work at scale. The remainder of the present thesis describes research devoted to exploring this key question.

Chapter 7. Experimental set-up for main study

7.1 Introduction

Until now, global research into the agricultural farming of *Sphagnum* has largely been centred on sites located in continental Europe and Canada. These research efforts have begun to increase the knowledge and understanding of *Sphagnum* farming (Chapter 4) although the topic remains a largely unproven form of paludiculture in terms of its commercially viable development and market penetration.

Commercially viable *Sphagnum* farming needs to produce large volumes of *Sphagnum* biomass to provide a credible replacement for fossil peat in growing media. Therefore, the production area must be sufficient to produce volumes capable of satisfying market demand, which in the UK alone equates to c 3.35 million cubic metres of peat-based growing media per year, shown in Table 6. Glasshouse-grown *Sphagnum* will never be capable of meeting this demand, so the question is whether *Sphagnum* can be grown outside the glasshouse environment in sufficient quantities to meet market needs.

Until 2017, the only fully commercial operation in the world devoted to growing *Sphagnum* as a crop was that produced by MPS for use in peatland habitat restoration schemes. Meanwhile the *Sphagnum* farming trials so far undertaken in Germany and Canada using the MLTT approach have not yet demonstrated viable agricultural practices capable of widespread adoption nor commercial viability of the crop. The pilot studies described in Chapter 6 represented one part of the first-ever research programme looking into the potential for *Sphagnum* farming within the UK lowland agricultural landscape. Having obtained promising results from these pilot investigations using the novel MIFA approach to *Sphagnum* farming, the obvious next step was to expand these trials into something closer to real-world agricultural field trials.

Based on initial pilot projects described in Chapter 6, it has been shown that *Sphagnum* from micropropagated stock can grow in an interior and exterior environment with irrigation systems in place on a shallow <4 cm layer of peat growing media. The pilot studies showed that when top-down irrigation was used with and without various mulch covers, PWP remained under 100 hPa, a key ecohydrological threshold for *Sphagnum* growth (Hayward and Clymo, 1982; Price, Rochefort and Quinty, 1998). However, the previous pilot studies took place in artificial environments, in the absence of a water table and were performed on a small scale, for short time periods and not on true peat soils.

To achieve real world proof of concept for the top-down irrigation method, two main research sites were chosen as representative of the typical areas currently used for *Sphagnum* farming experiments in temperate Europe. The site selection also reflected the most common land uses for degraded lowland peat soils within the UK, therefore representative of sites that are the most likely candidates for conversion to paludiculture in future. These sites, and the trials undertaken on them, form the core of the novel research undertaken and described within this thesis.

The overall conceptual framework for the MIFA approach developed in the main research study is presented in Figure 14. This reflects the broader conceptual framework for paludiculture crops shown in Figure 5 but focuses specifically on *Sphagnum* farming and highlights the areas of novelty this research study aims to investigate.

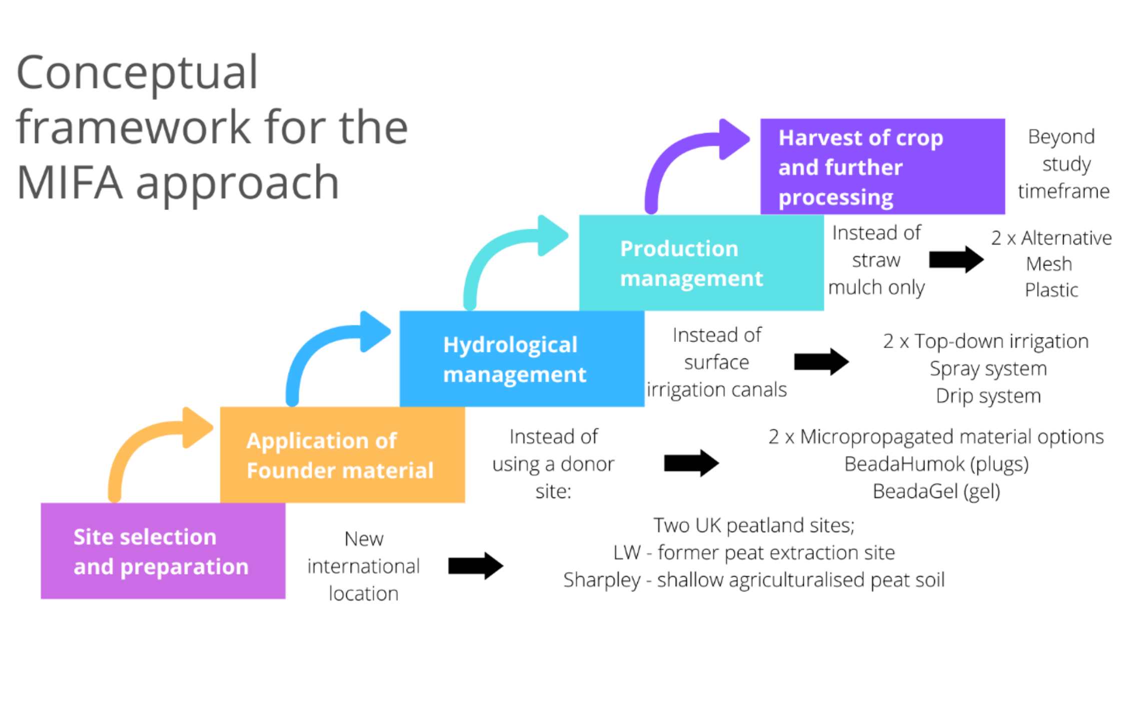


Figure 14: Conceptual Framework for the PhD study. Black arrows indicate the key areas of novelty investigated in this study, generating new knowledge building on *Sphagnum* farming projects published to date.

The novel aspects of this research build upon the pilot studies that suggest at a small scale that irrigation from above can support *Sphagnum* growth in established moss carpets. However, the further questions raised about establishing *Sphagnum* via the proposed MIFA approach can be summarised as follows:

- providing two new international locations for *Sphagnum* farms in lowland Britain;
- investigating the use of micropropagated *Sphagnum* material as founder material instead of harvesting *Sphagnum* from a donor site;
- applying two novel top-down irrigation treatments for hydrological management;
- new methods for measuring *Sphagnum* growth; and

- the use of novel mulch covers to provide protection for *Sphagnum* post-application and beyond.

7.2 Locations and site descriptions

Following extensive exploration and negotiation, facilitated by Micropropagation Services Ltd. (MPS), two sites were identified and agreed with the landowners as suitable for this first field trial of the MIFA approach to *Sphagnum* farming. The two sites were chosen in part because of their contrasting characteristics, but also because they sat at opposite ends of the spectrum of what type of land might feasibly offer potential for *Sphagnum* farming in the UK lowlands. One site was located in the Midlands of England while the second site is located in NW England (see Figure 15).

Figure 15: Site locations for the experimental sites, Little Wooden (upper star) and Sharpley (lower star).

7.2.1 Site 1 – Sharpley, Leicestershire

The Sharpley site is an area with a shallow peat soil which has been subjected to conventional agricultural activities for at least three generations. As such, it is typical of the most extensive potential areas for paludiculture in the UK (potential areas are discussed in Chapter 1). Shallow peat soils are often classified as ‘wasted’ when the peat depth is less than 40 cm as they lose their productivity. This is because it is the presence of organic-rich peat that provides good soil structure and valuable capacity for moisture retention. *Sphagnum* farming on shallow wasting peat will potentially reduce further soil losses, whilst maintaining agricultural productivity into the long term by preventing soils from becoming ‘wasted’.

The experimental site occupies the southeast corner of an agricultural field positioned on a watershed located on one of the highest summits within Charnwood Forest, Leicestershire. The summit consists of a granitic outcrop, as a result, the peat of the experimental site had a complex mix of granitic fragments within a clay matrix. This combination proved extremely difficult to penetrate when installing equipment, but which probably forms a reasonably impermeable sub-peat layer.

The field sits within a wider landscape mosaic consisting of agricultural land (Figure 16) which has experienced gradual agricultural intensification since the Middle Ages (Keil and Wix, 2014). Remnants of Charnwood Forest woodland also occupy the less-tractable portions of this landscape. It is situated at an altitude of between 190 m and 180 m above sea level (Ordnance Survey, 2023) with a gently sloping gradient of 2 degrees from the NW corner towards the *Sphagnum* farm area, which is located in the SE corner of the field at UK National Grid Reference: SK 44952 16697, shown in Figure 16 and covers an area of 50m by 10m (0.05 ha).

Figure 16: Sharpley *Sphagnum* farming site location circled, within a wider mosaic of land use. Map at 1:10000 scale. Source EDINA, Digimap. The agriculturalised peat site is surrounded by agricultural land – representing a conflict in water use within the landscape – retaining water in the moss area but removing it in surrounding areas.

Although the area of peat soil is too small to register on the LandIS Soil Maps, the inherent peat-forming conditions of the area are indicated by the Environment Agency (EA) Flood Risk maps for surface flooding, which clearly highlights the area of existing peat soil (see Figure 17 below).

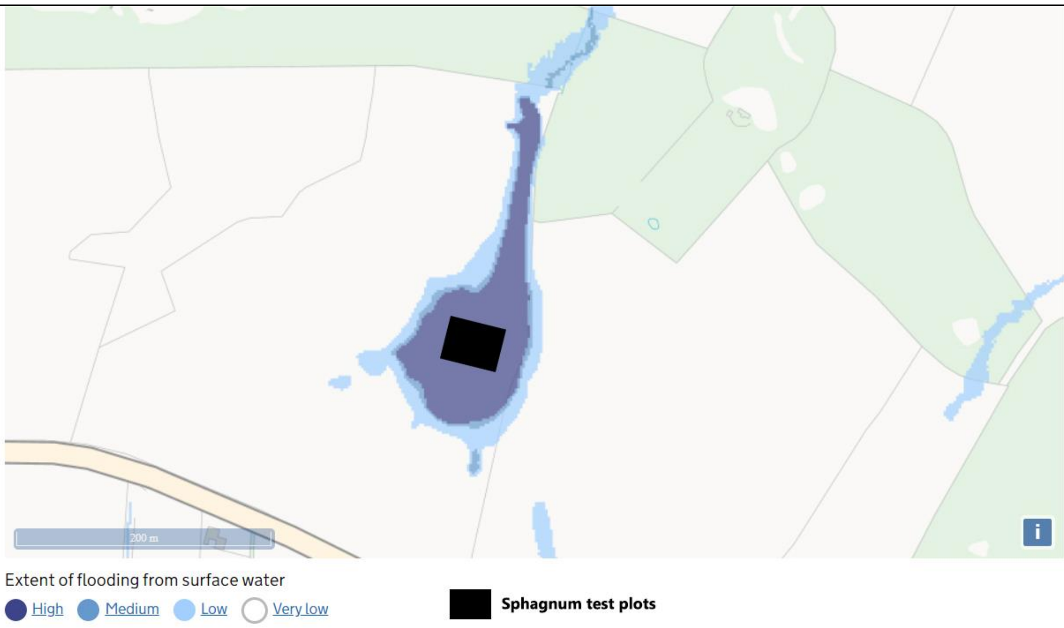


Figure 17: Environment Agency flood risk map for surface flooding for the area of the Sharpley experimental plots.(Environment Agency, 2023)

The peat here is not truly ombrotrophic for several reasons. Firstly, the peat is shallow and less than 30 cm in some places and is underlain by the matrix of granitic fragments and marine clay. Secondly, the site lies in something of a saddle and is identified by the EA as subject to surface flooding. Therefore, the peat at Sharpley is more likely to be minerotrophic as it has some interaction with the underlying water table.

Historically, the Sharpley site has been subject to agricultural usage for a considerable period of time. A network of field drains, installed underneath the organic soil layer, can be discerned in the pattern of vegetation cover and regular slight undulations of the ground surface. According to the current farmer these drains were installed approximately 60 years ago, but it is difficult to establish the full history of the site beyond the knowledge of the current landowners. The area selected for the *Sphagnum* farm was described by the current landowner as a ‘problem area’ because the area tends to remain wet throughout the year. The site has been regarded by the farmer as suitable only for grazing or haymaking rather than arable crops. Such ‘problem areas’ for conventional agriculture represent precisely the kind of ground which, with the adoption of paludiculture, has the potential to be transformed into what becomes regarded as ‘productive ground’.

The site has had a mixed history of agricultural use, having previously been enriched with fertiliser, ploughed, cultivated for crops and been subject to cattle grazing. The landowner anecdotally recalled that, as a child, he could stand at the bottom of a deep excavated channel within the peat during installation of the field drains. This suggests the original peat depth has undergone considerable losses following drainage. This history of usage and the consequent impact on the peat soil layer means that today this layer is probably better termed an organo-mineral soil rather than a pure peat soil, but such organo-mineral soils are extensive within the lowland agricultural landscape of the UK.

Agriculturalised peat soils account for c.13% of the UK peat soil area, and have been identified as a key area for intervention to achieve the required reduction of 35-85% of agricultural emissions (Chapter 1). The Sharpley experimental site is thus a good testbed for investigation of *Sphagnum* farming potential on such soils. If *Sphagnum* can be cultivated on such a thin agriculturalised peat soil, this substantially increases the potential area available for *Sphagnum* farming in the UK.

Climatic data

A private weather station based in East Leake (8 miles to the NE of the study site) was used to ascertain average climatic data for the region. Monthly data is available from 2014 to present (East Leake Weather, 2018). The climate of the study region is temperate with an annual temperature range of -7˚C to 40˚C, the mean annual temperature is 10.7˚C, average rainfall is 891.5 mm yr^{-1}, with a minimum rainfall of 48 mm falling per month.

Soil characteristics

Soil samples were collected by Manchester Metropolitan University across the proposed site following site-levelling works. The remaining soil had an average mineral content of 63% and an organic content of 37%, based on 53 samples from across the site. The soil pH was weakly acidic, with an average pH of 6.49 based on 76 test samples. Soil nutrient analysis evaluated 26 nutrients within the soil samples. The highest mean values, perhaps not surprisingly given the history of the site, were nitrate (268ppm) calcium (151 ppm) chloride (57 ppm) and potassium (46 ppm).

7.2.2 Site 2 – Little Woolden, Greater Manchester

The Little Woolden site was established on a lowland raised bog which had been subject to commercial peat extraction, using the milling method for many decades. The site represents one small part of what was once the huge Chat Moss raised bog complex formed on the lowland coastal plain associated with the River Mersey and lying between the cities of Liverpool and Manchester in NW England. Since the 1840s, extensive agricultural land-claim and various peat extraction schemes have combined to reduce the once-large Chat moss complex to a few remnant fragments, of which Little Woolden Moss is one, albeit with no surviving natural bog habitat (Bragg, Lindsay and Robertson, 1984).

Commercial peat extraction operations on Little Woolden ended in 2017 with expiry of the planning permission, leaving large expanses of reasonably level 'milling fields' of bare peat. The residual peat depth varies from 1.8 – 2 m and is underlain by marine clay. Restoration of the site to bog habitat is being undertaken by the Lancashire Wildlife Trust, although the site is surrounded by commercial agriculture (Figure 18), which makes integrated landscape-scale hydrological management difficult.

Several former peat extraction sites have been trialled in the wider literature and are typical locations for *Sphagnum* farms. This site will allow close comparison between *Sphagnum* farming in the UK and other *Sphagnum* farms on commercially extracted peatlands around the world. Former extracted sites are ideal candidates for *Sphagnum* farming for several reasons. Firstly, less site preparation is needed as access infrastructure is readily available, the surface level of the former peat fields is often close to level. They may also provide donor *Sphagnum* for further restoration work across their locale in future.

Figure 18: Little Woolden *Sphagnum* farming site location circled, with wider land use context. Map at 1:10000 scale. Source EDINA, Digimap. Note the large area of dark, bare peat surface remaining after peat extraction on Little Woolden Moss. The site is surrounded by agricultural land – representing a conflict in water use within the landscape – retaining water in the moss area but removing it in surrounding areas.

Climatic data

A weather station based in Salford (8 miles to the NE of the study site) was used to determine average climatic data for the site. Publicly available data are available from a privately owned weather station from 2002 to 2021 (Paul Unknown, 2023). The climate of the study region is temperate. Over the period of 2002 – 2021 a temperature range of -9.3˚C to 38.3˚C was observed, with a mean annual temperature of 10.62˚C. In terms of precipitation, mean rainfall is 984.2 mm yr $^{-1}$ while the minimum rainfall in any one month is 47.9 mm.

Soil characteristics

A total of 10 soil samples were collected by Manchester Metropolitan University (MMU, unpublished data) across the proposed site following site-levelling works. Soil pH was found to be markedly acidic at pH 4.52, while the mean mineral content was determined at 9 % and the organic content of 91%.

Site preparation

The Little Woolden site differed markedly from the Sharpley site. Cessation of peat extraction occurred shortly prior to the *Sphagnum* farm development. The area chosen for the *Sphagnum* farming trials was located within the centre of a peat milling field which formed part of a site-wide habitat restoration plan. A hardcore access track was constructed around three sides of the peat milling field, then a series of bunds made from compacted peat was constructed within the milling field to create a set of restoration ‘cells’. Two of these bunded cells were allocated to the *Sphagnum* farming trials. Preparation of the chosen area involved removal of the dried, oxidised layer of surface peat, followed by levelling, raking to a fine tilth, then addition of micropropagated *Sphagnum* founder material.

7.3 Description of experimental sites

Both sites followed largely the same experimental layout, with an aerial view presented in Figure 19. Two irrigation areas were established at each site, each irrigation area consisting of 36 plots measuring 2 m by 1.7 m (3.4m^2). These plots were arranged in a 4 x 9 grid with 50 cm between each plot for ease of access (Figure 20 and Figure 21).

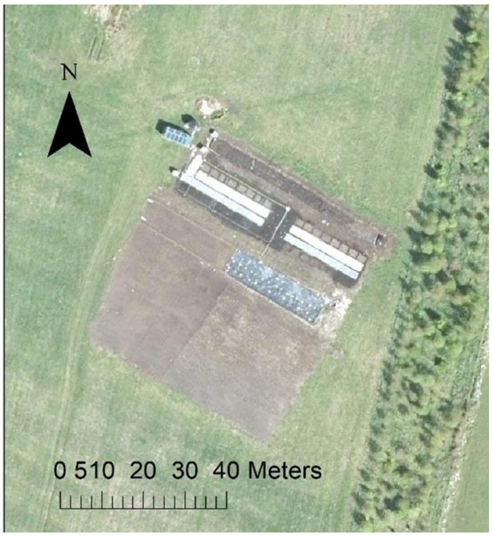


Figure 19 Aerial views of the Sharpley site (left) and the Little Woolden site (right) produced from Edina Digimap aerial imagery at 25 cm resolution, with a map scale of 1:10000.

7.3.1 Founder *Sphagnum* material

Two different forms of micropropogated *Sphagnum* were supplied by the commercial partner (MPS) and tested. The use of micropropagated founder material is novel, as all previously published *Sphagnum* studies have used founder material obtained by wild harvesting from donor sites or from *Sphagnum* farming sites which have themselves relied on wild-harvested founder material. As discussed earlier, the benefit of the MIFA approach used here, based on micropropagated founder material, is that the founder material consists only of the crop species and is free from other plant material. Micropropagation also allows the active selection of single- or multi-species founder material, giving a greater precision in species selection than taking material from a donor site. It also removes the need to deal with legislative, permit and logistical issues associated with wild harvesting.

The two types of micropropagated founder material were *Sphagnum* plugs of mature plants (BeadaHumok™) and a sprayed gel of immature plant fragments (BeadaGel™), both types being created and sold by MPS.

Single species plugs consist of mature *Sphagnum* plants which are grouped by hand at MPS into plugs of approximately 5 cm diameter. The advantage of the plugs is that the plants are already approaching maturity and are bunched together as they would be in a natural *Sphagnum* carpet, which is likely to confer increased drought resistance to the plug.

Single species *Sphagnum palustre* plugs were planted at a planting frequency of 36 plugs per m^2. This resulted in planting in an arrangement of 12 x 12 plugs per plot to achieve an MPS recommended density of 36 plugs per m^2. Plugs were planted by hand, using a handheld dibber. The dibber made a 3 cm hole in the peat soil, allowing plugs to be inserted. The plugs were secured in the planting location by backfilling the hole with peat and gently compacting the peat around the plug.

In contrast, Single species *Sphagnum palustre* gel was applied to the appropriate plots at a rate of 2 litres per m^2 using a hose applicator. The gel can be applied using a backpack applicator, so application is potentially more efficient in terms of material, time and cost. However, the material may initially be more prone to drought and flooding-outwash issues due to the smaller fragment size compared to plugs and the fact that the *Sphagnum* fragments are not held in place as the plugs are.

Evaluation of the relative performance observed for these two types of micropropagated material forms one of the novel aspects of the present thesis research programme because the two possible MIFA approaches to the application of founder material had never before been simultaneously tested under field conditions.

7.3.2 Irrigation

As discussed earlier, the MIFA approach to irrigation aims to mimic rainfall and supply water directly to the *Sphagnum* surface where it is needed for growth. The pilot studies described in Chapter 6 investigated three top-down irrigation systems, namely mist (in the glasshouse) and spray and drip systems (in exterior settings), to deliver irrigation water on a small scale within a highly controlled and artificial system. The main study on the field sites aimed to expand these in terms of scale and apply them to real world experiments on peat soil.

All irrigation methods across the field sites were controlled by an evaposensor identical to those used in the pilot studies and in the MPS greenhouse. Unfortunately, the total irrigation volume applied was not quantified in this study, although MPS estimates for water input using the evaposensor control based on previous use, were an irrigation input of 1 to 2 mm (1000 – 2000 m^3 ha^{-1} $yr^{-1)}$

The mist irrigation method as used in the MPS glasshouse was deemed unsuitable for use at scale, as the mist emitted was very fine and therefore likely to suffer large distribution losses from wind. Drip irrigation and overhead spray irrigation systems were therefore selected as the two novel irrigation methods to be tested as part of the MIFA approach, as shown in Figure 20 and Figure 21. Such systems are already used at scale in a variety of other crop cultivation models (Knox and Weatherhead, 2006; van der Kooij *et al.*, 2013; Wang *et al.*, 2022) but have never been trialled on a *Sphagnum* farm.

The spray irrigation infrastructure consisted of plastic piping surrounding the perimeter of the combined irrigation area. Spray nozzles were distributed along these plastic pipes, with nozzles located at each corner by plots AS1, DS1, AS9 and DS9 with 4 additional nozzles distributed along the outermost edges of both columns A and columns D.

The drip irrigation areas were created by installing continuous lengths of dripline tape along the length of each column (i.e., along the long axis of the area, from Plots 1 to 9), with 30 cm spacing between each dripline. The dripline tape was laid on the peat surface with the expectation that it would become buried beneath the *Sphagnum* crop over time. Six individual dripline tapes ran through each column of plots.

	AS	BS	CS	DS
1	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
2	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
3	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
4	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
5	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
6	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
7	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
8	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
9	No planting No mulch	No planting No mulch	No planting No mulch	No planting No mulch

	AD	BD	CD	DD
1	No planting No mulch	No planting No mulch	No planting No mulch	No planting No mulch
2	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
3	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
4	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
5	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
6	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
7	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
8	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
9	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw

Figure 20: Sharpley experimental layout showing the Spray irrigation area (left) and Drip irrigation area (right). Mulch covers are distributed from left to right as Columns A to D (No mulch, Mesh, Plastic, Straw), followed by an S to denote plots in the spray irrigation area, or a D for the Drip irrigation area. Rows are distributed from top to bottom as 1 to 9.

	AD	BD	CD	DD
1	No planting No mulch	No planting No mulch	No planting No mulch	No planting No mulch
2	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
3	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
4	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
5	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
6	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
7	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
8	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
9	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw

	AS	BS	CS	DS
1	No planting No mulch	No planting No mulch	No planting No mulch	No planting No mulch
2	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
3	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
4	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
5	Gel No mulch	Gel Mesh	Gel Plastic	Gel Straw
6	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
7	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
8	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw
9	Plug No mulch	Plug Mesh	Plug Plastic	Plug Straw

Figure 21: Little Woolden site experimental layout showing the Drip irrigation area (left) and Spray irrigation area (right). Mulch covers are distributed from left to right as Columns A to D (No mulch, Mesh, Plastic, Straw), followed by an S to denote plots in the spray irrigation area, or a D for the Drip irrigation area. Rows are distributed from top to bottom as 1 to 9.

Source of irrigation water

Irrigation water was delivered mainly by collected rainwater or in dry conditions from reverse osmosis from mains water at the Sharpley site. Irrigation water was provided only by collected rainwater at the Little Woolden site, as the surrounding site provided ample water storage, and reverse osmosis was deemed logistically and financially unviable for Little Woolden.

7.3.3 Mulch cover selection

Three mulch cover treatments used in the earlier pilots were applied to the *Sphagnum* plots for both plugs and gel. The cover applications consisted of a synthetic woven mesh ‘Wondermesh’ (column BD or BS), perforated plastic mulch (column CD or CS) and the standard *Sphagnum* straw mulch (Column DD or DS) as used in the MLTT method (Chapter 4). The selected covers were applied to eight plots per irrigation area. One column (AD or AS) at each irrigation area was planted but given no mulch as a control, while one row in each irrigation area was designated for unplanted, no mulch control plots.

The mesh mulch used consisted of the extra finest insect mesh provided by Wondermesh. The characteristics of this material are a mesh size of 0.3 by 0.7mm, a weight of 120 g per m^2, and the provision of 20% shading. The product is advertised as suitable for protection from hail, heavy rain, wind, birds and a range of insect sizes (aphids, flies, beetles and moths). Wondermesh is made of High-Density Polyethylene (HDPE) and is UV treated which provides a high quality, stable & durable net. The expected lifespan of the product is given as c. 10 years.

The plastic mulch used consisted of a white perforated plastic cover supplied by LBS horticulture and was the micro-perforated plastic option. This material consists of a thin 0.05mm polythene film that has been hot needle perforated to leave holes of 2.0mm diameter, with approximately 10,000 holes/m^2. The claimed benefits of this plastic mulch are reduced evaporation, increased lifespan of the material, and reduced algal growth. Factors which should improve crop hygiene, without impairing water availability.

The Straw mulch was applied at a rate equivalent to 3000kg/Ha field dry straw. Or 0.3kg per m^2. This is the same application rate of straw recommended for use in the MLTT as set out by Quinty and Rochefort (2003). Which has been shown to be a key component of peatland restoration (along with *Sphagnum* introduction and water management). At this application rate the greatest

Sphagnum percentage covers have been achieved in Canadian peatland restoration. Too much straw can be detrimental to plant establishment, while too little straw cannot provide adequate protection for establishing *Sphagnum* fragments and can cause failure of plant establishment. However, a disadvantage of straw, is that it can account for a large part of the cost of restoration, it will degrade over time, and can add 'new' carbon into the peat equation which may liberate CO_2 via increased microbial activity at the peat surface.

7.3.4 Control plots

One column (Column A or AS) for each irrigation area was planted but given no mulch as a mulch control. In addition, one row in each irrigation area (Row 9 spray and Row 1 drip at Sharpley, and Rows 1 at Little Woolden) remained unplanted with no mulch as a full control – or at least as much of a full control as was possible within the constraints imposed by the partnership-defined layout.

7.3.5 Fixed peat anchors

Fixed peat anchors were installed to act as fixed markers for terrestrial laser scanning (TLS). Fixed peat anchors, constructed from 8 mm stainless steel rod, were embedded into the underlying mineral ground at the four corners of each irrigation area. Smaller 6 mm stainless steel rods were installed in similar fashion across the *Sphagnum* planting areas. In both cases, a washer was fixed flush with the peat surface to accurately record the surface level immediately after *Sphagnum* application.

7.4: Partnerships involved - and associated constraints

Several partners were involved in the design and delivery of the site:

- Micropropagation Services Ltd. (MPS) provided the *Sphagnum* founder material, and delivered overall project management, site development and technical plans.
- The Stanley family and the Lancashire Wildlife Trust provided access to land at the Sharpley, and Little Woolden sites respectively.
- Manchester Metropolitan University were an academic partner who delivered the GHG monitoring, water, and soil chemistry analysis, MMU also installed dipwells and weather stations.
- UEL were involved in monitoring *Sphagnum* growth, monitoring PWP, and installing automatic water level recorders.
- Melcourt Industries provided professional horticulture insight and experimented with novel growing media using dried micropropagated *Sphagnum* produced in the MPS greenhouse.

While partnership working has many advantages including shared ideas, expertise, and support, it also comes with practical constraints resulting from the need to negotiate and come to compromise solutions which introduce limitations on the experimental design.

Firstly, the plot layout was beyond my control as the researcher. An ideal study would have higher levels of replication and randomisation within each irrigation area to increase the study accuracy and minimise any confounding treatment effects. At both sites, the maximum replication provided for each founder material and mulch cover pairing is limited to four. Mulch covers were not randomised across each irrigation area, with the same layout maintained across both irrigation areas. This potentially introduces biases due to distance from irrigation input as well as edge effects. However, the same plot layout offered the advantage of increasing practical aspects such as removing and replacing cover materials for measurement and identifying plots as the study went on.

For true statistical validity, and the avoidance of any pseudo-replication risk, multiple independent experimental areas would need to have been established across each site (Hurlbert, 1984). This would have involved creating multiple hydrologically independent replicates of drip and spray irrigation areas at each site. However, the resources in terms of cost would have made this prohibitive within the grant funding awarded.

It has been argued that an undue concern about potential pseudo-replication can be damaging to ecological studies, as a pedantic approach is likely to ‘slow the pace of ecological research and limit the scope of management case studies, natural events studies, and reduce the dissemination of valuable data available to form evidence-based solutions’ (Davies and Gray, 2015). As the project was aiming to provide proof of concept for the MIFA approach to *Sphagnum* farming using top-down irrigation under real world conditions on peat, proof of concept was deemed an important step forward in terms of generating exploratory understanding in the absence of an ideal study design at this stage.

Secondly, at each site, the plug and gel plots were established at different times. Some plots therefore experienced different weather conditions during their establishment phase. This additional variation at time of establishment could make it difficult to unpick genuine differences introduced by irrigation, founder material or mulch cover treatments when comparing plug and gel plots against each other. The adoption of standardising yield comparisons based on annual figures were one way of potentially overcoming this. This has been used in other SF studies (Gaudig *et al.*, 2014), to divide total yield via the number of years to obtain an annual figure of dry mass (t ha-1 yr-1) for comparisons across different sites, species, management and hydrological regimes.

Thirdly, if direct comparison is to be made between the benefits of the micropropagated founder material compared to donor-site founder material, it would have been beneficial to compare donor material plots against micropropagated *Sphagnum*. However, this rather defeats the MIFA purpose of using micropropagated material in order to avoid impacting natural donor sites. If such a comparison were subsequently considered necessary, it could be more effectively run as a mesocosm study using small amounts of material.

7.5 Experimental hypotheses

The **Core Hypotheses** for this main study are:

- **Core Hypothesis 1**: The water table at both sites will be kept close to the peat surface with top-down irrigation in combination with natural rainfall.
- **Core Hypothesis 2**: Top-down irrigation will prevent critical *Sphagnum* PWP thresholds being exceeded at both experimental sites
- **Core Hypothesis 3**: Alternative mulches will result in greater growth than the traditional straw mulch. Additionally mulch treatments will provide greater growth than the no mulch plots.
- **Core Hypothesis 4**: the growth of *Sphagnum* plugs will exceed that of *Sphagnum* gel.
- **Core Hypothesis 5**: The *Sphagnum* yield produced by top-down irrigation systems (measured in tonnes of dry biomass per ha) will be comparable to previously published data from *Sphagnum* farming sites that have used donor *Sphagnum* for founder material.

Chapter 8. Hydrological investigations

8.1 Underpinning thinking/ hypotheses

The water relations of this project concern two areas of *Sphagnum* water relations:

- Pore water pressure (PWP), which represents soil moisture availability held within the pore spaces found in peat soils; and
- Height of the *Sphagnum* carpet above the water table, which is the anaerobic, water saturated level below the peat surface.

8.1.1 Hydrology in natural systems

PWP and water table are closely linked, primarily this is because a conflict of approaches surrounds hydrological variables within the peatland literature. Hydrological studies primarily utilise variables such as hydraulic head and PWP; whereas ecological studies use 'coarser' variables such as Water table (WT), volumetric water content (VWC) or gravimetric water content (GWC) (Thompson and Waddington, 2008). However, as *Sphagnum* primarily takes up water via capillary action, the extent to which *Sphagnum* can access water is more ecologically meaningful.

It has long been established that for living *Sphagnum*, dessication is prevented when pore water pressures are between 100 hPa and 600 hPa (Hayward and Clymo, 1982; Lewis, 1988; Schipperges and Rydin, 1998; McCarter and Price, 2014). Beyond this range, the hyaline cells (dead-end cells that retain water in *Sphagnum*) begin to drain, causing the moss to desiccate and cease to photosynthesize.

Exceeding the PWP range is problematic as recovery from desiccation is often poor in *Sphagnum* mosses, due to irreversible cell damage, the rupture of cell membranes, degradation of chlorophyll cells and leakage of cell compounds (Gerdol *et al.*, 1996). Long term desiccation is also fatal to *Sphagnum*, and it cannot survive for more than 14 days of complete desiccation (Sagot and Rochefort, 1996). The PWP range of 100 to 600 hPa is influenced directly by the physical size of the pore openings of the hyaline cells and is a result of the physical adaptations and limitations of the *Sphagnum* plant, therefore these PWP values indicate the biological limit of *Sphagnum* to PWP changes.

A pressure head of -100 cm is equivalent to a PWP of 100 hPa as recorded on the tensiometers used in this PhD study. This 100 hPa reflects the lower established boundary for *Sphagnum* desiccation as reported in the seminal papers of Hayward & Clymo, 1982 and Lewis, 1988. Other key studies that have utilised the 100 hPa threshold include Mcarter and Price (2014) and Gauthier, McCarter and Price, (2018).

The importance of using the lower part of the threshold is that when *Sphagnum* is establishing on a bare peat surface, the growth form typically results in increased moss height at the expense of high bulk density (Turetsky *et al.*, 2008; Waddington, Lucchese and Duval, 2011). As a result, newly established moss layers are looser, with a higher proportion of large diameter pores (McCarter & Price, 2014), making them more vulnerable to changes in water availability within the unsaturated zone in the upper few cm's of the moss carpet. Denser, established moss carpets have fewer large pore spaces so are less vulnerable with time.

Due to larger pore spaces between *Sphagnum* individuals it was expected that the newly created *Sphagnum* plots would have a lower dry bulk density (DBD), lower water retention capacity, and a higher risk of desiccation compared to more established *Sphagnum* carpets. The prevention of desiccation is highly desired to maximise productivity, therefore applying the conservative, lower PWP threshold of 100 hPa as the threshold for success of the irrigation regime was chosen. As an optimised growing system would seek to minimise the loss of water and hence a reduction in photosynthesis and growth at the *Sphagnum* capitula.

The higher threshold of 600 hPa may have been appropriate if the *Sphagnum* had been purposefully compressed to increase water retention, had a very dense growth form, or the tensiometers had been buried deeper. As all these aspects could increase the relative bulk density of the *Sphagnum* crop and decrease the pore size between individual mosses. However, burying the tensiometers deeper into the developing *Sphagnum* crop doesn't inform us about what is happening at a capitula level, and the bulk density of a developing crop Is problematic to acquire due to the destructive nature of the testing.

8.1.2 Top-down irrigation vs a natural system

These two hydrological features are important to monitor because although water table is closely correlated with peatland condition in a natural system, water table performance may be less influential compared to irrigation in terms of PWP when water is applied directly on a bare peat or *Sphagnum* surface.

The use of top-down irrigation is highly novel and could enable the expansion of *Sphagnum* farming onto marginal peat sites. This is urgently required, as the impact is great - across the c.325,000 ha of lowland peatland remaining in the UK, 60% is designated as cropland (Trenbirth and Dutton, 2019). Conventional drainage-based agriculture is damaging to peat soils (Chapter 1), making Cropland on peat inherently unsustainable in the long term.

The drainage required to make peat soils productive under conventional agriculture has had devastating effects, as the vast majority of cropland on peat

(132,100 ha) is classified as 'wasted'. Peat is classified as wasted when the peat layer is < 40 cm deep. A smaller area of cropland (50,600 ha) is found on deep peat (Trenbirth and Dutton, 2019) , but the long-term trajectory is inevitable. Without a change in farming practice, this will become wasted peat. This must change. Expanding paludiculture and *Sphagnum* farming onto former agriculturalised soils in England can play a role in this transition.

The top-down irrigation method may offer opportunities for wider uptake as it utilises technology that is both commercially available and familiar to farmers. Furthermore top-down irrigation can be turned off quickly. The ability to reduce water on site could increase harvesting options, reduce surface flooding, and allow the re-use of equipment and conversion to another land use more readily than surface irrigation. Finally, the direct delivery of irrigation water to the growing *Sphagnum* layer may reduce drought stress and increase yields compared to typical surface-based irrigation studies.

To date, no *Sphagnum* farming studies have investigated the MIFA approach of top-down irrigation at scale, in the field. The commercial partners MPS, utilise top-down irrigation to cultivate *Sphagnum* in a greenhouse setting, but it has not been attempted on peat soils 'in the field'. To assess the hydrological implications of this approach for *Sphagnum* growth, four hypotheses focused specifically on hydrological behaviour were formulated:

Hydrological hypothesis 1:

Top-down irrigation is an effective mechanism for maintaining the water table at a level between 5 to 10 cm below the soil surface. A water table -10 cm from the surface has been identified as a key factor for success in other *Sphagnum* farming sites (Gaudig *et al.*, 2014).

Hydrological hypothesis 2:

Top-down irrigation will prevent PWP from exceeding 100 hPa for >90% of the experimental time. Water will be applied at the capitula level and is likely to prevent a high proportion of days where *Sphagnum* faces desiccation.

Hydrological hypothesis 3:

Plug plots will have greater desiccation resistance (experience fewer days where PWP is > 100 hPa during establishment compared to gel plots. This is likely as plug plants are more established, while gel fragments are smaller and more dispersed when initially applied.

Hydrological hypothesis 4:

The Little Woolden site will be more hydrologically suited to *Sphagnum* farming and will experience a greater frequency of high-water table conditions, as well as fewer days exceeding the 100 hPa threshold for *Sphagnum*, than the Sharpley site. This is expected due to a greater peat depth and higher annual rainfall.

8.2 Hydrological monitoring methods

A full description of the layout for the experimental treatments has been provided in Chapter 7. The present section is concerned with methods used to obtain hydrological data.

Hypothesis 1 and Hypothesis 4 are related to water table position relative to the peat surface. Peat water table position was recorded at the centre of each irrigation area. The loggers used were Aqua Troll 500 vented water level loggers, which feature automatic barometric compensation, making them stand alone units to balance efficiency and cost. These were installed in the centre of each irrigation system area, at both the Sharpley and Little Woolden sites. The

Aqua troll loggers automatically record the water table level at a predetermined frequency, store thousands of data points and have a long battery life. A recording interval of every 15 minutes was selected as the best balance of recording power, battery life, and memory use.

The Aqua Troll loggers were housed within a length of perforated plastic pipe having a porous drain cap attached at the base, the whole assembly then being sunk into the substrate until only a small lip of pipe protruded above the ground surface (see Figure 22). The perforations and porous drain cap allowed uninterrupted flow of groundwater around each logger.

The perforated pipes to house the Aqua Troll dataloggers were installed to 1 m depth below the surface at the Little Woolden site, as the peat depth was great enough to allow this. This depth was chosen as it was expected that the water table variation would not exceed a depth greater than 1 m from the peat surface when irrigation methods were active.

Figure 22: Schematic of the Aqua Troll 500 level loggers used. Note 3 cm of protruding down pipe above the surface. This allows the logger to be cable tied and fixed into position. A porous drain cap was fixed to the end of the pipe to enable unimpeded water flow through the peat soil.

