Literature Review: Defra Project SP1218

An assessment of the potential for paludiculture in England and Wales

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EXECUTIVE SUMMARY

In England, there are approximately 325,000 ha of lowland peatlands, with 240,000 ha (74% of the total stock) used for farming and food production. Much of this has been drained to maximise yields of high value fresh produce crops. Sixty nine percent of the cropped peatlands in England are in regular use for horticulture crop production, with the remainder being used for arable/cereal rotation. Peat wastage under cropland is typically 10 to 30 mm yr^{-1}, and this land has the highest greenhouse gas (GHG) emissions of any UK land-use on peat (>10 times higher than emissions from upland peat).

As a consequence of these issues, there is a growing interest in ameliorating peat and associated CO_2 loss through 'paludiculture' – namely farming and agroforestry systems designed to generate a commercial crop from wetland conditions using species that are typical of (or tolerant of) wetland habitats. The prospect of raising water levels to reduce emissions in peatlands managed for production, demands new ways of growing existing crops, or new crops capable of thriving when subject to the elevated water tables. Where trials have been undertaken, findings suggest that paludiculture has the potential to reduce CO_2 (and overall GHG) emissions relative to conventional drainage-based agriculture or peat extraction. This mitigation potential largely takes the form of avoided present-day CO_2 emissions from deep-drained peat cropland, which can be as high as 25-30 tonnes CO_2 equivalent per hectare per year. A number of studies, however, suggest that paludiculture sites could become net CO_2 sinks, thereby helping to remove GHGs from the atmosphere. Sequestering CO_2 by adoption and uptake of paludiculture techniques has the potential to make an important contribution to achieving the UK's commitment to net zero GHG emissions by 2050.

Wide scale production of paludiculture bioenergy crops, such as *Phragmites australis* (Common Reed) can sequester in the region of 4–13 $tCO_2e\ ha^{-1}\ y^{-1}$. As these crops will be used to produce energy, sequestered CO_2 will be re-emitted, resulting (if all sequestered CO_2 is removed from the site as biomass) in a neutral carbon balance at the stand scale. If the bioenergy crops produced substitute for fossil fuel use, the net CO_2 balance, however, will be negative. Other uses of harvested biomass, including incorporation into building materials or the return of biomass carbon to the soil in unreactive forms, have the potential to contribute directly to long-term CO_2 sequestration.

The majority of peat used in horticulture consists of decomposed *Sphagnum* moss. One proposed form of paludiculture is the production of *Sphagnum* crops which can be harvested and processed to create growing media. *Sphagnum* grown at high water tables (e.g. on former peat extraction sites) could be potentially utilised in blended growing media products. Significant research and development will be required however, before *Sphagnum* can be considered viable for containerised plant raising. There is also a need to demonstrate economic viability, since alternative processed media must be offered at a comparable and affordable price point, assuming a product can fulfil the required physical, chemical and biological requirements for containerised production.

Paludiculture is a land-use focused on farming and agroforestry as the primary activity, and biodiversity benefits therefore arise as an 'added-value' of the activity. Rewetting of lowland peat for paludiculture could nevertheless generate a number of co-benefits for the environment, wildlife and human wellbeing. Although likely to be less than those that could be achieved through full peatland restoration, the combination of direct financial returns from crop production, climate change mitigation and biodiversity benefits could help to make paludiculture economically viable provided that appropriate financial mechanisms are put in place which recognise these broader societal benefits.

There remain significant practical, economic and societal challenges for the large-scale implementation of paludiculture, including the need to support rural economies, maintain national food security, develop markets and supply chains, manage water within complex and heavily modified landscapes, and avoid displacement of emissions associated with food production to other areas. Facilitating the wider adoption of paludiculture is likely to require the development of financial incentive schemes for farmers, landowners and investors, new regulatory approaches and investment in supporting infrastructure. This is turn requires a stronger evidence base, both to develop viable paludiculture systems and to accurately quantify the associated benefits and trade-offs. Compared to conventional crops grown under higher water tables, paludiculture crops may offer lower but more reliable economic yields, and by protecting the soil from ongoing loss may help to maintain the productive lifetime of the and. The high water demand of paludiculture crops presents some challenges in water-scarce regions, but well-deisigned areas of paludiculture within farmed landscapes could also provide effective water storage within the landscape, holding flood water during winter and releasing some of this to adjacent farmland during summer. The incorporation of paludiculture areas within farmed landscapes may therefore enhance their overall resilience to climate change.

As yet, paludiculture does not offer a comprehensive economic, large-scale, immediately implementable solution to the challenge of high GHG emissions from cultivated lowland peats, and other forms of emissions mitigation such as high water level management of conventional agricultural land are likely to be needed. However, with further development of crops, water management systems and markets, paludiculture has the potential to become to make a valuable contribution to the development of more sustainable and resilient peatland farming systems in future, and to contribute to delivering the UK's net zero emissions target.

1 INTRODUCTION

This chapter considers the importance and extent of peatland agriculture in the United Kingdom and introduces seasonal/continuous rewetting of peat soils as an alternative production practice in former wetlands. **Section 1.1** contextualises the drainage of lowland peat soils, **Section 1.2** highlights the agricultural importance of peatlands, and explores their condition and extent, **Section 1.3** considers the act of rewetting and its potential to limit GHG emissions, and **Section 1.4** introduces paludiculture as a productive use of cropland, under high water table conditions.