The Little Woolden site also benefited from additional manual dipwells which were monitored regularly by MMU (Chapter 7). Five were installed per irrigation area and monitored approximately monthly. However, dipwells were not installed at the Sharpley site due to logistical and physical limitations of shallow peat and underlying rocky geology.

At the Sharpley site, because the peat depth was shallow and the underlying mineral base was a granitic-clay mix, the perforated plastic pipes housing the Aqua Troll loggers could not be installed to a depth of 1 m as they were at Little Woolden. They were installed at different depths below the peat surface in the two irrigation areas, specifically to a depth of 43 cm in the drip irrigation area and at 29 cm within the are subject to spray irrigation. The consequence of this much shallower installation of the Aquatroll logger is that that when the water table fell below the base of the pipe, water table depth was not recorded.

At all sites, approximately 3 cm of perforated plastic pipe remained above the surface. This provided space to secure the Aqua Troll logger in position at the surface using cable ties. This protruding portion of pipe was judged enough to prevent peat or debris from falling into, or being washed into, the pipe, which could then have disrupted measurements.

Water table data were downloaded in the field using the manufacturer’s Bluetooth communications device and VuSitu mobile phone app. This meant that there was no need to physically remove the datalogger from the perforated pipe when downloading the stored data. This limited potential error caused by movements in water logger position during data collection. Water table data were physically stored on the mobile phone and emailed to the researcher’s account to provide constant data backup.

8.2.1 Water table data analysis methodology

Water table data were received in .csv files and subsequently converted into an Excel spreadsheet. Initial data analysis used pivot-table tools within Excel to group data gathered in 15-minute intervals into daily mean, maximum and minimum positions.

8.2.2 Pore water pressure methodology

Prior to any tensiometer installation on site, the tensiometers were evaluated in the lab, cleaned, and refilled with degassed distilled water, as per manufacturer instruction. Data loggers were set up and checked for proper function prior to deployment in order to ensure operational effectiveness in the field.

On deployment in the field, the tensiometers were fixed in a vertical position using cable ties attached to thin metal rods inserted into the peat layer beneath. Fixing tensiometers in a vertical position prevented air bubbles reaching the pressure transducer within the tensiometer, a phenomenon that can occur during dry periods if the tensiometer is installed at an angle. Air bubbles in the tensiometer tube can lead to inaccurate PWP readings as they prevent the internal PWP in the tensiometer from equalising with the pore water in the surrounding substrate.

A tensiometer ‘system’ consists of a datalogger, and eight tensiometers. Across the entire experiment, six full ‘systems’ were used: System A and B, purchased in the earlier pilot studies, and System 1, 2, 3 and 4 purchased when the main study began.

8.2.3 A phased approach

The Tensiometer experiments were divided into two phases at each *Sphagnum* farming site. Phase 1 involved inserting the tensiometer bulbs into the peat soil at a depth of 2 cm to record PWP in the upper peat surface. This was necessary as the newly planted *Sphagnum* layer was not deep enough to accommodate the tensiometer bulb entirely.

Phase 2 involved redeploying the tensiometers with the tensiometer bulbs placed directly into the *Sphagnum* layer at a depth of 2 cm once a suitable depth of *Sphagnum* had been reached. This ensured that the impact of top-down irrigation on PWP within the living *Sphagnum* could be monitored.

8.2.4 Sharpley – Phase 1 logger locations

At the Sharpley site, Phase 1 ran from the 22/08/2018 to 16/04/2019. Plugs were planted on the 9 - 17 August 2018 while the gel was applied on 3rd September 2018. Gel was also reapplied on selected plots on the 11th May 2019. For a full site timeline, see Appendix A3.

Tenisometers were located as shown in Table 13 and Table 14:

Table 13: Phase 1: Tensiometer locations, Sharpley spray irrigation area. Data recorded on data logger system A, from 22/08/2018 to 16/04/2019.

Data logger System A	Sharpley Spray irrigation area		
Tensiometer	Tensiometer Plot location	Cover treatment	Founder material:
T0	AS4	No mulch	Plug
T1	AS4	No mulch	Plug
T2	BS4	Mesh	Plug
T3	BS4	Plastic	Plug
T4	CS4	Plastic	Plug
T5	DS4	Straw	Plug
T6	BS6	No mulch, unplanted during Phase 1	none
T7	CS6	No mulch, unplanted during Phase 1	none

Table 14: Phase 1: Tensiometer locations and cover treatments, Sharpley Drip irrigation area. Data recorded by data logger system B, from 22/08/2018 to 16/04/2019.

Data logger System B	Sharpley Drip irrigation area		
Tensiometer	Tensiometer plot location	Cover treatment	Founder material:
T0	AD5	no mulch	Plug
T1	AD5	no mulch	Plug
T2	BD5	mesh	Plug
T3	CD5	plastic	Plug
T4	CD5	plastic	Plug
T5	DD5	straw	Plug
T6	BD7	No mulch, unplanted	none
T7	CD7	No mulch, unplanted	none

8.2.5 Sharpley – Phase 2 logger locations

Phase 2 ran from 12/05/2019 to 06/11/2019. Additional tensiometer equipment was purchased, expanding the monitoring programme. System 3 and System 4 were deployed at Sharpley. The System A logger set used during Phase 1 was repurposed to monitor the unplanted, no mulch plots which acted as controls within the irrigation areas at Sharpley. System B encountered issues and was removed for repair before deployment at the Little Woolden site. The tensiometer plot locations are presented in Table 15, Table 16 and Table 17.

Table 15: Phase 2 tensiometer locations, Spray irrigation area. Data recorded by data logger system 3 from 12/05/2019 – 6/11/2019.

Data logger System 3	Sharpley, Spray irrigation area		
Tensiometer	Tensiometer Location	Cover treatment	Founder material
T0	AS4	No mulch	Plug
T1	BS4	Mesh	Plug
T2	CS4	Plastic	Plug
T3	DS4	Straw	Plug
T4	AS5	No mulch	Gel
T5	BS5	Mesh	Gel
T6	CS5	Plastic	Gel
T7	DS5	Straw	Gel

Table 16: Phase 2 tensiometer locations, Drip irrigation area. Data recorded by data logger system 4 from 12/05/2019 – 6/11/2019.

Data logger System 4	Sharpley, Drip irrigation area		
Tensiometer	Tensiometer location	Cover treatment	Founder material
T0	AD5	No mulch	Plug
T1	BD5	Mesh	Plug
T2	CD5	Plastic	Plug
T3	DD5	Straw	Plug
T4	AD6	No mulch	Gel
T5	BD6	Mesh	Gel
T6	CD6	Plastic	Gel
T7	DD6	Straw	Gel

Table 17: Phase 2 tensiometer locations, unplanted, no mulch controls – Both irrigation areas. Data recorded by data logger system A from 12/05/2019 – 6/11/2019.

Data logger System A repurposed for 'Sharpley bare'	Deployed across both irrigation areas	
Tensiometer	Tensiometer location	Irrigation area
T0	AS9	Spray
T1	BS9	Spray
T2	CS9	Spray
T3	DS9	Spray
T4	AD9	Drip
T5	BD9	Drip
T6	CD9	Drip
T7	DD9	Drip

8.2.6 Little Woolden – Phase 1 logger locations

Phase 1 ran from 4/12/2018 to 17/04/2019. Plugs were planted during late October 2018. Gel was applied to a single row of plots, Row 5, covering plots AM5-DM5 during November 2018. A data logger and tensiometer set was deployed to capture hydrological performance during this phase.

Table 18: Phase 1: Tensiometer locations and cover treatments, Little Woolden Drip irrigation area. Data recorded by data logger system 1, from 4/12/2018 to 17/04/2019.

Data logger System 1	LW, Drip irrigation area. Phase 1		
Tensiometer	Tensiometer Location	Cover treatment	Founder material:
T0	AD6	No mulch	Plug
T1	AD8	No mulch	Plug
T2	BD7	Mesh	Plug
T3	BD8	Mesh	Plug
T4	CD7	Plastic	Plug
T5	CD9	Plastic	Plug
T6	DD8	Straw	Plug
T7	DD9	Straw	Plug

Table 19: Phase 1: Tensiometer locations and cover treatments, Little Woolden Spray Irrigation area. Data recorded by data logger system 2, from 4/12/2018 to 17/04/2019.

Data logger System 2	LW, Spray irrigation area. Phase 1		
Tensiometer	Tensiometer Location	Cover treatment	Founder material:
T0	AS8	No mulch	Plug
T1	AS9	No mulch	Plug
T2	BS6	Mesh	Plug
T3	BS9	Mesh	Plug
T4	CS6	Plastic	Plug
T5	CS7	Plastic	Plug
T6	CS5	Plastic	Gel
T7	DS5	Straw	Gel

8.2.7 Little Woolden – Phase 2 logger locations

Phase 2 at Little Woolden ran from 17/04/2019 to 06/11/2019, with further gel also being applied on the 17th April 2019 to the remaining gel plots of Rows 2,3 and 4. In Phase 2, the plug plots had grown sufficiently for the tensiometers to be inserted directly into all plugs. Tensiometers were also installed directly into the developing *Sphagnum* gel carpet of Row 5 in the spray irrigation area. This phase aimed to investigate the water supply directly within the *Sphagnum* rather than the water relations of the peat substrate, although to investigate the newly applied gel plots on both irrigation areas, tensiometers were necessarily installed directly into the peat substrate.

Table 20 Phase 2: Tensiometer locations and cover treatments, Little Woolden Drip Irrigation area. Data recorded by System 1, from 17/04/2019 to 5/11/2019.

Data logger System 1	LW, Drip irrigation area. Phase 2		
Tensiometer	Tensiometer Location	Cover treatment	Founder material:
T0	AD5	No mulch	Gel
T1	BD5	Mesh	Gel
T2	CD5	Plastic	Gel
T3	DD5	Straw	Gel
T4	AD6	No mulch	Plug
T5	BD6	Mesh	Plug
T6	CD6	Plastic	Plug
T7	DD6	Straw	Plug

Table 21: Tensiometer locations and cover treatments, Little Woolden Drip Irrigation area. Data recorded by System 2, from 17/04/2019 to 5/11/2019.

Data logger System 2	LW, Spray irrigation area. Phase 2		
Tensiometer	Tensiometer Location	Cover treatment	Founder
T0	AS5	No mulch	Gel
T1	BS5	Mesh	Gel
T2	CS5	Plastic	Gel
T3	DS5	Straw	Gel
T4	AS6	No mulch	Plug
T5	BS6	Mesh	Plug
T6	CS6	Plastic	Plug
T7	DS6	Straw	Plug

8.3 Results

8.3.1 Water Table: Sharpley site

The water table at the Sharpley site was recorded continuously for the period 1st September 2018 to 1st November 2019, with readings were logged every 15 minutes. The mean daily water table position relative to the peat surface is presented in Figure 23 and Figure 24. Daily rainfall appears to be the major driver of water table behaviour despite the irrigation supply, as the pattern of water table fluctuation mostly, though not always, closely follows that of precipitation inputs.

At both irrigation areas, water logger installation depth dictated the maximum peat water table recorded. The maximum water table depth was therefore -34 cm at the spray irrigation area and -43 cm at the drip irrigation area.

The highest recorded water table differed between the two irrigation types. The highest recorded water table for the spray irrigation plot was -1.18 cm from the surface, whereas in the drip irrigation area the highest water table recording was +6.96 cm, suggesting there was some surface pooling. There was no outflow control at Sharpley, so surface pooling was perhaps to be expected. The data suggest, however, that surface pooling occurred on only five days out of the 442 days monitored, so the impact on the cultivated *Sphagnum* was likely to be minimal.

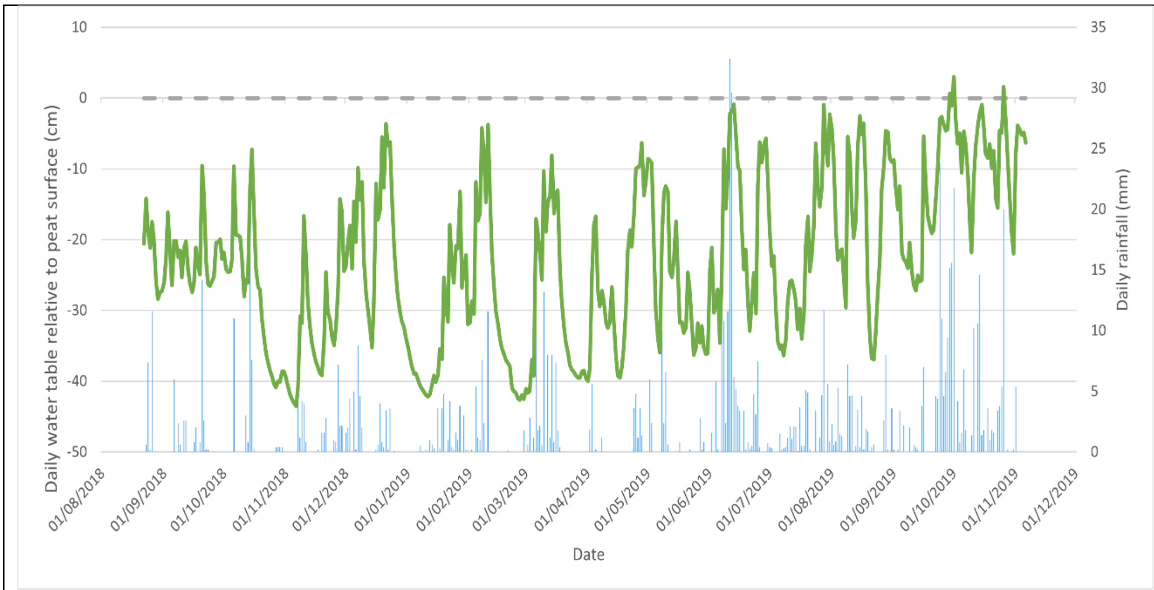


Figure 23: Water Table behaviour at the Sharpley site, Drip irrigation area from 22/08/2018 to 06/11/2019. Line Data represent mean daily water table position relative to the surface (left axis). Bar data represent daily rainfall totals in mm (right axis). The dashed grey line represents the peat surface at 0 cm. Flat areas of data represent periods where the water table is at the lowest limit of the Aqua troll logger (fixed at -43 cm). any data points below this have been removed as at these points the water table is below the peat layer.

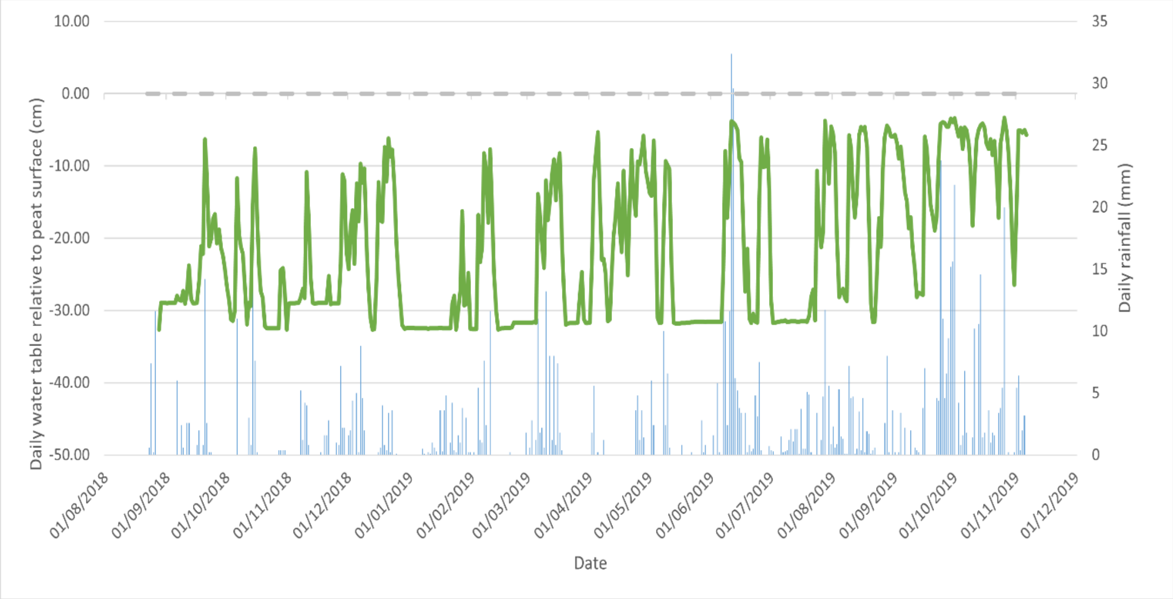


Figure 24; Water Table behaviour Sharpley site for the Spray irrigation area from 22/08/2018 to 06/11/2019. Line Data represent mean daily water position relative to the peat surface (left axis). Bar data represent daily rainfall totals in mm (right axis). The dashed grey line represents the peat surface at 0 cm. Flat areas of data represent periods where the water table recording is reported at the lowest limit of the Aqua troll logger (fixed at -32 cm). WT recordings < -32 cm have been removed as at these points the water table is below the peat layer.

Table 22 and Table 23: show the daily mean water table position (grouped into 10 cm range values) as a percentage of days present in these zones and their relevance for the MIFA approach to *Sphagnum* farming.

Table 22: Shapley Drip irrigation area, Water table position groupings, mean water table levels, 22/08/18 to 06/11/19.

WT range relative to peat surface	Number of days	% time in this range	Impact of this range
greater than 0 cm	3	0.68	not optimal for long periods
0 to -10 cm	80	18.10	optimal range for MLTT
less than – 10 cm but greater than -20 cm	87	19.68	within capillary water range
less than -20 cm but greater than -30 cm	136	30.77	within capillary water range
less than -30 cm	136	30.77	reduced capillary water range

Table 23: Sharpley Spray irrigation area, Water table position groupings, 22/08/18 to 06/11/19

WT range relative to peat surface	Number of days	% time in this range	Impact of this range
greater than 0 cm	0	0.00	not optimal for long periods
0 to -10 cm	90	20.36	optimal range for MLTT
less than – 10 cm but greater than -20 cm	88	19.91	within capillary water range
less than -20 cm but greater than -30 cm	122	27.60	within capillary water range
less than -30 cm	142	32.13	reduced capillary water range

Table 22 and Table 23 demonstrate that the water table position followed a similar pattern in both irrigation areas. The optimal WT range for *Sphagnum* under the MLTT with surface irrigation approach had been identified by Gaudig *et al.*, (2014) as -10 cm. The water table recorded, remained in the optimal 0 to -10 cm range for 18% within the drip irrigation area and 20% in the spray irrigation area. The capillary fringe range at which *Sphagnum* can access water via capillary action is generally regarded as -30 cm and for the remainder of the time the water table very rarely fell below -40 cm, which is another threshold regarded as critical by the MLTT approach to successful *Sphagnum* restoration and *Sphagnum* farming (Quinty and Rochefort, 2003).

Of particular note, is the fact that the lowest water tables and longest relatively dry periods occurred in late winter and early spring of 2019. In contrast, whilst still displaying considerable fluctuations, water tables slowly rose for the whole of the summer period in 2019 and never fell to levels observed earlier in the year.

8.3.2 Pore water pressure: Phase 1 - Sharpley site

During Phase 1 at Sharpley, data for plug plots in both irrigation areas showed that PWP within the surface peat was kept below the threshold of 100 hPa for most of the time as seen in Figure 25 and Figure 26, demonstrating that water availability was rarely a limiting factor for *Sphagnum* growth in the drip irrigation area. Although, this is caveated as the drip irrigation system has less data available following a short circuit issue, it is assumed that it followed a similar pattern during times of data gaps.

For the majority of the time, the spray irrigation area had PWP below 100 hPa, however some tensiometers recorded several days when maximum PWP crossed the 100 hPa threshold, whereas the drip system (System B) showed no such simultaneous peaks during the periods when the tensiometers were operative. The peaks in the spray system amounted to a cumulative total of 17 and 27 days recorded by tensiometers T0 and T1 respectively. However, the greatest run of consecutive days where PWP exceeded 100 hPa was four days, which is fortunately still less than the critical six days of continuous drought for many *Sphagnum* species, as discussed earlier. All periods where PWP exceeded 100 hPa corresponded to periods of lower water table, as revealed by the data shown in Figure 23 and Figure 24.

The relatively abrupt change in tensiometer values within the spray sector, when the critical threshold of 100 hPa was quite suddenly and very considerably exceeded, may point to a technical issue. Interestingly, this same large spike was observed simultaneously in the straw much plot whereas the mesh and plastic showed no such sudden peak. Perhaps the angle of the tensiometer was altered by the action of crows or hares (both being observed to have actively explored or otherwise disturbed the experimental infrastructure during the study). While the continuous mesh and plastic provided effective protection, the no mulch plot and straw-covered plot offered little or no protection from such disturbance.

Although the data from the drip irrigation plots are fragmentary so commenting extensively on them is not possible, the pattern of behaviour appears to show much less variability in the pore water values over extended periods compared with values obtained under spray irrigation.

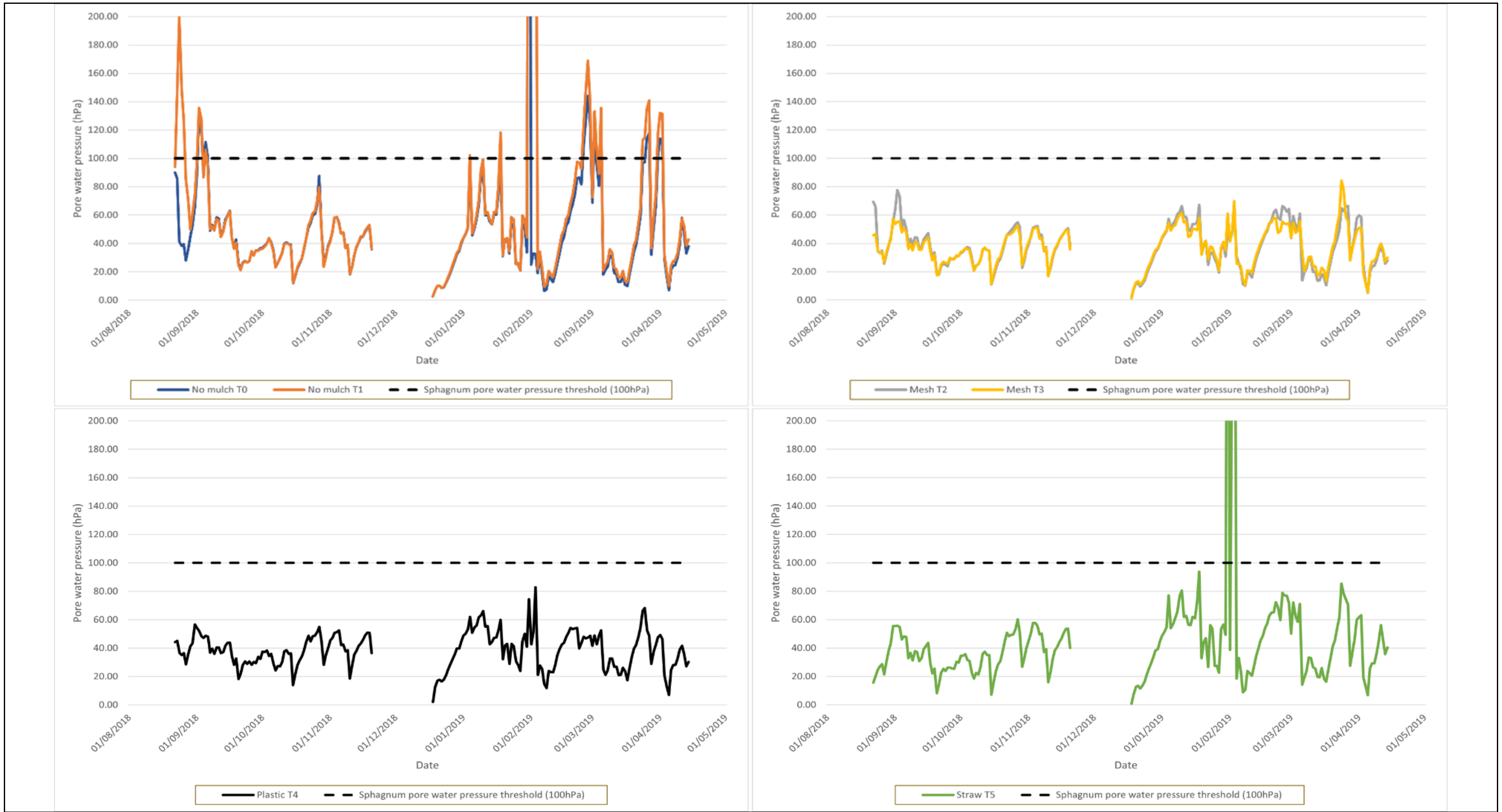


Figure 25: Maximum daily PWP recorded in the Sharpley spray irrigation area, by mulch cover type. Plugs were the only founder material used in Phase 1. Tensiometers were installed in the top 2 cm of peat substrate. Note the data gap at 21/11/18 to 21/12/2018, due to data overwrite on the logger between visits. The peaks of large data spikes are not visible because of axis standardisation for direct comparison across all plots. The black dashed line represents the key pore water threshold of 100 hPa.

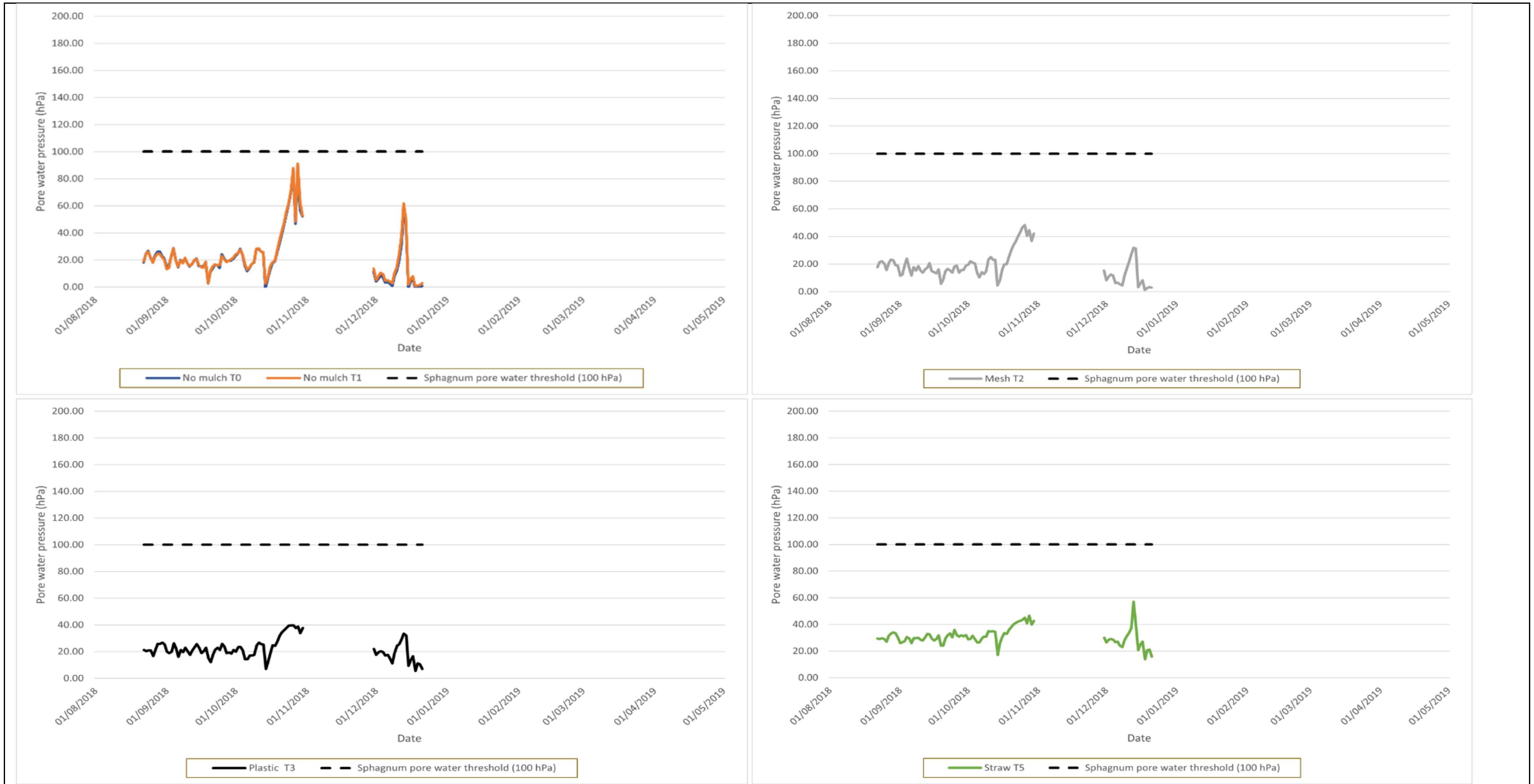


Figure 26: Maximum daily PWP recorded in the Sharpley drip irrigation area, by mulch cover type. Plug plants were the only founder material used in Phase 1. Tensiometers were installed in the top 2 cm of peat substrate. There is a data gap from 01/11/18 to 1/12/18 due to a data overwrite. Tensiometer T4 was damaged by rodent activity so has been removed, this damage caused the datalogger to short circuit - data therefore only runs to the 22nd December 2018. The graph duration and axis are standardised for comparison with the spray irrigation area.

Given the critical nature of the 100 hPa pore water threshold and the critical length of physiological drought that exceeding this threshold represents, the tensiometer data have been summarised in Figure 27 and Figure 28 to indicate the percentage of time over the monitoring period that individual experimental treatments have spent in three defined conditions – namely:

- functioning, or ‘optimal’ pore water pressure (i.e. between 0 and 100 hPa);
- non-functional pore water pressure (i.e. exceeding the 100 hPa critical threshold);
- flooded/inundated (i.e. pore water pressure is negative <0 hPa).

The resulting bar charts provide a convenient and visually standardised view of pore water performance across the various experimental treatments. For Phase 1, plugs appear to benefit from mulch cover during the establishment phase under spray irrigation achieving optimal conditions for 87 to 92% of the time, where covered plots for spray achieved this for 98 to 100% of the time. and that mulch covers may not guarantee the necessary establishment conditions but can achieve them for very high proportions c. 96 to 100%. Otherwise, the MIFA approach appears to provide suitable establishment conditions at Sharpley for both plugs and gel, irrespective of whether irrigation is provided by dripline or overhead spray.

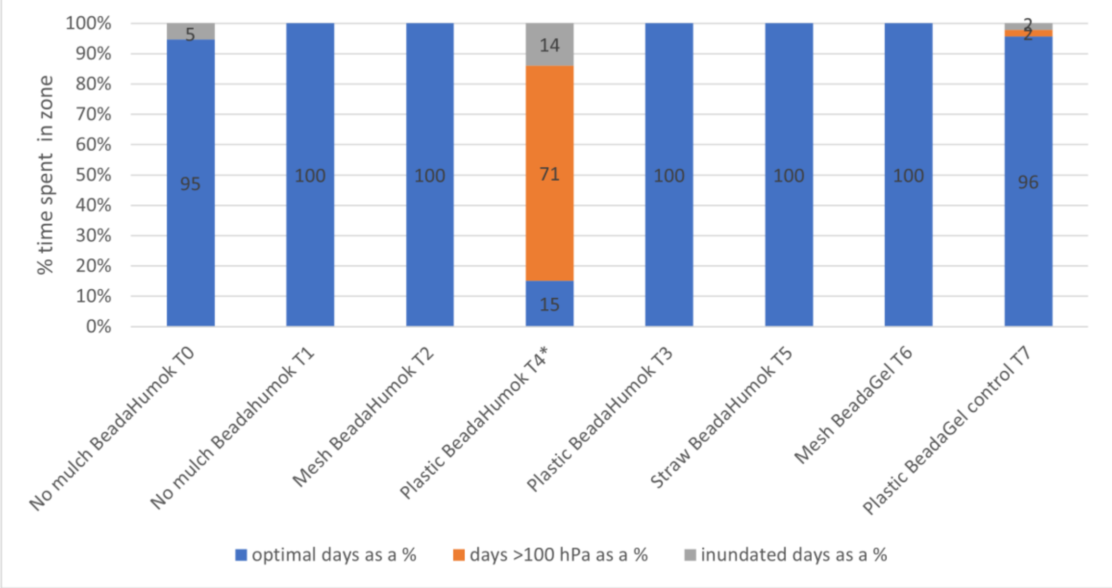


Figure 27: Sharpley Drip irrigation area, Phase 1 monitoring. Bars show time spent in each PWP zone, indicated by colour coding and legend. * T4 is almost certainly a faulty tensiometer.

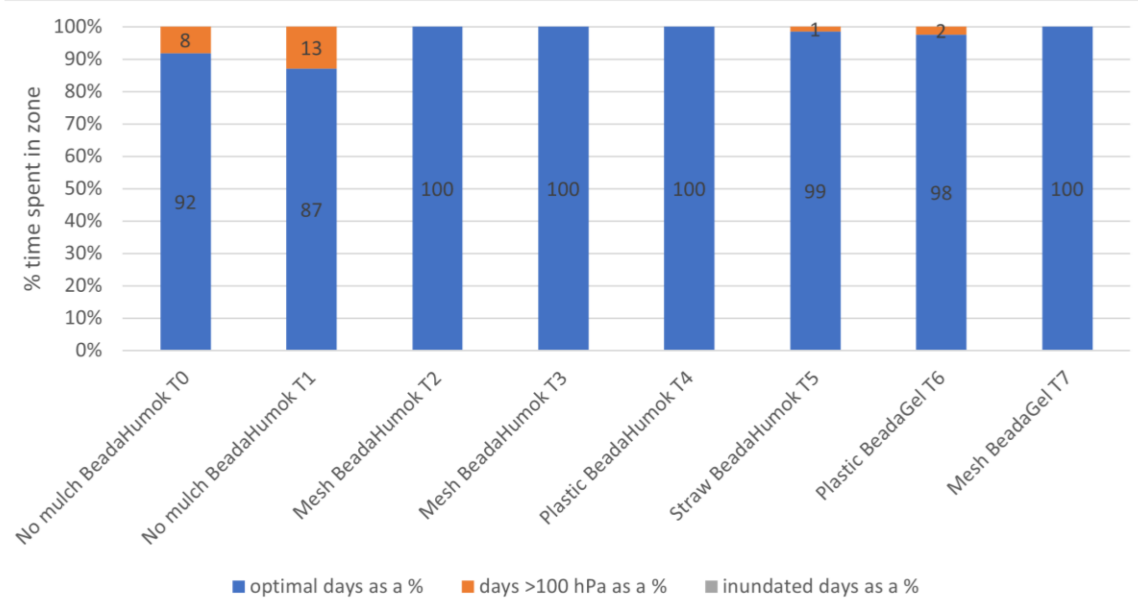


Figure 28: Sharpley spray irrigation area, Phase 1 monitoring. Bars showing time spent in each PWP zone indicated by colour coding and legend.

8.3.3 Pore water pressure: Phase 2 - Sharpley site

There are two important differences between conditions prevailing while Phase 1 monitoring was undertaken and those prevailing during Phase 2:

- the crop had grown significantly by Phase 2 so it was possible to obtain readings from within the *Sphagnum* layer; and
- Phase 2 recording spanned the summer months whereas Phase 1 monitoring occurred during the preceding winter months.

The implications for these points were that larger pore spaces exist between *Sphagnum* individuals than in peat soil (Thompson and Waddington, 2008; McCarter and Price, 2014) therefore when tensiometers are placed in the *Sphagnum* carpet instead of peat more frequent PWP threshold breaches could be expected, until a closed *Sphagnum* carpet was achieved. Also phase 2 taking place in the summer months meant that the irrigation system was operating under drier conditions, and that if PWP could remain under 100 hPa, it would demonstrate that *Sphagnum* could be cultivated under difficult conditions.

Spray irrigation

It is revealing to look first at the pore water pressures observed within the unplanted, no mulch controls under the spray irrigation. This provides contextual soil-water background in terms of the baseline against which the performance of the irrigation and the *Sphagnum* crop can be assessed.

From Figure 29 it can be seen from the unplanted, no mulch control plots that even without any mulch cover or vegetation cover, the irrigation system was able to keep pore water pressures below the critical threshold of 100 hPa for much of the late summer and autumn. During May and July of 2019, however, pore water pressures rose well above this threshold, reaching 600-700 hPa at times, with the period in May, for example, lasting some 20 days. July also saw an extended period for some tensiometers where values reached 400-600 hPa for again a duration of around 20 days. These events represent periods when conditions are extremely hostile to *Sphagnum* survival and growth.

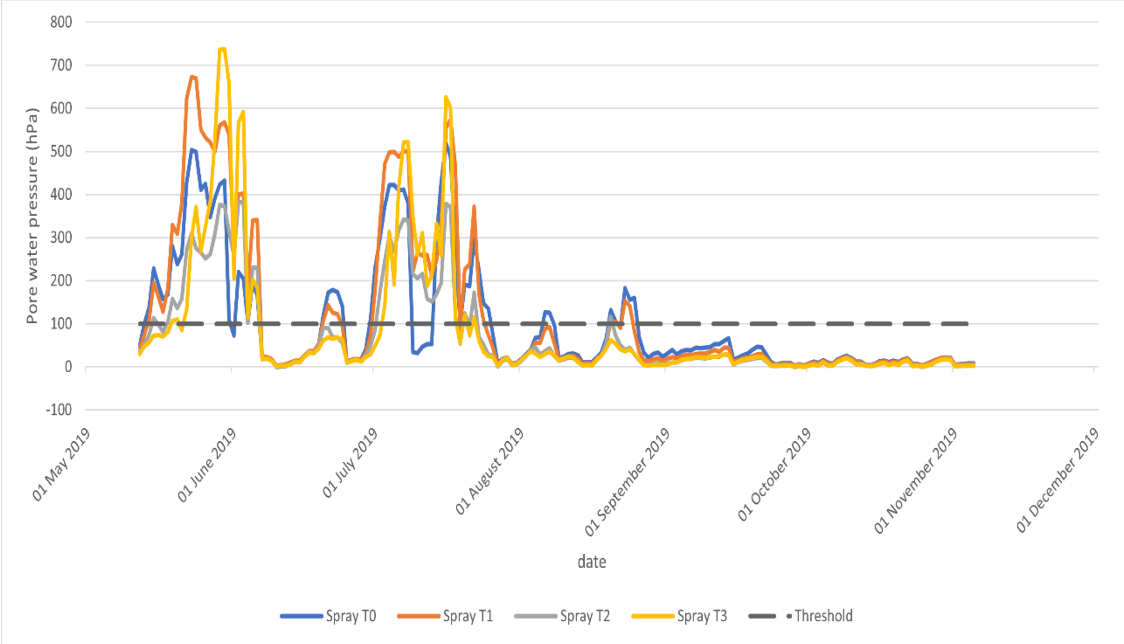


Figure 29: Pore water pressures for unplanted, no mulch controls in the spray irrigation zone during Phase 2 of Sharpley monitoring.

It is instructive, therefore, to consider the pore water pressures obtained from both plugs and gel during these same periods when irrigated by overhead spray. It can be seen from Figure 29 that both the plugs and gel which lacked any form of mulch cover (T0 and T5) showed a very similar response to the harsh conditions of May and July 2019 as seen in the bare unplanted controls. In the case of the *Sphagnum* plugs, straw mulch also responded poorly, with pore water pressures rising above the critical threshold in both May and July, while the mesh also performed rather poorly, similarly crossing the threshold on both occasions if not displaying values quite as high as those seen with the straw mulch. Only the plastic sheeting remained below the critical threshold throughout the summer.

Results from the gel subjected to spray irrigation show that the no mulch, straw, and plastic gel plots all resulted in similar responses as those observed for the plugs, but the mesh gave rise to a dramatically different response during May and July. Pore water pressures rose to between 300 and 400 hPa, and, in the case of July, remained above the critical threshold for some 15 days.