1.1 History of Peatland Agriculture in the United Kingdom

More than two millennia of wetland drainage and conversion to other habitats have resulted in the widespread loss of wetlands throughout the global terrestrial and coastal environment (**Davidson, 2014**). The driving force behind much of this conversion, at least in the west, has been the concept of agriculture as an essentially 'dryland' activity. This is because a large proportion of crop species that underpin western agriculture have their origins in the semi-desert regions of the Middle East; which included grass species such as, emmer wheat (*Triticum dicoccoides*), einkorn wheat (*Triticum monococcum*) and barley (*Hordeum vulgare*) (**Skoglund *et al.*, 2012; Yavas, Unay and Aydin, 2012; Sheenan, 2018; Shiono *et al*., 2019**).

Consequently, even while agricultural practices until the modern post-war period were giving rise to increased biodiversity associated with open ground habitats, wetland biodiversity was declining because wetlands were seen as a hindrance to productive farming. Huge tracts of land in the lowlands were drained by investors seeking to convert land that was agriculturally low-value in the conventional sense into highly profitable farmland – perhaps the most well-known example of this being the draining of the East Anglian Fenland in the 17th-century (**Darby, 1940**; **Darby, 1956**; **Sheail and Wells, 1983**). However, the most extreme of these occurred during WW2, in which the War Agricultural Executive Committees catalogued every aspect of all farms with a view to maximising food production (**Darby, 1956**; **Humphries and Hopwood, 1999**).

The results achieved by wetland drainage have undoubtedly been extraordinary. The UK's cultivated lowland peatlands contribute substantially to regional and national economic output, food security and livelihoods. The Fens of East Anglia now provide around 50% of all ALC Grade 1 agricultural land in England, they supply 33% of England's fresh vegetables (**NFU, 2019**), yet the peatlands of the Fens, the Somerset Levels and the Humberhead Levels together represent only 1.6% of all utilised agricultural area (UAA) devoted to cropping or lowland grassland in England (**Morris *et al*., 2010; Defra, 2019**). Indeed, **Williams (1990b)** observes that wherever wetland soils are converted to agricultural use, they usually contribute to agricultural productivity out of all proportion to the area involved. As such, the large-scale cessation of agricultural activity on lowland peats and flood-prone ground would have potentially severe regional, and potentially national, social and economic impacts, if not replaced with alternative sources of income and food production (**Morris et al., 2010**).

Wetland drainage has been driven forward to such an extent that in England every floodplain has been converted to some form of drained, productive agricultural land (as well as, to a lesser extent, urban or industrial use). Less than 1% of the original 4,000 km^2 East Anglian Fens continue to support wetland habitat (**Sheail and Wells, 1983; Rotherham, 2013; Fens for the Future**), while agriculture now represents the major land cover type for more than half the area of the original 37,500 ha of lowland raised bogs in England (**Lindsay and Immirzi, 1996**). This picture is mirrored elsewhere around the world; an estimated 48% of former wetlands in the United States had been lost by 1975 with around 90% of this loss resulting from agricultural development (**Williams, 1990b**). Estimated losses of peatland habitat exceed 50% for a number of Central European nations (**Bragg and Lindsay, 2003**), while globally, the available evidence points to wetland loss of more than 50%, with losses occurring four times faster in the 20th Century than any time prior (**Davidson, 2014**).

Modern drainage continues and is organised by a series of Internal Drainage Boards (IDBs), many of which were founded in the 18th century (**Ely Group of IDB's, 2016**). According to the Institute of Civil Engineers, at least two major periods of drainage infrastructure development took place during 1829-1845 and 1964-1974 (**ICE, 2018**). IDBs manage the drainage of around 1.2 million ha of agricultural grade land in England, with pumped drainage necessary for around 50% of this drained land (**Roca et al., 2011**).

1.2 Extent and Status of Lowland Peatlands in the United Kingdom

Across the UK, there are about 325,000 ha of lowland peatlands, with around 194,100 ha (60% of the total stock) designated as cropland (**ONS, 2019**). Of this area, the large majority (182,700 ha) is located in England. Sixty nine percent of this cropped peatland is assigned to horticulture, with the remainder being used for arable/cereals production. The largest area of cultivated lowland peat in England (and the UK as a whole) is the Fenlands of East Anglia, which harness 7% of England's agriculture production on less than 4% of England's farmed area **(NFU, 2019)**. Growers in the Fens produce a significant percentage of the country's cereal, oilseed rape and protein crops; as well as one fifth of England's production output for each of the vegetables, potatoes, flowers and bulbs, and sugar beet sectors **(NFU, 2019)**. Other areas of cropland on lowland peat include parts of the Humberhead Levels, Lancashire Mosses and north Nottinghamshire (Isle of Axholme).

Approximately 72% (132,100 ha) of the cropland is classified as 'wasted' (retaining a peat layer of < 40 cm), and the remainder (50,600 ha) is on deep peat. Deep peat is generally the highest grade (ALC Grade 1) agricultural land, whereas wasted peat (also referred to as 'skirtland' in the Fens) is of lower agricultural value and is less suitable for the cultivation of high value vegetable crops (**Rob Parker, G's Fresh pers. comm.**). The current distribution and extent of wasted peat is, however, open to interpretation. The term is most often applied to areas within the East Anglian Fens and the Humberhead Levels, based largely on former accounts of deep peat and its distribution. However, this tends to overlook other parts of the UK where lowland peat is, or was formerly, present; with little mapping of either deep or wasted peat distributions in recent decades.

Large areas of lowland peat are also drained to form grassland for livestock rearing, with around 141,000 ha of intensive grassland on deep peat, and a further 35,000 ha on wasted peat (**Evans et al. 2017**). The area of lowland peat under intensive grassland is more broadly distributed across the UK, with the largest areas in Scotland and Northern Ireland, and substantial areas also occurring in Wales and in some parts of England such as the Somerset Levels. The majority of the UK's 8000 ha of industrial peat extraction is on lowland raised bog in England and Scotland, with a smaller area in Northern Ireland.

Impacts of drainage-based agriculture on peat include high rates of carbon loss through peat oxidation and consequently high greenhouse gas emissions. These have been estimated to be in the region of 12 Mt CO_2e yr^{-1} from cropland and intensive grassland combined, and account for over half of all estimated UK peatland emissions (**Evans et al., 2017**). Over half of these emissions are associated with wasted peat, although these emissions are uncertain and will be quantified more accurately via a new project for BEIS ("Toward an accurate estimate of wasted peat GHG emissions in England", Project reference TRN 2177/12/2019). The oxidation and shrinkage of wetland soils has also led to a significant and ongoing process of land subsidence which has presented increasing challenges to farming communities in terms of continual and increasing drainage costs, coupled with increasing levels of flood risk. Rates of subsidence in cultivated UK lowland peatlands are believed to be in the range of 10 to 30 mm yr^{-1}, resulting from oxidation and compaction (**ONS, 2019**). Issues related to drainage and subsidence are examined in more detail in a separate review for Work Package 2 of this project (**Page et al., 2020**)

1.3 Peatland Rewetting

As a result of the contribution of drained peatlands to GHG emissions and other environmental impacts, there has been a growing policy drive towards peatland rewetting. To date, this has been focused mainly on upland blanket bog, where the opportunity costs of reduced agricultural output are small (especially after accounting for subsidies), but the overall level of emissions reduction that can be attained is also relatively small. In the lowlands, on the other hand, the potential emission reductions, opportunity costs and potential socio-economic consequences of rewetting are all much higher, representing a significant policy challenge. Consequently, although some wetland restoration projects have taken place on lowland peat, these are limited in scale at present. Nevertheless, the UK Government's 25 Year Environment Plan does recognise that current agricultural practices on drained peatlands are inherently unsustainable (**Defra, 2018**).

Table 1.1** Committee on Climate Change key recommendations on lowland peat to deliver net-zero. Adapted from: **CCC (2020)

Recommendation	Proposed Date
Ban peat extraction and its sale, including of imports: Since 2012, no new licences for extraction have been granted. Nevertheless, some existing licences are not due to expire until the early 2040s - *a voluntary phase out scheme for peat is in place for horticulture use in England by 2020 for the amateur market and by 2030 for the professional market*. (See **Section 3.1**)	Before 2023
Regulate that peat soils are not left bare: Where cover crops are used between rotations (in winter months), potential of some nitrogen fixing crops to increase N_2O emissions should be taken into consideration when selecting crops.	From 2021
Require internal drainage boards to maintain optimal water table levels: The overriding control on CO_2 emissions from lowland peat is mean water-table depth. It is estimated that for every 10 cm increase in the water table, there is a corresponding reduction in emissions of 3 tCO_2e/ha.	Before 2023
Public funding for sustainable management practices, and restoration of low value land (e.g. grasslands): These include switching to continuous or seasonal management of the water table and the use of cover crops. Public funding can be used to restore agricultural land with lower opportunity costs such as grassland, of which there are 2,000 ha in the Fens and 4,250 ha in Somerset.	From 2021
Research to improve verification and, in the longer-term, use of market mechanisms to pay for carbon benefits: Restoration of more valuable cropland could be funded by the private sector through the purchase of carbon credits. The Peatland Code would have to be extended to capture the different types of degradation and restoration methods in the lowlands.	By mid-2020s

Overall, the Committee on Climate Change (CCC) has incorporated a restoration target of 25% of cropland on lowland peat by 2050 **(CCC, 2020)**. Although, comparatively modest in terms of the overall reduction in emissions that rewetting would generate, the socio-economic impacts of taking a quarter of high-value farmland on lowland peat soils completely out of production, coupled with a pre-emptive ban on peat extraction and its use in growing media, would be substantial. This would also carry the risk of displacing emissions associated with food production to other areas, including other countries. The underpinning analysis for the CCC study (**Thomson et al., 2018**) also considered the potential for seasonally raised water level management of cropland systems on peat, and suggested a potential emission saving of around 1.5 $MtCO_2e\ yr^{-1}$. However, the practicability of this approach has yet to be demonstrated and work is ongoing to evaluate the impacts of raising water levels in these systems. An alternative approach, considered here, is paludiculture.

1.4 Paludiculture

Paludiculture is the term used to describe farming and agroforestry systems designed to generate a commercial crop from wetland conditions using species that are typical of (or tolerant of) wetland habitats. The Ramsar Convention definition of wetlands (see Article 1.1 of **Ramsar, 1994**) provides for a wide range of conditions under which paludiculture may be undertaken.

In Germany the practice is largely confined to those areas having a peat soil, but it may be extended to other wetland soils, including those in the UK where conventional agriculture may no longer be viable in the future. In some wetland settings, paludiculture could offer continued commercial cropping from ground where conventional agriculture is no longer possible or desirable. This may involve areas of peat soil (peatlands), areas that have lost their peat soils ('wasted' peat or 'skirtland'), or areas of mineral soil subject to permanent or periodic waterlogging ('marsh'). The dominant feature of all areas with potential for paludiculture is that they are prone to waterlogging, and thus often artificially drained for conventional agriculture or to reduce flood risk.

Paludiculture does not seek to displace conventional agriculture, rather it represents one possible option from an assortment of solutions where conventional agriculture is already being rendered non-viable as a result of internal factors or external forces. There may also be potential to transform isolated areas within an agricultural holding which are currently not productive, as a result of a high water table or its tendency to flood, into areas of higher productivity by farming plant species adapted to waterlogged conditions.

Although paludiculture may also be practiced in surviving wetland habitats (and to some extent is, through activities such as reed-cutting), these remnants are often so limited in extent that they are highly valued for their biodiversity and other ecosystem benefits; to the extent that more intensive commercial exploitation via paludiculture may not be permitted.