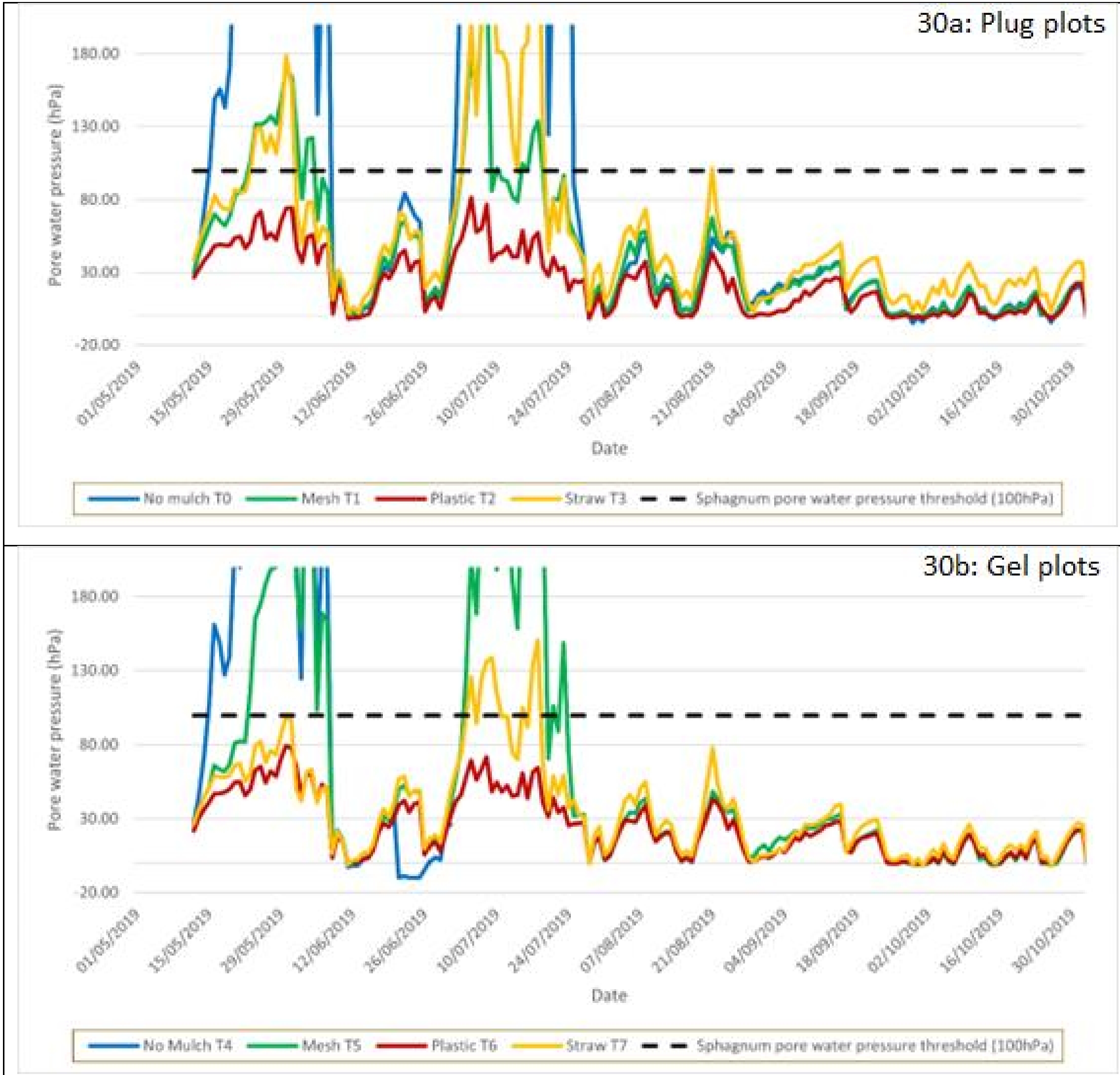


Figure 30a and 30b: Pore water pressures in the Sharpley spray irrigation area for both plugs and gel: Phase 2 experiment where tensiometers are installed directly into established plug and gel surfaces. 30a (Top) Plugs and 30b (Bottom) Gel. Maximum pore water pressures are presented by mulch type. Note tensiometer malfunction (No mulch T4, gel plot). The tensiometer was found to broken upon retrieval; data beyond 1/07/2019 for T4 are invalid.

Drip irrigation

As before, it is helpful to consider the PWP conditions prevailing under a regime of drip irrigation when neither mulch nor a growing crop is present. Figure 31 below, yet again shows a substantial rise in PWP during May, again reaching values of between 500 and 700 hPa, then another, though significantly smaller, peak in July. An additional dramatic peak was recorded by tensiometer T4 in late August, and although all other tensiometers recorded a relatively small rise, none responded as dramatically as tensiometer T4.

Other than these peaks in PWP, the drip irrigation proved capable of maintaining conditions well below the critical threshold of 100 hPa required for *Sphagnum* survival and growth.

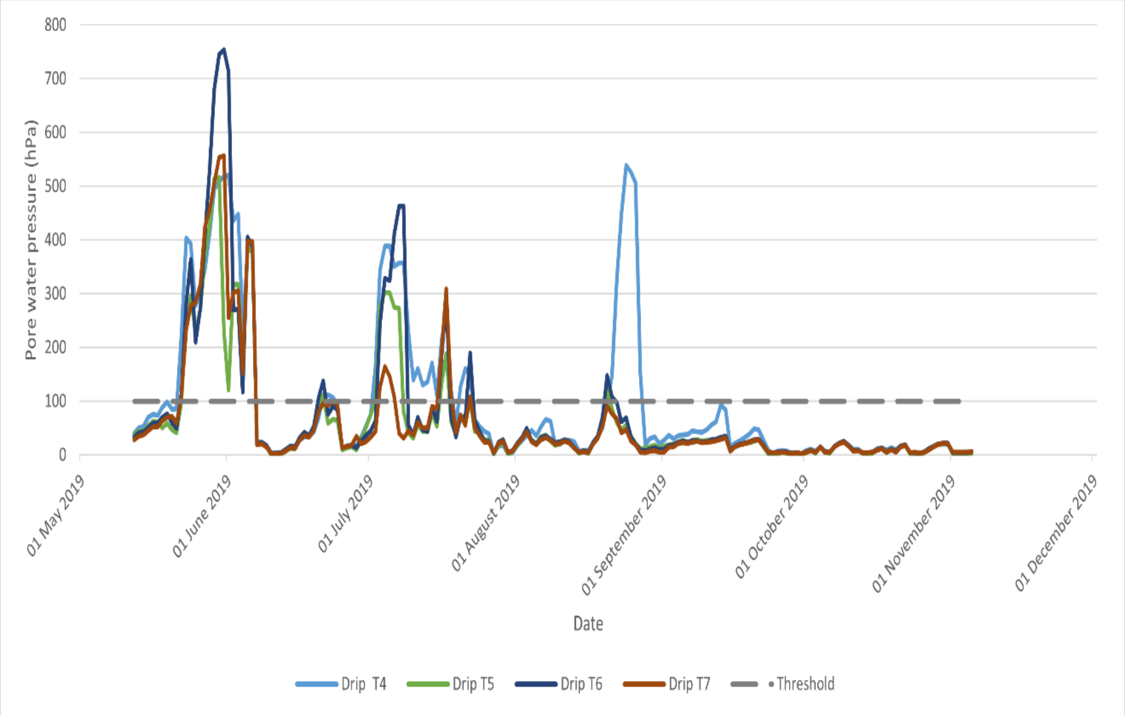
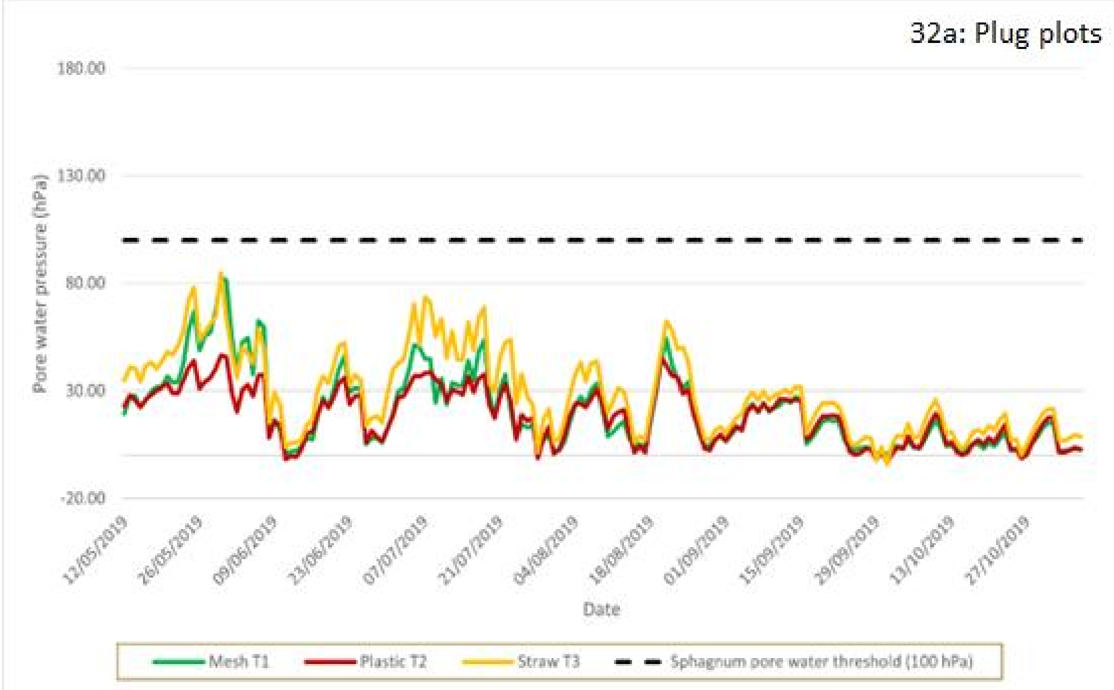


Figure 31: Pore water pressures for unplanted, no mulch controls in the drip irrigation zone during Phase 2 of Sharpley monitoring.

Under drip irrigation, the *Sphagnum* plugs maintained pore water pressures well below the critical threshold of 100 hPa for the whole summer. As shown in Figure 32a on the next page, PWP peaks occurred in May, July and August, but only in May did the PWP approach the critical threshold, reaching around 80 hPa with both mesh and straw mulches, but for the remainder of the summer the values for all mulch types ranged between 0 and 40 hPa.

For the gel, the straw mulch once again proved to be least satisfactory, with pore water pressures exceeding the critical threshold four times as shown in Figure 32b, albeit for only a few days each time. Mesh mulch just reached the 100 hPa threshold briefly in May, but for the much of the summer, mesh, plastic and even straw maintained pore water pressures of between 0 and 40 hPa.



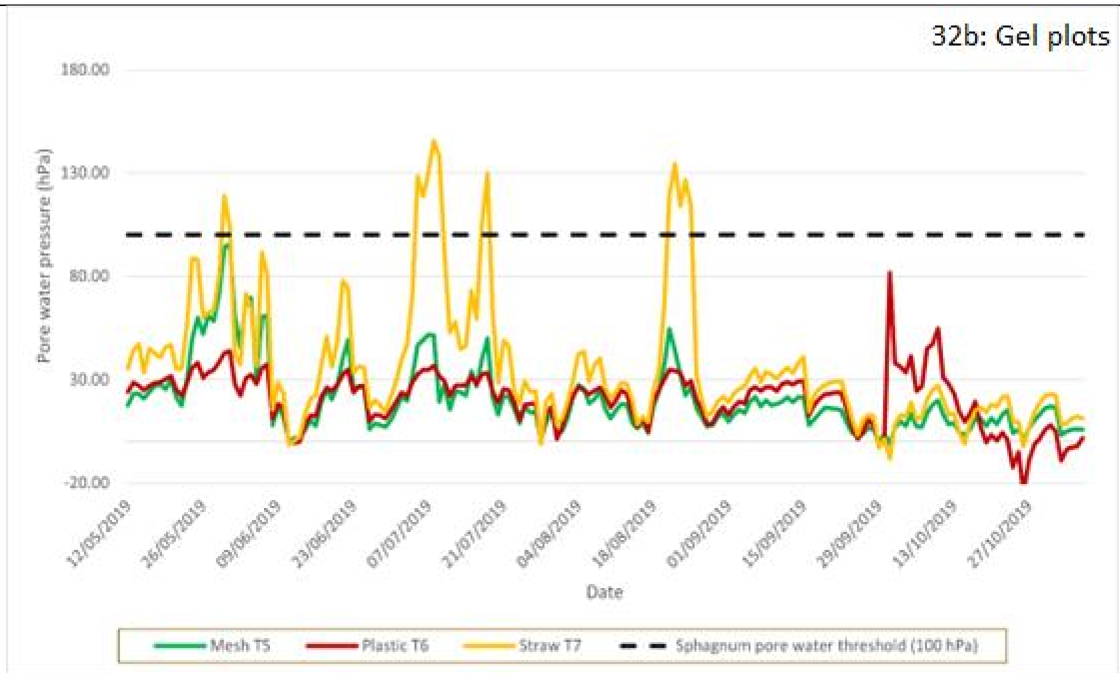


Figure 32a and 32b: Pore water pressures in the Sharpley drip irrigation area for both plugs and gel: Phase 2 experiment where tensiometers are installed directly into established plug and gel surfaces. 30a (Top) Plugs and 30b (Bottom) Gel. Maximum pore water pressures are presented by mulch type.

Figure 29 and Figure 31 show the unplanted, no mulch control plots and both highlight the fact that the complete absence of protective mulch on bare peat resulted in significant periods when the PWP exceeded the critical threshold for *Sphagnum* survival and growth. This is particularly critical for gel fragments as they are smaller and likely to have a poorer recovery during establishment.

Overall performance of the differing mulch types reveals straw and mesh to be potentially less reliable than plastic mulch in terms of maintaining constancy of suitable PWP, though both plastic and mesh may occasionally retain too much moisture, represented by the negative PWP values, which suggest saturation in the *Sphagnum* crop.

At this stage of development, based on the percentage times presented in Figures 33 to 36 drip irrigation provides optimal conditions for higher percentages of time for mesh, plastic and straw plugs at 96 to 99 % of the time, compared to spray irrigation plots with the same mulch covers 87 to 90%. while spray irrigation appears to be preferable for plastic and straw gel plots at 92% to 96% of the time.

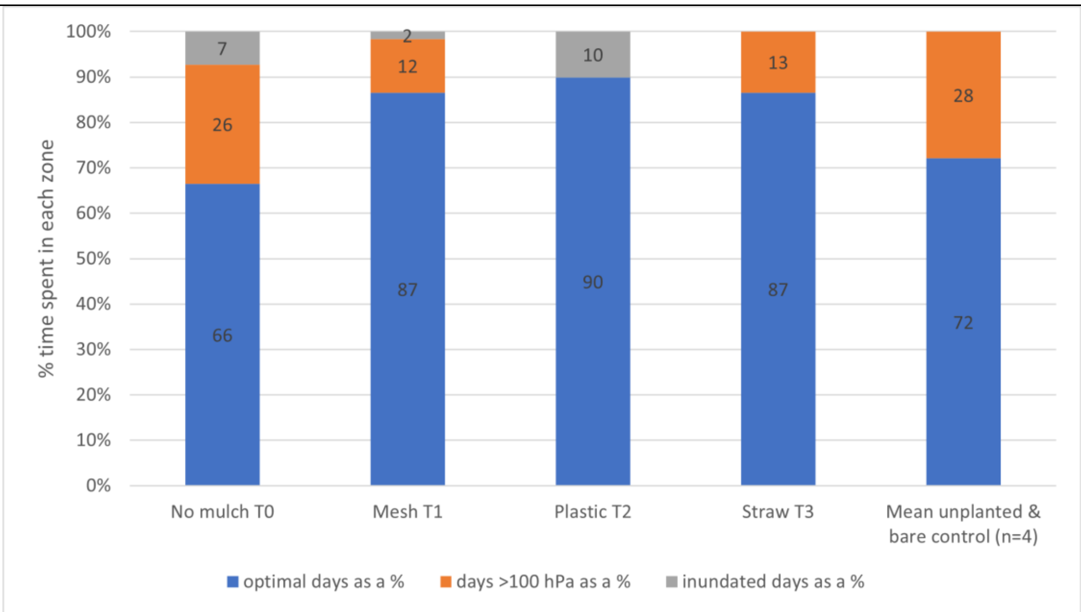


Figure 33: Sharpley spray irrigation area, Plug plots, Phase 2 monitoring. Bars showing time spent in each PWP zone: optimal, too dry >100 hPa or too wet <0 hPa by mulch type.

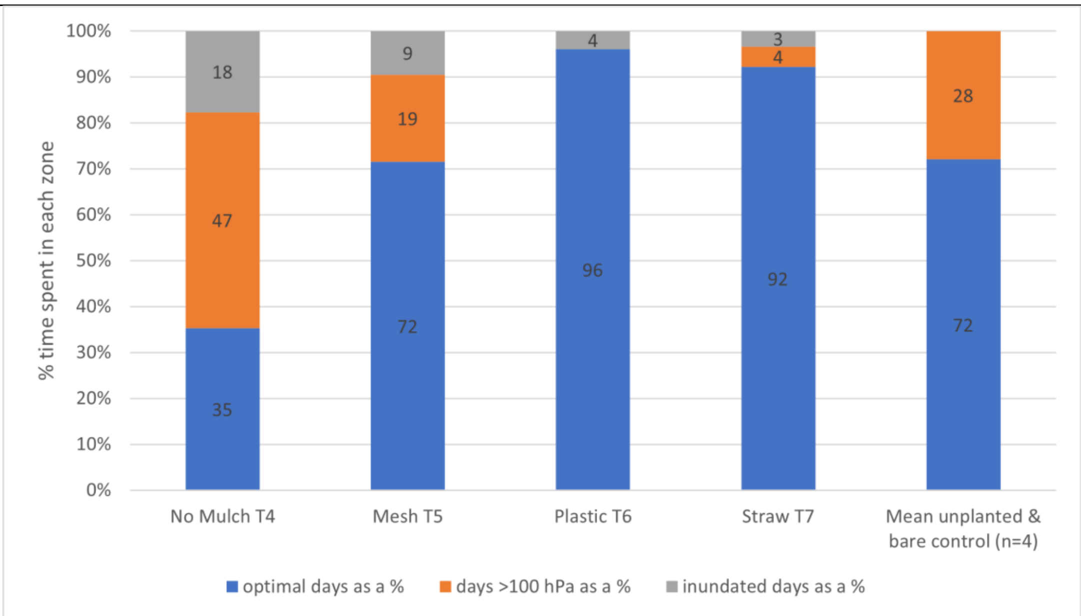


Figure 34: Sharpley spray irrigation area, Gel plots Phase 2 monitoring. Bars showing time spent in each PWP zone: optimal, too dry >100 hPa or too wet <0 hPa by mulch type.

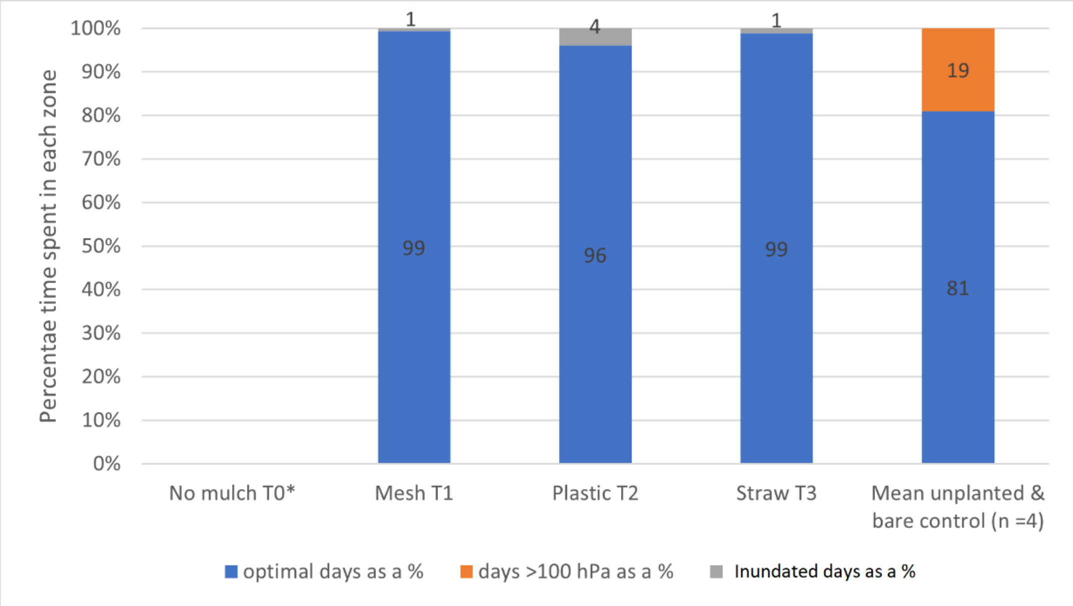


Figure 35: Sharpley Drip irrigation area, Plug plots, Phase 2 monitoring. Graph showing time spent in each PWP zone: optimal, too dry >100 hPa or too wet <0 hPa by mulch type.

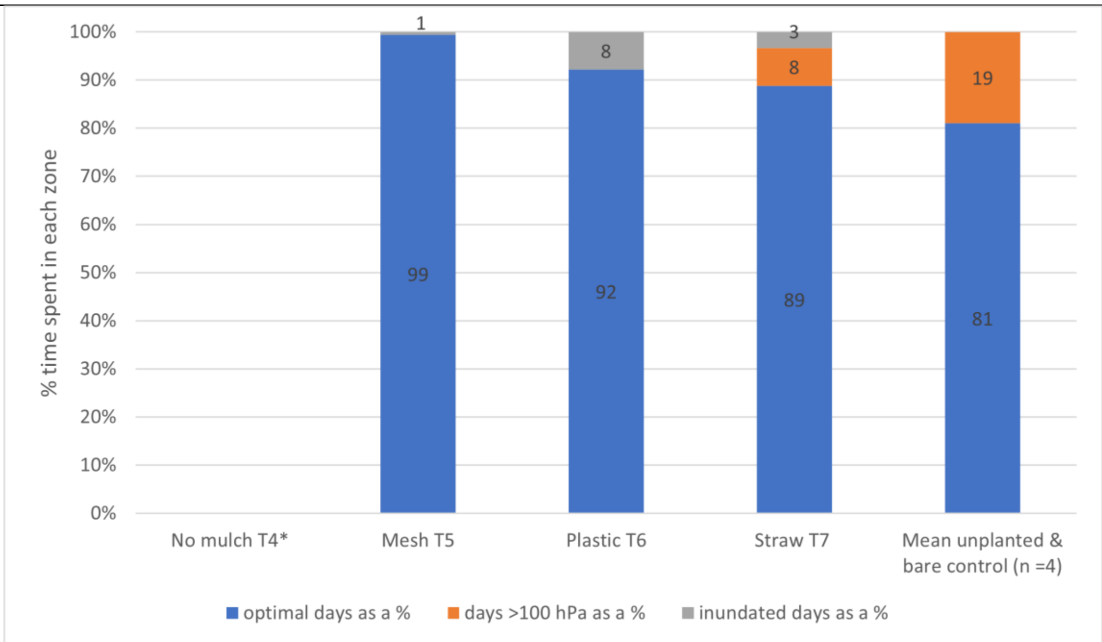


Figure 36: Sharpley Drip irrigation area, Gel plots, Phase 2 monitoring. Graph showing time spent in each PWP zone: optimal, too dry >100 hPa or too wet <0 hPa by mulch type.

8.3.4 Water table: Little Woolden

The water table was recorded at the Little Woolden site continually for the period: 21st November 2018 to 5th September 2019, shown in Figure 37 and Figure 38 on the next page.

In the drip-only area the water table ranged from a maximum depth of -46.35 cm below the surface to a height of +17.08 cm above the surface. In the combined drip-and-spray area the water table ranged from a maximum depth of -42.21 cm below the surface to a height of +24.99 cm above the surface. As at Sharpley, the water table appeared to be largely influenced by daily rainfall, closely following the rainfall pattern for much or the time, as shown in Figure 37 and Figure 38.

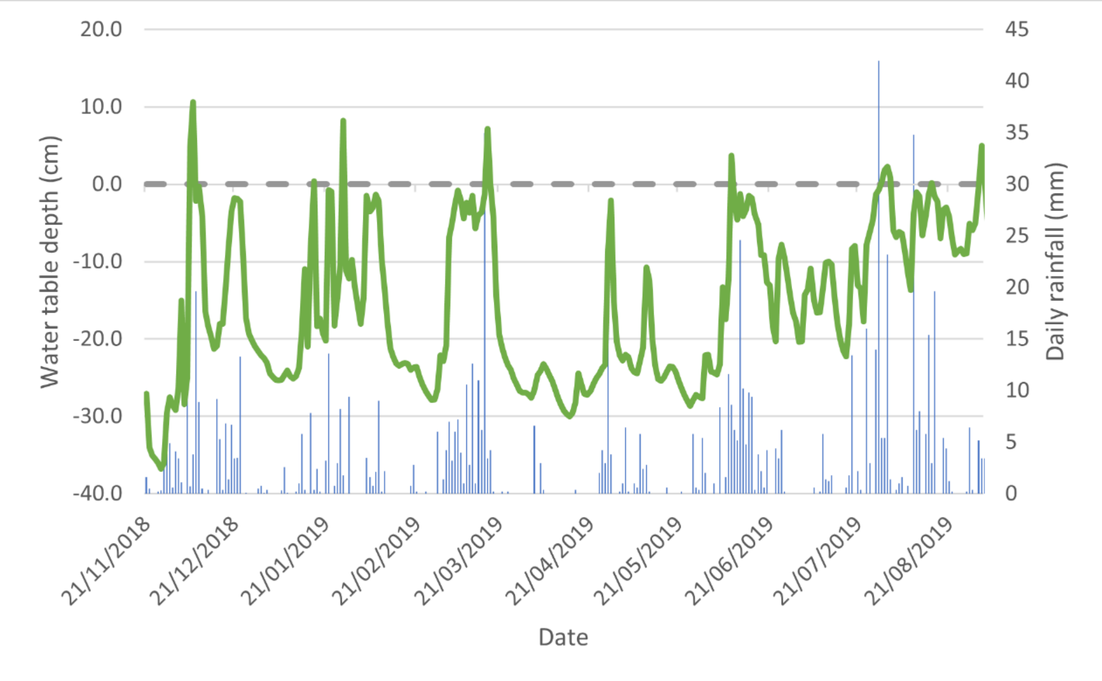


Figure 37: Water Table behaviour Little Woolden site for the Spray irrigation area. Line Data represent mean daily water position relative to the surface (left axis). Bar data represent daily rainfall totals in mm (right axis). The dashed grey line represents the peat surface.

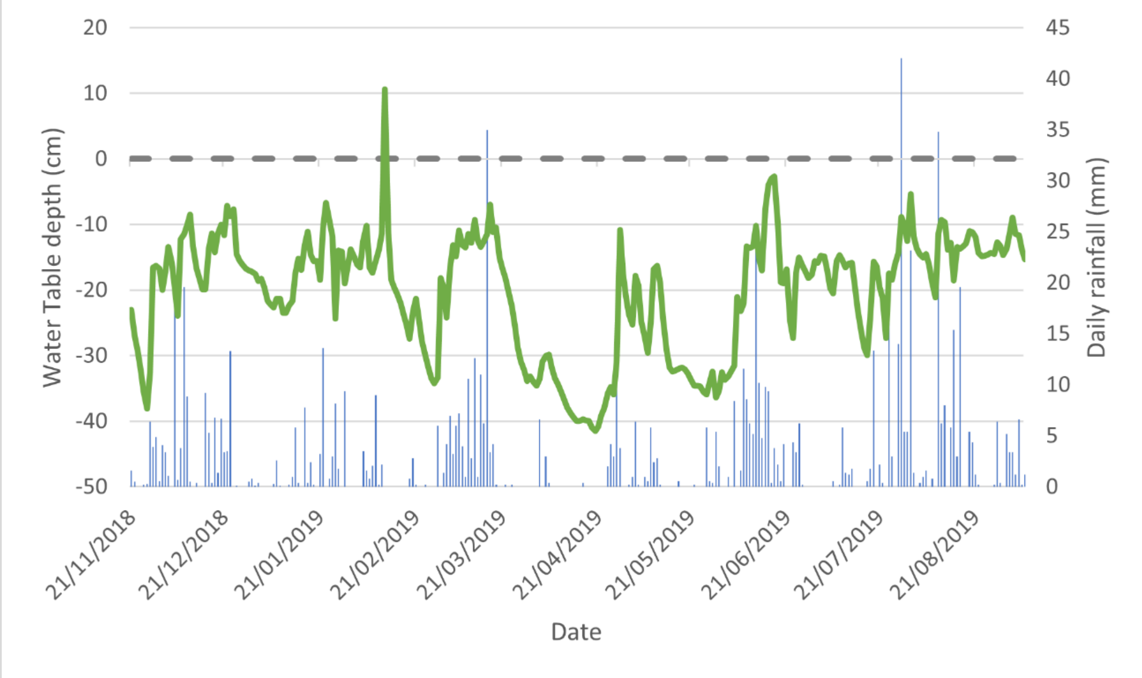


Figure 38: Water Table behaviour Little Woolden site for the Drip irrigation area 21/11/18 to 05/09/19. Line Data represent mean daily water position relative to the surface (left axis). Bar data represent daily rainfall totals in mm (right axis). The grey line represents the peat surface.

Table 24: show the daily mean water table position at the Little Woolden site, split into 5 cm zones as a percentage of days present in these zones.

Little Woolden Drip irrigation area, Mean water table levels, 21/11/18 to 09/09/19		
Based on average water table depth (cm)	Days	Percentage of total days
Greater than 0	1	0.34
less than 0 but greater than -5	3	1.02
less than -5 but greater than -10	15	5.12
less than -10 but greater than -15	79	26.96
less than -15 but greater than -20	80	27.30
less than -20 but greater than -25	35	11.95
less than -25 but greater than -30	18	6.14
less than -30	62	21.16

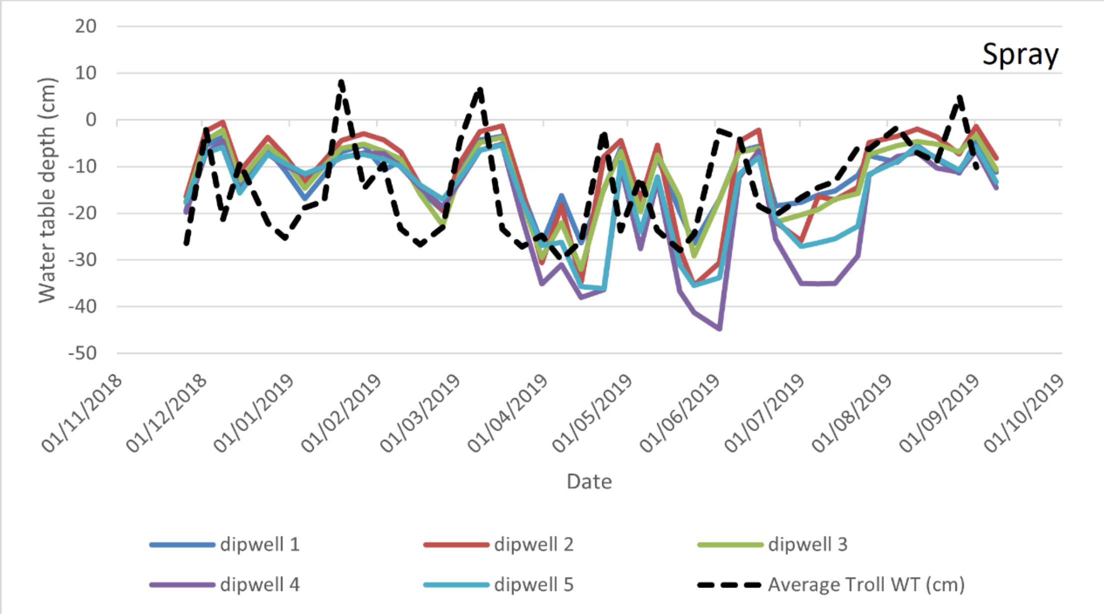
Table 25: show the daily mean water table position split into 5 cm zones as a percentage of days present in these zones.

Little Woolden Spray irrigation area, Mean water table levels, 21/11/18 to 09/09/2019		
Based on average water table depth (cm)	Days	Percentage of total days:
Greater than 0	13	4.44
less than 0 but greater than -5	52	17.75
less than -5 but greater than -10	39	13.31
less than -10 but greater than -15	30	10.24
less than -15 but greater than -20	29	9.90
less than -20 but greater than -25	72	24.57
less than -25 but greater than -30	52	17.75
less than -30	6	2.05

8.3.5 Dipwells and data comparisons

The little Woolden site also benefited from additional manual dipwells, that were monitored regularly across the site. This allows for data validation with the automatic troll loggers (presented in Figure 39).

The automatic loggers generally follow the same pattern as the manual dipwells (n = 5 dipwells per irrigation area). The Drip irrigation area Aqua troll logger follows the dipwell water table very well and is generally slightly lower than the manual dipwells. The spray irrigation area also followed the general water table trend as monitored by the manual dipwells. However, the water table recorded by spray area aqua troll logger is often higher than the reference dipwells. This is most likely due to its central position within the experimental layout, and the fact that the lower lying spray irrigation area is generally wetter – experiencing more surface flooding than the slightly higher drip irrigation area.



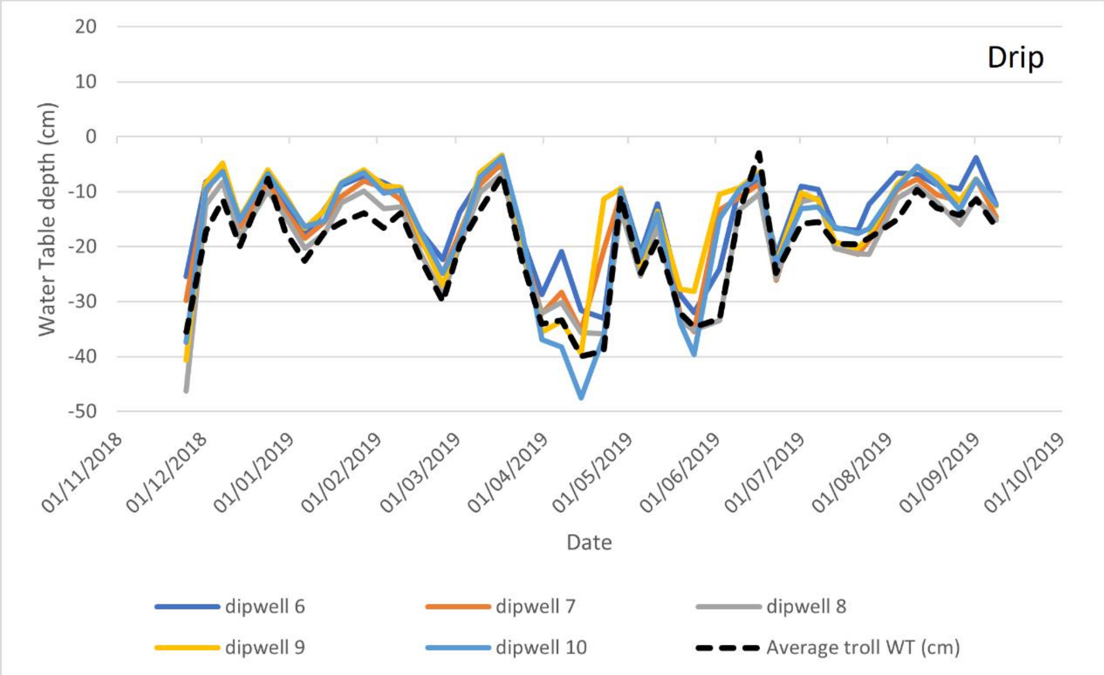


Figure 39 a and b: Aqua Troll automatic water level logger data (black dashed line) compared to data gathered from manual dipwells (n = 5 per irrigation area) at the Little Woolden *Sphagnum* farming site. The Spray irrigation area is presented in the upper graph (39a) and the Drip irrigation area is presented in the lower graph (39b); data from 25/11/18 to 08/09/19.

8.3.6 Pore water pressure: Phase 1 – Little Woolden

The Phase 1 monitoring at Little Woolden took place between the 4th December 2018 and the 17th April 2019 although the irrigation system was not operational until the 5th April 2019. During this extended period of no irrigation, there was frequent rainfall shown in Figure 37 and Figure 38 and both irrigation areas maintained a high-water table. The drip-only irrigation area sustained a water table within 30 cm of the peat surface for over 80% of the time (Table 24) while the spray irrigation maintained the water table within 30 cm of the peat surface for 97% of the time (Table 25).

Unfortunately, despite best efforts to provide insulating covers, several tensiometers did not survive this winter period (loggers T0, T1, T2, T7 in System 1) and (Loggers T0 and T1 in System 2). This was due to ground frosts and areas of surface-water pooling. These cold events caused the tensiometer tubes to freeze, crack and lose their integrity. Most damaged tensiometers failed in early December and were subsequently replaced on the first site visit in February 2019.

Notwithstanding these difficulties, Phase 1 revealed some intriguing behaviour in relation to the irrigation methods and the mulch covers, given that the plugs were newly planted and the gel had only recently been applied.

Drip irrigation (plugs only)

Tensiometer failure was obviously a major factor influencing data capture within the zone of plug planting. The extent to which individual tensiometers started recording suspect values is difficult to pin down, but the very high values recorded for tensiometer Mesh T2 (Figure 40 below) for example, may fall within this category and should therefore probably not be taken as a true indication of the performance of the mesh at that location.

The tensiometer pair recording conditions in the absence of any mulch (top left in Figure 40) follow a similar pattern to each other, initially with both giving values below the critical threshold of 100 hPa, but the dramatic rise of tensiometer T1 beyond this threshold, and the fact that it gives readings consistently higher than those obtained from tensiometer T0, may indicate that there is potentially a technical problem similar to that highlighted for tensiometer Mesh T2.

However within the results shown in Figure 40, if it is assumed for the moment that the readings for tensiometers No mulch T0 and Mesh T3 are correct, it can be seen that the no mulch condition displays greater variability than when either mesh or plastic mulch are employed. The plastic mulch (Plastic T4 and T5), results in the lowest range of pore water pressures, barely reaching 50 hPa, whereas the mulch peaks at a slightly higher 60 hPa, but both remain well below the critical threshold of 100 hPa.

The straw mulch, in contrast, resulted in both tensiometers exceeding the critical threshold towards the end of the data run. Once again, this mulch type which forms a key part of the MLTT approach is shown to be less effective than mesh or plastic mulch during the critical establishment phase.

No data were obtained for Phase 1 within the drip irrigation sector for gel performance because at that stage gel had only been applied to some parts of the spray irrigation sector.