It is important to be clear from the outset that paludiculture is a land-use focused on farming and agroforestry as the primary activity and is not directly concerned with questions of biodiversity, habitat loss or nature conservation (**Wichtmann et al., 2016**). Biodiversity benefits arise as an 'added-value' consequence of paludiculture rather than as an intended product of the activity. In this, it is similar to the concept of '*satoyama*' in Japan, where traditional farming of rice paddies is being actively maintained or re-introduced as a deliberate decision to provide sustainable management and livelihoods for local communities, resulting in the production of high-quality and high-value rice. A fortunate and unplanned by-product of this move towards *satoyama* rice farming has been a dramatic increase in the wetland biodiversity in those areas (**Watanabe et al., 2012**); including the first breeding success of storks (*Ciconia boyciana*) in several generations associated with the huge increase in wetland species that form the prey of this top predator (**Hasegawa, 2010**).

Despite the focus on crop production, rather than habitat conservation or restoration, it is clear from various examples around the world that paludiculture has the potential to support and enhance both these environmental activities as well as generating a wider range of ecosystem benefits associated with wetland habitats (**Wichtmann et al., 2016**; **Tanneberger et al., 2016**; **Dommain, 2016**). Potential co-benefits of paludiculture management of lowland peat are considered in **Section 5** of this report.

2 PRODUCTIVITY AND SUITABILITY OF LAND FOR PALUDICULTURE

This chapter considers the productivity of wetlands, the key physiological traits of its plants, and identifies under which conditions paludiculture can be practiced. **Section 2.1** compares the dry-matter productivity of plants under wetland and conventional systems, **Section 2.2** explains the constraints imposed by waterlogged soils on crop production, and the properties of wetland species which allow them to thrive under wetland conditions, and **Section 2.3** develops a broader definition of potential paludiculture sites, including methods for their identification.

2.1 Wetland Productivity

The assumption that successful crop production is only possible from dry ground, although true for many conventional agricultural crops and associated agricultural infrastructure, there is nothing inherently unproductive about wet ground. In fact, wetlands are some of the most productive ecosystems on Earth; some wetland types are capable of matching the dry-matter productivity of such high-intensity farming systems as those of the Midwestern United States (**Williams, 1990a**).

Some wetland species, notably rice, have driven the rise of civilizations (**Te-Tzu Chang, 1987**). Other wetland species or species assemblages provide food indirectly by generating and maintaining conditions that support environments which are themselves then the source of high-yielding foodstuffs, such as riverine or coastal fisheries (**Ringler and Cai, 2006; Setiadi, 2014**). **Oehmke and Abel (2016)** give figures for dry matter yields of various wetland plant species harvested from 'natural vegetation' or examples of 'spontaneous succession'. These yields are compared in **Figure 2.1** with benchmark values for some conventional grassland and arable crops.

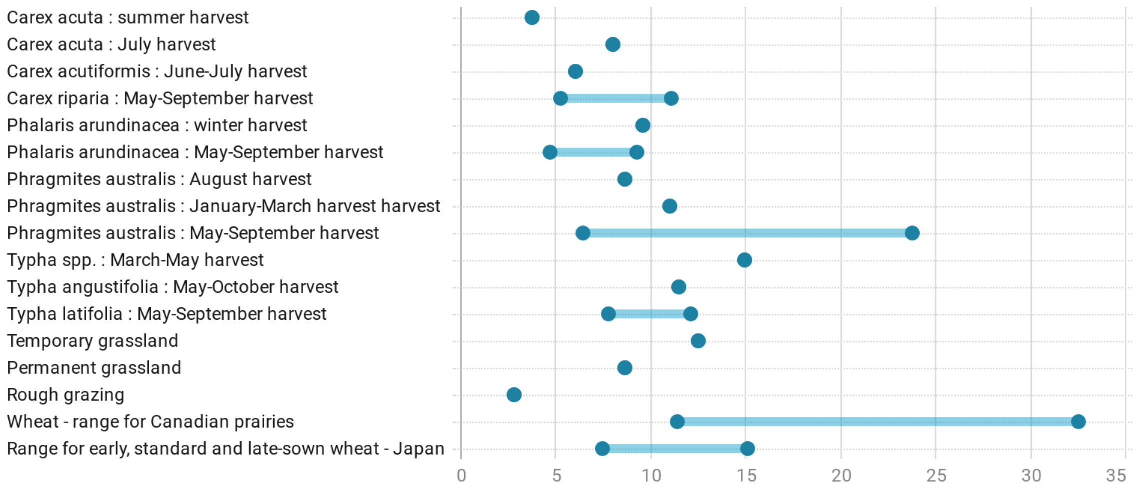


Figure 2.1 *Average annual dry-matter yields (t ha^{-1} yr^{-1}) of selected wetland plants harvested at differing times for examples of natural or spontaneous vegetation stands, taken from **Oehmke and Abel (2016)**, compared with dry-matter yields from grassland (**Qi et al., 2018**) and wheat (**Huffman et al., 2015; Sawada et al., 2019**)*

From **Figure 2.1** it is clear that some wetland plants can match the dry matter productivity of much agricultural grassland and some arable wheat fields. Although a direct comparison between arable and biomass crops can only be partially indicative of land productivity under divergent systems, turning this natural productivity into a modern, commercially viable form of land management requires a number of substantial challenges to be overcome. These challenges are considered within various sections of the present report, along with the opportunities offered to maintain agricultural production from areas that might otherwise be forced by external forces to cease or substantially curtail current conventional agricultural production.

2.2 Wetland Plants

Oxygen diffuses through water 10,000 times slower compared with air (**Clymo, 1983**). Organic matter requires large quantities of oxygen to break down and decomposition rates are greatly reduced under the anaerobic conditions that typically prevail below the water table. This anaerobic environment is hostile to many plant species, because the root system must be able to survive without oxygen and seeds must be able to germinate and produce viable seedlings.

The key to competitive success of wetland plants is the presence of aerenchyma; these are tissues containing enlarged pore spaces, generally in stems and roots, to enable gas exchange between different parts of a plant (**Evans, 2003**). This allows oxygen to be transported to the root system, enabling root cells to function despite being immersed in an oxygen-free environment. The presence of aerenchyma is therefore characteristic of wetland species and is even used in Danish law to define wetland plants. It is therefore one way of identifying paludiculture crop species, with the exception of lower plants (such as mosses), which have no internal transport system. Some non-wetland species do however have the capacity to develop aerenchyma under waterlogged conditions and may therefore also have paludiculture potential.

The preponderance of relatively narrow leaves with an upright habit amongst wetland plant families, such as the sedges and rushes, reduces heat stress during the hottest part of the day (**Gates, 1980**) within the open landscapes typical of many wetlands. The resulting frequently dense growth of such species results in competitive advantage being conferred more by access to light than to any influence of the water table (**Kotowski et al., 2001; Kotowski and van Diggelen, 2004**). In many cases the leaves or stems also have a degree of stiffening (e.g. *Juncus*, *Carex*, *Cladium*) that enables the leaves to be self-supporting above the water table (**Godwin, 1978**) and renders plant stands more resistant to wind-blow; a desirable trait for a commercial crop species.

Trees also offer a number of potential end-products from paludiculture. While almost all native British trees are intolerant of salt water incursion, most will tolerate freshwater flooding (**Rackham, 1986**) and some will continue to thrive under conditions of almost constant waterlogging. The array of potential plant species for use in paludiculture thus ranges from full-grown trees down to some of the most primitive plant groups.

Products such as common reed (*Phragmites australis*) have a continuous history of use that goes back to the Mesolithic, while others such as sweet grass (*Glyceria fluitans*), were used until just a century ago (**Łuczaj et al., 2012**), and *Sphagnum* bog moss saved potentially hundreds of thousands of wounded allied soldiers from death by gas gangrene during World War 1 (**Ayres, 2013**). Conventional agriculture also has a history of embracing entirely new crops, such as quinoa and echium, as well as rediscovering old forms of farm produce such as oilseed rape ('col-seed' – **Godwin, 1978**). Both 'novel' and 'old' crops may therefore have potential for paludiculture.

2.3 Land Suitability for Paludiculture

A key part of agricultural production is scale. Where a crop can only be grown on a single small patch of ground and nowhere else, there is no commercial or practical incentive to develop the machinery, infrastructure and markets necessary to make the enterprise of growing the crop worthwhile. The larger the area taken up by a crop the greater the incentive and need for associated machinery, infrastructure and markets to be developed. Scale is thus an important consideration for adoption and development of paludiculture. Identification of the UK land stock having the potential to support paludiculture represents a critical step in determining whether it is worth investing time and effort in developing this form of agriculture beyond small trial plots.

The distribution of extant deep peat soils does not define the limits for paludiculture, although it is likely to represent the core area of large-scale paludiculture activities; in part to ensure that major carbon emissions from land use are halted. Where remaining (deep and wasted) peat deposits are discontinuous or fragmented, however, it may be more economical and practical to include areas of mineral soil within the paludiculture system, particularly if the crop species performs well on wet mineral soils (e.g. reed canary grass (*Phalaris arundinacea*) – **Oehmke and Abel, 2016**). In such cases, the wet mineral ground has the potential to accumulate waterlogged carbon as crop residues, forming new carbon stores. Many of these mineral soils also fall within the broad definition of 'wetland' as defined in the IPCC Wetland Supplement as Inland Wetland Mineral Soils (**IPCC, 2014**).

The total area of land in England currently in agricultural use under crops, temporary and permanent grass, excluding permanent rough grazing in the uplands, is 7.9 million ha (**Defra, 2018**). Of this utilised agricultural land, 1.2 million ha lie within floodplains (**Table 2.1**) and therefore have a natural tendency to wetland conditions, as also indicated by the fact that the same total area of land (1.2 million ha) is deemed to be at flood risk directly from rivers or from the combined effects of river and sea. Agricultural Land Classes (ALC) 1-3 occupy 1 million ha of this land, with 187,000 ha regarded as ALC Grade 1, while 851,000 ha is classed as ALC Grade 2 or Grade 3. Of ALC Grades 1-3, 73,500 ha is considered likely to experience a flood-return period of 5-10 years, with the bulk of this (71,300 ha) occurring on Grade 2 and Grade 3 land (**Roca et al., 2011**). Meanwhile **Roca et al. (2011)** identified the financial benefits of a change to agricultural flood alleviation to be £5 million annually, the bulk of which relates to ALC Grade 1. Paludiculture may thus have a part to play in offering an attractive alternative form of economic land use, within this 71,300 ha, if other factors render conventional agriculture increasingly unattractive.

For Wales, **Roca et al. (2011)** calculate that 214,600 ha of land lie within floodplains. They also state that 111,100 ha of this land is within agricultural use (**Table 2.1**), the majority (91,000 ha) being set to grass, while 19,400 ha is devoted to arable farming. The remainder is assigned to horticulture or set-aside. In terms of ALC quality, Grade 1 land occupies only 1,500 ha of agricultural land in the floodplain, suggesting that the majority of arable farming on floodplains is undertaken on Grade 2 land at best (of which there is 8,900 ha), while the remaining 9,000 ha is probably undertaken on Grade 3 land (of which there is 41,900 ha). The remaining Grade 3, 4 and 5 land in the floodplain is presumably largely under grass.