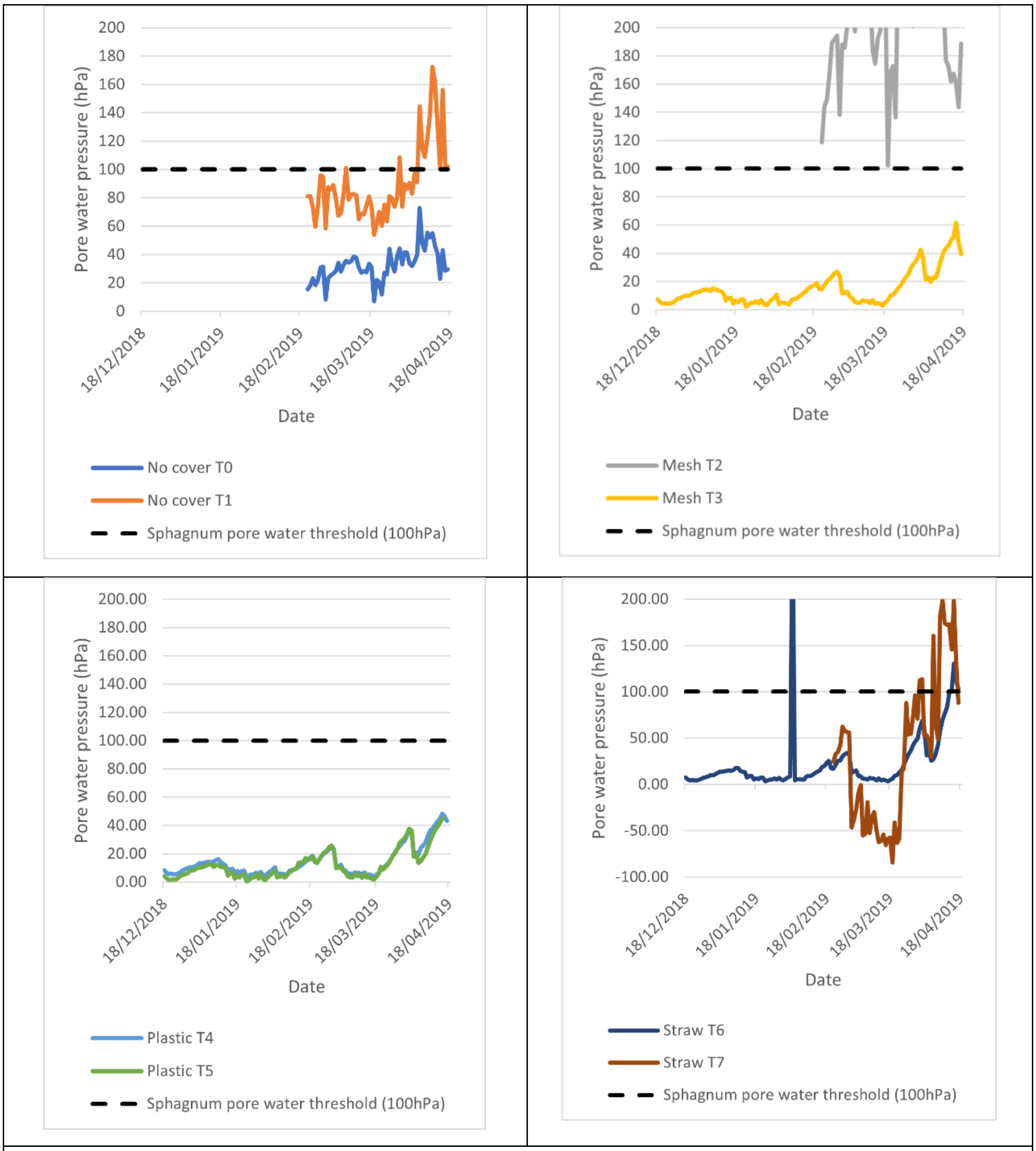


Figure 40: Little Woolden Drip irrigation area as recorded by System 2. Graphs show the Maximum daily PWP recorded, presented by mulch cover type. Data runs from 18/12/2018 to 17/04/2019. Note Data gap in the no cover, mesh, and plastic plots, due to tensiometer failure. All plots use plug founder material.

Spray irrigation

For the *Sphagnum* plugs, once again, the tensiometer pair located in the no-mulch plots returned different readings (No mulch T0 and T1), with T0 showing a PWP consistently above the critical threshold of 100 hPa, while T1 appears to indicate that it was flooded for most of its working period, with pore water pressures below zero (such values generally being an indication that the tensiometer bulb is under water). It is difficult to know what reliance can be placed on these no mulch tensiometer readings, but if T0 is correct, it would seem that where the peat surface was not saturated the exposed peat matrix had a PWP well above that capable of sustaining *Sphagnum* survival.

In stark contrast, both forms of mulch – mesh and plastic - performed well, maintaining pore water pressures well below the critical threshold sought by the MIFA approach, with three tensiometers (Mesh T3, Plastic T4 and T5) recording lower pore water pressures throughout the recording period than was observed for the drip irrigation sector.

For the gel plots, straw mulch proved to be as effective as other mulch types, specifically mirroring the performance of the consistently successful plastic mulch, albeit with an odd spike which may or may not be a technical issue rather than a genuine reading.

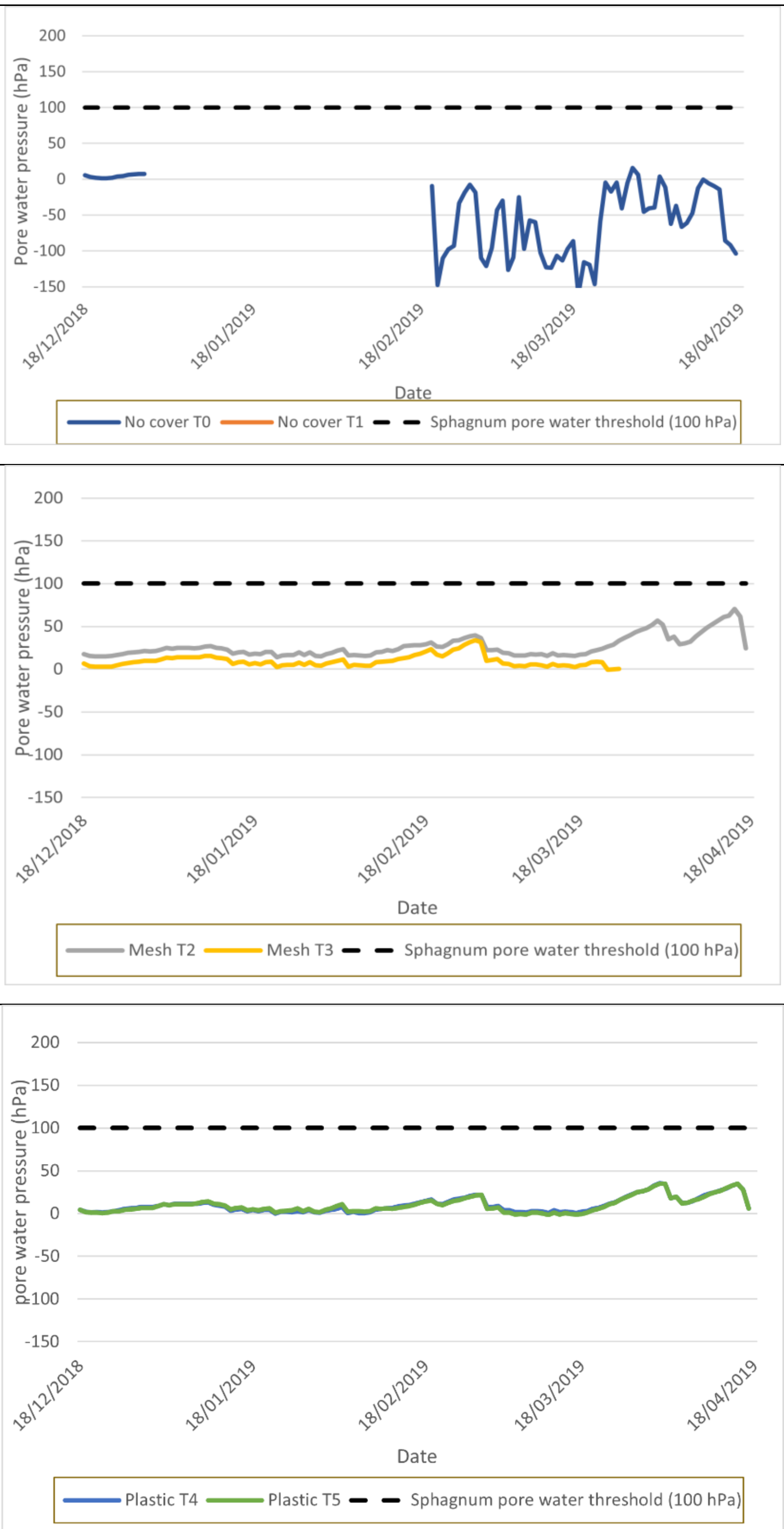


Figure 41: Maximum daily PWP recorded in plug plots planted on the Little Woolden Spray irrigation area recorded by System 2. Presented by mulch cover type: No mulch (top) Mesh (middle) Plastic (bottom). Data runs from 18/12/2018 to 17/04/2019. Note Data gap in the no mulch plots (top, due to tensiometer breakage.

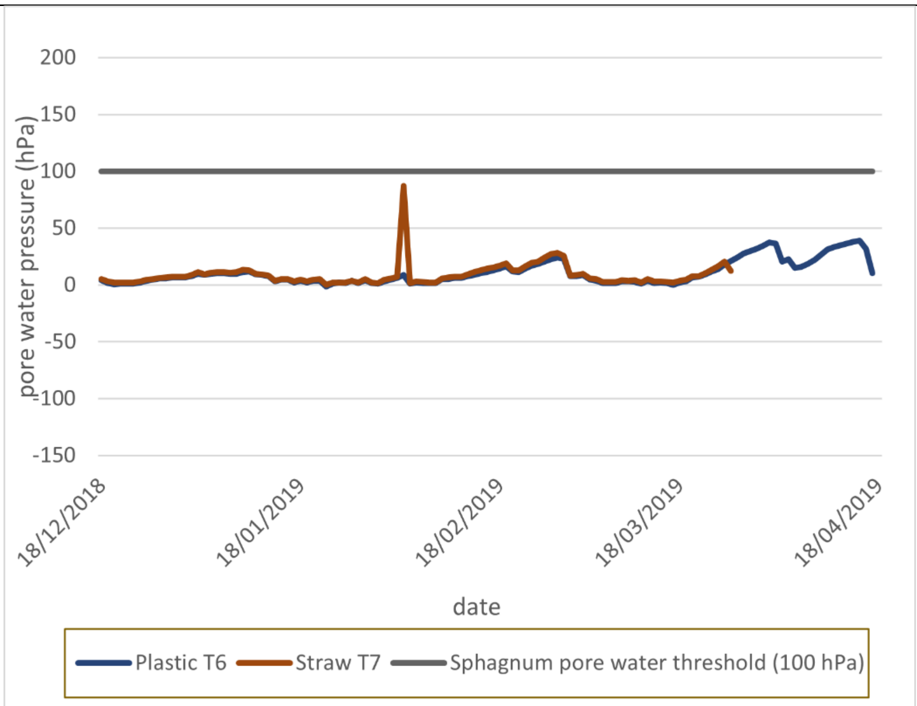


Figure 42: Maximum daily PWP recorded in Gel plots planted on the Little Woolden Spray irrigation area (row 5) during Phase 1. Due to equipment limitations, only 2 mulch types, plastic and straw were monitored. Data runs from 18/12/2018 to 17/04/2019.

During the establishment phase, and ignoring what appear to be faulty tensiometer, it seems from Figure 43, that use of a mulch at this stage is essential if using overhead spray or drip irrigation when planting *Sphagnum* plugs into ombrotrophic bog peat at the Little Woolden site. The use of either plastic or mesh in concert with either drip or spray, will nevertheless ensure that pore water conditions are maintained in a suitable condition for *Sphagnum* to survive and grow. However, If using drip irrigation with straw, this cannot always be guaranteed.

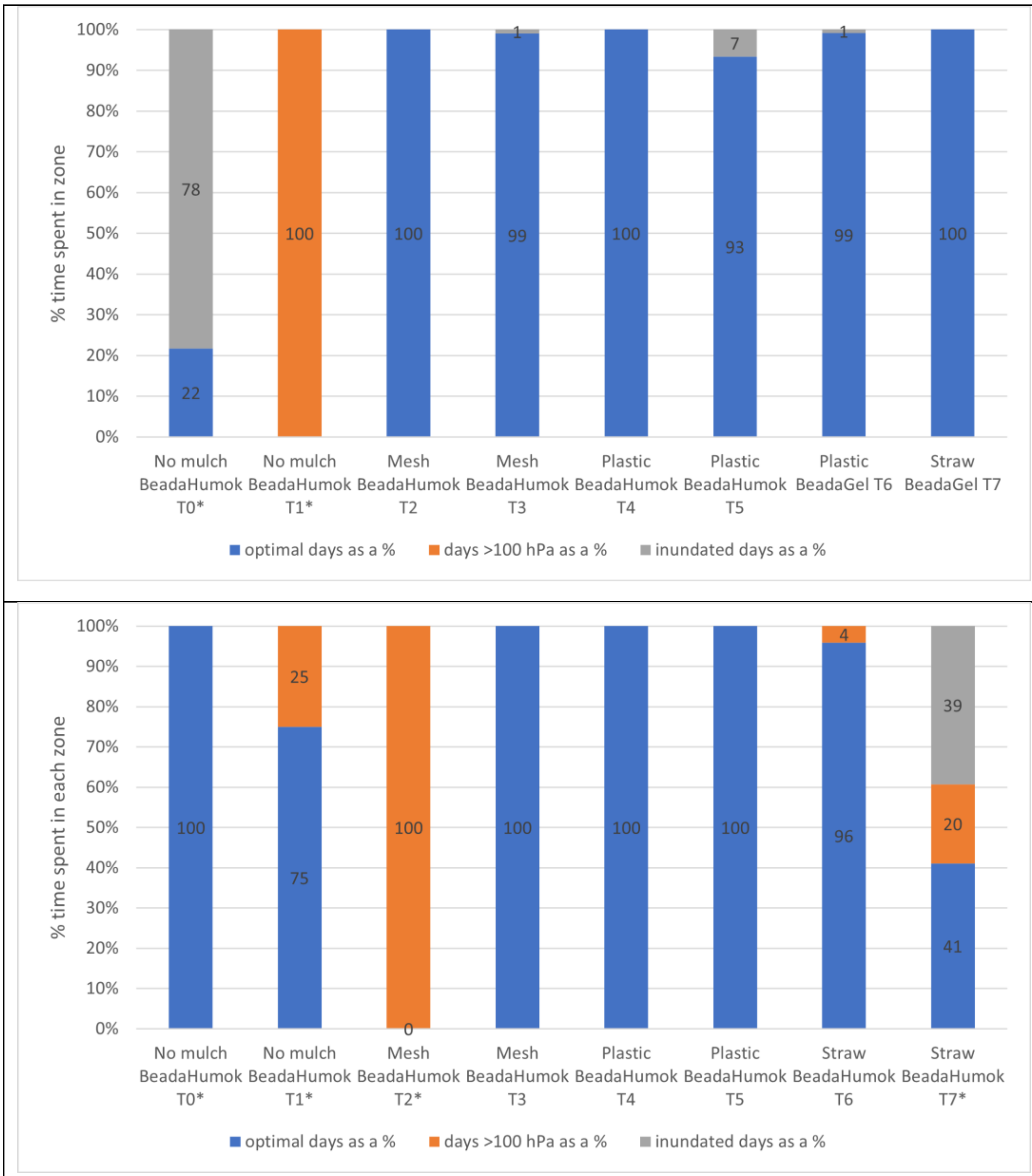


Figure 43: LW spray irrigation area, Phase 1 monitoring. Graph showing time spent in each PWP zone: optimal, too dry >100 hPa or too wet <0 hPa by mulch type and founder material. Spray irrigation area (top) Drip irrigation area (bottom).

8.3.7 Pore water pressure: Phase 2 – Little Woolden

The Phase 2 monitoring at Little Woolden took place between 27th April and 3rd November 2019. By this time, the *Sphagnum* plugs had become established and begun expanding, while the gel plots no longer resembled bare peat, instead starting to come together as a continuous *Sphagnum* carpet. As such, one might expect pore water pressures to reflect the fact that the tensiometers were now positioned within a developing crop rather than in the bare-peat substrate of a freshly planted/applied surface.

Unfortunately, no tensiometers were available to be placed in the unplanted, no-mulch control plots so it is not possible to compare (as was possible for Sharpley), this control environment against the environment prevailing within the crop.

As with the Sharpley Phase 2, this period of monitoring was undertaken during the spring and summer months. As is clear from the water table records for Little Woolden, as shown in Figure 37 and Figure 38 earlier, the spring and early summer period saw two major extended dry periods, one extending through almost the whole of April while the second period, though shorter, still lasted for much of late May into early June and overlapped with the beginning of Phase 2. After those events the water table, though fluctuating, showed a slow and steady rise into the autumn. This context is important when considering the responses of the developing crops.

Drip irrigation

The dry period in May corresponds with marked increases in PWP across all treatments, shown in Figure 44 with the 'no mulch T0' tensiometer in the gel treatment showing the highest and most extended rise, mostly exceeding the critical 100 hPa threshold for much of May. The plastic mulch showed the most muted response, rising only to about 50 hPa in both plugs and gel, while the mesh briefly breached the critical threshold in both plug and gel, rising to 120 hPa, before falling back to the same very low levels as the plastic mulch. Straw in the gel plots also crossed the critical threshold very briefly during the May dry period, and although it fell back to levels close to those of plastic and mesh, it also remained more sensitive to individual dry events which are indicated by occasional peaks in the 'no mulch' tensiometer.

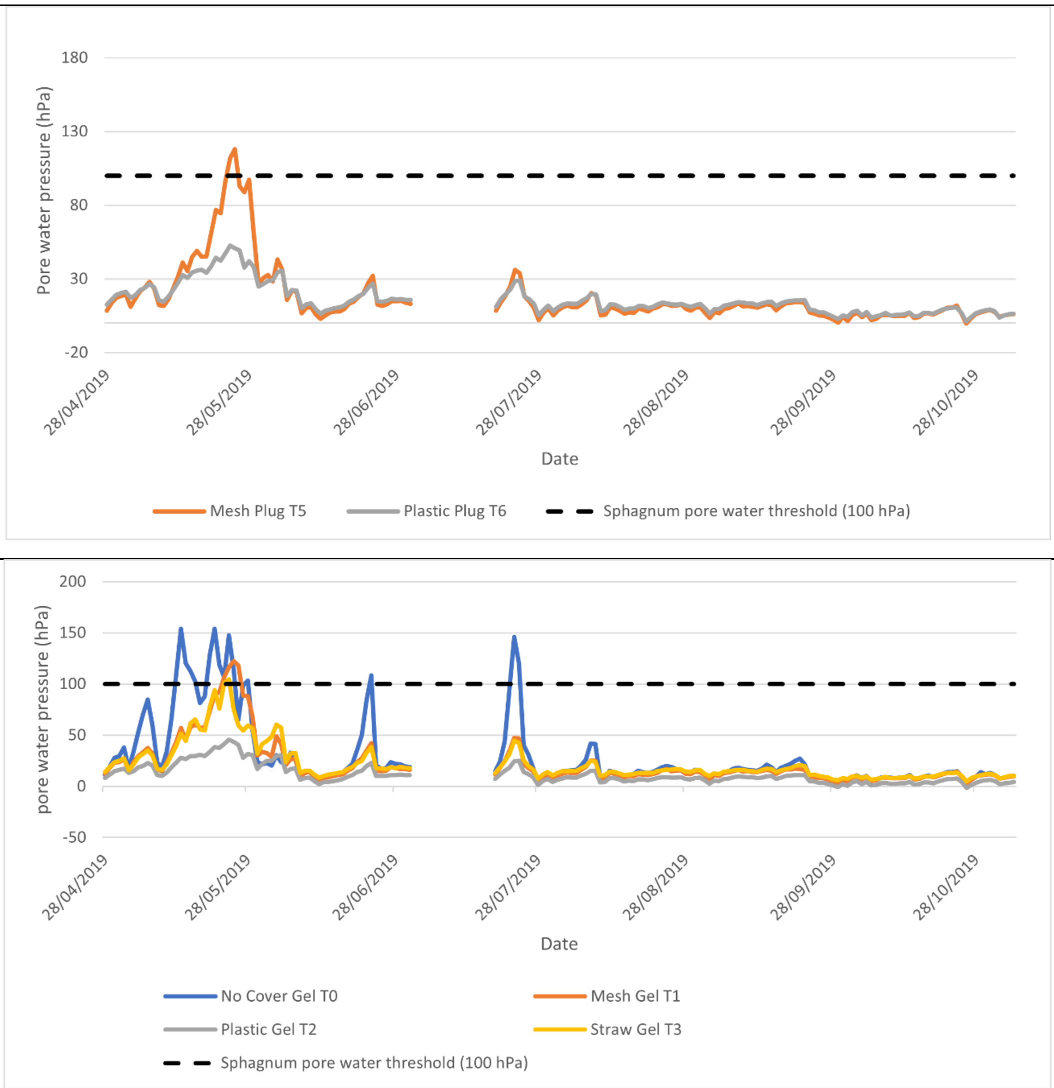


Figure 44: Little Woolden Drip irrigation area phase 2 monitoring, Maximum daily PWP recorded in plug plots (top) gel plots (bottom). Data gaps due to memory overwrite.

There was no discernible difference between the performances of plug and gel throughout the recording period, which perhaps reflects the fact that expanding plugs and developing gel carpet in effect both reflect the environment of a *Sphagnum* dominated surface.

Spray irrigation

Spray irrigation enabled all treatments, including the plots lacking a mulch, to remain below the critical threshold of 100 hPa even during the dry period in May as seen in Figure 45. The ‘no-mulch’ plot almost reached the critical threshold, as did the straw-mulch plot, and in fact with the gel even the plastic mulch rose significantly higher than it did during this period under drip irrigation. Gel pore water pressures in general remained more variable under spray irrigation than did the plugs, perhaps because the gel carpet was still consolidating whereas the plugs were already a consolidated entity.

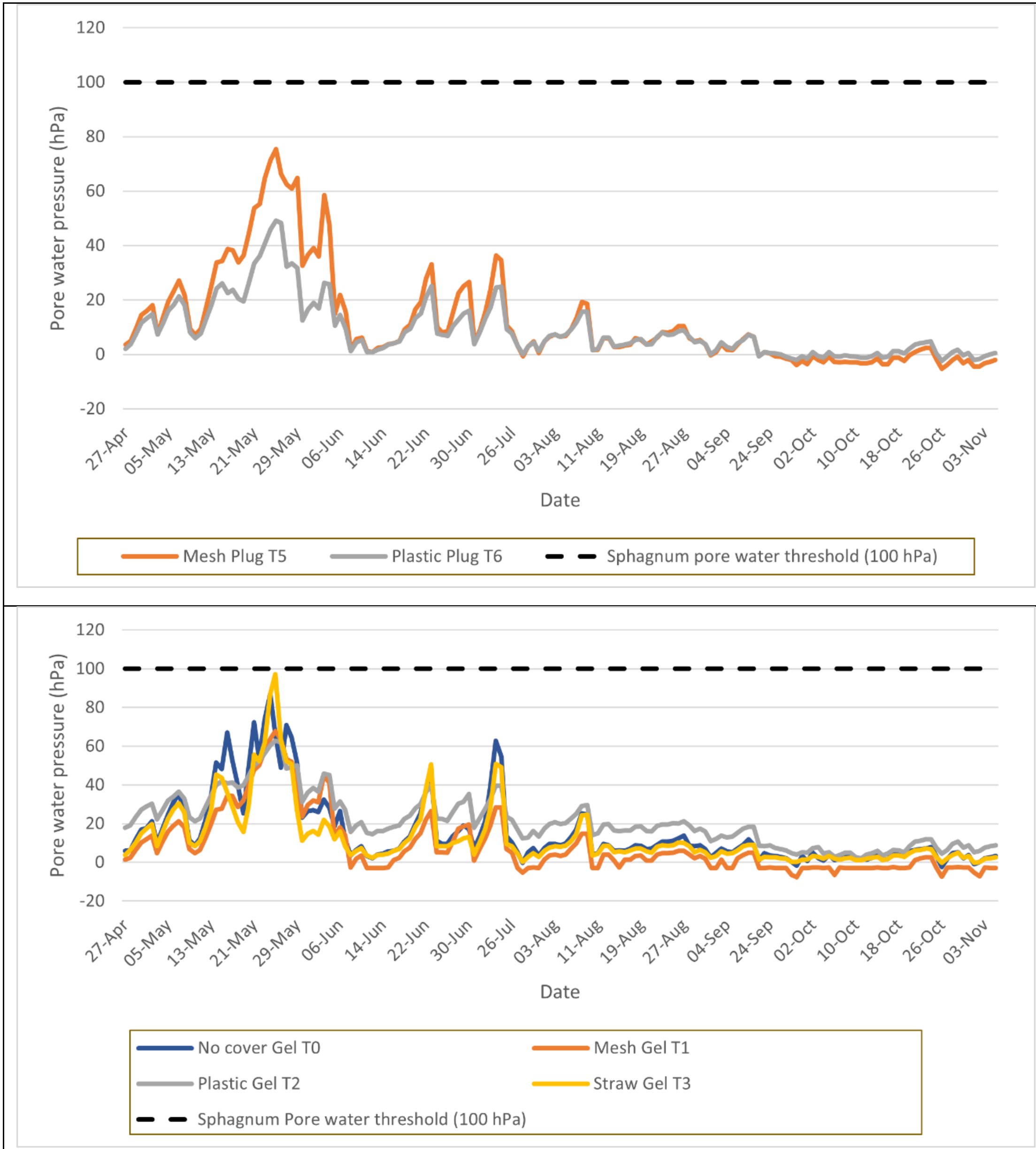
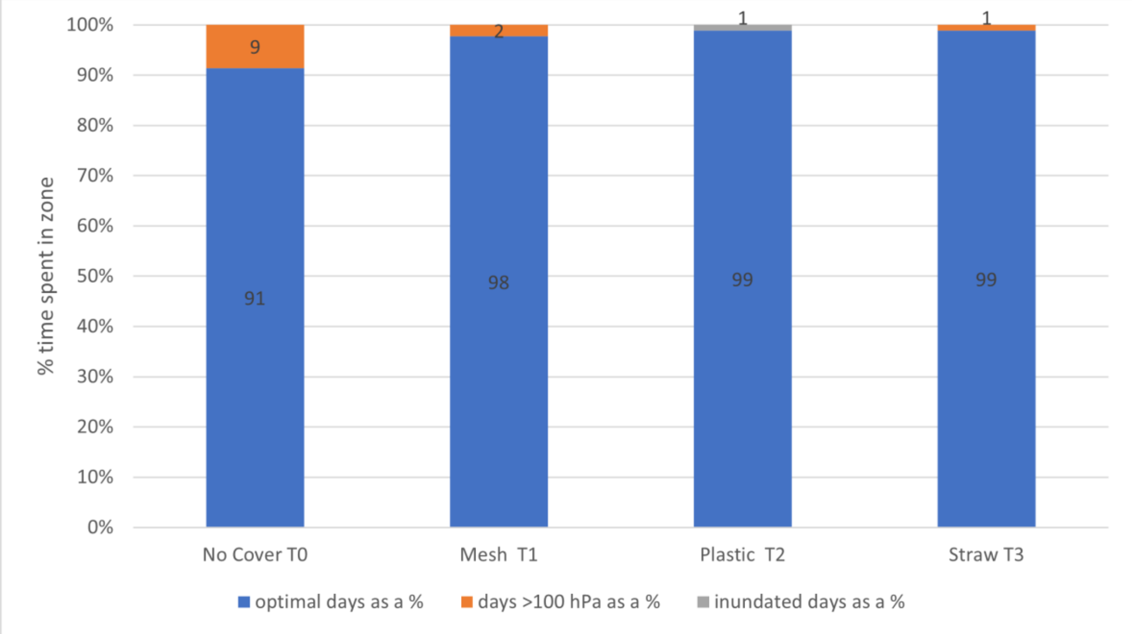


Figure 45: Little Woolden Spray irrigation area phase 2 monitoring, Maximum daily PWP recorded in plug plots (top) gel plots (bottom).

It is apparent from Figure 46 and Figure 47 below, that once the *Sphagnum* crop has become established and begun to form extensive carpets of *Sphagnum* on an ombrotrophic peat base, choice of mulch, or even (to an extent) dispensing with mulch, and choice of irrigation system – all have relatively little bearing on PWP.

Indeed, the focus of concern becomes more one of surface pooling or acrotelm saturation when using overhead spray in combination with mesh and plastic mulch covers, indicated by the negative PWP readings shown in grey in Figure 47, which may potentially reduce crop productivity, whereas this phenomenon does not seem to occur when using drip irrigation.



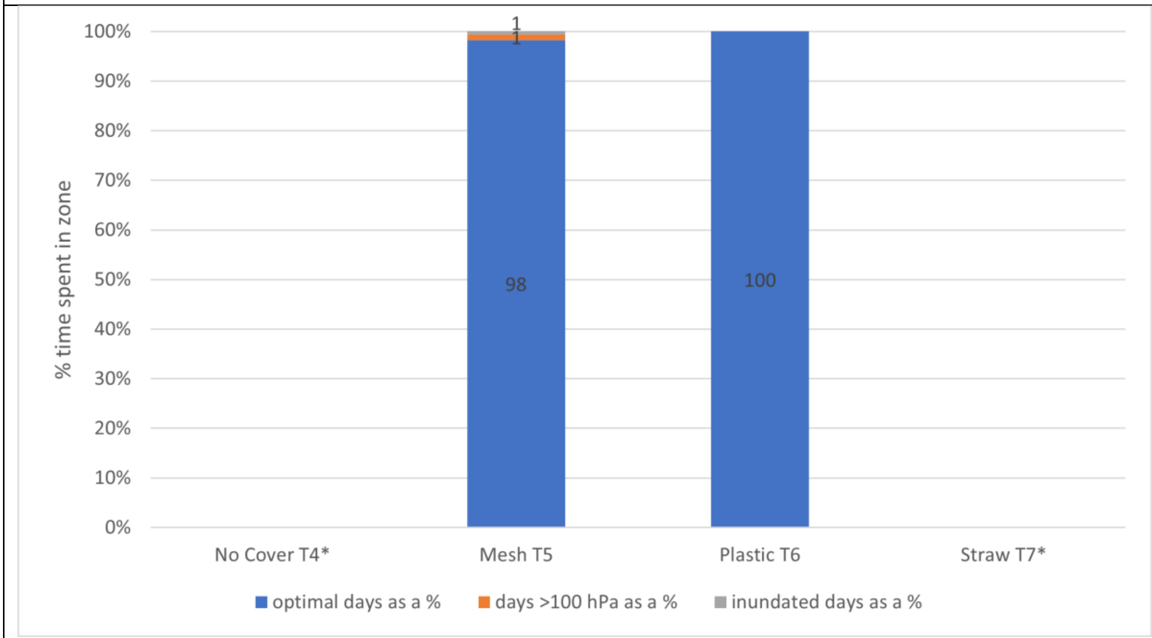
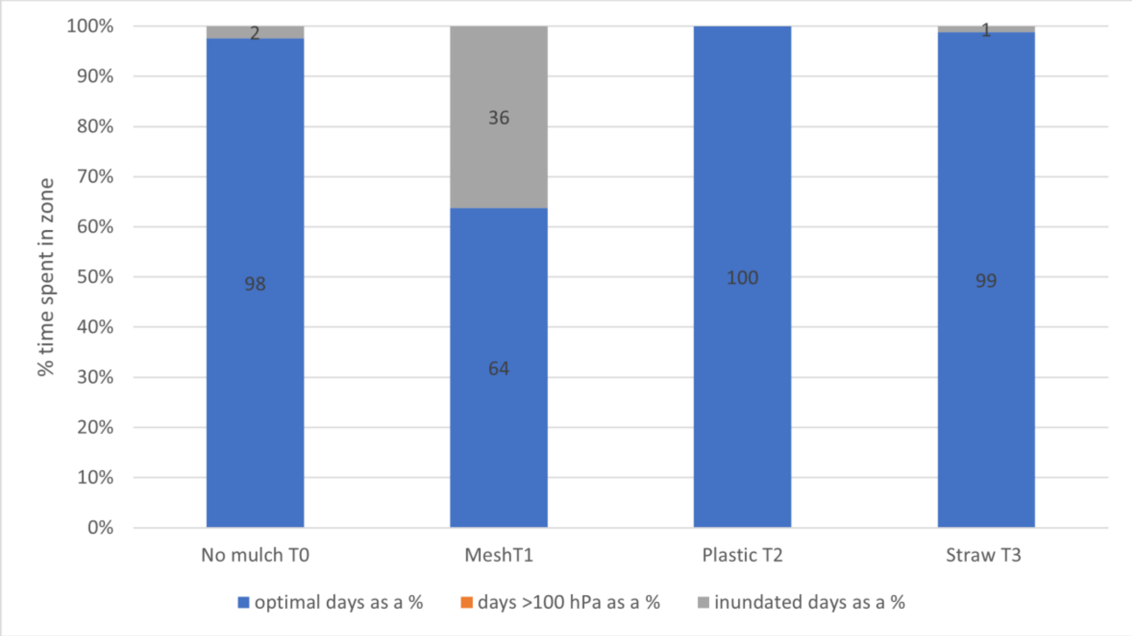


Figure 46 LW Drip irrigation area, Phase 2 monitoring. Graph showing time spent in each PWP zone: optimal, too dry >100 hPa or too wet <0 hPa by mulch cover type for Gel plots (top) and Plug plots (bottom)



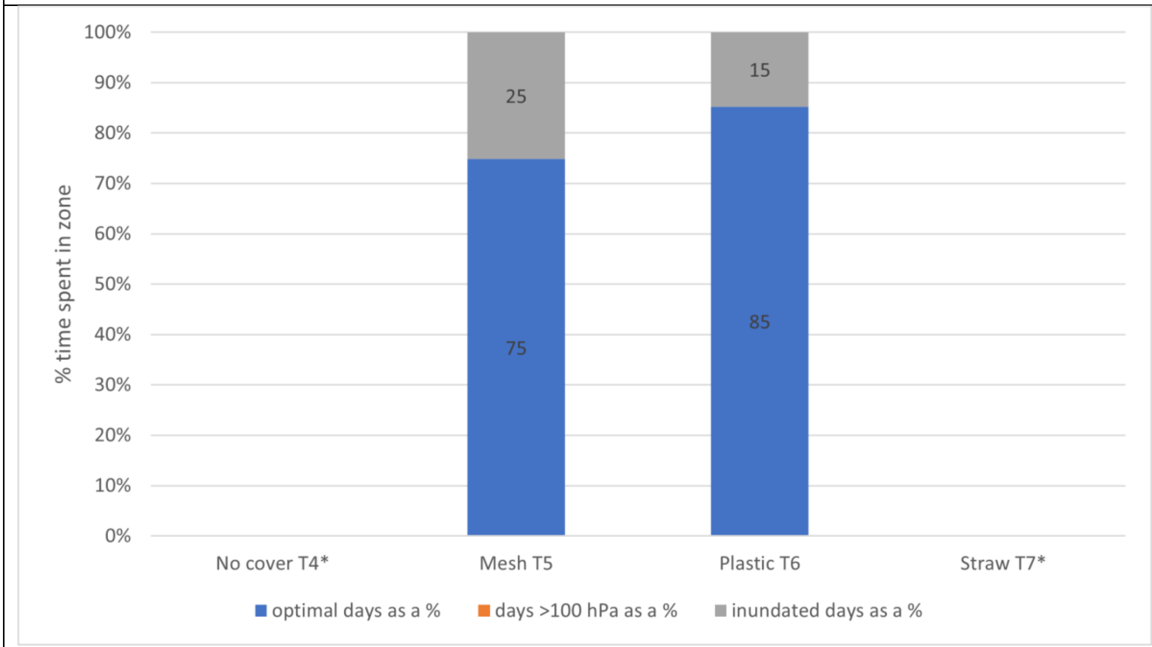


Figure 47 LW Spray irrigation area, Phase 2 monitoring. Graph showing time spent in each PWP zone: optimal, too dry >100 hPa or too wet <0 hPa by mulch cover type for Gel plots (top) and Plug plots (bottom)

8.4 Discussion about hydrological evidence

The MIFA approach is based on the idea that a *Sphagnum* crop not derived from wild harvesting can be grown on almost any surface if micropropagated material is irrigated from above. The Sharpley site is probably almost the most challenging starting-surface possible for such an approach, and it is clear from Figure 27 and Figure 28 that PWP experienced considerable fluctuation during the initial establishment phases, but a mulch cover did much to reduce this degree of fluctuation and keep it below critical thresholds. Compared to the extremely hostile conditions prevailing in the absence of mulch and crop cover, conditions became steadily more benign as the crop developed, even to the point where absence of a mulch cover only became a critical issue during long periods of dry weather.

The Little Woolden site, being a more typically ombrotrophic bog surface, did not appear to offer any substantial benefits compared to the Shapley site during the establishment phase. PWP was maintained well below the critical threshold in some cases but showed considerable variability in others. As time went on, however, and as the crop developed, conditions became even more benign than those at Sharpley, with even fewer occasions when the critical threshold for pore water was exceeded, even in the absence of a mulch cover.

In terms of mulch type, straw may be the favoured mulch for the MLTT approach and indeed it performed well in many of the settings here, but overall it was occasionally less effective than either mesh or plastic mulch, particularly when drip irrigation was employed.

Of perhaps even greater significance for the proposed MIFA approach, however, is the fact that although the water table experienced very considerable fluctuations during the study period, measured PWP in the surface peat and then in the developing crop generally remained much less variable, almost independent of the water table except during extreme dry periods, and largely below the critical pore water threshold. The type of irrigation did not alter this pattern significantly, both drip and spray performing well most of the time.

It might be argued that the *Sphagnum* crop grew successfully and pore water was maintained because the water table itself never fell below a critical threshold of -40 cm proposed by (Schouwenaars, 1988), this critical threshold also now forming part of the MLTT approach. The deepest mean water table recorded for Little Woolden (where, unlike Sharpley, the sensor depth was not limited by the nature of the sub-surface material) was 41 cm, for a single day, and studies have found that *Sphagnum* is not limited by water table position when the water table remains closer to the peat surface than -40 cm (Price and Ketcheson, 2009; Taylor and Price, 2015).

Such an argument could only be resolved by further testing on a surface where the water table was well below, and maintained below, this -40 cm threshold. However, the application of top-down irrigation may mean that maintaining a water table at this depth is not feasible anyway – it is indeed likely that water table and top-down irrigation are intimately linked in such a way that it would be difficult to maintain very low water tables under conditions of top-down irrigation.

Despite the various arguments proposed, critical PWP thresholds have been mostly prevented to the extent that *Sphagnum* can be grown on surfaces which have been subject to agricultural or industrial drainage until very recently, and this surface PWP has been maintained despite considerable fluctuations in the water table.

From the data available, more days where PWP exceeded 100 hPa were experienced by the drip-irrigated gel plots at Sharpley than by the plug plots. At

Little Woolden, the gel was more established which then allowed comparison of mature plug and mature gel plots. The results from suggest that when gel is more established there is little or no difference in susceptibility to drought events using the spray system for plug and gel treatments.

Both sites struggled to maintain the desired water table for *Sphagnum* farming using the MLTT method with irrigation canals, which aims for a water table within 10 cm of the peat surface for the majority of the time. The choice of irrigation method made little difference at the Sharpley site as both irrigation sectors retained a water table within 10 cm of the peat surface for 18 – 20% of the time. At Little Woolden, the water table was maintained within 10 cm of the peat surface for just 6% of the time in the drip-irrigation sector and 31% of the time in the spray sector.

This trend was repeated when 0 to 30 cm below the surface is considered, both irrigation sectors at Sharpley remaining in this zone for 68% of the time, while at Little Woolden it was 78 and 98% of the time for the drip and spray sectors respectively.

In terms of tensiometer results, focusing particularly on conditions once the *Sphagnum* founder materials were becoming well established, the following can be noted:

- Little Woolden drip irrigation had 23 out of 1050 tensiometer observation days of more than 100 hPa;
- Little Woolden spray irrigation had 0 out of 978 tensiometer observation days of more than 100 hPa;
- Sharpley drip irrigation had 14 out of 1074 tensiometer observation days of more than 100 hPa;
- Sharpley spray irrigation had 158 out of 1253 tensiometer observation days of more than 100 hPa;

In terms of tensiometer data, therefore, Little Woolden experienced fewer total ‘tensiometer observation days’ where the maximum PWP recorded exceeded 100 hPa, with 23 days at Little Woolden and 176 days at Sharpley.