Roca et al. (2011) only consider land lying within floodplains, whereas there is much wet ground – or potential wet ground – that lies beyond the floodplain valley bottoms. Outside floodplains the main cause of waterlogged ground comes from surface-water flooding. This type of flooding is widespread and is considered to threaten more properties than flooding from river and sea (**Bevan, 2018**), but consists of many smaller scattered areas of ground.

Although Flood Risk Maps for surface flooding are now available, the total area uniquely associated with surface flooding was not quantified by **Roca et al. (2011)** and so far remains unpublished, though the information may be available from the Environment Agency. It would seem, however, that whatever the cumulative total area, the contribution of land subject to surface flooding is likely to be considerable but consist of many comparatively small areas set within a patchwork of intervening drier ground, the whole of which would then require a managed water regime.

The surface flooding map thus has the potential to add significantly to the stock of land with potential for adoption of paludiculture. Where such areas are relatively small they may be more suited to the cultivation of low volume-high value paludiculture crops, while also potentially transforming a 'problem area' within a farm holding into productive land.

The Flood Risk Map for surface water flooding, for example, accurately identifies an area currently being used as a test plot for *Sphagnum* farming paludiculture (**Figure 2.2**) and which now has a shallow organo-mineral soil but which within living memory had a substantial depth of peat (**J. Stanley, farmer, pers. comm**.). It does not feature on any of the other map types currently used to identify flood risk, peat soils or wasted peat.

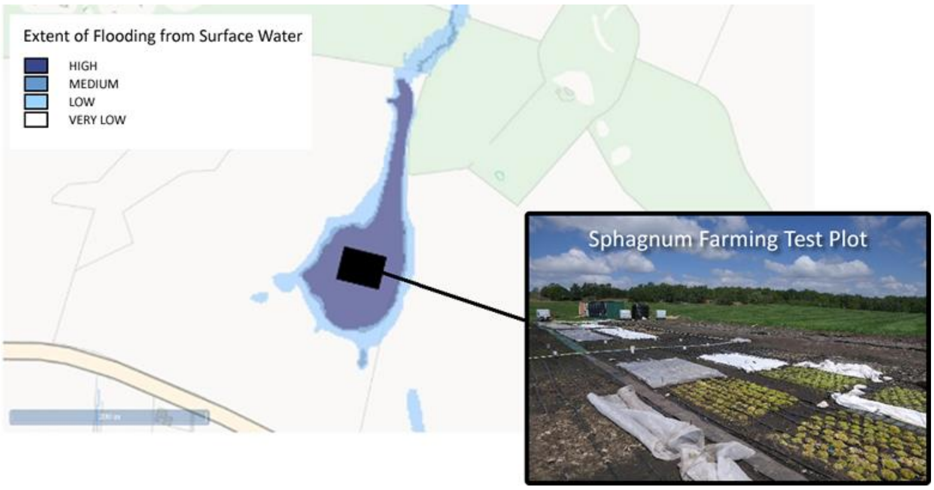


Figure 2.2 *Extract from Environment Agency Flood Risk Map for surface flooding, indicating direct correspondence between high-medium risk of surface-water flooding and the locality of former deep peat (now shallow organo-mineral) soil used as test-bed for Sphagnum farming paludiculture with irrigation from above.*

A combination of sources, including those discussed above, might be used to generate a map of ground having the potential for the adoption of paludiculture, areas which might benefit indirectly from paludiculture, or areas not appropriate for paludiculture:

- maps of peat and peaty soils in the lowlands;
- topographic map indicating areas of low gradient;
- flood risk map – river flooding;
- flood risk map – coastal flooding;
- flood risk map – surface-water flooding;
- maps of Utilised Agricultural Area (UAA);
- maps of Agricultural Land Class (ALC);
- areas of urban or industrial infrastructure;
- areas subject to nature conservation designation.

This approach would broadly mirror the exercise already undertaken for the Mecklenburg-West Pomerania region of Germany (**Schröder and Schroeder, 2016**). The result of such a decision-support exercise, as applied to Cumbria, can be seen in **Appendix III**.

For lowland England and Wales as a whole, the figures can be summarised (**Table 2.**), but a complete assessment of paludiculture potential would require a full analysis of the various datasets listed above. In the interim, it seems that the potential area for which paludiculture may be an option is likely to lie somewhere between the 81,000 ha of grassland on peat and the 1.5 million ha area in agricultural use within floodplains plus some areas of surface-water flooding. These totals are roughly equal to the agricultural areas in England currently under main-crop potatoes and wheat respectively (**Defra, 2019**).

Table 2.1 *Areas of Land (ha) in England and Wales Relevant to Possible Use by Paludiculture.*

Land Category	England	Wales
Utilised agricultural area (UAA) under cropland and grassland (temporary and permanent) excluding upland rough grazing and commons[1,2]	7,944,000	1,359,000
Area of <u>all</u> land at flood risk from river or sea[3]	1,655,400	111,100
Area of <u>all</u> land at flood risk from surface water	unknown	unknown
Area in agricultural use within floodplains[3]	1,224,900	111,100
Agricultural Land Classes 1-3 in floodplains[3]	1,038,100	52,300
ALC Classes 2-3 in floodplains[3]	851,100	50,800
Deep and wasted peat in the lowlands[4+5]	326,936	8,000 – 15,000
Croplands and grassland on deep and wasted peat in the lowlands[4]	256,382	6,679
Deep and wasted peat under agricultural production in the lowlands (England only)[6]	240,000	-
Cropland on deep and wasted peat in the lowlands[4]	182,382	102
Cropland on wasted peat in the lowlands (England only)[4]	132,107	-
Grassland on deep and wasted peat in the lowlands ('intensive grassland' only, in Wales)[4]	74,000	6,577
Cropland on deep peat in the lowlands (England only)[4]	50,594	-
Area of peat <40cm ('peaty soils' and 'peaty pockets'[5]) in the lowlands	unknown	unknown
Land relying on pumped drainage within IDB districts (predominantly England)[3]	635,722	

[1] **Defra, 2018**; [2] **Welsh Government, 2019**; [3] **Roca et al., 2011**; [4] **Evans et al., 2017**; [5] **JNCC, 2011**; [6] **Morris et al., 2010**

3 OPTIONS FOR PALUDICULTURE

This chapter explores potential paludiculture crops and current knowledge of their production under waterlogged conditions. We consider their intended end-use, markets and known barriers for their immediate uptake. **Section 3.1** explores farmed *Sphagnum* as a growing media constituent within the context of the current growing media industry and wider research into peat-free and peat reduced media blends, **Section 3.2** considers the use of tree and non-tree species as bioenergy crops, **Section 3.3** evaluates the potential of foodstuffs to be produced under high water table conditions, and **Section 3.4** considers alternative uses not covered by other sections, including as a raw material for fabrics and for their medicinal use.

3.1 Paludiculture and the Growing Media Industry

Although extracted peat has had many uses, including as a soil improver and as a fuel source, currently extracted peat is primarily used as a raw material for growing media. This section provides some context to the use of peat in horticulture, and then explores the extent to which paludiculture products may be suitable as raw materials for containerised crop production.

3.1.1 Peat Extraction & Consumption in the UK

Over the last 60 years, peat has been developed and refined as a substrate for containerised production and is currently the most widely used raw material by professional growers in the UK. Given concerns around the environmental impacts of peat extraction, there has been a push for alternatives to peat as a substrate (**Bragg, 2018; Barrett et al., 2016**; **Carlile and Coules, 2013; Defra, 2011b, 2013a, 2013b**). The growing media industry itself has also recognised that reliance on a single raw material is also risky in terms of availability and cost (**Defra, 2009a**, **2011a, 2013a, 2013b**). In 2011, Defra introduced a voluntary target for amateur gardeners to phase out the use of peat by 2020 and a final voluntary phase-out target of 2030 for professional growers of fruit, vegetables and ornamentals (**Defra, 2018**). Although there is an aspiration to phase out peat use in horticulture as soon as 2023 **(CCC, 2020)**, **Gunn (2020)** reiterates the wider concerns with phasing out peat, in all its forms for the horticulture sector, and the Government's overarching commitment to the voluntary 2030 phase out scheme.

Progress in replacing or reducing peat content of growing media has, however, been limited because of the lack of viable alternatives within the supply chain. Alternatives to peat are often costlier, as the material has to be processed for use and can be subject to performance issues (e.g. poor crop quality), either alone or in a blend with peat, and may have elevated water and nutrient demand / loss because of increased drainage (**AHDB, 2016**; **Defra, 2011a; Mulholland et al., 2019**). The last reported figures (from 2015) on the proportion of peat in growing media (**AHDB, 2016**) indicated that there has been a gradual decrease in the use of peat across all sectors (56% of volume since 1990), with much of this recent volume reduction being ascribed to changes made by professional growers, with relatively little reduction in peat use by amateur gardeners over the same period.

Peat extracted in England and Scotland has decreased considerably since 1991; originally accounting for 58% (234,150 tonnes) of UK consumption, in 2014 it accounted for only 22% (119,250 tonnes) of consumption. This shortfall however, has been fulfilled by an increase in imports, primarily from Ireland. In GHG emissions terms, this is essentially equivalent to displacing the emissions from food production on peat to other countries, i.e. it produces no net benefit and may even lead to higher emissions linked to transportation. Furthermore, over the same observed period, the volume of peat consumed has risen from 402,774 to 539,213 tonnes; with a peak of 782,804 tonnes in 2003. Although there has been a reduction in the proportion of peat used in growing media, the total volumes consumed have risen as the growing media industry has expanded. The increased use of growing media is due to the vast increase in yield, reduced transmission of root-zone disease and minimal water and nutrient use compared with traditional soil-based production systems.

Demand for peat across Europe (for all end-uses) is also predicted to increase, from 26 to 60 million cubic metres between 2017 and 2050; with demand in Asia predicted to increase from 7 to 80 million cubic metres over the same period. Much of this latter increase is attributed to China, which is expected to increase its demand from 6 to 35 million cubic metres. In total, global demand for peat is predicted to rise from 59 to 244 million cubic metres from 2017 to 2050 (**Blok, 2018**).

As a result of the decrease in UK peat extraction, only 50 people are now employed in peat extraction across Great Britain (**Business Register and Employment Survey, ONS nomis, official labour market statistics**; available from: http://www.nomisweb.co.uk), although 70 people are still employed in Norther Ireland, which is now the largest peat supplier (**Neil Bragg, pers. comm.**). The remoteness of many extraction sites, especially in Northern Ireland, limits the creation of new industries, particularly crop production, which require sufficiently short supply chains to population centres, with access to processing and/or packing facilities. These factors may be less of an issue for fibre crops grown on former extraction sites, including eventual peat alternative raw materials, as they could benefit from existing peat trade routes.

In overall economic terms, peat replacement has had an estimated cost to the horticulture industry in the order of £100 million **(Defra, 2009b**), associated with product development, capital investments in new machinery, and the increased reliance on internationally sourced raw materials (most of which are more expensive than peat). From a consumer perspective, peat-free blends are on average 21% to 44% more expensive compared with standard peat-based mixes. Any poor performance of low-peat or peat-free blends in the horticulture sector also imposes costs on producers. The growers which were expected to be hit the hardest are those in the mushroom, forced bulb and pot plant sectors, as they are more exposed to international competition. The estimated annual cost of phasing out peat in hardy nursery stock (HNS) sector by 2020 could be around £39.5m; which represented 8.4% of the sector's value (**Defra, 2011a)**.