The summary numbers above include records for plots which had no mulch cover, the same summary was repeated with the exclusion of the no mulch plots. The new summary suggests:

- Little Woolden drip irrigation had 8 out of 1050 tensiometer observation days of more than 100 hPa;
- Little Woolden spray irrigation had 0 out of 978 tensiometer observation days of more than 100 hPa;
- Sharpley drip irrigation had 14 out of 1074 tensiometer observation days of more than 100 hPa;
- Sharpley spray irrigation had 87 out of 1432 tensiometer observation days of more than 100 hPa.

Consequently, even when only the mulch treatments are considered, the pattern remains the same. Spray performs better at Little Woolden, while drip irrigation performs better at Sharpley in terms of tensiometer observation days where the maximum PWP recorded is > 100 hPa.

In terms of the Core Hypotheses and the Hydrological hypotheses, therefore it is possible to state the following:

Core Hypothesis 1: The water table will be kept close to the peat surface with surface irrigation.

> **Hydrological hypothesis 1**: Top-down irrigation is an effective mechanism for maintaining the water table at a level between 5 to 10 cm below the soil surface.

No - the water table at Sharpley remained in the 0 to – 10 cm zone for only 18% and 20% of the time monitored for the drip irrigation and spray irrigation areas (see Table 22 and Table 23). The surface irrigation methods applied in combination with natural rainfall did **not** maintain a water table close to the peat surface at either site.

Core Hypothesis 2: Top-down irrigation will provide enough water to keep critical *Sphagnum* pore water pressure thresholds being exceeded at both sites.

Hydrological hypothesis 2: Top-down irrigation will prevent PWP from exceeding 100 hPa for more than 90% of the experimental time.

Yes (mostly) – provided a mulch cover is provided for the establishment phase, with a slight preference for mesh or perforated plastic mulch, and if the base ground is former ombrotrophic bog, then a slight preference for drip irrigation in order to avoid flooding of the crop.

Hydrological hypothesis 3: Plug plants will have greater desiccation resilience during establishment compared to gel plots.

Yes (to an extent) – the late application of gel means that it is not possible to make a direct comparison between early establishment plugs and early establishment gel, except for plots with plastic and straw mulches at Little Woolden, where the gel performed as well as plugs, but during later establishment phases, gel with straw mulch generally performed less well than plugs, whereas there were no cases of plugs performing *less* well than gel.

<u>Hydrological hypothesis 4</u>: The Little Woolden site will be more hydrologically suited to *Sphagnum* farming than the agricultural Sharpley site, experiencing a greater frequency of high water table conditions as well as fewer days exceeding the 100 hPa threshold for *Sphagnum* survival and growth.

> **Yes** (to an extent) - On first visiting the Sharpley site, expectations were extremely low that there was any prospect of growing *Sphagnum* within the designated field. It was a grassy pasture with almost none of the classic signs of a peatland subject to agricultural activity reverting back to a form of peatland habitat. One of the unexpected results of this study is the fact that such an unpromising starting point was indeed able to produce good growth of *Sphagnum*, equal to that grown at Little Woolden if provided with the appropriate mulch. However, it is true that water tables tended to be somewhat lower and pore water pressures higher at Sharpley, probably reflecting both the differences in climate and the fact that the Little Woolden plots sit on a deep layer of ombrotrophic bog peat whereas Sharpley is a thin, organomineral peat soil.

8.4.1 Top-down irrigation, a new, or perhaps complementary, approach to *Sphagnum* farming?

As part of the evidence gathering to support a new form of *Sphagnum* farming using the MIFA approach, the water table and soil moisture data gathered by this project also have wider implications and application.

Firstly, it is expected that in the short-to-midterm, agreement to raise water tables within what are currently conventional agricultural landscapes will be difficult to achieve. Surface irrigation methods may therefore be a low hanging fruit for pioneer *Sphagnum* farmers in the UK. Use of surface irrigation allows farmers to begin *Sphagnum* farming without raising water tables and impacting nearby landowners. The reduced need for earthworks as shown in the Sharpley site, has also shown that cultivation via surface irrigation can take place with no impact on neighbouring fields and even within parts of the same field not being used for *Sphagnum* farming.

On a more general note, failure to account for soil moisture availability within climate projection models could impact key peatland management decisions in northern and temperate peatlands. Nijp *et al.*, (2017), found that predicted drought frequency was reduced by up to 52% for northern peatlands when water storage potential within a living moss layer and peat volume were included in hydrological models (Nijp *et al.*, 2017). The authors make the point that this is a serious omission from many climate projections models on peatlands.

For example, some models link moss photosynthesis to groundwater table only (Eppinga *et al.*, 2009; Frolking *et al.*, 2011; Heijmans *et al.*, 2013). This approach fails to account for direct rewetting of the living moss in the acrotelm through precipitation inputs. Precipitation input has a direct impact in capillary water transport, moss growth and carbon uptake (Thompson and Waddington, 2008) so will influence *Sphagnum* growth. Future climate modelling for establishing the long-term viability of *Sphagnum* farming should include all forms of precipitation input (rain, snow, mist, dew) together with irrigation inputs, whilst also considering the impact that a developing acrotelm surface may have on limiting the effects of drought events through desiccation resistance.

How do the pore water pressure results from this research study compare to a natural peatland? In a natural peatland, measured pressure heads in *Sphagnum* hummocks ranged between -5 and -50 cm H_2O (Lindholm and Markkula, 1984), this is comparable to 5 to 50 hPa as presented for the tensiometers in this thesis. In this natural site, a mean PWP of 20 hPa was recorded. At a nearby drained peatland with remaining *Sphagnum* cover the median pressure heads recorded by Lindholm and Markkula (1984) increased to a mean of 36 hPa and a maximum pressure head to of 60 hPa was recorded. This suggest that most PWP recordings made in the course of the present research programme, when under mulch covers, were equivalent to those of a natural peatland system and better than those recorded for a drained peatland that was nevertheless still capable of supporting a *Sphagnum* community.

8.5 Conclusions

This study has shown that top-down irrigation in combination with natural precipitation can maintain WT levels within 30 cm of the peat surface for 68% time at a shallow peat and 78 -93% of the time at a deep peat site. The porewater pressure at both sites has generally remained below key ecohydrological thresholds, ensuring that soil moisture conditions are not prohibitive to *Sphagnum* growth or cultivation.

Future work will be needed to optimise the surface irrigation system for *Sphagnum* farming. Future studies should investigate the full water balance, the impact of mulch covers on evapotranspiration rates, and monitor the water inputs via a meter. Additional ecosystem service benefits may arise from expanding surface water irrigation with the standard method of raising the water table. This approach has the potential to reduce carbon emissions and peat subsidence.

The impact of the two surface irrigation methods on *Sphagnum* growth will be explored in the next chapter.

Chapter 9. Growth studies

9.1 Underpinning thinking/hypotheses

Sphagnum yield is central to the success of *Sphagnum* farming as a concept. High quality biomass must be produced at scale to provide the raw material for growing media and provide enough material to enable the use of *Sphagnum* biomass in a wider range of material uses. As described in Chapter 4, initial *Sphagnum* farm studies utilising surface irrigation suggest that *Sphagnum* will be cultivated on a 3-to-5-year rotation. Harvests will occur at an optimum balance between annual growth and decomposition. The growth of *Sphagnum* under the new MIFA approach may produce differing growth and yield results than these early studies, so must be investigated.

Within the *Sphagnum* literature, *Sphagnum* growth metrics currently used are percentage cover, depth of *Sphagnum* carpet (cm) and dry bulk density (DBD) (g cm^3) and biomass yield (tons dry mass per ha per year). However, all these units can be converted mathematically to allow comparison across other literature values (Clymo, 1970, 1973; Gaudig *et al.*, 2014). *Sphagnum* growth has been measured over large time periods (years), allowing annual yields to be obtained. Yield is likely to be off most interest for future *Sphagnum* farmers, as this will influence the economic viability of the practice and higher yields will bring higher income. A key component suggested for a successful yield is the rapid development of high percentage cover because this increases resistance to desiccation (a key factor when starting *de novo* with a bare peat surface). A high *Sphagnum* cover also results in carbon sequestration and high *Sphagnum* productivity. It was therefore essential that *Sphagnum* growth was monitored.

As described when setting out the MIFA experimental layout in Chapter 6, two novel methods were used to provide founder material: BeadaHumok™ which are plug plants, and BeadaGel™ a *Sphagnum* rich hydrocolloidal gel. Growth rates were expected to differ with these two founder materials, due to the contrasting fragment size of the *Sphagnum* material provided by these methods. These are whole moss individuals within plugs vs smaller fragments within a gel. Fragment size is likely to have a significant impact on desiccation resistance and therefore upon growth.

Core hypothesis 4 posits that plug plots will show a greater rate of growth than gel plots.

The central hypotheses for the growth studies are:

Growth hypothesis 1: No mulch plots will produce the lowest yield of *Sphagnum* compared to covered plots;

Growth hypothesis 2: Mesh and plastic mulches will result in greater yields than traditional straw mulch;

Growth hypothesis 3: The yield of plugs will exceed that of gel;

Growth hypothesis 4: The *Sphagnum* yield produced by the MIFA top-down irrigation systems (measured in dry biomass per ha) will be comparable to previously published *Sphagnum* farming sites that have used donor *Sphagnum* for founder material.

9.2 *Sphagnum* growth methods

9.2.1 Recording Percentage cover of *Sphagnum* crop during establishment: Method development.

Measures of plant productivity have traditionally relied upon what are essentially destructive methods of assessment, specifically harvesting and measuring the biomass obtained from defined areas of the crop. This is partly because simple height measurements must be correlated with measures such as stem density in order to calculate biomass per given volume, and often stem density is highly variable.

A *Sphagnum* carpet, on the other hand, is a continuous carpet having an entwined structure resulting from the interplay between the spreading branches of the individual *Sphagnum* plants. As a consequence, it is easier to obtain a reliable measure of plant density, and thus DBD, for a *Sphagnum* carpet than it is to calculate DBD for a stand of, for example, common reed (*Phragmites australis*). Indeed, a great many figures already exist in the literature for typical dry bulk densities of *Sphagnum* carpets. It is therefore possible to consider using non-destructive methods of measuring *Sphagnum* productivity based on various combinations of visual, or optical, assessment combined with standard values of DBD obtainable from published literature.

The initial method selected for quantifying percentage cover of the *Sphagnum* crop in this study was therefore image-analysis from fixed point photography. The method allowed rapid data collection in the field, provided a visual record of plot development and generated a digital database of measurements. The collected images created a dataset available for future additional analysis.

Plugs were the first founder material type to be planted at both study sites. In total, 120 plugs were planted per designated plot, the plug sizes being such that each was clearly visible in a photograph taken at eye height.

9.2.2 Method 1: Handheld image capture

Images were taken by hand from eye height using a Canon EOS 1200D camera, equipped with an EFS 18-55mm lens. Images were captured in RAW format to ensure the greatest data capture was obtained, whilst maximising image versatility for post processing.

Images were taken of every plot (n = 36) within a given irrigation area. A 1 m ruler or 1 m tape was placed within the plot, on top of the *Sphagnum* surface. The addition of an object of known dimensions ensured that measurement tools used in image post-processing could be calibrated against this object prior to taking image-based measurements. This calibration was necessary as there was potential for every image to be slightly different depending on natural variation introduced during image capture such as the angle the camera was held at, slight variations in lens zoom and the height at which the camera was held.

Due to the risk of lens distortion when using the widest possible camera lens-angle, it was deemed unwise to attempt to capture the entire plot within a top-down view in one image. A photo capture method was devised to avoid this. with images taken at all four corners of each *Sphagnum* cell to capture the overall development of the *Sphagnum*. A fifth image, with a top-down view focusing on a single corner, was captured as a sub-sample for pixel analysis. The fifth photo was taken in the lower right-hand corner of each cell.

Method 1 image analysis

All images were post processed in Adobe Lightroom CC 2019. Images were corrected for lens distortion effects using the Lens correction tool within Adobe Lightroom. The lens correction tool was set to the specific profile of the Canon EFS 18-55mm lens while performing these corrections. This process reduces any image distortion, particularly at the edges of images.

Image based measurements were performed in Adobe Photoshop CC 2019. Each individual image had a measurement scale calibrated against the 1m ruler captured in the image. The lasso selection tool was used to select individual plugs by eye. Selection measurements were then exported, allowing the average plug size to be calculated. The percentage cover for each subsample was then calculated based on the mean plug size multiplied by the original number of plugs planted. These measurements could then be compared against the size of the average plugs when planted and the initial percentage cover.

Method 1 limitations

This method seemed to work well, and useful data were gathered for plugs. As described in the site descriptions, on planting, each plug represented a discrete unit. However, at later site visits it became apparent that Method 1 for counting had limitations. As plugs grew, they merged with other individual plugs. This made it difficult to define and compare individual plugs objectively. This demonstrated a basic flaw in Method 1 as a means of measuring long-term growth, because it relied on measuring individual plugs, but when these individuals could no longer be identified the method was no longer applicable.

Method 1 was not suitable for analysis of gel plots. This was due to the high dispersal of very small *Sphagnum* founder material, which made it very difficult to identify and select *Sphagnum* by eye within the image using the Photoshop software.

9.2.2 Method 2: Alternative handheld camera image analysis

A new method was required to overcome the limitations of Method 1. As the main objective of monitoring during the establishment stage was firstly to quantify percentage cover of the developing *Sphagnum* carpet, a method that could capture the increasing percentage cover of *Sphagnum* within an image was considered appropriate.

The second image analysis method attempted to increase the objective nature of assessing *Sphagnum* cover using pixel counts across the whole image. This removed the problem of defining individual plugs and also made the method applicable to gel plots. It could also be retrospectively applied to earlier images.

The concept behind this method is that every image is constructed from pixels, and each pixel is assigned a colour value. If certain pixels can be related to *Sphagnum*, they can be selected, quantified and used to calculate the area of *Sphagnum* present in an image. As in Method 1, the fifth photo image was used as a subsample.

Pixels were selected using the selection tool within Adobe Photoshop CC 2019. Following testing, the colour tolerance was set to 40%, as this was judged through an initial pixel selection process to be suitable for capturing the range of *Sphagnum* colour against a contrasting dark peat surface. Pixels were selected by zooming into an area of *Sphagnum*, holding the Ctrl button and clicking on pixels within individual *Sphagnum* plants, until the entire *Sphagnum* plant had been encompassed by the selection mask. See Figure 48 for an example of a pixel selection mask.

Once a selection mask had been created, the selected pixel count was displayed within the image histogram. The pixel count within the *Sphagnum* area was recorded, and a percentage area was calculated mathematically by using the total pixel count in the histogram with no selection mask present to represent the plot area.

Method 2 limitations

Pixel selection is based on colour, and this worked well when *Sphagnum* was weed-free because the main colour selection was green *Sphagnum* vs brown/black peat. A high contrast in colour allowed clear selection differences between the two different variables.

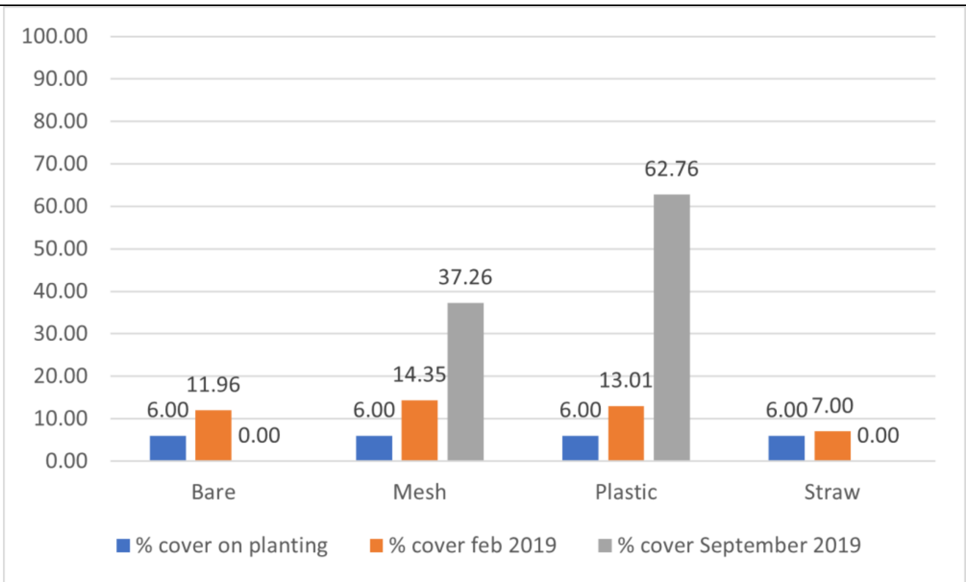
However, with time, vascular species such as *Cirsium vulgare, Juncus effusus and Urtica dioica*) colonised the *Sphagnum*. As vascular species were also green, the pixel count was unable to differentiate between the two reliably. This reduced the reliability of this method when weeds were present.

In addition, some areas of *Sphagnum* in were not selected if they were in areas shaded by weeds within the image, see Figure 48. The pixel counts obtained by Method 2 should generally thus be viewed as underestimates of the area covered by *Sphagnum* once any significant weed load develops within the crop.

Figure 48: Plot CS1 - Sharpley site, Spray irrigation area. Photo taken in September 2019 showing *Sphagnum* specific pixel selection (area within the hashed line) The darker bare peat areas and drip line are not selected by this process. However not all areas of *Sphagnum* are selected due to shading.

Notwithstanding these limitations (which are not so significant if the weed load can be kept low) Method 2 can provide a reasonably reliable record of crop growth, at least for plugs and even for gel if there is little or no weed load.

Figure 49 illustrates the figures obtained for the spray irrigation plug and gel plots at Sharpley, from which plug growth is clearly demonstrated whereas with the gel it became increasingly difficult to distinguish, or even see, gel fragments as the weed load developed a significant canopy.



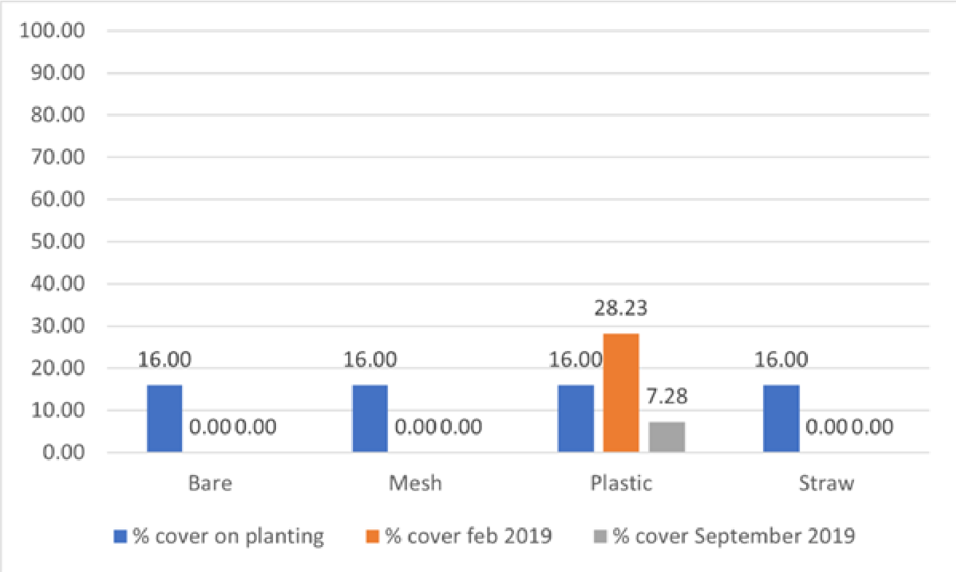


Figure 49: Pixel counts of the spray-irrigated *Sphagnum* crop for three points in the growing timeline. (Top) Plugs; (Bottom) Gel. Note: no gel measurements were taken for straw-mulch plugs in September 2019, and measurements at all three dates were only viable for plastic-mulch gel due to weed pressure. Bare refers to the no mulch plots.

However, provided these limitations are recognised, Method 2 can certainly provide a quick (to record) and relatively cost-effective way of measuring growth until such time as the bare planted surface is completely covered, particularly if the weed load can be kept low during this phase. As highlighted above, it also provides an objective record of growth during this early phase – an archived record which may lend itself to further analyses in the future.

Methods 1 and 2 nevertheless have one fundamental weakness once the *Sphagnum* crop has closed over the whole plot to form a continuous carpet – simple image analysis cannot measure the increase in crop thickness. In theory, this might be possible using stereo-photogrammetry procedures but the technical challenges of this approach, particularly when differences in crop thicknesses are likely to be quite subtle, renders it a method of last resort.

While standard photogrammetry may not be a realistic option for measuring crop growth, more technically advanced approaches which use some of the same principles do offer methods which are practical, objective and suitably high-resolution.

9.2.3 Method 3: Terrestrial Laser Scanning (TLS)

Terrestrial laser scanning (TLS) was chosen as the third method of establishing *Sphagnum* cover and volume change over time. TLS provides a versatile methodology for capturing ultra-high-resolution point data, (defined as < 2 mm point spacing) and can achieve accuracies of 1 mm at 10–15 m from the scanner (Idrees and Pradhan, 2016). Laser scans can capture spatial data with x, y and z coordinates. As a result, TLS scans can accurately capture surfaces with complex morphology (Lague, Brodu and Leroux, 2013; Cabo *et al.*, 2018; Ordóñez *et al.*, 2018).

TLS has been used to calculate the aboveground biomass in tree's such as Black Spruce. Wagers *et al.*, (2021), captured and fed TLS data into computer models to calculate predicted tree attributes such as DBH, crown diameter, crown area, height, tree volume over a wider scale. The authors evidence that their model produced results that were more accurate than previous models and reduced time-consuming manual measurements of tree attributes increasing data capture efficiency. Other studies have used TLS to assess additional tree features such as Leaf area index (Seibert *et al.*, 2022).

However even in trees that are static TLS estimates when used in isolation can be prone to disadvantages regarding data quality; This is affected during data capture by scanner limitations, scanning design errors and the prevailing conditions during data capture such as wind or rain. While data analysis can be affected due to the limits of automatic software extraction techniques. Poor data quality can reduce the effectiveness of the data analysis and post analysis use (Pitkänen *et al.*, 2021). Therefore, careful experimental design is needed to maximise data quality within the limitations of TLS data capture and analysis.

Due to the natural variation found in peatlands, TLS has been used to assess pattern and microtopography and the data has been usefully evaluated against pre-determined vegetation classes (Anderson, Bennie and Wetherelt, 2010). TLS has also been used to produce digital terrain models for the identification of discrete peatland features such drainage ditches (Stenberg *et al.*, 2016) and gullies (Forbriger *et al.*, 2012; Höfle, Griesbaum and Forbriger, 2013). By capturing multi-temporal scans, erosion rates have also been quantified (Chico et al, 2019). TLS has also been used to assess microtopographical features (Lovitt *et al.*, 2018; Stovall *et al.*, 2019; Graham *et al.*, 2020).

The Advantages of TLS in peatland areas over traditional techniques have been stated as applicability, rigour, and ease of acquisition of TLS data, with disadvantages arising when pooling surface water prevented data capture (Clutterbuck *et al.*, 2018) and (Pitkänen *et al.*, 2021).

Ecohydrological relationships in upland peatlands have been investigated using satellite sensors in combination with TLS at ground level (Luscombe *et al.*, 2012) TLS was used as a supporting feature to assess fine scale vegetative response to water table. This study found strong spatial dependence between structurally variable minerotrophic vegetation communities and high-water tables.

Furthermore TLS data capture has been deployed in peatland areas to quantify peatland surface change due to erosion pressures at millimetre accuracy (Chico *et al.*, 2019). By capturing millimetre scale changes in complex surfaces, TLS therefore offers the possibility of obtaining an accurate quantified comparison of change both in cover and volume of a *Sphagnum* crop.

For these reasons, TLS provided a valuable development of Methods 1 and 2 used in the present research programme. The TLS scans could be used to track percentage cover and *Sphagnum* carpet height change across scans. This coupled with literature values for DBD would allow volume change and yield to be calculated. To the best of the authors knowledge no published studies have used TLS to assess *Sphagnum* cover or volume change over time in a *Sphagnum* farming scenario.

To achieve a full and accurate TLS data capture, a series of fixed markers around each site was established to act as ground-control markers for use during subsequent data analysis and comparison of sequential *Sphagnum* surfaces. These ground-control markers were constructed of 8mm threaded steel rod with a 5 mm diameter disk at their uppermost part and a small connector attached immediately above that to give a small fixed structure into which the spiked end of a Differential GPS could be placed to provide an accurate position of the marker. The rods of the ground-control point were sunk into the underlying substrate (clay or mineral soil) below the peat to anchor them securely. The ground control markers were installed at the corners, mid points and centres of each irrigation area. A gap of no more than 15m existed between each ground control marker to ensure that in subsequent TLS data analysis multiple scans could be stitched together accurately.

Data capture by the TLS was performed on three occasions: Feb 2019, Sep 2019 and August 2020. Scan data for Scans 1 and 2 were obtained by Dr Ben Clutterbuck from Nottingham Trent University with assistance from myself as part of my PhD research training, using a Faro focus X330. Scan 3 data series was obtained by myself and Dr Clutterbuck using a Faro focus X330 and a Faro focus S350 scanner in combination working as an integrated team.

Multiple scanning angles are required to reduce the impact of any physical objects blocking a laser beam and preventing data capture during the scanning process. Prior to scanning a minimum of 12 reference spheres were fixed systematically across the target scan area using the ground-control points. These reference spheres were used to register and place scans in their correct orientation during data processing because the ground-control points were increasingly difficult to identify in the scans as the *Sphagnum* crop developed.

Consequently, due to the size of the target area, twelve individual laser scans were performed for each of the irrigation areas at each site. The scan profile selected was the Outdoor-Far profile, The second highest resolution available, resolution ½ (a setting without units) was used while data quality of 2x was selected resulting in 174.8 million laser points per scan. This was selected to balance a high density of spatial resolution, with a manageable scan duration of approximately 7 minutes each and a scanner accuracy of ±1 mm. Each individual scan was given a unique name and timestamp to provide a catalogue of scans to ensure that all locations were scanned. The Faro focus scanners recorded their own position using an internal GPS and captured 360-degree colour images. The colour images allowed colour to be applied to laser points data during processing and provided an extra sense-check for scan position to take place.

Scan positions were distributed evenly across each irrigation area being scanned to guarantee full coverage of the *Sphagnum* plots as shown in Figure 50, This also captured geospatial data for the fixed ground control points.

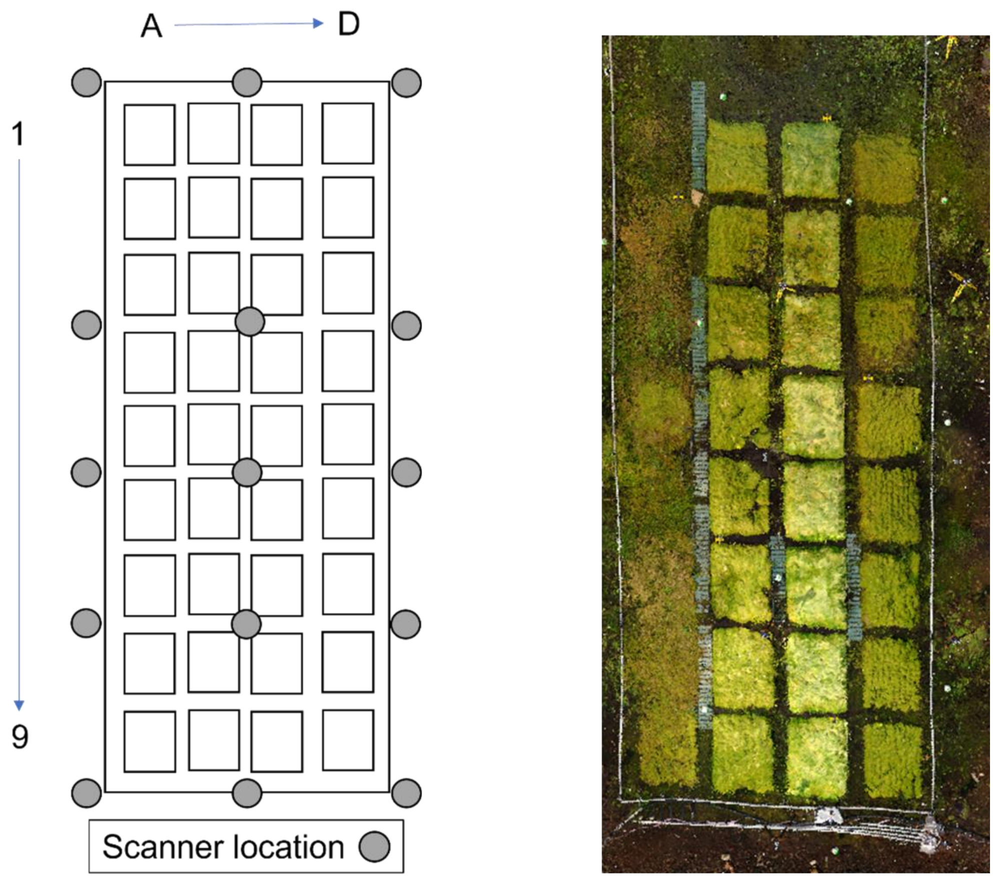


Figure 50: Typical TLS scanner set up to enable complete and accurate data capture (left) with example TLS output viewed from above (right).

Two key limitations of the method are that the scanner cannot be used in the rain as raindrops on the TLS camera, mirror and laser emitter reduce the accuracy, and pooled water in plots would reflect laser beams and show as a blank area – so the scans could not be carried out immediately following heavy rainfall. This proved to be something of an issue at Little Woolden, given the rainfall pattern within the Manchester area, thereby influencing days or times suitable for scanning, but it proved possible to work round this issue by undertaking other tasks while conditions were unsuitable for scanning. It is worth noting, however, that scanning is possible in the dark, so delays which led to fieldwork extending into late evening did not prevent scanning from being completed successfully.

TLS data analysis

All data processing took place in Faro Scene 7.5.3.610, Arc Map 10.8, LAStools and Cloud Compare v2.

Point-cloud assembly in Faro Scene

3D point-clouds were assembled for each irrigation area, at each site, for each of the three scanning occasions using Faro Scene software. Once the scans were loaded into the software, the small reference spheres were identified using the identify function and manually checked. This manual approach was used in preference to relying solely on the identify function provided by the software because occasionally the software misidentified another rounded object as a sphere (such as a person's head). Edge artifact, and Stray and Distance filters were applied to each scan to further reduce any scan errors from stray points in the scan. The filters were applied using default settings for edge and stray. However, a distance filter of 25m was applied as this retained a good overlap between scanner locations while also removing points far outside the area of interest. An additional benefit of doing so was a reduced project file size and increased data storage efficiency.

Once filters were applied, scans were ‘placed’ using the reference spheres. This snapped all the individual scans into the correct location and orientation. This process gave a mean error of, typically, less than 3 mm when multiple scans were combined using the reference spheres.

Following the placement of scans, colour was applied to the point-cloud using the 360^{o} photos taken by the scanner. An initial point cloud was then created.

3D point clouds operate in an arbitrary space with x, y and z coordinates for each laser point obtained. The three scans for February 2019, September 2019 and August 2020 were transformed into a common coordinate space relative to the initial baseline scan coordinates for February 2019. The timeline of each scan and the age of Sphagnum in each are presented in Appendix A3. This was achieved by taking additional manual reference points from fixed objects in Scan 1 which were then used to ‘force a correspondence’ by applying the Feb 19 coordinates to matching locations in the Sep 19 and August 2020 scans within Faro Scene. The mean scan-point error arising from forced correspondence was also calculated to ensure that the distortions arising from this transformation were both small and acceptable.

Once the 3D point clouds had a common coordinate system, areas of interest (individual *Sphagnum* plots as described in Chapter 6) could be extracted from the point cloud using the clip box tool in Faro Scene. The same coordinates for individual clip boxes were used across all three scans for their corresponding plots. This allowed for direct comparison of spatially accurate data across multiple years.

Individual plots were then exported as .las files, coded by plot location, site and date, .las files are the industry standard for laser datasets (ESRI, 2020). Cloud Compare was used as a final sense-check to confirm individual plots had been exported with the correct location code, this removed any risk of mislabelled plots and subsequent errors in additional analysis.

De-weeding plots using LAStools

Like many other *Sphagnum* farming projects to date weed growth in the *Sphagnum* plots became a minor issue. Despite regular weeding across all sites, at the point of data capture, some individual plots were found to have a high vascular plant cover. Vascular plants (such as *Cirsium vulgare, Juncus effusus and Urtica dioica*) were typically seen to grow through the established *Sphagnum* carpet, thereby blocking the view of *Sphagnum* beneath. This was a common problem already referred to when describing Methods 1 and 2. Fortunately, the TLS scanning approach offers a possible solution to this challenge. Specifically, a method was devised to 'strip' the vascular weeds from the scans using a .las file manipulator called LAStools.

LAStools is a software package designed to analyse aerial based LiDAR data (laser data obtained from the air, via satellite, plane or UAV). The 'lasground tool' within LAStools enables laser-based data to be classified into laser points that are associated with ground points or objects above ground level.

As *Sphagnum* in the experimental plots had a low growth form, it was possible to classify the *Sphagnum* areas as 'ground points'. Most weeds sitting above the *Sphagnum* layer could be classified as 'objects above ground level'. This was achieved through fine tuning of the lasground tool. The step size (output resolution) and offset (height above ground points) were found to be the key aspects of the algorithm manipulation process to achieve this. Finally, the las to las tool within LAStools was used to remove the vascular plant cover. The result left a clearer dataset with reduced vascular plant cover (for example see Figure 51)

Figure 51 Plot BS7 in September 2019 showing de-weeding results, the original plot as scanned (top) compared to de-weeded plot (bottom) the de-weeding process removes vascular species from scan data, in this case *Juncus effusus*. This reduces the impact of vascular plant heights on mean vegetation height during analysis in Arc GIS.

Growth Results using Arc GIS

The TLS data of themselves do not generate volume data for the *Sphagnum* crop, they merely create a 3D surface for each plot. To generate volume data, it was decided that the most appropriate method would be to import the TLS results into ESRI's ArcMap as las datasets. This is because ArcGis software can calculate volumes between differing layers. Every plot at each site was therefore imported into ESRI's ArcMap.

Given that the combined use of TLS data in combination with ArcMap has not until now been used as a method of determining paludiculture crop volume, it is probably helpful to set out the precise sequence of this novel method.

Las datasets were converted to Raster images using the conversion tool 'las dataset to raster'. This was conducted using the following settings:

Value field: Elevation

Interpolation type: Binning

Cell assignment: Average

Void fill: linear

Output data: float

Sampling type: cell size

Sampling value: 0.01 (1 cm^2)

Z factor:1

The output of this conversion step was a raster dataset for each plot, at each scan occasion with a spatial resolution of 1 cm^2

As mentioned in the TLS data processing section earlier, the mean scan error of an individual scan was typically less than 3 mm. An additional check was used to ensure an acceptable margin of error when comparing multiple scans. This was conducted by comparing the ground level from a subsample of plots at each irrigation area. To achieve this comparison, point shapefiles were created in Arc Map for the selected scans. These were placed along the outermost edges of the plots. Ten points per side were used, and occasionally a side was omitted from the point creation if a clear obstruction was visible in the raster file – for example one side could be obscured by portable boardwalk in one or more scans for a plot.

By using the 'extract multi-values to points' tool in Arc Map Spatial Analyst, the elevation of each ground level point could be obtained across multiple scans.

The difference between Scans 1-2 and Scans 2 – 3 were calculated using the 'field calculator' function. It was determined that if the total mean difference between scans was greater than 10 mm then the transformation within Faro Scene was considered to have generated an unacceptable level of distortion and thus the forced data correspondence step within the TLS software may have to be revisited. However, following checks this was not necessary for any scans at Little Woolden, but Scan 1 for Sharpley at both irrigation areas could not be aligned correctly (see Table 26)

Once satisfied that the ground level was within an acceptable margin of error, the difference between scans were calculated using the 'Map Algebra' function. Differences were calculated with the functions:

Scan 3 (August 2020) minus Scan 1 (February 2019)

Scan 2 (September 2019) minus Scan 1 (February 2019)

Scan 3 (August 2020) minus Scan 2 (September 2019)

The outputs of these functions were new raster images, 'difference layers'. These difference layers provided the height-change for every square cm within each plot across each scan.

The gel plots were, however, necessarily processed differently. At the Little Woolden site the founder material was applied later than the plugs, and therefore did not appear in Scan 1, only appearing in Scan 2 and 3. As scan 1 was used for the baseline scan for analysis, a proxy Scan 1 was obtained by creating a 1 cm^2 resolution interpolation of the September 2019 ground level for the gel plots based on the reasonable assumption that the sprayed gel in effect had zero thickness when first applied. Changes from this ground level were used to account for the growth since planting in April 2019.

A similar process was undertaken for the straw covered plug plots in February 2019, where an interpolation of the February 2019 ground surface was used as a proxy for Scan 1 as the subsequently applied straw cover prevented an accurate capture of plugs beneath the straw mulch layer during Scan 1.

Once the difference calculations had been performed, a polygon shapefile was created to map the *Sphagnum* extent in September 2019 and August 2020. The final shapefile(s) were then used as a mask layer within the extract by 'mask function' in Arc Map to clip out just the area of *Sphagnum* within the chosen difference layer.

Finally, the classification statistics for each of the difference clip layers could be extracted based on the *Sphagnum* extent. The resulting Digital Elevation Model (DEM) allowed mean height change for the *Sphagnum* to be expressed for each 1 cm^2 pixel within the defined *Sphagnum* extent. The classification statistics were used to calculate the mean percentage area and the growth rate (volume of *Sphagnum*) in each plot.

The TLS/Arc analysis provides a volume of fresh *Sphagnum* within each plot. The overall volumetric yield obtained across each plot up to Scan 3 (August 2020) was then converted to a dry mass yield in tonnes of *Sphagnum*. This was based on a conversion using standard DBD values from Clymo and Hayward (1982) for natural acrotelm vegetation, together with values from Wichmann *et al.* (2020) for *Sphagnum* processed for growing media post-harvest.

Clymo (1973) challenged the assumption that an increased *Sphagnum* length results in a direct increase in bulk density, an important factor when converting volume to yield. Figure 1 in Clymo's paper shows that *Sphagnum* moss with the greatest length increment did not achieve the highest bulk density measurement. The two measures of growth vary in different ways. In general, if the plants were shaded and/or the water table was high, growth in terms of weight was lower than growth in length. Growth in length, however, was reduced only when the water table was low, and the shade was dense. The combined effect was that the plants became ‘stragglier’ in shade and/or high-water table. Therefore, the use of length alone without considering plant bulk density can lead to erroneous conclusions.

Clearly, *Sphagnum* weight and volume relationships must be considered when calculating yields of *Sphagnum* biomass. The standard method for reconciling weight and volume is bulk density – i.e. the weight per volume of the *Sphagnum*, expressed in grams per unit volume. In undisturbed acrotelm peat the bulk density falls into the range of 0.03 – 0.09 g cm^{-3} , equivalent to 30 – 90 g per litre (Lindsay, 2010). The lower value of this range, g cm^{-3}, is used to determine ‘in the ground’ yield calculations, to provide a conservative estimate.

The overall *Sphagnum* yield was then standardised into the yield in tonnes per ha per year to allow comparison across all datasets. This is in keeping with the wider literature (Gaudig *et al.*, 2014; Pouliot, Hugron and Rochefort, 2015; Wichmann, Prager and Gaudig, 2017) and allows for comparison across a number of *Sphagnum* farming sites where *Sphagnum* yield in tonnes ha^{-1} yr^{-1} are presented. A potential limitation of this approach is that it assumes *Sphagnum* growth is linear and distributed evenly per month. It may also result in a bias towards higher yields where plots have experienced more than one growing season.

9.3 Results

Many results were generated via the pixel count method. However, these were superseded in terms of accuracy and usefulness by the TLS data and have therefore only been presented earlier in Figure 49 as examples to show the capabilities, but also the limitations, of the method. The pixel count method was also limited in that it could only provide a measure of percentage cover and therefore an area measurement. Without the addition of a height measurement it would have been impossible to calculate a volume and an indicative yield of *Sphagnum*. It may however be a useful method where a TLS is not available, and in combination with *Sphagnum* depth measurements could be used to calculate a volume based on area numbers.

9.3.1 How accurate was the TLS method?

For each scan period, multiple laser scans were combined to produce a TLS dataset. TLS Scanners capture distances with mm accuracy within scans, which allowed entire monitoring areas to be assessed. Each TLS scan consisted of millions of individual data points; however, it is important to quantify the scan error to have confidence in the measurements. The TLS software reports the error within each individual scan; similarly, when multiple scans are processed into one point cloud, a mean scan alignment error is generated. The mean error recorded for each point cloud at each scanning occasion was used as the original scan alignment error. When multiple scans at the same irrigation area were forced into a common coordinate space and used for volume calculation, a new mean error was calculated which is used to calculate the percentage error.

The formula below was used to calculate the maximum, relative and percentage error based on the mean scan error for each TLS dataset, when multiple scans were forced into a common coordinate space and used for volume calculation.

Side length **x** (mm)

Volume **v** = x^3 (mm^3)

Max error **dv** = 3$\mathbf{x}^2$**dx** where **d** is the deviation i.e. mean error in mm

Relative error = **dv/v**

Percentage error = **dv/v** * 100

The error results are presented in Table 26 below.

Table 26: Percentage error introduced by comparing TLS Scans across multiple timeframes when forced into a common coordinate space.

scan number and date	original scan alignment error (mm)	**(dx)** mean error for snapped scans	**x** (1m side length in mm)	**v** **(**volume in mm^3)	**dv** (mean error)	**dv/v** (relative error)	**%** error on 1 m^3
Little Woolden Spray							
Scan 3 - Aug 20	2.36	7.00	1000	10^9	21000000	0.02100	2.10
Scan 2 - Sep 19	2.32	5.84	1000	10^9	17520000	0.01752	1.75
Scan 1 - Feb 19	2.73	used for coordinates	n/a	n/a	n/a	n/a	n/a
Little Woolden Drip							
Scan 3 - Aug 20	2.36	7.01	1000	10^9	21030000	0.02103	2.10
Scan 2 - Sep 19	2.98	4.63	1000	10^9	13890000	0.01389	1.39
Scan 1 - Feb 19	2.96	used for coordinates	n/a	n/a	n/a	n/a	n/a
Sharpley Spray							
Scan 3 - Aug 20	2.45	6.37	1000	10^9	19110000	0.01911	1.91
Scan 2 - Sep 19	4.47	used for coordinates	n/a	n/a	n/a	n/a	n/a
Scan 1 - Feb 19	n/a	base scan problems – not used	n/a	n/a	n/a	n/a	n/a
Sharpley Drip							
Scan 3 - Aug 20	3.05	6.24	1000	10^9	18720000	0.01872	1.87
Scan 2 - Sep 19	3.61	used for coordinates	n/a	n/a	n/a	n/a	n/a
Scan 1 - Feb 19	n/a	base scan problems – not used	n/a	n/a	n/a	n/a	n/a

Table 26 shows that completed TLS datasets for single scans were aligned with an error of 2.36 to 4.4 mm. Forcing datasets into a coordinate space relative to the first scan increased the mean margin of error slightly. The mean error in the TLS datasets with forced coordinates used for volume calculations ranged from 4.47 mm to 7.01 mm.

This resulted in an overall percentage error from 1.39% to 2.10 % for volume measurements per m^3. As this is a percentage, not a fixed numerical value, the same can be applied throughout the percentage covers and yields as they are all based on simple mathematics and conversions. Therefore, the TLS method accuracy can be stated as 97.8-98.6% in terms of area and volume measurements used for yield calculations – an acceptable degree of accuracy.

Bråkenhielm and Qinghong, (1995), compared three other techniques for percentage cover, namely visual estimation (VE), point frequency and sub-plot frequency. VE was found to have the highest accuracy, precision, and sensitivity over the other methods, but the authors found inter-person error with VE was slightly greater for small and wide-spread plants, especially mosses, than for other life-forms. It has been suggested that the accuracy of the visual estimation is reliable when plot size is less than 1 m^2 as larger areas are difficult to mentally integrate (Dethier *et al.*, 1993).

Other monitoring methods that have been used for vegetation are line-point intercept (LPI) and the permanent plot (PP) method. When compared directly on a restored Canadian peatland, the LPI was found to consistently overestimate percentage cover, whereas permanent plots were more useful over longer time periods requiring fewer monitoring and analysis days. The authors state that it is better to estimate cover values to the nearest percentage for accuracy (Rochefort *et al.*, 2013).

However, a quantified percentage accuracy for the above methods is not provided. Sutherland suggests that VE for percentage cover may result in interpersonal differences of < 20% (therefore accuracies are c. 80%) (Sutherland, 2006). So, it is reasonable to assume that the TLS accuracy of >95% is more accurate than the other methods, as it removes subjectivity, inter and intrapersonal bias.

Remote sensing of bryophytes is an emerging field, and as the interest in their ecosystem service potential has increased, so too has the need to observe and quantify their extent, condition, and productivity. Many peatlands are extremely large and therefore remote sensing is a sensible approach to deliver additional data. Peckham, Ahl and Gower (2009) have used airborne lidar and multispectral sensor imaging to assess bryophyte cover in boreal forested peatlands using multiple linear regression models. The study found that the models could explain 63–79% of ground truthed feathermoss cover and 69–92% of the *Sphagnum* cover. The error within the models was calculated at 3–15% making them 85-97% accurate at predicting feathermoss, *Sphagnum*, and total moss ground cover.

This adds legitimacy to the TLS accuracy estimates of more than 95% as the TLS method is, in essence, ground based lidar so the accuracy should be broadly comparable. Within the relatively simpler study design of the present study.

9.3.2 *Sphagnum* cover development across both sites

The *Sphagnum* plots generally all increased in percentage cover compared to the percentage covers at establishment, which were estimated based on the pixel counts referred to earlier. Percentage cover relative to plot size is important as the time to develop ‘canopy closure’ is more important for long term success than an individual *Sphagnum* plug/gel fragments growth.

The percentage cover at establishment was 6% for plugs when 120 plugs were applied per 3.4 m^2 plot, and 17-18% for gels when applied at a volume of 2 L per m^2 (6.8 L per plot). The difference in establishment percentage cover is due to the innate differences in the BeadaMoss products – plugs are compact, discrete raised mini hummocks, whereas gel on application is sprawling, with many small individual *Sphagnum* fragments covering a larger area.

By scan 3, September 2020, all covered plots at Little Woolden generally had a high percentage cover for both Plug (plugs) and Gel (gels).

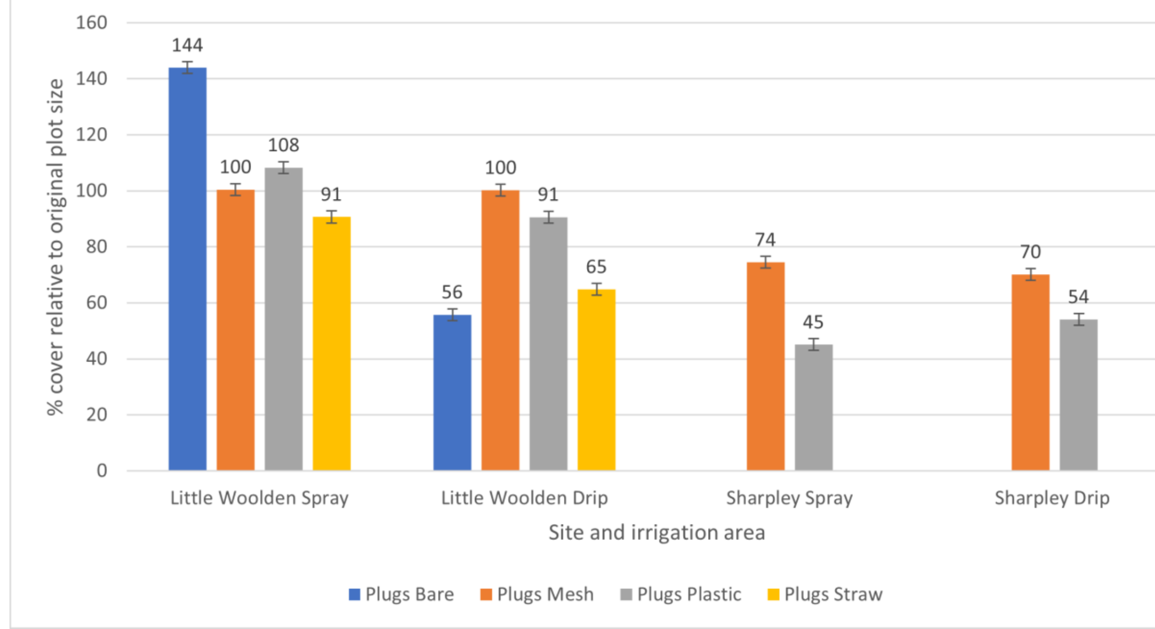


Figure 52: Mean Plug percentage cover relative to the original plot size of 3.4 m^2 across all sites by Scan 3. Please note that there are temporal differences between irrigation areas – with time post planting of 22 months and 24 months for Little Woolden and Sharpley respectively.

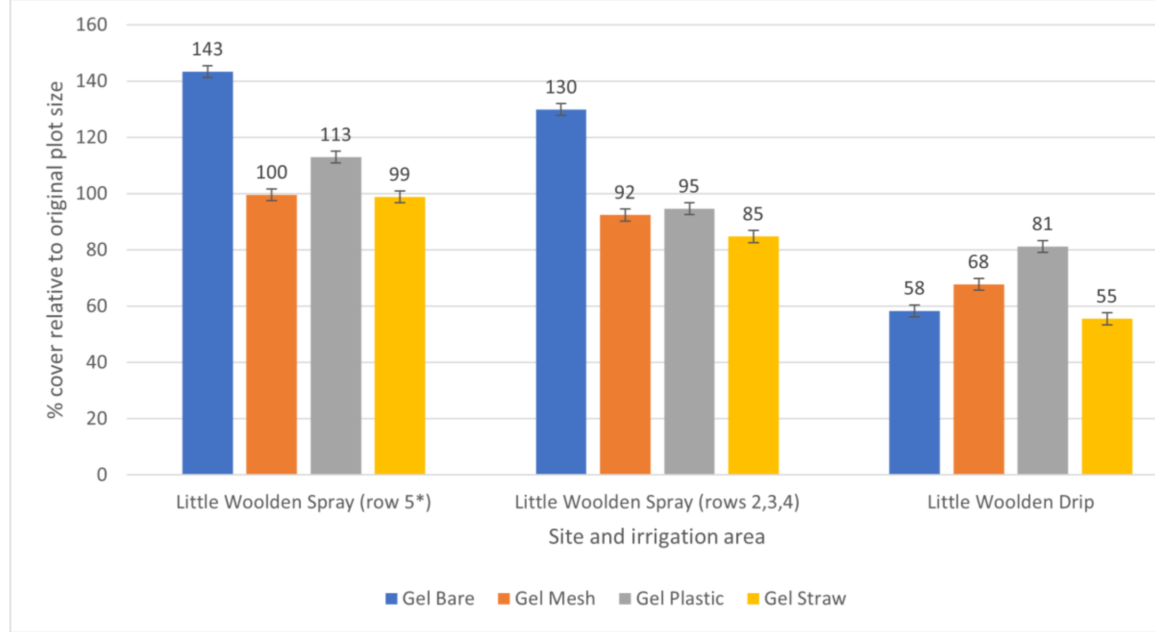


Figure 53: Mean Gel percentage cover relative to the original plot size of 3.4 m^2 across all sites by Scan 3. Please note that there are temporal differences between irrigation areas – with time post planting of 21 months (row 5) and 16 months for Rows 2,3,4 at Little Woolden as no Gel plots survived to the end of monitoring at Sharpley.

Sphagnum percentage cover is a key component for successful cultivation, with rapid establishment of cover an essential step to improved microclimate, drought resistance and a high biomass accumulation. The values obtained from the TLS data are expressed in percentage terms relative to the original planted plot area of 3.4m^2 shown in Figure 52 and Figure 53. The percentage area results should be regarded as conservative, as when defining *Sphagnum* areas within Arc GIS, great care was taken to create polygons within the *Sphagnum* area, leaving a small *Sphagnum* boundary outside the polygon. An example of this approach is given in Figure 54 for plot BS9, a plug plot with a mesh cover at the spray irrigation site at Little Woolden.

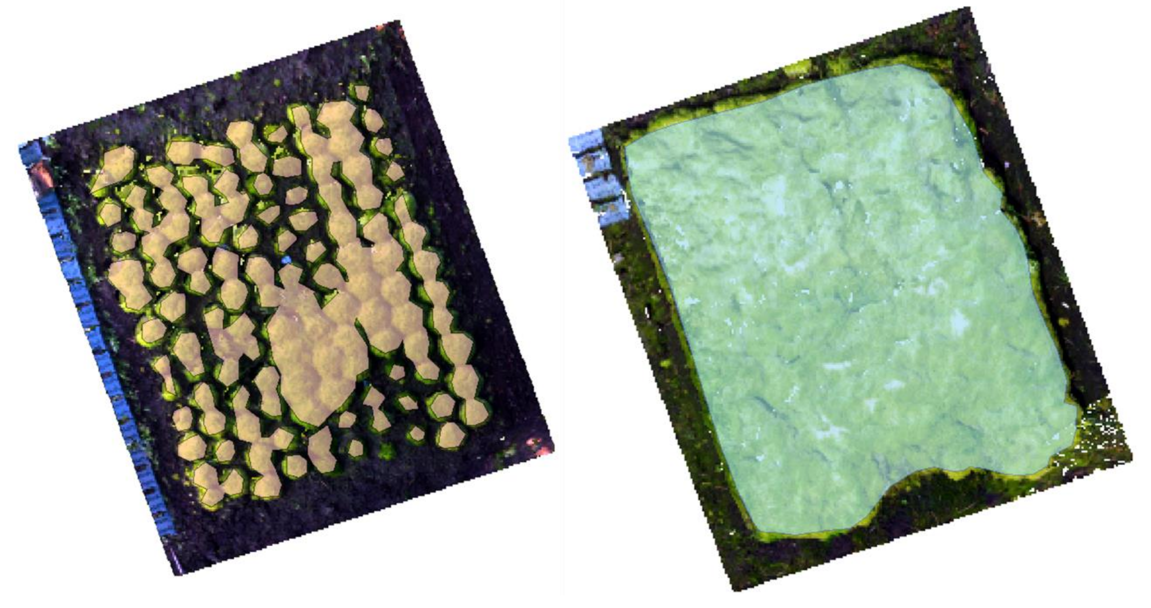


Figure 54: BS9, a Plug plot with a Mesh cover at the Spray irrigation site at Little Woolden showing conservative cover estimates when extracting *Sphagnum* extent via polygon mask. Scan 2 taken in September 2019 is on the left and Scan 3 taken in August 2020 is on the right. The percentage cover at Scan 2 for plot BM9 was 52.81% and 104% at Scan 3. Percentages greater than 100% are where the *Sphagnum* has outgrown the original plot area of 3.4 m^2.

At the Little Woolden site, the covered plug plots in the spray irrigation area achieved a high percentage cover after 11 months, the range across all mulch covers being 38-67% cover, with a mean of 53%. By 22 months all plots had a percentage cover greater than 85%. Some plots exceeded their original planted area, with mulch-covered plots as a whole achieving a range of 88 – 110% (mean of 99.81%) The drip irrigation area had a similar pattern, but slightly lower percentage cover results at 11 months. The range across all drip-irrigated mulch-covered plots was 13 – 57% (mean of 34%) and by 22 months this increased to 55 – 109% (mean of 85%) when compared to the original planting area.

Looking at individual cover types specifically for plug plots at Little Woolden, across both irrigation systems the no mulch plots had the highest percentage cover (143-144%). However, this is artificially high due to the lack of clear plot delineation in the uncovered plots by the final scan and should be regarded as overestimates. Typically, *Sphagnum* in covered plots had greater growth and was easier to demarcate. This gave a higher confidence in achieving reliable area calculations. *Sphagnum* performance is ultimately decided by volume of biomass and dry mass yield, which were substantially lower in the no mulch plots compared to covered plots. This was because *Sphagnum* carpet heights were lower, or thinner, in the no mulch plots.

On average, Mesh, Plastic- and Straw-covered plots achieved a slightly higher percentage cover in the spray irrigation area compared to the drip irrigation area at the Little Woolden site. This was not true for Sharpley, however, where plug plots achieved a greater % cover under mesh covers in the spray area, but under drip irrigation the plastic covers achieved higher percentage growth.

At the Sharpley site, only mesh- and plastic-covered plots seeded with plug material remained at the time of the final scan in 2022. At 24 months the percentage cover achieved a range of 12.33 – 86% and a mean of 60% cover in the spray irrigation area, while the drip irrigation area had a range of 31 – 99% with a mean of 62%. The mean percentage covers achieved by covered plots when looked at overall were similar. However, the mesh-covered plots achieved the higher percentage covers in both the drip and spray irrigation areas when compared to mesh at 74% vs 45% and 70% vs 54% respectively. Unfortunately, it is not possible to give a comparable data point for straw or no mulch plug plots at Sharpley for as by Scan 3 these plots had been cleared and pre-prepared for another round of planting.

9.3.3 Increase in Carpet thickness over time

The greatest increases in *Sphagnum* carpet thickness were found at the Little Woolden site, as seen in *Figure 55* where a maximum increase of 16 cm was observed at a mesh-covered plug plot (BS9).

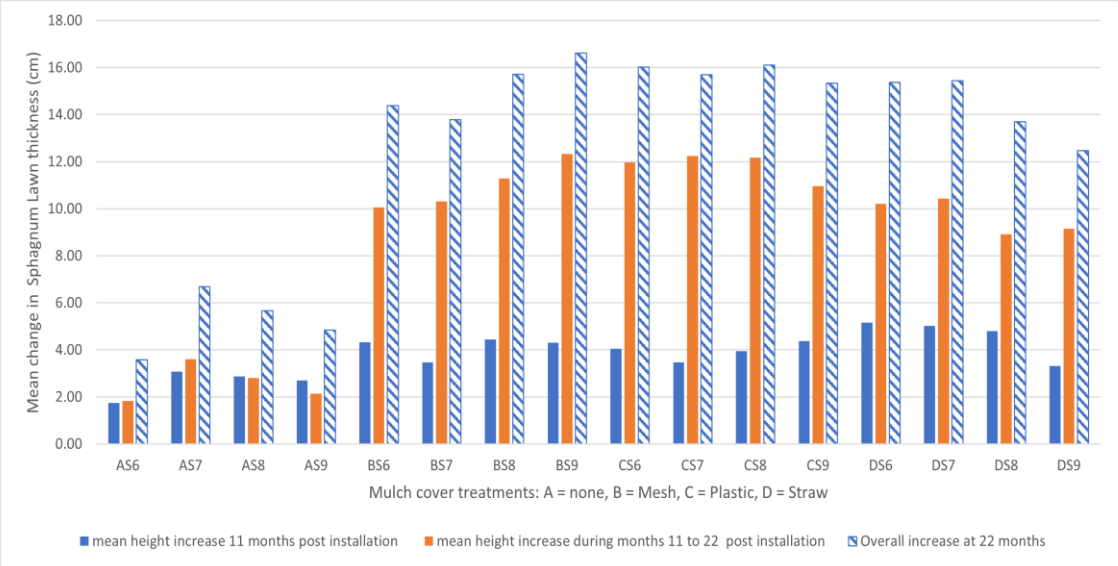


Figure 55: Mean *Sphagnum* carpet thickness change in cm for all plug plots at the Little Woolden site, Spray irrigation area by cover type. Plots coded A (No mulch) B (Mesh cover) C (Plastic cover) D (Straw cover). Height change is shown at 11 months post installation (change in mean height from Scan 1 to Scan 2), height change from Scan 2 to Scan 3, and the overall carpet thickness change since Scan 1, or an interpolated surface (straw plots only).

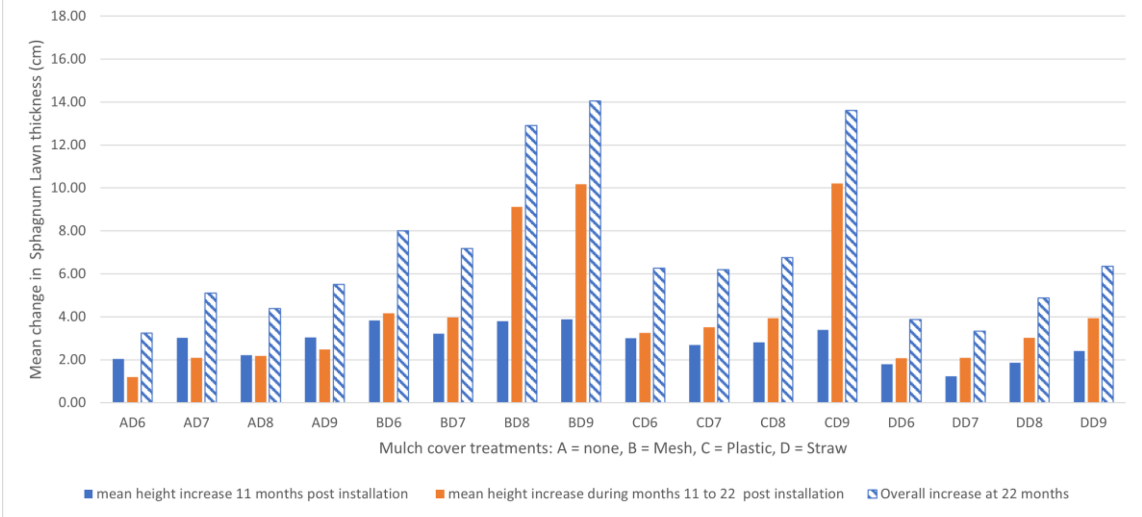


Figure 56: Mean *Sphagnum* carpet thickness change in cm for all plug plots at the Little Woolden site, Drip irrigation area by cover type. Plots coded A (No mulch) B (Mesh cover) C (Plastic cover) D (Straw cover). Height change is shown at 11 months post installation (change in mean height from Scan 1 to Scan 2), height change from Scan 2 to Scan 3, and the overall carpet thickness change since Scan 1, or an interpolated surface (straw plots only).

At Little Woolden the drip irrigation sector achieved a lower overall increase in *Sphagnum* carpet height compared to the spray irrigation sector shown in Figure 55 and Figure 56. Within the drip irrigation area, the three plots with the largest increase in overall *Sphagnum* height at 22 months were mesh plots BD8, BD9 and Plastic plots CD9 and were comparable in height-increase to the mesh, plastic and straw plots in the drip irrigation area. All other plots achieved a lower *Sphagnum* height-increase compared to their counterparts in the spray irrigation area. The averages (together with standard deviations) for plots in the drip-only irrigation area at 22 months were No mulch 4.56±0.86, Mesh 10.53± 2.99, Plastic 8.20± 3.12 and Straw 4.60±1.15. The no mulch plots at both sites had a similar mean height increase at 22 months achieving 5.19 cm vs 4.56 cm for the spray and drip irrigation sectors respectively. In contrast, the mulch covered plots in the spray irrigation area consistently produced greater height gains at 22 months compared to the drip irrigation.

9.3.4 Calculated *Sphagnum* yield

Tables 27 to 32 below, display the growth data in full as obtained by the TLS and GIS work for all the plots, with the in the ground yields determined mathematically by converting volume to mass using a range of DBD measurements. And annualising these results.

For example, during five years of cultivation at the Rastede site in Germany, the dry mass productivity at the Rastede *Sphagnum* farm in Germany resulted in mean values of 24 t ha-1 with a range of 15 to 34 t per ha. This gives annualised yields of 4.9 t ha-1 yr-1, with a range of 3.1 to 6.8 t per ha per year.

The theoretical 'in the field' yields given in this study are based on the lower estimates for natural acrotelm bulk densities for a conservative mathematical conversion from volume to yield.

A primary rationale for literature values being used are that final DBD values for each of the *Sphagnum* plots cultivated successfully to scan 3 have not been obtained. This is because testing for DBD is a destructive testing process, which requires at least 30L of material to undertake. Such an approach would require large portions of the *Sphagnum* plots to be removed. As the study the aims were to exploratively assess the potential for new irrigation methods and growth of micropropagated *Sphagnum*, and the study period did not cover the three-to-five-year timescale for harvest, destructively sampling the plots in the study period was not an option.

Furthermore, the act of converting *Sphagnum* from a raw material into a processed component of growing media (or other use) will involve steps such as drying and chopping which can substantially alter the finished products DBD.

For example, Wichmann (2020) reports a range of bulk densities for both Fresh and Dry *Sphagnum* taken from various *Sphagnum* species across multiple sites, gathered using a variety of processing and drying stages. The Fresh *Sphagnum* (fresh mass or fm) bulk densities vary considerably from 31 – 282 g FM per L, while the DBD vary from 12.3 to 47.8. the mean for fm bulk density is 200 g per L, while the mean DBD was 29 g per L.

For this reason, the processed estimate was calculated using the low and high dry bulk densities of processed *Sphagnum* from Wichmann *et al.* 2020. This approach has led to yields that appear higher than the published dry mass yields within the *Sphagnum* farming literature. So, the full data analysis is presented throughout tables 27 to 32 below.

Table 27 Little Woolden Spray irrigation area, TLS results for plug plots with percentage cover and yield data based on ‘in the ground’ and ‘processed’ DBD

Little Woolden Spray Irrigation area plug plots, percentage cover and yield data											
Sphagnum treatment	Mulch cover	Plot code	Original plot area size (m^2)	*Sphagnum* area count in September 2019 (m^2)	*Sphagnum* area count in August 2020 (m^2)	*Sphagnum* % cover in September 2019 as a % of original plot area	*Sphagnum* % cover in August 2020 as a % of original plot area	Total carpet thickness change by August 2020 (m)	Overall yield in tonnes per ha if using Acrotelm DBD of 30g per L	'In the Ground' Annual yield in T per ha using acrotelm DBD of 30g per L	'Processed' annual yield in T per ha (low DBD of 20g per L)	'Processed' annual yield in T per ha (high DBD of 38g per L)
Plug	No cover	AS6	3.40	0.4677	4.65	13.76	136.80	0.02	7.40	4.04	2.69	5.11
Plug	No cover	AS7	3.40	0.6361	4.35	18.71	127.99	0.04	10.82	5.90	3.93	7.47
Plug	No cover	AS8	3.40	0.7169	5.11	21.09	150.16	0.04	11.86	6.47	4.31	8.20
Plug	No cover	AS9	3.40	0.5477	5.47	16.11	160.84	0.05	14.35	7.83	5.22	9.92
Plug	Mesh	BS6	3.40	1.4537	3.03	51.00	89.00	0.13	38.53	21.02	14.01	26.62
Plug	Mesh	BS7	3.40	1.6652	3.42	48.98	100.59	0.13	38.47	20.98	13.99	26.58
Plug	Mesh	BS8	3.40	2.0581	3.68	60.53	108.33	0.14	43.37	23.65	15.77	29.96
Plug	Mesh	BS9	3.40	1.7957	3.54	52.81	104.00	0.15	45.54	24.84	16.56	31.46
Plug	Plastic	CS6	3.40	2.296	3.73	67.53	109.79	0.15	45.16	24.63	16.42	31.20
Plug	Plastic	CS7	3.40	2.1569	3.76	63.44	110.66	0.15	44.63	24.35	16.23	30.84
Plug	Plastic	CS8	3.40	1.9164	3.61	56.36	106.04	0.15	45.31	24.72	16.48	31.31
Plug	Plastic	CS9	3.40	2.003	3.62	58.91	106.60	0.14	42.66	23.27	15.51	29.48
Plug	Straw	DS6	3.40	1.3235	3.36	38.93	98.90	0.14	41.39	22.58	15.05	28.60
Plug	Straw	DS7	3.40	1.4129	3.00	41.56	88.17	0.14	41.65	22.72	15.14	28.77
Plug	Straw	DS8	3.40	1.2973	2.94	38.16	86.54	0.13	37.74	20.58	13.72	26.07
Plug	Straw	DS9	3.40	2.0588	3.03	60.55	89.03	0.12	35.52	19.37	12.91	24.54

Table 28: Little Woolden Spray irrigation area, TLS results for gel plots with percentage cover and yield data based on 'in the ground' and 'processed' DBD .

Little Woolden Spray irrigation area gel plots, percentage cover and yield data												
Sphagnum treatment	Mulch cover	Plot code	Original plot area size (m^2)	*Sphagnum* area count in September 2019 (m^2)	*Sphagnum* area count in August 2020 (m^2)	*Sphagnum* % cover in September 2019 as a % of original plot area	*Sphagnum* % cover in August 2020 as a % of original plot area	Total carpet thickness change by August 2020 (m)	Overall yield in tonnes per ha if using Acrotelm DBD of 30g per L	'In the Ground' Annual yield in T per ha using acrotelm DBD of 30g per L	'Processed' annual yield in T per ha (low DBD of 20g per L)	'Processed' annual yield in T per ha (high DBD of 38g per L)
Gel	No mulch	AS2	3.40	n/a	4.79	n/a	140.79	0.00	0.76	0.57	0.38	0.72
Gel	No mulch	AS3	3.40	n/a	3.65	n/a	107.22	0.02	6.80	5.10	3.40	6.46
Gel	No mulch	AS4	3.40	n/a	4.81	n/a	141.54	0.02	6.09	4.57	3.04	5.78
Gel	No mulch	AS5	3.40	1.85	4.87	54.30	143.33	0.07	21.56	12.32	8.21	15.61
Gel	Mesh	BS2	3.40	n/a	3.33	n/a	98.08	0.11	32.15	24.11	16.08	30.54
Gel	Mesh	BS3	3.40	n/a	2.71	n/a	79.80	0.10	30.65	22.99	15.33	29.12
Gel	Mesh	BS4	3.40	n/a	3.38	n/a	99.29	0.13	39.36	29.52	19.68	37.39
Gel	Mesh	BS5	3.40	2.24	3.39	65.92	99.56	0.13	38.72	22.13	14.75	28.03
Gel	Plastic	CS2	3.40	n/a	3.28	n/a	96.56	0.11	33.93	25.44	16.96	32.23
Gel	Plastic	CS3	3.40	n/a	3.20	n/a	94.16	0.07	20.46	15.34	10.23	19.44
Gel	Plastic	CS4	3.40	n/a	3.17	n/a	93.15	0.10	30.13	22.59	15.06	28.62
Gel	Plastic	CS5	3.40	3.05	3.84	89.81	112.93	0.16	47.72	27.27	18.18	34.54
Gel	Straw	DS2	3.40	n/a	3.21	n/a	94.31	0.10	28.70	21.53	14.35	27.27
Gel	Straw	DS3	3.40	n/a	3.19	n/a	93.78	0.09	26.13	19.60	13.07	24.83
Gel	Straw	DS4	3.40	n/a	2.25	n/a	66.15	0.08	24.40	18.30	12.20	23.18
Gel	Straw	DS5	3.40	3.05	3.36	89.73	98.81	0.14	42.28	24.16	16.11	30.60

Table 29: Little Woolden Drip irrigation area, TLS results for plug plots with percentage cover and yield data based on ‘in the ground’ and ‘processed’ DBD.

Little Woolden Drip Irrigation area plug plots, percentage cover and yield data												
Sphagnum treatment	Mulch cover	Plot code	Original plot area size (m^2)	*Sphagnum* area count in September 2019 (m^2)	*Sphagnum* area count in August 2020 (m^2)	*Sphagnum* % cover in September 2019 as a % of original plot area	*Sphagnum* % cover in August 2020 as a % of original plot area	Total carpet thickness change by August 2020 (m)	Overall yield in tonnes per ha if using acrotelm DBD of 30g per L	'In the Ground' Annual yield in T per ha using acrotelm DBD of 30g per L	'Processed' annual yield in T per ha (low DBD of 20g per L)	'Processed' annual yield in T per ha (high DBD of 38g per L)
plug	No mulch	AD6	3.40	0.45	1.33	13.29	39.09	0.03	9.71	5.30	3.53	6.71
plug	No mulch	AD7	3.40	0.84	2.71	24.79	79.79	0.05	15.30	8.35	5.56	10.57
plug	No mulch	AD8	3.40	0.34	1.25	10.05	36.64	0.04	13.15	7.17	4.78	9.09
plug	No mulch	AD9	3.40	0.54	2.30	15.76	67.53	0.06	16.54	9.02	6.01	11.43
Plug	Mesh	BD6	3.40	1.51	3.37	44.50	99.21	0.08	23.99	13.09	8.72	16.58
Plug	Mesh	BD7	3.40	0.91	3.00	26.66	88.29	0.07	21.55	11.75	7.83	14.89
Plug	Mesh	BD8	3.40	1.81	3.70	53.26	108.77	0.13	38.71	21.12	14.08	26.75
Plug	Mesh	BD9	3.40	1.93	3.56	56.63	104.61	0.14	42.15	22.99	15.33	29.12
Plug	Plastic	CD6	3.40	1.35	2.95	39.82	86.74	0.06	18.80	10.25	6.84	12.99
Plug	Plastic	CD7	3.40	0.89	3.00	26.17	88.25	0.06	18.58	10.13	6.76	12.84
plug	Plastic	CD8	3.40	0.97	2.85	28.39	83.71	0.07	20.25	11.05	7.37	13.99
Plug	Plastic	CD9	3.40	1.96	3.53	57.66	103.84	0.14	40.80	22.26	14.84	28.19
Plug	Straw	DD6	3.40	0.47	1.87	13.79	54.92	0.04	11.59	6.32	4.22	8.01
Plug	Straw	DD7	3.40	0.73	2.12	21.50	62.48	0.03	9.97	5.44	3.63	6.89
Plug	Straw	DD8	3.40	0.87	2.39	25.71	70.39	0.05	14.62	7.97	5.32	10.10
plug	Straw	DD9	3.40	0.59	2.43	17.47	71.53	0.06	19.03	10.38	6.92	13.15

Table 30: Little Woolden Drip irrigation area, TLS results for gel plots with percentage cover and yield data based on ‘in the ground’ and ‘processed’ DBD.

Little Woolden Drip Irrigation area gel plots, percentage cover and yield data												
Sphagnum treatment	Mulch cover	Plot code	Original plot area size (m^2)	*Sphagnum* area count in September 2019 (m^2)	*Sphagnum* area count in August 2020 (m^2)	*Sphagnum* % cover in September 2019 as a % of original plot area	*Sphagnum* % cover in August 2020 as a % of original plot area	Total carpet thickness change by August 2020 (m)	Overall yield in tonnes per ha if using acrotelm DBD of 30g per L	'In the Ground' Annual yield in T per ha using acrotelm DBD of 30g per L	'Processed' annual yield in T per ha (low DBD of 20g per L)	'Processed' annual yield in T per ha (high DBD of 38g per L)
Gel	No mulch	AD2	3.4	n/a	1.33	n/a	29.14	0.01	-6.53	-4.90	-3.27	-6.21
Gel	No mulch	AD3	3.4	n/a	2.71	n/a	79.42	0.03	-4.00	-3.00	-2.00	-3.80
Gel	No mulch	AD4	3.4	n/a	1.25	n/a	66.02	0.01	-6.95	-5.21	-3.47	-6.60
Gel	No mulch	AD5	3.4	n/a	2.30	n/a	58.92	0.02	-4.65	-3.49	-2.33	-4.42
Gel	Mesh	BD2	3.4	n/a	3.37	n/a	81.54	0.03	9.05	6.78	4.52	8.59
Gel	Mesh	BD3	3.4	n/a	3.00	n/a	74.59	0.03	13.91	10.43	6.95	13.21
Gel	Mesh	BD4	3.4	n/a	3.70	n/a	48.33	0.04	7.66	5.74	3.83	7.27
Gel	Mesh	BD5	3.4	n/a	3.56	n/a	66.51	0.04	5.69	4.26	2.84	5.40
Gel	Plastic	CD2	3.4	n/a	2.95	n/a	91.30	0.03	18.95	14.22	9.48	18.01
Gel	Plastic	CD3	3.4	n/a	3.00	n/a	77.67	0.03	15.16	11.37	7.58	14.40
Gel	Plastic	CD4	3.4	n/a	2.85	n/a	80.14	0.03	8.27	6.20	4.13	7.86
Gel	Plastic	CD5	3.4	n/a	3.53	n/a	75.79	0.04	12.35	9.26	6.17	11.73
Gel	Straw	DD2	3.4	n/a	1.87	n/a	71.21	0.02	12.77	9.57	6.38	12.13
Gel	Straw	DD3	3.4	n/a	2.12	n/a	40.04	0.02	6.71	5.03	3.35	6.37
Gel	Straw	DD4	3.4	n/a	2.39	n/a	46.75	0.02	4.85	3.64	2.43	4.61
Gel	Straw	DD5	3.4	n/a	2.43	n/a	63.97	0.02	9.67	7.25	4.84	9.19

Table 31: Sharpley site, Spray irrigation area, TLS results for plug plots with percentage cover and yield data based on ‘in the ground’ and ‘processed’ DBD.

Sharpley site Spray irrigation area, plug plots, percentage cover and yield data												
Sphagnum treatment	Mulch cover	Plot code	Original plot area size (m^2)	*Sphagnum* area count in September 2019 (m^2)	*Sphagnum* area count in August 2020 (m^2)	*Sphagnum* % cover in September 2019 as a % of original plot area	*Sphagnum* % cover in August 2020 as a % of original plot area	Total carpet thickness change by August 2020 (m)	Overall yield in tonnes per ha if using acrotelm DBD of 30g per L	'In the Ground' Annual yield in T per ha using acrotelm DBD of 30g per L	'Processed' annual yield in T per ha (low DBD of 20g per L)	'Processed' annual yield in T per ha (high DBD of 38g per L)
Plug	Mesh	BS1	3.4	n/a	2.793	n/a	82.16	0.12	34.69	17.35	11.56	21.97
Plug	Mesh	BS2	3.4	n/a	2.925	n/a	86.04	0.10	29.54	14.77	9.85	18.71
Plug	Mesh	BS3	3.4	n/a	2.047	n/a	60.21	0.06	18.16	9.08	6.05	11.50
Plug	Mesh	BS4	3.4	n/a	2.361	n/a	69.43	0.04	13.46	6.73	4.49	8.53
Plug	Plastic	CS1	3.4	n/a	2.650	n/a	77.95	0.14	42.78	21.39	14.26	27.09
Plug	Plastic	CS2	3.4	n/a	2.005	n/a	58.97	0.14	42.93	21.46	14.31	27.19
Plug	Plastic	CS3	3.4	n/a	0.419	n/a	12.33	0.10	29.93	14.96	9.98	18.95
Plug	Plastic	CS4	3.4	n/a	1.053	n/a	30.96	0.11	31.76	15.88	10.59	20.11

Table 32: Sharpley site, Drip irrigation area, TLS results for plug plots with percentage cover and yield data based on ‘in the ground’ and ‘processed’ DBD.

Sharpley site, Drip irrigation area, plug plots, percentage cover and yield data												
Sphagnum treatment	Mulch cover	Plot code	Original plot area size (m^2)	*Sphagnum* area count in September 2019 (m^2)	*Sphagnum* area count in August 2020 (m^2)	*Sphagnum* % cover in September 2019 as a % of original plot area	*Sphagnum* % cover in August 2020 as a % of original plot area	Total carpet thickness change by August 2020 (M)	Overall yield in tonnes per ha if using acrotelm DBD of 30g per L	'In the Ground' Annual yield in T per ha using acrotelm DBD of 30g per L	'Processed' annual yield in T per ha (low DBD of 20g per L)	'Processed' annual yield in T per ha (high DBD. of 38g per L)
Plug	Mesh	BD2	3.4	n/a	2.010	n/a	59.11	0.08	24.33	12.16	8.11	15.41
Plug	Mesh	BD3	3.4	n/a	3.390	n/a	99.71	0.06	18.64	9.32	6.21	11.81
Plug	Mesh	BD4	3.4	n/a	3.067	n/a	90.21	0.11	31.65	15.82	10.55	20.04
Plug	Mesh	BD5	3.4	n/a	1.074	n/a	31.59	0.05	16.41	8.20	5.47	10.39
Plug	Plastic	CD2	3.4	n/a	0.868	n/a	25.52	0.01	3.62	1.81	1.21	2.29
Plug	Plastic	CD3	3.4	n/a	2.876	n/a	84.58	0.11	32.30	16.15	10.77	20.46
Plug	Plastic	CD4	3.4	n/a	2.010	n/a	59.12	0.08	24.29	12.15	8.10	15.39
Plug	Plastic	CD5	3.4	n/a	1.603	n/a	47.14	0.05	16.44	8.22	5.48	10.41

9.4 Discussion about Growth data

9.4.1 Novel mulch performance

Core hypothesis 3: Alternative mulches will result in greater growth than the MLTT straw mulch. Additionally, cover treatments will provide greater growth than the no mulch plots.

> **Growth hypothesis 1**: The no mulch plots will produce lower yield of *Sphagnum* compared to mulch covered plots.
>
> **Growth hypothesis 2**: Mesh and Plastic mulch covers will result in greater yields than the MLTT straw mulch.

ANOVA tests were used to compare the mean annual 'in the field' yields in dry matter tonnes ha^{-1} yr^{-1} across irrigation sites and mulch covers. This test was chosen as ANOVA allows for the comparison of more than two groups at a time and determines whether there are any statistically significant differences between the means of the multiple groups. However, additional post hoc testing is required to determine which specific group's means when compared to each other are different. Due to the low volume of statistical tests required, ANOVA tests were performed on the online platform www.astatsa.com designed for quick, but rigorous statistical output. The online calculator also provides the MS Excel steps or R code required to verify their statistical tests. The low sample numbers, i.e., no more than 4 plots per mulch cover at each irrigation area meant that there may be less statistical power than in a large sample, though they offer useful discussion points.

For plug plots at Little Woolden under spray irrigation, ANOVA results shown in Table 33 revealed that the No mulch plots had significantly lower yields than Mesh, Plastic and Straw covered plots ($P < 0.05$), whereas the three mulch types (Mesh, Plastic and Straw) had mean yields that were not significantly different from each other. This same relationship was found for the Little Woolden gel plots across both drip and spray irrigation areas (P <0.05).

Interestingly, the plug plots under drip irrigation at Little Woolden revealed a different relationship. The no mulch plots were not significantly different from straw or plastic covered plots, whereas mesh plots achieved a significantly higher yield in dry matter tonnes ha^{-1} yr^{-1} than no-mulch control plots.

At Sharpley a full and balanced comparison was not possible as the No mulch plots and Straw plots did not survive to the final scan. However, a reduced ANOVA to compare Mesh and Plastic plug pairs was performed. The small number of independent treatments means that the calculator acts as a t test. There were no significant differences between the yield of mesh and plastic covers under drip or spray irrigation treatments, which mirrors the results of Little Woolden plugs under spray irrigation, Little Woolden gel subject to spray irrigation and Little Woolden gel under drip irrigation.

Table 33 One way ANOVA results comparing the mean yield in DM t ha^{-1} yr^{-1} for all mulch cover treatments against each other across both irrigation areas at Little Woolden.

Irrigation area Little Woolden	Mulch type (over plugs)	Tukey HSD inferfence based on mean yield DM t ha-1 yr -1				ANOVA results
		vs gel bare	vs gel mesh	vs gel plastic	vs gel straw	
Spray	No mulch	insignificant	** p<0.01	** p<0.01	** p<0.01	F value: 39.2292 ***p<0.001
Spray	Mesh	** p<0.01	insignificant	insignificant	insignificant	
Spray	Plastic	** p<0.01	insignificant	insignificant	insignificant	
Spray	Straw	** p<0.01	insignificant	insignificant	insignificant	
Drip[	No mulch	** p<0.01	insignificant	insignificant	insignificant	F value: 12.38 ***p<0.001
Drip	Mesh	** p<0.01	** p<0.01	insignificant	** p<0.01	
Drip	Plastic	** p<0.01	insignificant	insignificant	insignificant	
Drip	Straw	** p<0.01	insignificant	insignificant	insignificant	

Notes: 1: all plug treatments in both drip and spray sectors are significantly different compared to no-mulch gels with the exception of no-mulch plugs vs no-mulch gels under spray irrigation; 2: Plug no-mulch plots are significantly different to gel mesh, gel plastic and gel straw; 3: under drip irrigation there are no significant differences between plug and gel plots in terms of yield except for plug mesh vs gel mesh and plug mesh vs gel straw

Based on the one-way ANOVA statistics it is possible to say that in the majority of cases the no mulch plots produced a lower yield of *Sphagnum* than plots with cover treatments, thereby demonstrating the benefits of a mulch cover for maximising *Sphagnum* yield. Plugs under drip irrigation were an exception to this finding, which is interesting and worth exploring further if it means that plug-planting might be capable of doing well without the cost and time-consuming process of covering the crop with mulch. However, the result may simply have arisen from the large standard variation in yield for plastic covered plots combined with the low mean yield for straw plots, resulting in no significant difference in yields compared to the no-mulch control plots. This should be explored further, given the potential savings in production costs.

The Sharpley site sheds no light on this question, at least not statistically, because the plots became extremely weed-covered and were cleared by Micropropagation Services Ltd in preparation for a subsequent trial, thus being removed from the current experimental work.

9.4.2 How did the MLTT straw mulch perform?

At Little Woolden, Mesh and Plastic mulch covers were compared with straw mulch, based on plug plots under spray irrigation and gel under both drip and spray irrigation. No significant differences were recorded, except for the plug plots under drip irrigation, where mesh-covered plots showed significantly higher yields compared to straw covered plots. Unfortunately, Sharpley provides no information on this question because actions by our MPS partners resulted in loss of that particular experimental sector.

On balance, it appears that when the DBD of the *Sphagnum* layer is assumed to be 30 g per litre, there are no major differences between mean yields across mesh, plastic and straw covered plots in most cases. However, the MIFA irrigation system combined with the new mulch covers appear to offer a significant benefit on shallow peat (Sharpley) and for plug plots under drip irrigation.

9.4.3 Performance of drip irrigation versus spray irrigation

The question of whether drip irrigation or spray irrigation would prove to be the better system to support the MIFA approach was not a specified question in the original shaping of hypotheses. This was mainly because, so little was known about the relative benefits of drip irrigation versus spray. However, the present research programme is able to shed some light on this question.

From analysis of the data available, it appears that on thin peat such as Sharpley, the use of spray irrigation with plug founder material and a mesh mulch could be the best option (see Table 34)

Table 34 ANOVA results comparing Drip vs Spray for Sharpley Plugs

Irrigation	Mulch type	Sharpley mesh plugs	Sharpley plastic, plugs	F value
Spray	Mesh plugs	** $p<0.01$	insignificant	11.81
Spray	Plastic plugs	** $p<0.01$	insignificant	$p < 0.001$
		Sharpley mesh, plugs	**Sharpley plastic, plugs**	
Drip	Mesh plugs	insignificant	insignificant	1.50
Drip	Plastic plugs	insignificant	insignificant	$p = 0.266$

Core hypothesis 4: <u>The growth of *Sphagnum* plugs will exceed that of *Sphagnum* gel</u>.

> **Growth hypothesis 3**: The yield of *Sphagnum* plugs will exceed that of *Sphagnum* gel.

Plugs are larger and more established when first planted. For this reason, it was assumed that they would have greater resilience to changing environmental conditions and therefore achieve a higher yield compared to gel plots.

One-way ANOVA tests were conducted to compare the mean yields in dry mass t ha^{-1} yr^{-1} for Plug vs Gel founder material and plot mulch cover material. The use of dry mass t ha^{-1} yr^{-1} was necessary to attempt to control the time differences between plots, which were 22 months for Plug and 16 months for

Gel plots. Separate ANOVA tests were used for each irrigation area at Little Woolden and are presented in Table 35 with the Tukey HSD test inferences.

Table 35: Results from ANOVA and Tukey HSD post hoc test when comparing Plug and Gel mean yields in Dry mass t ha^{-1} yr^{-1} across all cover types at the Little Woolden spray and drip irrigation areas.

Irrigation area	**Mulch type**	**Tukey HSD inference based on mean yield DM t ha^{-1} yr^{-1}**				**ANOVA results**
		vs gel, no mulch	**vs gel, mesh**	**vs gel, plastic**	**vs gel, straw**	
Spray	No mulch	insignificant	** p<0.01	** p<0.01	** p<0.01	F value: 39.2292 ***p<0.001
Spray	Mesh	** p<0.01	insignificant	insignificant	insignificant	
Spray	Plastic	** p<0.01	insignificant	insignificant	insignificant	
Spray	Straw	** p<0.01	insignificant	insignificant	insignificant	
Drip	No mulch	** p<0.01	insignificant	insignificant	insignificant	F value: 12.38 ***p<0.001
Drip	Mesh	** p<0.01	** p<0.01	insignificant	** p<0.01	
Drip	Plastic	** p<0.01	insignificant	insignificant	insignificant	
Drip	Straw	** p<0.01	insignificant	insignificant	insignificant	

The one-way ANOVA for plug plots vs gel plots at the Little Woolden site, spray irrigation sector showed significant differences between no mulch plots and the three cover plots (P<0.001), however the Tukey HSD results suggest that there were no significant differences between the annual yields of plugs or gel plots when Mesh, Plastic or Straw mulch covers were used.

This relationship is mostly the same under drip irrigation, with no significant differences between plugs and gel plots in terms of yield when covers were used. The exceptions to this were for plug mesh vs gel mesh and plug mesh vs gel straw which show significant differences with Tukey HSD testing ($P<0.01$).

Finally, for no mulch plots, there were no significant differences between no mulch plug and no mulch gel plots under spray irrigation, whereas in the drip-irrigation sector this appears to show a significant difference ($p<0.01$).

In summary, under spray irrigation the use of founder material introduced no significant differences in terms of yield when using mesh, plastic, or straw covers. However, under drip irrigation, mesh-covered plug plots had a significantly higher mean yield in dry mass tonnes ha^{-1} yr^{-1} than mesh-covered gel plots under the same irrigation system (** $p<0.01$), with a mean of 17.24 tonnes dry matter ha^{-1} yr^{-1} vs 6.81 tonnes dry matter ha^{-1} yr^{-1} respectively.

9.4.4 Performance of the MIFA approach compared with the MLTT approach

Core hypothesis 5: The *Sphagnum* yield produced by MIFA-style irrigation will be comparable to previously published data from *Sphagnum* farming sites that have used donor material.

Growth hypothesis 4: as above

The calculated TLS volumes and DBD of 30 g per L were used to estimate an 'in the ground yield' for this research study. This was necessary as the *Sphagnum* plots were not harvested during this research study. Considering all covered plots together, the plugs at Little Woolden under spray irrigation achieved a mean of 22.73 t dry matter ha^{-1} yr^{-1}. Plugs under drip irrigation achieved a similar mean of 22.75 t dry matter ha^{-1} yr^{-1}. In terms of gel plot yields, spray irrigation plots achieved a mean yield of 12.73 t dm ha^{-1} yr-1 vs 7.81 tonnes dry matter ha^{-1} yr^{-1} for plots under drip irrigation.

At Sharpley the mean yield of all covered plots was 15.2 t dm ha^{-1} yr^{-1} in the spray irrigation area, while 10.48 t dm ha^{-1} yr^{-1} was achieved in the drip irrigation area.

The estimated yields at Little Woolden for *Sphagnum* produced under the MIFA approach are substantially higher than the natural productivities of *Sphagnum* and the yields achieved in the *Sphagnum* farms listed in Table 36. The estimated yields for Sharpley are also higher than those presented in Table 36. However these are only estimates based on literature values. An actual measured value will be possible at harvest but is beyond the scope of this research study.

Table 36: Natural productivities of *Sphagnum* in natural conditions[1] & cultivated conditions[2], expressed as Dry Mass in tonnes per ha per year. Adapted from (Mulholland *et al.*, 2020)

***Sphagnum* measured:**	**Location**	**Productivity in Dry mass tonnes ha-1 yr -1**	**Source**
Global *Sphagnum mean productivity*[1]	Global mean non cultivated sites	2.59	Gunnarson 2005
S. palustre[1]	Kolkheti lowlands (Georgia)	5.75	Krebs *et al.*, 2016
S. cristatum[1]	New Zealand	8.4	Stokes *et al.*, 1999 & Gunnarson 2005
S. falcatulum[1]	New Zealand	7.7	Stokes *et al.*, 1999 & Gunnarson 2006
S. subnitens[1]	New Zealand	5.9	Stokes *et al.*, 1999 & Gunnarson 2007
S. fuscum[1]	Germany	8	Overbach and Happach 1957
S. magellanicum[1]	Germany	7.9	Overbach and Happach 1958
S. rubellum[1]	Germany	9.6	Overbach and Happach 1959
S. palustre[2]	Rastede (Germany)	6.5	Temmink *et al.*, 2017
S. papillosum[2]	Ramsloh (Germany)	3.6	Gaudig *et al.*, 2014
S. Fuscum[2]	Shippagan (Canada)	1.12	Pouliot *et al.*, 2015
S. palustre[2]	Drenth (Germany)	1.12	Grobe, Tiemeyer and Graf, 2021
S. palustre[2]	Provinzialmoor (Germany	6.29	Grobe, Tiemeyer and Graf, 2021

The shallow peat study (Grobe, Tiemeyer and Graf, 2021) tested geotextile mulch covers vs straw mulch covers. In contrast to this research study, the authors suggested that geotextile covers offering 50 % shade are not a viable alternative to straw mulch in terms of microclimate, shade and peat wetness. The Boosted Regression Trees used in Grobe's study show that the moss fragments for both species established better when covered with straw applied with a density of 80 %. The authors state that the main issues with geotextile in their study is that the geotextile became saturated and increased anoxic conditions in wetter weather, and in windy weather the geotextile flapped about lifting moss fragments off the peat – both combining to provide a poor microclimate for *Sphagnum* establishment when developing mosses are most vulnerable.

Initial losses during establishment are a challenge, as in this study even when geotextile covers were removed the moss did not regain these losses (Grobe, Tiemeyer and Graf, 2021). Overall straw mulch is put forward as a suitable material to promote establishment of *Sphagnum* fragments if constant water supply cannot be ensured (Quinty and Rochefort, 2003; Pouliot, Hugron and Rochefort, 2015; Gaudig *et al.*, 2017). When the method of water provision is applied by canal irrigation straw may be more advantageous. However, this is not something found in the top down irrigation approach used in this research study as the Mesh and Plastic mulch covers achieved comparable, and often higher percentage cover and carpet thickness increases than straw plots.

While the present research programme has generated a great deal of data while investigating and testing the proposed MIFA approach to *Sphagnum* farming, it is clear that a great many factors remain to be satisfactorily explored and resolved. Some aspects are discussed next.

9.4.5 Validity of converting volumes into yields

The conversion factor relying on bulk density can be performed in several ways, and this may significantly impact the calculated yields.

Firstly, bulk density taken from standard acrotelm values provides an idea of the unprocessed, 'in the field' value for living moss layers in natural peatlands. Acrotelm values are often taken to be 0.03 grams per cm^3 based on extensive experiments (Clymo and Hayward, 1982; Clymo, 1992). The Clymo value is equivalent to 30 grams per L. However, the weight of vegetation at the acrotelm surface will compress the decaying vegetation underneath it, with consequent increases in bulk density with depth. When accounting for this compression effect on DBD values across the entire acrotelm depth, which may be up to 30 cm deep, a new overall figure of 0.06 g per cm^3 has been suggested (Lindsay, 2010) this provides a bulk density of 60 g per cm^3. This value can be expected to provide the most realistic value of *Sphagnum* in the field, however the lower value was used for yield estimates in this research study to provide conservative estimates.

Sphagnum yield is often not just dependent on the 'in the ground' value. The preparation of harvested *Sphagnum* into Growing Media requires harvesting, drying and chopping, which will result in altered 'processed' DBD values. A standardised method of volume calculation will need to be used where *Sphagnum* is mixed with other components as a mixed growing media to determine an 'in the bag' value. This process is often used in the growing media industry with products being sold with labels stating the 'number of Litres when filled' (British Standards, 1999).

Within the *Sphagnum* farming literature attempts have been made to standardise the DBD values used for processed harvested values. A recent paper by Wichmann *et al*. (2020) identified a low range for DBD and a high range for DBD based *Sphagnum* post-harvest taken from several *Sphagnum* farming projects in Germany. The range was determined via the mean processed DBD ± 1 standard deviation (Wichmann *et al.*, 2020). The DBD range for processed *Sphagnum* was presented as a low of 20 g per L and a high of 38 g per L respectively.

Ultimately the yield must be presented in tonnes of dry mass per ha per year (t ha^{-1} yr^{-1}). Studies reporting *Sphagnum* productivity can also be used to compare growth across *Sphagnum* farm sites and natural sites. Productivity is defined as the dry biomass that is produced per square metre per year (Gunnarsson, 2005), but these units can be converted mathematically to present a yield in tonnes per ha for comparison.

Few studies have addressed natural productivity and growth in length of *Sphagnum palustre* specifically (Lütt, 1992; Fukuta, Sasaki and Nakatsuba, 2012; Krebs, Gaudig and Joosten, 2016). In the Kolkheti lowlands a large productivity of 387–788 g m^2 yr^{-1} was observed for *S. palustre* across a range of lowland sites (Krebs, Gaudig and Joosten, 2016). The mean S. *palustre* productivity result in the Kolkheti lowlands was 575 g m^2 yr^{-1} which is in the middle of the global productivity range from 8 to 1,450 g m^2 yr^{-1}, but almost twice as high as the global mean *Sphagnum* productivity of 259 g m^2 yr^{-1} (± 206, SD) (Gunnarsson, 2005). The global mean would equate to 2.59 ± 2.06 tonnes dry mass per ha per year.

9.4.6 Dry bulk densities of processed *Sphagnum* for growing media

The processed *Sphagnum* for growing media will typically have a lower DBD than the *Sphagnum* ‘in the field’. Within the literature, typical processed figures based on the CEN EN 13041 test are given for raised bog peat across the whole range of decomposition can be found. Low decomposition *Sphagnum* peat, defined as H2 to H4 on the Von Post scale can be expected to have a DBD of 50 to 80 kg per m^3, while more decomposed peat at the opposite end of the scale (H6 to H8) is denser and has a range of 160 to 220 kg per m^3 (Schmilewski, 2008). These processed bulk densities are higher in comparison to the DBD of typical acrotelm vegetation of 0.03 g per cm^3 which is 30 kg per m^3 when converted to equivalent units for comparison.

The lower the decomposition level, the lower the DBD. The CEN 13401 testing involves homogenising the processed material and free pouring a subsample into a 30L measuring cylinder through a mesh screen (CEN, 1999b, 1999a; Blok, Eveleens and van Winkel, 2019). The method introduces a small amount of compression with the use of a 10 cm collar that sits above the main measuring cylinder and is filled with material from the subsample, but there is no additional weight added. Typically, at lower humification levels, *Sphagnum* peat presents with larger and more rigid fragments of moss. These act as scaffolding and provide more pore spaces within the 30 L sampling chamber – resulting in a lower mass per fixed volume and a lower DBD. Peat with a low humification level H2 to H4 typically has a total pore space of 95 - 97%, while H6 – H8 peat has a pore space of 87 - 91% (Schmilewski, 2008).

9.4.7 Yield information and long-term performance

Assessing the *Sphagnum* yield and the long-term performance of a site over time are also key areas of importance for study. Farmers will need to know the optimum time taken from planting to harvest, the expected yield at regular intervals, and any interventions or likely outcomes arising from their decision to begin a *Sphagnum* farm.

Sphagnum productivity is a balance of growth vs decay. Typically, under naturally functioning conditions fresh *Sphagnum* in the acrotelm experiences mass loss of 10 - 20% per year, with most decay occurring in the first 4 to 6 months, from thereon mass loss is much slower (Rochefort, Vitt and Bayley, 1990). This is a remarkably low loss of mass compared to vascular plant leaves deposited in a peatland environment, which may lose 40 - 80% of their mass per year (Rochefort, Vitt and Bayley, 1990). The lower decomposition rate in *Sphagnum* is largely due to *Sphagnum*'s adaptation to acidic environments (Clymo and Hayward, 1982). One implication of expressing yield in tonnes of dry mass per ha per year is that *Sphagnum* farming experiments presenting yields averaged over many years may well smooth out these initial losses and show a greater yield, whereas *Sphagnum* farming experiments with a dataset of lower duration <12 months may be disproportionately affected by earlier losses. Data based on shorter growth durations may also be biased towards larger yields if the losses of 10 - 20% are not considered.

9.5 Conclusions

This Growth study has resulted in 3D point cloud data of the Sharpley and Little Woolden *Sphagnum* farms. The data has been used to calculate both the percentage area and height change of developing *Sphagnum* carpets under two different irrigation regimes, and three mulch cover types. These physical observations have also been combined with literature values for dry bulk density of a natural *Sphagnum* acrotelm to calculate estimated *Sphagnum* yields in t dm ha^{-1} yr^{-1}.

The MIFA approach has been successful, resulting in calculated *Sphagnum* yields that are substantially higher than the natural productivities of *Sphagnum.* In some cases, the calculated yields exceed the physical yields achieved in other *Sphagnum* farms within the SF literature. However, as the *Sphagnum* plots were not harvested during the study period. An actual measured value for dry bulk density will be obtained at harvest to fully verify the yields, but this is beyond the scope of this present research study.

Further Discussion about the present study continue in the next chapter.

Chapter 10. General Discussion and Project Impact

10.1 Can we directly link pore water pressure to *Sphagnum* growth?

The number of consecutive days of drought impacting recovery provides the best framework to link the impact of hydrology on growth. Clymo and Hayward (1982) showed that when *Sphagnum* capitula were completely desiccated and subsequently re-wet, 100% of the S. palustre individuals tested recovered when drought lasted 6 days or less, at 10 days 50% of the capitula recovered, 5% recovered at 13 days and no S. palustre survived a drought of 16 days. Different *Sphagnum* species have different recovery rates as shown in Figure 9 earlier in Chapter 3.

As discussed earlier in Chapter 3, *Sphagnum* hyaline cells begin to lose water when pore water pressure (PWP) increases to between 100 and 600 hPa. This can be used as the range for desiccation (although complete desiccation occurs when PWP is > 600 hPa). Being conservative, the 100 hPa threshold has been used as the proxy for drought stress in this thesis when the daily maximum PWP recorded by a tensiometer exceeds 100 hPa.

Within this framework, *Sphagnum* growth and recovery could be impeded where the maximum PWP exceeds 100 hPa for more than six consecutive days. The data for these is presented in Table 37 below, for both sites and irrigation areas. Plots meeting this criteria are compared against the mean growth data for comparable plots. Comparable plots, which were defined as the same irrigation area, mulch cover, and founder material.

Irrigation area, phase and plot details	Number of consecutive days where PWP exceeded maximum value > than 100 hPa for > 6 days in a row	*Sphagnum* % cover at final scan	Overall height increase of *Sphagnum* carpet (m) at final scan	Annual 'In the field' yield dry mass (t ha^{-1} yr^{-1})
Sharpley Phase 1 Drip irrigation				
CD5 (Plastic plug)	43	47	0.05	8.22
Sharpley Phase 2 Spray irrigation				
AS4 (No mulch plug)	24	Not present at final scan		
BS4 Mesh plug)	10	69	0.04	6.73
BS5 (Mesh gel)	16	Not present at final scan		
DS4 (Straw plug)	10	Not present at final scan		
Little Woolden Phase 1 Drip irrigation				
AD8 (No mulch plug)	12	37	0.04	7.17
BD7 (Mesh plug)	56	88	0.07	11.75
DD9 (Straw plug)	8	72	0.06	10.38
Little Woolden Phase 1 Spray irrigation				
AS9 (No mulch plug)	57	160	0.06	7.83
Shading key:		Above average for comparable plots	Below average for comparable plots	Plot removed before final scan

Table 37: The potential impact of exceeding the ecohydrological PWP thresholds for more than 6 days on Sphagnum growth. The table presents all the plots across the entire study where the maximum PWP recorded exceeded 100 hPa for more than 6 consecutive days, indicating the possibility of desiccation damage and reduced recovery to a Sphagnum crop.

Across the entire PWP monitoring period at Sharpley, five plots were of relevance. Four of these were monitored with some certainty, which were Mesh plot BS4 (10 days) and mesh plot BS5 (16 days) Straw plot DS4 (10 days) and No mulch AS4 (24 days). Plastic plot CD5 (47 days) was monitored by a tensiometer that was found to be faulty in the Phase 1 monitoring, so there is less confidence around this high duration of PWP threshold being breached, especially as in the Phase 1 PWP monitoring, the PWP range was deemed optimal for >96% of the time for plots with a mulch cover when a faulty tensiometer is excluded (Figure 27 and 28).

As shown in Table 37 at Sharpley, plots CD5 (plastic plug) and BS4 (mesh plug) achieved a lower percentage cover and a lower carpet increase than comparable plots' this suggests that the periods where the maximum daily PWP exceeded 100 hPa for more than 6 days may have reduced growth metrics.

During the phase 2 monitoring, only spray irrigation plots had threshold breaches over 6 days in duration. The plots, AS4 (no mulch plug), BS5 (mesh gel) and DS4 (straw plug) unfortunately have an incomplete growth record, as all of the no mulch and straw plots across both irrigation areas for both gel and plugs were completely removed at Sharpley.

However as shown earlier in figure 33 and 34, the optimal PWP based on comparable plots were achieved for AS4 (66% of the time), BS5 (87%) and DS4 (875). This suggests that the no mulch plots achieved poor growth because of hydrology, but factors other than hydrology impacted the removal of the mulch covered plots. As plots AS4, BS5 and DS4 did not survive to the end of the study it is impossible to comment on the impact of PWP thresholds being exceeded and assess the impact of these on growth.

At Little Wolden, all plots survived to the final scan in August 2020, allowing for a more rounded comparison. Plots AD8 (no mulch plug) and BD7 (mesh plug) both achieved lower than average growth results. While DD9 (straw plug) and AS9 (no mulch plug) achieved higher growth results when compared to their comparable plot counterparts. Looking at the full growth data in Table 29, (Chapter 9) BD7 had the highest drought stress at 56 days, and subsequently had the lowest dry mass yield of all the mesh plug plots under drip irrigation at 11.75 t ha^{-1} yr^{-1}, and DD9 had the highest annual yield of the straw gel plots (10.38) suggesting that recovery and subsequent *Sphagnum* growth was possible in plot DD9 as only 8 days of drought were experienced, this aligns with the data presented in Clymo and Hayward (1982), that suggests that 50% recovery is possible for *S. palustre* after 10 days of drought.

It is therefore a mixed picture when trying to directly link the recorded yields with the hydrological impacts of consecutive days where the PWP threshold of 100 hPa is breached across the plots at Sharpley and Little Woolden.

This was partially limited by the number of Tensiometers available as only a subsample of plots were monitored, but also due to temporal resolution differences between tensiometers and the TLS scans. This is because the short-term nature of PWP threshold breaches is recorded in days, whereby the measurement of growth is measured at intervals of months. Future work may overcome this as discussed later.

10.1 New Knowledge

The conceptual framework for this research project was presented in Figure 14, which set out the novel areas of investigation. At the end of the project the following aims presented in the conceptual framework have been achieved:

- The Project provided two new international locations for *Sphagnum* farms in lowland Britain;
- The use of micropropagated *Sphagnum* material as founder material instead of harvesting *Sphagnum* from a donor site has been proven to work successfully for the first time;
- The use of Dripline and Spray irrigation treatments have been proven to be a successful option for hydrological management for *Sphagnum* farming under the new MIFA approach;
- The use of TLS has provided a new methodology for measuring *Sphagnum* growth;
- The use of novel Mesh and Plastic mulch covers have been shown to provide effective protection for *Sphagnum* founder material post-application and beyond and offer alternatives to the Straw mulch used in the MLTT.

The Micropropagation with Irrigation From Above (MIFA) approach is capable of producing a good *Sphagnum* crop even on sub-optimal sites, and offers a viable alternative to the conventional MLTT approach for *Sphagnum* farming. Paludiculture as a concept is very new and to become accepted as a standard part of agriculture will require a radical change in attitudes to land drainage – a change which reverses centuries, if not millennia, of agricultural practice. Acceptance of high water tables and even flooding as an agricultural practice will take time, possibly requiring generational change. Consequently, adoption of the current 'standard' approach to *Sphagnum* farming, namely supplying *Sphagnum* via the Moss Layer Transfer Technique (MLTT) and raising water tables, is likely to be perceived as a threat to conventional agricultural practices and may receive significant push-back from both the farming community and potentially the wider community. The MIFA approach offers a way forward using methods that are already familiar within rural communities and poses no threat to adjacent farming practices. The MLTT by its very definition requires the use of a donor moss layer to transfer, therefore the MIFA technique has the added advantage of using micropropagated founder material and so poses no threat to remaining natural stocks of *Sphagnum* and provides a 'clean' founder material free from other non-crop species. A future research programme could, however, investigate a hybrid of the two approaches, where founder material is provided by micropropagation instead of the MLTT, water tables are raised, and top-down irrigation is used in times of drought or low water tables.

Top-down irrigation prevented micropropagated *Sphagnum* from exceeding ecohydrological thresholds for large percentages of time on both deep peat and shallow organo-mineral soils. This suggests that with further improvements (in water availability and irrigation techniques) this method will enable *Sphagnum* farming to take place in areas where raising the water table is unfeasible or undesired. This opens the prospect of *Sphagnum* farming to a wider range of farmers and land types beyond peat soils alone. The rationale for this is that this further reduces the demand on commercial extraction and wild harvest across the globe, resulting in reduced wetland degradation, and improved ecosystem services (ES) in near natural areas.

Top-down irrigation does not, however, result in a high and stable water table, which is a benefit in terms of fitting the MIFA approach to *Sphagnum* farming into the current agricultural landscape.

Mulch covers such as perforated plastic and woven mesh are suitable for *Sphagnum* cultivation when paired with irrigation from above. Alternative mulches to the straw used in the MLTT approach do not appear to limit growth, with eventual growth rates comparable to that achieved using straw cover. Indeed straw seems to be slightly less successful than mesh or plastic under some conditions. Mulch covers offer increased shade and humidity to the *Sphagnum* crop which may be beneficial to growth, although the constant level of humidity may give rise to algal infestation, and mulch may also lead to flooding as a result of excess water retention. While including an outflow option to drain away excess water is one solution, an alternative may be to dispense with mulch covers once the crop is well established because some no-mulch plots proved capable of maintaining the necessary hydrological conditions once well established, although growth was undoubtedly much reduced in the absence of mulch cover.

Terrestrial laser scanning (TLS) can be used successfully to measure *Sphagnum* crop growth. *Sphagnum* is a crop that forms a continuous carpet with an undulating surface. Most methods rely on sub-sampling parts of the *Sphagnum* carpet, but the undulating nature means that large areas must be sampled in order to obtain a reasonably representative average across a sampled area. This is often done destructively, removing a substantial area, particularly if done repeatedly during crop development. Resulting in a significant portion of the crop being lost to such sampling. Terrestrial laser scanning (TLS) permits substantial areas to be measured repeatedly without harm to the crop in order to produce crop-volume data over time. TLS worked well in the study scenarios as the surface was levelled prior to *Sphagnum* application and baseline TLS scans occurred close to founder material application. The TLS method will have broader application for paludiculture crop monitoring providing baseline scans are taken.

10.2 Future work needed to develop the MIFA approach further

This research project has provided ‘proof of concept’ for the top-down irrigation method of *Sphagnum* farming on degraded lowland soils. However, there are several areas for further investigation that should be explored to deliver this method as a commercially viable *Sphagnum* paludiculture.

10.2.1 Linking pore water pressure impact to growth

As discussed earlier, bringing the temporal resolution of PWP measurements and growth measurements closer together may help to better understand the impact of PWP thresholds being exceeded and their impact on *Sphagnum* growth and recovery.

Future work could benefit from more frequent site visits and data recording, as areas withing plots showing signs of desiccation or ‘bleaching’ could then be marked out and on subsequent visits assessed for green or ‘vital’ *Sphagnum* within the marked area. With the TLS scans, marked areas could then be assessed for a reduction in *Sphagnum* volume, height, or percentage cover.

Additionally changes in *Sphagnum* growth form may arise due to prolonged periods where PWP was exceeded, as *Sphagnum* typically prioritises density over increased shoot growth when experiencing dry conditions, and the growth form is more compact as a result. Though measuring growth form under this approach would require frequent DBD analysis and careful *Sphagnum* removal techniques to ensure the accuracy of this approach.

This would be something that future work could investigate as it was not undertaken in this study.

10.2.2 Water Budget

A full water budget should be developed for the MIFA study sites, as this was not undertaken in this study. The irrigation water input was not quantified during the study as a water meter was not available as part of the irrigation supply infrastructure. Evapotranspiration, water deficits or surplus were also not quantified. However, MPS estimates for water input using the evaposensor control based on previous use, were an irrigation input of 1 to 2 mm (1000 – 2000 m^3 ha^{-1} $yr^{-1)}$. Water budgets are likely to be site specific, so future research projects and commercial *Sphagnum* farms will need to calculate these prior to their development. Water budgets will have implications for the volumes of irrigation water required, source of water supply and influence the choice of water storage.

The only study to date that investigates water budgets on a *Sphagnum* farm is presented in Brust *et al.* (2018) for the Rastede *Sphagnum* farm in NW Germany. The area is temperate with an annual rainfall of 849 mm, which is similar to the annual rainfall at the Sharpley site (844 mm) and lower than the long term average for the Little Woolden site (1198 mm). At the Rastede site, an annual irrigation volume of 160 mm to 360 mm (1600 - 3600 m^3 ha^{-1} yr^{-1}) was required to maintain the water table at 5 cm below the *Sphagnum* surface. The key recommendation from the water budget at Rastede is that the water demand must be considered, and an appropriate water supply must be guaranteed.

Furthermore, with climatic change, additional water storage capacity is likely to be required for drier years or extremes of weather where a higher evaporative demand would require more irrigation water to prevent *Sphagnum* desiccation, a conclusion shared by Oestmann *et al*., (2022a) where passive warming mesocosm plots on SF sites suggested climatic warming will increase GHG emissions when adequate water supply is not maintained.

Furthermore, it is likely that as the peat soil within the MIFA field trials may have experienced additional water losses from the un-irrigated areas surrounding the MIFA sites as evapotranspiration due to advection and as horizontal seepage may have increased due to steepening of hydraulic gradients between the irrigated and non-irrigated areas. Brust *et al.,* (2018) point out that evapotranspiration and seepage losses can decrease with an increased size of rewetted area, clearly, larger scale trials with a greater planted area of *Sphagnum* under the MIFA approach will need to be studied to investigate and quantify the implication of larger areas on water budget issues.

It is possible that the MIFA approach may require comparatively less water than the MLTT with irrigation canals, however this cannot be confirmed until a water budget is determined. Furthermore, as these initial MIFA field trials were small, and the individual plots not connected in one large *Sphagnum* carpet, the lateral transport of water across an interconnected *Sphagnum* carpet could alter the volume of irrigation water required. So, the water balance in an upscaled MIFA trial may differ with a larger planted area.

10.2.3 Carbon Budget

In addition to a water budget, a full carbon budget for this novel irrigation method should be developed. The LCA should encompass the embodied and operational carbon of the irrigation infrastructure, energy use and long-term impacts of the synthetic novel mulches under the MIFA approach.

In Germany GHG analysis for a *Sphagnum* farm using surface irrigation canals suggests that *Sphagnum* production strips are a net sink of carbon dioxide, absorbing 5 – 9 t CO_2e ha^{-1} yr^{-1} but the irrigation canals themselves are a net source of 11 t CO_2e ha^{-1} yr^{-1} (Günther *et al.*, 2017), the study in Germany did not investigate carbon release in dissolved organic carbon (DOC) or particulate organic carbon (POC) which would be needed for a full carbon budget.

More recently, Oestmann *et al.* (2022b) reported GHG balances from two *Sphagnum* farming sites on highly degraded cut-over bogs in Germany vs a near natural control site, the CO_2 exchanges for the SF sites were both positive and negative, ranging between -0.6 and 2.2 t CO_2 ha^{-1} y^{-1}. The authors state that these carbon emissions were mostly influenced by water table depths. Emissions of CH_4 were low across the sites, while N_2O was only elevated in the irrigation canals. Interestingly one site, Drenth, utilised drip irrigation after the initial sub surface pipe irrigation failed. Unfortunately, not enough detail about the drip system is given within the paper to make any comparisons with the study sites in this thesis.

In a review paper by Bianchi *et al.*, (2021). the GHG balance for SF is shown to be negative at -2.8 t CO_2 eq. ha^{-1} yr^{-1} when the harvested biomass is not included in the GHG balance. However, when the harvested biomass is included in the GHG balance, restored peatland sites and SF sites were found to be broadly similar at +1 T CO_2 eq. ha^{-1} yr^{-1} (restored) vs 3 t CO_2 eq. ha^{-1} yr^{-1} (SF). Emissions of CH_4 were also negligible across both scenarios but N_2O fluxes were higher under restoration. Although the literature values for GHG emissions from paludiculture sites are still relatively scarce, it is important to recognise that the current published results represent significant GHG mitigation potential for drained peat sites, especially when compared to drained agricultural peatlands which involve emissions in the range 25 – 35 t CO_2 eq. ha^{-1} yr^{-1} (IPCC 2014).

Most restoration and SF studies to date have raised water tables, with the goal of a high c.10 cm from the surface being the aim for most *Sphagnum* farms (Gaudig *et al.*, 2014). As the water table in both study sites presented in this thesis was not consistently high and biomass harvest was not performed, it is likely that the GHG performance will be different to the above published studies. A full carbon budget therefore needs to be calculated to assess the net GHG benefit of the MIFA approach.

10.2.4 Potential environmental impacts of mulch covers

Although not investigated as part of this thesis study, plastic mulches have the potential to contribute to the plastic pollution problem if not used appropriately. This is primarily due to degradation across their usable life and poor recovery of all mulch fragments from the soil after use which releases plastic pollutants to the agricultural soil. Additionally, the mismanagement of plastic mulches at their end of life can lead to more plastic accumulating in the environment when it does not enter an appropriate waste stream. Generally, the end-of-life waste stream option are recycling, incineration for energy recovery or landfill (Serrano-Ruiz, Martin-Closas and Pelacho, 2021).

There are also concerns that long term plastic mulching from fossil and biobased plastics may negatively affect crop yields due to increased plastic accumulation in soil, while also potentially impacting ecosystems beyond the agricultural systems where they are used (Serrano-Ruiz, Martin-Closas and Pelacho, 2021). This can lead to the reduction of nutrient availability, and microorganism diversity, which in turn also alters soil structure, nutrient dynamics, and moisture retention in soil. Further concerns surround reduced root development and altering the GHG balance for soil emissions (Gao *et al.*, 2019). Microplastics resulting from the degradation of plastic mulches are also a potential problem as up to 32% of plastics may remain in the environment following their usable life (Kumar *et al.*, 2020). Future studies should seek to investigate these concerns under the MIFA approach on peat soils.

The environmental impact and LCA of the carbon emissions of the alternative mulch covers should also be investigated as this was beyond the scope of this PhD study. The mesh and plastic covers used for alternative mulches are fossil fuel based (HDPE and polythene) so have the potential for longer term impacts, while the straw mulches almost completely degraded by the end of the PhD study, so have had a shorter-term impact. Future studies should seek to investigate these concerns surrounding the short and long-term use of HDPE and polythene mulches under the MIFA approach.

10.2.5 Planting and Husbandry

Plug plants were planted by hand, at a density of 36 plugs per m^2, but this is not likely to be viable in the commercial world where sites will be significantly larger than the 50 m x 10 m irrigation areas employed in this study. Gel offers more promise for quick application over a large area, but this has yet to be trialled on a large scale.

Engagement will be necessary with mechanical engineers to design and adapt machinery suitable for *Sphagnum* planting, maintenance and harvesting, as commercially available machines will increase the farming efficiency. For example, developing a vehicle mounted method of application for micropropagated founder material will be essential for wider uptake in a commercial world. Straw mulch has been applied via machine and by hand at scale on larger *Sphagnum* farm sites already (Gaudig *et al.*, 2014) so future work should also report on the practicalities of applying mesh and plastic covers at scale using machinery.

Weed pressure from vascular plants was experienced at the study sites, as on other *Sphagnum* farms. These were dealt with by hand weeding using volunteers in this study. In the commercial farming world this is unlikely to be a satisfactory solution. Future work will need to develop a form of herbicide or other treatment to limit vascular plant competition.

10.2.6 Harvesting and economic use

Issues surrounding the harvest of *Sphagnum* were not addressed in this study due to the long rotation times expected to maximise the *Sphagnum* yield. However, the top-down irrigation method will offer the potential for mechanised harvest as the irrigation could be turned off quickly, allowing the moss surface to dry out prior to harvest. Future studies should investigate the latter stages of *Sphagnum* farming shown in the conceptual diagram Figure 14, including testing harvesting methods and *Sphagnum* processing for future use.

Finally, the initial economics of this new irrigation method should be assessed to compare with the costs for other experimental *Sphagnum* farms such as those reported in Germany (Wichmann *et al.*, 2020). It is likely that the costs per area of establishing and running the small experimental sites used in this study will be higher than those of future *Sphagnum* farmers, who will benefit from economies of scale and increased optimisation of the method.

10.2.7 What challenges must be overcome to achieve commercial *Sphagnum* farming success in lowland Britain?

To be truly sustainable *Sphagnum* farming will have to achieve the human wellbeing benefits and values set out by TEEB framework shown in Figure 1. Delivering economic, social, and ecological sustainability. Thereby helping to deliver key UN Sustainable Development Goals and deliver positive ecosystem benefits. Many of these recommendations will likely apply to most paludiculture crops.

Beyond the technical advances that need to be explored, society will need to be given the opportunity to learn about the concept of paludiculture, develop awareness surrounding the benefits of this method of farming, and explore the opportunities and limitations. This will require inclusion in academic settings such as agricultural colleges, universities and courses investigating ES provided by wetlands and paludiculture set out in Table *3*. Additionally public campaigns like the Peat campaign shown in Figure 3 may be needed to promote farmed *Sphagnum*.

Engagement with the farming community and investing in early adopters will be necessary to showcase *Sphagnum* as a crop, drive farmer led innovation and ensure acceptance of the crop. Work should be undertaken to investigate a *Sphagnum* farming rotation system or the use of *Sphagnum* in a mixed farm approach to maximise crop diversity and opportunity for future paludiculture farms. Currently most paludiculture options, including *Sphagnum* farming, are biomass focused rather than food focused, so the farming community and society will need to prepare for a potential trade-off with reduced food production on peat soils for greater ES benefits via a paludiculture system. Of course, future research should investigate perennial paludiculture food crops which would also help to solve this problem. One such paludiculture food crop of the future could be Glyceria fluitans, a food crop harvested in the wild for human consumption. This was briefly explored as an early part of the research programme (Clough, unpublished) and can be found in the appendix of this study.

It will be important to build on the growing political interest for paludiculture systems to support necessary policies, support systems and other mechanisms to provide paludiculture with adequate attractiveness to wider commercialisation. For example, the paludiculture-subgroup of the DEFRA Lowland Agricultural Peat Task Force has developed a 10 year policy roadmap for paludiculture in England. This has led to the 'Nature for Climate: Paludiculture Exploration Fund', which was launched on the 22nd December 2022, and aims to provide grants to help unlock barriers to the development of commercially viable paludiculture (Natural England, 2022).

Further work with companies producing growing media will be necessary. Melcourt industries Ltd. performed growth trials on different mixes and found very promising results, (Melcourt, unpublished). However additional trials with a wider group of growing media manufacturers are needed. Amateur consumer trials may also be needed to assess consumers reaction to a *Sphagnum* based growing media mix as they comprise two thirds of the market. Solid promotional campaigns will be needed to reassure the public and professional growers that *Sphagnum* based mixes are just as good as peat, and to avoid the repeat of bad experiences with peat-free growing media as seen in the 1990's. This will require coordination and buy in from key horticultural trade associations such as the Growing Media Authority, Horticultural Trade Association and Royal Horticultural Society, to whom growers will look to for advice.

10.3 Wider challenges to widespread wise use of lowland peat

Peatland restoration offers clear ecosystem service benefits by enhancing ES shown in Table *3* which are reduced when peatlands are degraded. Therefore, restoration will play a role in delivering policy objectives linked to the policy context referred to above. However, peatland restoration faces many challenges.

Peatland restoration is defined as actions taken to restore degraded peatlands to a functioning, healthy state. Restoration involves the exclusion or reduction of damaging practices on a peatland area. While management interventions are applied to reduce the negative impacts of past land use (Andersen *et al.*, 2017).

There is academic consensus that peatlands provide ES (Barbier, 2011; de Groot *et al.*, 2012; Costanza *et al.*, 2014). However, attitudes and perceptions surrounding peatlands within the wider public are less understood. One study in Scotland found four key attitude narratives across stakeholders. Peatlands were categorised as wonderful wildernesses, cultural landscapes, degraded nature or wastelands (Byg *et al.*, 2017). Wider stakeholders including farmers, landowners and the public mould their perceptions based on personal experience and interaction with different ecosystems (Collier and Scott, 2008; Bennett, James and Klinkers, 2017). Therefore, activities to provide ongoing awareness raising and education about peatlands is the key to ensure public support for peatland restoration.

A key challenge is resolving the conflicting aims between peatland restoration and alternative land use on or near peatlands. Conflicts surrounding land use change, biodiversity and ES are primarily conflicts between people (White *et al.*, 2009). A range of measures are needed to mediate these human conflicts and communicate different points of view. Conflicting land use requirements must be resolved through achieving compromise, implementing measures to compensate for the removal of damaging activity, or by negotiating win-win scenarios.

Agriculture close to peatland sites poses a challenge. The water and soil management regime required by conventional agriculture is not compatible with the conservation of peat soils (Joosten and Clarke, 2002) as one land user wants to pump water in, and the other wishes to pump water away. This diametrically opposed desire to manage the land is a key issue that needs resolving. Very few stakeholders value peat soils intrinsically, but value the services they provide (Rawlins and Morris, 2010). The policy direction that the UK is moving towards suggests that payment for public goods may be one way in resolving such conflict, which is important as landowners will adopt potentially sustainable practices on peatlands if they are practical and financially viable (Rawlins and Morris, 2010). Changes to conventional agriculture such as raising water tables while maintaining normal production (Van Den Akker and Hendriks, 2017) or paludiculture (Wichtmann, Joosten and Schröder, 2016) may be one way to achieve a genuine win-win scenario for this type of conflict between peatland restoration and agriculture.

The scale and costs of the restoration interventions required for peat soils are a key challenge. Following 22 years of restoration activity through the EU life + programme, in western Europe, some 913 km^2 of peatland habitat has been restored at a cost of 167.6 million euros (Andersen *et al.*, 2017). This represents less than 2% of the total remaining peatland area within the regions under the EU Life + programme. So much greater ambition, funding, and restoration success is needed.

Finally, there is an issue of environmental designation. Designation provides a mechanism for delivering peatland restoration. However not all peat soil sites are designated, this ensures that large areas of peat soil will be omitted from restoration plans and funding. A restoration framework and incentive options must be developed to accommodate these areas to realise the improved ecosystem service benefits that peatland restoration will achieve.

10.4 Future policy and funding options to overcome challenges on lowland peat

From the policy schemes mentioned in section 1.5 and the challenges identified in 1.6, new management of peat soils is necessary to prevent current, and future carbon loss. Resulting adaptation and mitigation measures will deliver enhanced ES and drive the sustainable use of peat soils.

Restoration of degraded peat soils: Peatland restoration has the potential to store and sequester large volumes of atmospheric carbon. Through the UNFCCC, at COP 17, peatland drainage and rewetting was recognised as an activity pertinent to the Kyoto protocol (Bonn *et al.*, 2014). This ensured that carbon emissions could be accounted for on a voluntary basis. This enhanced peatland restoration as a climate mitigation strategy. Due to the large volumes of carbon involved, restoration is a key policy instrument in mitigating climate change (Griscom *et al.*, 2017).

The financing of restoration is likely to come from public and private funders.

Peatland restoration is being funded through active carbon markets. These markets allow carbon polluters to fund peatland restoration. The aim is that peatland restoration will offset the volumes of carbon emitted by the polluting company. This may allow a company to become carbon neutral or carbon negative (Bonn *et al.*, 2014). Should peatlands be included formally within national GHG inventory schemes and within accredited carbon markets their restoration value may rise even further.

Carbon markets are attractive tools to enable payment for ecosystem services, as benefits are felt globally rather than at a specific locality (Glenk and Martin-Ortega, 2018). However, carbon emission reductions require robust frameworks and verification to increase the reliability and trust in the volume of carbon abatement promised (Evans *et al.*, 2014). Default emission factors for organic soils were incorporated into the IPCC guidelines of 2006 (Eggleston *et al.*, 2006). However, the CO_2 emissions factors were felt to be too low (Couwenberg, 2011). As a result, updated emission factors were included in the 2013 IPCC guidelines update (Hiraishi *et al.*, 2014). This shows the importance of continually improving data and associated policy instruments relating to peatlands.

Carbon verification standards for peatlands already exist such as the Verification Carbon Standard (VCS) and the Climate and Community and Biodiversity Standard (CCCBS). Standards such as the VCS and CCBS have provision for verification of the GHG flux of pre and post restoration peatland sites (Bonn *et al.*, 2014). All verification methods should allow for a combination of repeat field monitoring, remote sensing and computer modelling to quantify carbon stock and flux pre and post restoration to ensure accurate verification (Smith *et al.*, 2020). Ultimately, verification standards will ensure robust and reliable restoration data and inform future restoration efforts.

There are examples of countries developing regional standards to leverage peatland restoration funding. In Germany the Moor Futures offsetting scheme, established in 2011 has c.147 ha of peatland either sold or awaiting sale for carbon offset. The Moor futures scheme adapted the VCS, as this would have been prohibitively expensive relative to the size of restored peatland sites: m=28ha within the scheme (Moor Futures, 2020).

Moor Futures utilised the Greenhouse Gas Emissions Site Types (GEST) methodology. GEST provides proxy carbon emission values by accounting for vegetation assemblages and their relation to water table, pH and land use. This allows carbon emissions pre and post re wetting to be calculated. Consequently, the offset scheme has secured 69,000 tonnes of CO_2e over 50 year timescales as Moor Future credits. The cost of credits ranged from €35 to €80 with a mean credit value of €55 per credit (Moor Futures, 2020).

The UK's Peatland code also acts as a private finance offset scheme. As of the 30th November 2020, 24 projects covering 4,232 ha of peatland restoration are registered on the UK land carbon registry. Four projects have been verified accounting for 101,944 tonnes of CO_2e units. The remaining 20 projects are awaiting verification by independent evaluators (IUCN UK PP, 2020b). Independent verification is an important step in providing confidence surrounding carbon offset figures.

Public funding schemes such as Agri-Environmental are being adapted to encourage peatland restoration and land use mitigation (Reed *et al.*, 2014). Agri Environmental Schemes will be essential in providing financial incentives to achieve transition towards sustainable peat soil management. There are uncertainties surrounding a transition to Payment for Ecosystem Services (PES), especially in relation to restoration. Landowners who have mismanaged their land could benefit more than landowners who have historically managed their land sustainably (Reed *et al.*, 2014; Hansda R *et al.*, 2020). It is highly likely that fostering wider collaboration and partnerships will be needed to achieve ecological restoration beyond financial incentives alone. Encouraging these additional measures will be essential.

The use of direct irrigation will also need evaluation against key *Sphagnum* farming metrics such as *Sphagnum* growth and yield. Additionally, research is required to evaluate the impacts of direct irrigation on water table and the greenhouse gas balance in comparison to studies that raise water tables from below.

A gap in knowledge surrounds the effective spatial distribution of a top-down irrigation system. Studies investigating how top-down irrigation on a bare peat surface affects the water availability for *Sphagnum* plants are required.

10.5 Barriers to uptake

10.5.1 Economics

A limiting factor to widespread adoption is whether *Sphagnum* farming could produce enough material to become a commercially viable alternative for peat in the growing media and horticultural markets. Studies on the economic viability of *Sphagnum* farming are limited. The establishment costs of each of three typical *Sphagnum* farming site types were first investigated in Germany (Wichmann, Prager and Gaudig, 2017) . This was followed up with a study presenting the profitability estimates for *Sphagnum* on former bog grassland, which identified key economic factors as: site establishment, site management, *Sphagnum* harvest and *Sphagnum* processing costs (Wichmann *et al.*, 2020).

Establishment costs at the Rastede site are reported as €128,000 per ha of *Sphagnum* production fields in 2011 (Wichmann, Prager and Gaudig, 2017) and the expansion of the site in 2016 was reported as €98,000 per ha (Wichmann *et al.*, 2020). The highest costs in 2011 were identified as the harvest of founder *Sphagnum*. However, in 2016, site preparation costs were the greatest cost burden. This was due to higher machinery use costs, a greater volume of peat removal, and more time on site required due to adverse weather (Wichmann *et al.*, 2020). The overall expansion of the Rastede site in 2016 was less costly as biomass harvested from the original *Sphagnum* farm was used as founder material which reduced founder material costs by 41%.

Operational costs include water management and crop maintenance. These varied over individual years, at the Rastede site the mean combined operation and maintenance costs for 2011 – 2016 were calculated at €8665 per ha. Site maintenance costs were the largest component.

The Rastede harvest cost calculations assumed a 5-year rotational harvest. Harvest costs were calculated based on labour rates, machinery GPS logs and machinery operation costs. The total field harvesting costs were €12,652 per ha in 2016.

Processing of the harvested *Sphagnum* biomass includes the off-field costs. These comprised loading, off site transport and costs for the cleaning, and screening of air-dried *Sphagnum* biomass. These costs totalled €7.43 per ha of harvested *Sphagnum*.

There may also be future costs following the initial *Sphagnum* harvest. During the establishment of the Rastede site additional *Sphagnum* founder material was applied after 11 months to fill in gaps and achieve a high percentage cover of *Sphagnum* moss (Gaudig *et al.*, 2017). These were not considered in Wichmann's analysis.

10.5.2 Technical Research Gaps

Sphagnum biomass is a suitable raw material for the replacement of peat. However, additional research is required to develop growing media mixes based on this material. work is needed to establish whether *Sphagnum* can be used as a 100% replacement for peat in horticulture. Additionally, a range of GM comprised of peat and *Sphagnum* blends may be possible, which would dilute the amount of peat present in growing media. Research investigating the potential materials for different growing media blends and applications is needed. However care must be taken that demand for a Peat and *Sphagnum* blend doesn't increase the overall volume of peat used in GM. *Sphagnum palustre* and S. papillosum have been identified as the most promising species for GM but other species remain to be tested (Gaudig *et al.*, 2014).

The greatest challenge identified in the boreal region is fine hydrological control (Pouliot, Hugron and Rochefort, 2015), However, this will beneficial for all *Sphagnum* farms. Wider exploration of automated irrigation systems that can simultaneously manage the water input and outflow has been identified as an area for additional research. Further studies are needed to develop balanced *Sphagnum* farming infrastructure in future.

Increased productivity may be achieved through further species selection or even through dedicated *Sphagnum* breeding trials. Focus on gametopyte sex and ploidy levels could offer the greatest potential for increased productivity and need to be investigated.

Commercial development of low-ground-pressure machinery capable of accessing a wet *Sphagnum* field for harvesting will also be necessary for widespread uptake of *Sphagnum* farming.

10.5.3 Legal and regulatory framework

Finally, beyond the immediate technical aspects, for *Sphagnum* farming to be viable, the current political and legal frameworks must be modified to ensure a paradigm shift in the way that peatlands are used for agriculture.

10.5.4 Future scalability needs to be determined

Approximately 145,000 ha of bog grassland has been identified as having the potential for paludiculture in Lower Saxony (Tanneberger *et al.*, 2020) . No such study has been undertaken in the UK to date, although a potential area of 1.2 million ha exists in England within floodplains (Mulholland *et al.*, 2020), and if paludiculture is to be used as a buffer zone, 5,000 ha would be suitable in Cumbria, (see Appendix A1 - Clough, 2017), which is further summarised in (Mulholland *et al.*, 2020).

10.5.5 Wider use of *Sphagnum* beyond growing media

There are potential uses in the biotechnology, medical, packaging, and personal care industries. Further research and innovation will be required to develop wider markets. These markets may further enhance the economic prospects of *Sphagnum* faming.

10.6 Overall project impact

At the start of the research project in 2016, the concept of paludiculture was not well known in the peatland community in the UK, nor was it a widely used term in public discourse. Initially the project partners sought funding from Horizon 2020 for a paludiculture based project and scored highly, but this bid was not successful; the research team felt that this was because it was considered by the evaluation team as almost too novel.

Some initial PhD time was devoted to developing a GIS based opportunity map for paludiculture sites in Cumbria presented at the UK's first ever paludiculture conference in 2017 (Clough, 2017 - see Appendix A2), as well as using some UEL internal funding to develop small scale *Glyceria fluitans* growth trials (Clough, unpublished) and develop the pilot trials for *Sphagnum* presented in Chapter 6.

However, the main focus of *Sphagnum* farming for this research study developed following our involvement within the first UK Research Council funding for a *Sphagnum* farming site which was awarded in 2018, via Innovate UK's Agri-Tech catalyst fund.

The *Sphagnum* farming sites at MicroPropagation services at Sharpley and Little Woolden provided the first opportunity to trial top-down irrigation techniques, combined with novel and conventional mulch covers across deep and shallow peat. At the beginning of the project there was a real possibility that the method could fail entirely, and numerous unknowns surrounded the use of top-down irrigation even basic questions surrounding ensuring water supply and *Sphagnum* survival.

The project was highly exploratory in nature and has certainly proved the concept of top-down irrigation. Through the research study opportunities to increase the interest in paludiculture and *Sphagnum* farming have presented themselves regularly (See Appendix A1). Conference presentations and poster sessions have generated interest with colleagues within the peatland research communities. Wider stakeholder engagement and education with statutory bodies such as DEFRA, Natural England and the Climate Change Committee have been essential in expanding the concept of paludiculture in the UK beyond the academic space.

The research study has led to collaboration with leading civil servants and peatland colleagues on the Paludiculture sub-group of DEFRA's Lowland Peat Agriculture Task Force, which is tasked with reporting to the Minister on the whole future of agriculture on lowland peat soils in England. UEL is currently the sole academic representative within the sub-group, and the new knowledge generated through the research study has been critical to driving the focus of the sub-group and achieving real-world.

The new knowledge generated in this *Sphagnum* farming research study has stimulated new funding opportunities. This has included underpinning the case for support of a £1million 'WaterWorks' project to expand *Sphagnum* farming into farmer-led trials. Trialling *Sphagnum* farming with farmers will be a key step in commercialising the approach in this research study and will lead to further innovation and fill in research gaps.

Chapter 11. Conclusions

11.1 Conclusion

This PhD thesis has explored the context of peatlands and their global importance for ecosystems, while setting out paludiculture as a practical solution to achieve sustainable biomass production on cultivated or extracted peat soils. In the introductory chapters *Sphagnum* farming was introduced as a paludiculture option, with most studies using the MLTT method for founder material with water provided via surface canal irrigation. The potential barriers to wider uptake were explored and the rationale for an alternative *Sphagnum* farming method was set out.

The overall aim of this thesis was to perform exploratory research into a new *Sphagnum* farming method. The Micropropogated with irrigation from above method (MIFA), has been shown to successfully produce cultivated *Sphagnum*. Therefore, the MIFA has a role as a future method of *Sphagnum* farming. The research was highly exploratory, this being the first time Micropropogated *Sphagnum* has been used as a founder material, and to the authors knowledge the first-time that irrigation from above had been used in the absence of active water table management for *Sphagnum* farming on peat soil.

Specifically, the experiments presented throughout this thesis made use of BeadaHumok plugs and Beadagel gel as founder material, while irrigation was supplied via Spray and Drip irrigation systems, commonly available agricultural mulch covers were used mesh and plastic, while the straw mulch as used in conventional *Sphagnum* farming trials was also used.

The broad conclusions are:

- The MIFA approach can produce a good *Sphagnum* crop on both cut over bog sites and shallow organo-mineral sites.

- Top-down irrigation in combination with natural rainfall prevented micropropagated *Sphagnum* from exceeding ecohydrological thresholds for large percentages of time at both sites.
- The use of novel mesh and plastic mulch covers was generally beneficial in the MIFA system but will benefit from future investigation.
- TLS was successfully used to measure *Sphagnum* crop growth non-destructively and could be used at larger scale: i.e., drone based lidar.

11.2 Scientific novelty

For researchers, we have identified that the Micropropagated with Irrigation from Above (MIFA) approach offers a new, promising method for *Sphagnum* farming. We intend this method to act as an intermediary step towards paludiculture, with a low barrier to entry compared to full rewetting as MIFA opens up the possibility of cultivating *Sphagnum* without fully re-wetting peat soils using irrigation canals.

One area of novelty is the use of micropropagated *Sphagnum* as a founder material, this was proven to be just as effective in both plug and gel form using a single species *Sphagnum* palustre compared to donor founder material gathered from natural sites. This method could be used in future to trial additional species, or the use of multi-species plugs as *Sphagnum* farming founder material.

The research highlighted that for the MIFA approach, PWP is the more useful hydrological metric to focus on rather than water table due to the lack of active water table management. This may have implications for future monitoring projects if the approach is adopted more widely.

The MIFA approach has merit, and produced good *Sphagnum* crops, with 'in the Field' yields of 10.48 to15.2 t dm ha $^{-1}$ yr $^{-1}$ at Sharpley and 7.83 to 22.75 t dm ha $^{-1}$ yr $^{-1}$ at Little Woolden across covered plots. Some but not all plots measured had yields higher than natural productivities and other *Sphagnum* farming studies to date. However, 'in the field' yields were calculated using literature values and will be updated at the point of harvest after 5 years.

The method needs further work and refinement to ensure wider uptake, and answering many of the technical questions identified in the aims and objectives chapter will be required to achieve commercial viability. Future work needs to investigate the water and carbon budgets of the MIFA method at field scale, while revealing further economic and logistical information at the point of harvest.

11.3 Practitioner novelty and impacts

Top-down irrigation as part of the MIFA approach was shown to be effective in delivering water to *Sphagnum* under cultivation on cut over bog and agricultural peat soil sites.

Standard agricultural mulch covers (Wondermesh and micro-perforated plastic) proved to be as effective as straw covers in terms of preventing *Sphagnum* desiccation and encouraging high yields. This suggest that plastic or mesh covers could be made available to practitioners, and a wider range of covers could also be investigated in future. The covers tested are potentially reusable and can remain on the crop for the duration of cultivation. Unlike the straw covers that degrade over time and offer little protection after 12 months or so.

The use of Micropropagated *Sphagnum* is new for *Sphagnum* farming but offers many practitioner benefits. In the long term, both Plugs and Gels performed well in terms of growth. However, there are some key differences to consider. Firstly, the BeadaGel allowed for rapid, single user application in the field compared to the BeadaHumok plugs which were planted by hand over several days. The trade-off for speed is that BeadaGel may be more susceptible to drought or flooding during establishment compared to plugs, so the use of either option may depend on confidence in a site’s infrastructure, the weather conditions at time of establishment and speed vs cost.

Practitioners may seek to develop more efficient methods of planting BeadaHumok plugs, potentially via adapting existing on farm machinery. This would develop faster planting of BeadaHumok plugs while giving additional drought resistance during *Sphagnum* establishment before a high percentage cover is reached after 6 to 9 months.

Practitioner led trials are something that need to take place in future, many aspects of commercially viable *Sphagnum* farming will only take place with wider uptake, the application of farm level knowledge and the willingness to trial and innovate. Practitioners will be best placed to lead these innovations, ideally backed up by scientific support and funding for early adopters, until *Sphagnum* farming and paludicultures are embedded within future Payment for Ecosystem Schemes such as ELM’s.

11.4 Final Statement

Overall, the MIFA approach shows good potential and offers an additional interim method for *Sphagnum* farming while legislation develops to support peatland re-wetting and paludiculture further. Therefore, researchers and practitioners should take confidence from this study and begin to explore the MIFA approach in future work; this will generate further confidence in the approach and support the wider uptake of *Sphagnum* farming across lowland peat sites in the UK.

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Sharpley location map

High Resolution (25cm) Vertical Aerial Imagery [JPG geospatial data], Scale 1:500, Tiles: sk4517,sk4516,sk4417,sk4416, Updated: 27 October 2021, Getmapping, Using: EDINA Aerial Digimap Service, <https://digimap.edina.ac.uk>, Downloaded: 2022-12-30 15:44:57.871

LW Location map:

High Resolution (25cm) Vertical Aerial Imagery [JPG geospatial data], Scale 1:500, Tiles: sj6995,sj6895,sj6894,sj6994, Updated: 27 October 2021, Getmapping, Using: EDINA Aerial Digimap Service, <https://digimap.edina.ac.uk>, Downloaded: 2022-12-30 16:22:12.667

Appendices

A1: Summary of paludiculture research project dissemination events

Date	Event	Contribution
29th November – 1st December 2016	IUCN UK Peatland Programme conference: developing a legacy for peatlands	Poster presentation
2017	Paludiculture UK conference	Oral presentation as conference speaker
29th January 2018	The Great Fen Joint Technical Advisory Committee	Oral Presentation
2nd – 4th October 2018	IUCN UK Peatland Programme conference: Peatland Connections: Building Prosperity	Oral presentation as conference speaker
26th September 2018	Introduction to paludiculture for DEFRA, BEIS, CCC and NE	Oral presentation as lead speaker
19th March 2019	Peoples Postcode Lottery, Dream fund: WaterWorks funded project	Oral presentation in funding bid (successful) and member of academic research team
April 2020	Literature Review: DEFRA Project SP1218 An assessment of the potential for paludiculture in England and Wales	Co-Author on report on the potential of paludiculture for DEFRA.
7th to 10th December 2020	IUCN UK Peatland Programme conference: Peatlands from Strategy to	Oral presentation and panel questions

	Action	
20th May 2021	In conversation with International Environmental Negotiations team DEFRA	Oral presentation introducing paludiculture to the IEN core team
15th June 2021 – present	Funded project GGR Peat – BBSRC/UKRI	Member of academic research team
21st July 2021	ITV Anglia news interview about WaterWorks project	Interview with journalist for live broadcast, public impact.
1st November – 12th Nov 2021	UNFCCC COP 26 – Peatland Pavilion	Core member of delivery team
22nd December 2021 - present	Member of the DEFRA paludiculture sub-group within the Lowland Agricultural Peat taskforce	Core Academic member of sub-group – developing the 10-year policy roadmap for England.
14th– 24th March 2022	Paludiculture sub-group stakeholder workshops and introductory webinar.	Oral presentation and workshop delivery.
September 2022	Applying as an academic partner in funding bid Farming Futures - Innovate UK	Member of academic research team in funding bid, pending.

A2: Paludicultures in Cumbria GIS mapping exercise

An opportunity mapping exercise took place early in the PhD programme to assess the potential of paludiculture sites as buffer zones around lowland peat SSSI's in Cumbria.

The results were as follows:

A Cumbria paludiculture map has been developed. This has been achieved by layering the following GIS datasets: SSSI site area and location, Agricultural land class categories, Land slope angle (created from a 50m resolution Digital Terrain Model) and the Natural England peaty soils layer.

From the map, Peat soil area in Cumbria was determined at approximately 217,000 ha. This area was then investigated to highlight areas with paludiculture potential. Suitable areas were defined as those on peaty soils, with a slope angle of 2 degrees or less, that are not in urban areas. As these sites could achieve the necessary level of water control with minimal groundworks compared to sites with greater slopes.

This resulted in an area of peat soils suitable for paludiculture of 108,000 ha across Cumbria.

However not all peat areas will be suitable for paludiculture based systems at current maturity levels – 'easy win' areas may offer the best opportunity for the initial development of paludiculture in the region to highlight the success of these new farming systems.

The Agricultural Land Classification system places land into a series of categories, being 1 excellent, 2 very good, 3 good –moderate, 4 poor, and 5 very poor. It makes sense in the first instance to keep paludicultures on the lower graded land (3,4,5) as this land is likely to be more amenable to land use change.

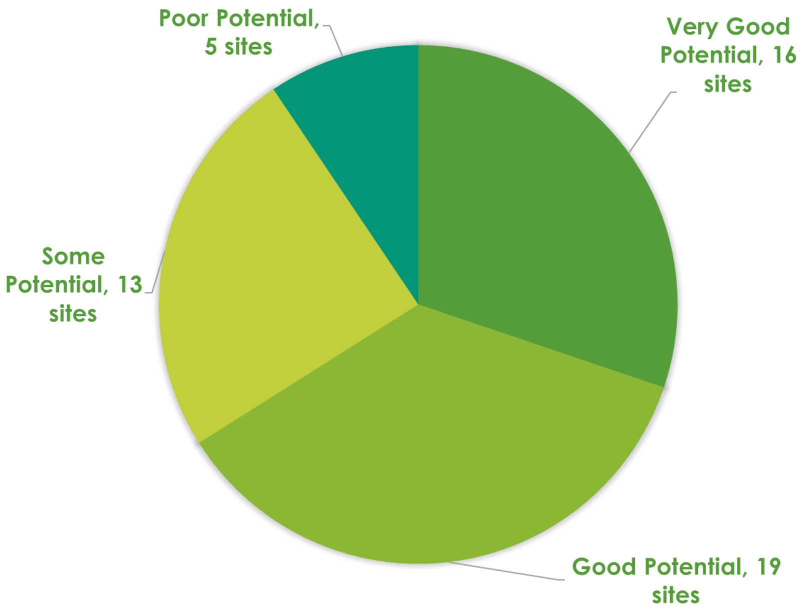
Paludiculture systems also have the potential to encourage more mutually beneficial land uses on the margin of currently protected wetland sites. Designated sites of a peatland interest are often surrounded by agricultural land on peaty soils. For conventional agriculture, the usual aim to keep this land dry, while the neighbouring designated land is being kept wet for conservation purposes.

Areas that offer further benefits are those in areas of higher flood risk (EA zones 2 and 3) or are immediately adjacent to areas under some form of environmental/conservation designation, as tying in paludicultures from surrounding areas may offer a conservation benefit to core reserve areas.

Subsequently a scoring system was developed for rating paludiculture potential within 1 km buffer zones surrounding Cumbria's lowland SSSI's with a peat component:

SSSI buffer - Paludiculture Potential Questions	Score
Is there unprotected peat within the SSSI buffer?	1
Is the peat within the buffer well connected?	1
Does at least some of the unprotected peat connect with the SSSI	1
Does the buffer area contain over 10% peaty soils, with a slope angle of 2 degrees or less?	2
Does the buffer area intersect with an EA flood risk zone?	2
Is the majority of the buffer area on land with an ALC of 4 and 5?	2
Is the majority of the buffer area on land with an ALC of 2 and 3?	1
total score (Max 9)	

The scores were as follows:



Summary of GIS layers used (under licence):

- Peaty Soils Locations (source: NE, BGS, NSRI and OS)
- SSSI Locations (source: data.gov.uk under open government licence).
- Cumbria County Administrative boundary (source: data.gov.uk under open government licence).
- Agricultural Land Classifications (source: data.gov.uk under open government licence).
- Environment Agency Flood Risk Zones 2 and 3 (source: data.gov.uk under open government licence).
-

The results of this work were presented in a talk titled 'Paludicultures in Cumbria' at the first UK paludiculture conference in 2017.

To be cited as:

Clough, J. 2017. 'Paludicultures in Cumbria' Paludiculture UK 2017: Working with our wetlands, 29-30 November 2017 Kendal, Cumbria. Available online: Paludiculture UK Conference 2017: Working with our wetlands - PUKC001 (naturalengland.org.uk) (Accessed: 21 January 2023)

The Cumbria paludiculture potential work is also referred to in Appendix iii within the following report:

Mulholland, B., Abdel-Aziz, I., Lindsay, R., McNamara, N., Keith, A., Page, S., Clough, J., Freeman, B. and Evans, C. 2020. Literature Review: DEFRA project SP1218: An assessment of the potential for paludiculture in England and Wales. UK Centre for Ecology & Hydrology.

And is presented below for convenience:

Appendix III GIS Desk Study Approach to Mapping Potential Paludiculture Areas in Cumbria.

There are large areas of peaty soils that could be converted into paludiculture within Cumbria, especially in undesignated areas. A 5m DTM may be useful to increase the accuracy of this potential area. The area referred to in this desk analysis should be taken as a starting point and can only be as accurate as the data used for the analysis.

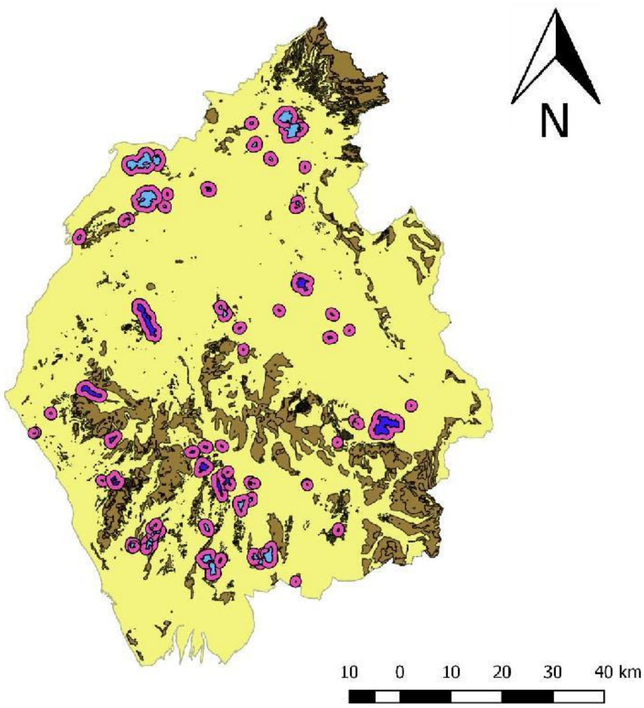


Figure 4: Basic paludiculture potential map for Cumbria – the light brown areas are peaty soils with a slope angle of 2 degrees or less – selected as suitable for paludiculture in this study. Areas of light blue are Lowland Raised Bog sites and dark blue are Lowland Fen sites. Pink shading illustrates a 1km buffer surrounding the SSSI areas. Please note that rivers designated as SSSI's have been omitted from this analysis.

Designated peatland sites are often surrounded by agricultural land on peaty soils. For conventional agriculture, the usual aim is to keep this land dry, while the designated land adjacent to it attempts to keep the land wet for conservation purposes. By encouraging paludiculture adoption around designated sites a working buffer zone could be established, bringing both areas of land use into a mutually beneficial system of infrastructure and water management.

Table AIII.1 – GIS Map Layers and their use in our Analysis

GIS MAP LAYERS	DETAILS
BASE	Cumbria administrative boundary map (source: data.gov.uk, under open government licence).
FLOOD RISK MAP	Environment Agency Flood Risk Zones 2 and 3 (source: data.gov.uk, under open government licence). Areas with a high flood risk downstream of peaty soils could benefit by the adoption of paludiculture practices which aim to rewet the peat, store additional water, and regulate the flow. All of which could lower flood risk. Flood Risk Areas also highlight areas that may stay wetter for longer and present opportunity areas for paludiculture set up. They also may reveal areas where peatland was once present but has since been lost to agriculture or other human uses.
PEAT SOILS	Taken from the Natural England 'Peaty Soils' dataset based on shallow, deep and peaty soils. Peaty Soils Locations (source: NE, BGS, NSRI and OS).
SLOPE ANGLE	This was created taken from a 50m resolution DTM and limited to slopes with an angle of two degrees or less. I. EDINA Digimap OS Service: OS Terrain 50 DTM [TIFF geospatial data], Scale 1:50000, Tile(s): Cumbria II. Updated: Aug 2017, Downloaded: Sept 2017 The 'peaty soils' dataset was then clipped to match the extent of the slope angle layer to create a potential paludiculture area layer.
LAND USE	Agricultural Land Classification (ALC) was used to assess land use - Urban areas were clipped from the potential paludiculture area layer, leaving agricultural areas and non-agricultural areas only (source: data.gov.uk, under open government licence). The final potential paludiculture area – deemed as areas that are not urban areas, with peat soil (of any depth) and a slope angle of two degrees to aid in infrastructure installation and access. Areas on ALC with a lower grade may offer an easier area to start paludiculture conversion on – though arguably the higher grades are likely to have a greater amount of peat depth remaining, so needs protecting via a change in farming practice too.
SSSI SITES	Designated SSSI sites were plotted on the opportunity map (created from all other layers which identified **approximately 108,000 ha**) - These were limited to Lowland Raised Bog sites and Lowland Fen sites based on Natural England's Broad Habitat categories. A 1km buffer zone was established around these sites. The potential paludiculture area within the SSSI buffer zones was revealed to be approximately **5000 ha**. SSSI Locations (source: data.gov.uk under open government licence).

A3: Timeline of Main Sphagnum project sites

The timeline for the Planting of *Sphagnum* and TLS scans has been presented below, to help explain the differences between Sphagnum ages across the main study sites.

Sphagnum farm project timeline						
Sharpley				Little Woolden		
Date	**Drip Irrigation area**	**Spray irrigation area**		**Date**	**Drip Irrigation area**	**Spray irrigation area**
Aug-18	All plugs planted	All plugs planted		Oct-18	Plugs planted	All plugs planted
Sep-18	All gels planted	All gels planted		Nov-18		Row 5 planted with gel
Feb-19	TLS Scan 1			Feb-19	TLS scan 1	
Sep-19	TLS Scan 2			Apr-19	Gels planted	Rows 2,3,4 planted with gel
Aug-20	TLS Scan 3			Sep-19	TLS Scan 2	
				Aug-20	TLS Scan 3	
Sphagnum age at TLS scans				Sphagnum age at TLS scans		
TLS Scan	**Drip Irrigation area**	**Spray irrigation area**		**TLS Scan**	**Drip Irrigation area**	**Spray irrigation area**
Scan 1	c.6 months for plugs at sharpley, and 5 months for gels though over winter little growth.	c.6 months for plugs at sharpley, and 5 months for gels though over winter little growth.		Scan 1	c. 4 months for plugs planted in October 2018. Though over winter so little growth. No gels present in this scan	c. 4 months for plugs planted in October 2018. Though over winter so little growth. No gels present in this scan
Scan 2	c. 13 months for plugs and 12 months for gels	c. 13 months for plugs and 12 months for gels		Scan 2	c. 11 months for plugs planted in October 2018, and 5 months for gels planted in April 2019	c. 11 months for plugs planted in October 2018, 10 months for row 5 gels planted in November 2018 and 5 months for gels planted in April 2019
Scan 3	c. 24 months for plugs planted in August, 23 months for Gels planted in September	c. 24 months for plugs planted in August, 23 months for Gels planted in September		Scan 3	c. 22 months for plugs planted in October 2018. 16 months for gels planted in April 2019	c. 22 months for plugs planted in October 2018. 21 months for row 5 gels planted in November 2018, and 16 months for gels planted in April 2020
Note Sphagnum ages are approximate, as planting took place over 1 to 2 weeks						