To date, grower perceptions of peat-free media have therefore largely been negative and transition to alternative growing media needs both to be economically competitive, and to be of sufficient quality to gain the confidence of the industry. In efforts to remove barriers associated with widespread use of peat-free and peat reduced blends, a programme of work has been exploring the use of growing media across horticulture sectors. Algorithms have been developed to predict indicator plant species performance based on measured physical properties of media blends (**Mulholland et al., 2019**). As modelling approaches develop, other factors such as nutrition, pH, root-zone crop protection products and interaction with growing media and plant root system therein can be manipulated to continually refine growing media for the horticulture industry. The techniques developed will assist the industries efforts to meet the zero peat use 2030 target for the horticulture sector. The work will also provide tools with which to evaluate and expedite the inclusion of future growing media constituents to ensure a sustainable future for intensive, containerised horticulture plant raising systems.

3.1.2 *Sphagnum* Farming as a Growing Media Constituent

Sphagnum farming aims to cultivate founder material for restoration purposes, or biomass as a raw material. Initially, much of the experience gained in *Sphagnum* farming stems from restoration research. A transition away from purely restoration work to commercial production has required development of research around species selection, crop management practices, processing of harvested material and supply chain analysis (**Gaudig et al., 2017**).

The natural productivity of *Sphagnum* varies widely between species (**Table 3.1**), and not all species can be used in the production of growing media or be grown successfully under a UK climate. The nutrient content of water supplying a Sphagnum farming area may also affect both the growth of the *Sphagnum* and its suitability for inclusion in growing media products. For example, sedge peat was phased out partly for commercial use because of high nutrient content – a growing media is best as a relatively low nutrient / inert product, so that the chemical properties can be augmented to match the needs of target plant species.

Table 3.1 *Productivity of Selected Sphagnum Species in Research Trials*

Sphagnum Species	Mean Biomass (g m^{-2} yr^{-1})	Climate	Location	References
S. cristatum	840	Hyper-oceanic	New Zealand	**[1]** & **[2]**
S. falcatulum	770	Hyper-oceanic	New Zealand	**[1]** & **[2]**
S. subnitens	590	Hyper-oceanic	New Zealand	**[1]** & **[2]**
S. fuscum	800	Humid	Germany	**[3]**
S. magellanicum	790	Humid	Germany	**[3]**
S. rubellum	960	Humid	Germany	**[3]**
S. palustre	575	Warm temperate, Humid	Georgia	**[4]**

[1] Stokes et al., 1999*;* ***[2] Gunnarsson, 2005****;* ***[3] Overbeck & Happach, 1957****;* ***[4] Krebs et al., 2016****.*

Numerous tests have been undertaken on the *Sphagnum* crop, and on living *Sphagnum* more generally, to determine the suitability of fresh, farmed *Sphagnum* as a growing medium (**Gaudig et al., 2008; Emmel, 2008; Kumar, 2017; Kämäräinen et al., 2018; Gaudig et al., 2018**). These experiments have suggested that selected species have some potential as a medium, but with variable results attributed to the proportion of different *Sphagnum* species used in the mix, crop, and crop development stage; see appendix of **Gaudig et al.** (**2017**) for an overview of some of the experiments being conducted. Additional research from Canada showed that adding *Sphagnum* fibre to peat substrate increased water retention and hydraulic conductivity of the growing media, but simultaneously, either reduced or had no impact on air-filled porosity (**Jobin et al., 2014**). Continued testing of different *Sphagnum* varieties, mixes and applications will be needed to establish the overall suitability of *Sphagnum* as a growing medium.

3.1.2.1 Sphagnum Farming in Germany

The most extensive growing trials of *Sphagnum* as a commercial crop to date have taken place in Germany (**Gaudig et al., 2014; Gaudig et al., 2017; Gaudig et al., 2018**). These trials have variously used bare peat surfaces left after peat extraction by commercial peat milling, former bog grassland, and floating rafts. For the land-based plots, the water table was raised close to the ground surface and maintained close to the surface through the use of sub-surface drainage pipes and irrigation ditches. Raw *Sphagnum* material was obtained by wild harvesting in the Netherlands from conservation sites where *Sphagnum* was overgrowing valued fen habitat, and then subsequently by growing up from *Sphagnum* taken from the established plots (**Gaudig et al., 2018**).

A major issue recorded during these growing trials was that the wild-harvested *Sphagnum* also brought with it a large number of other wetland plants, with soft rush (*Juncus effusus*) and the moss (*Polytrichum commune*) proving to be particularly vigorous contaminants. Monthly mowing kept the soft rush in check though the mown material had to be left on the moss surface, while the *P. commune* could only be removed after harvesting the moss crop (**Kumar, 2017**). After five years the complete carpet of *Sphagnum* appeared to be preventing continued growth of soft rush as it no longer formed part of the 'weed' plant cover. Its place was taken by common cotton grass (*Eriophorum angustifolium*), which is a constant species of bog vegetation and thus also of peat. Consequently, the presence of *E. angustifolium* within the *Sphagnum* crop was not regarded as an issue (**Gaudig et al., 2017**).

Growth rates observed at the two German sites saw close to 100% *Sphagnum* cover and a carpet thickness of 5-9 cm achieved within 1.5 years at one site where the water table was maintained within 10 cm of the ground surface and gaps in cover were filled at the end of the first year. While the second site, with a more variable water table and no gap-filling, achieved more than 90% cover within 3.5 years. Productivity ranged from 3.7 – 6.9 t dry matter (DM) ha^{-1} yr^{-1}, and after nine years the total dry biomass accumulation had reached 19.5 tonnes per hectare. Given the rate of *Sphagnum* growth, the German trials suggest feasibility of harvesting once every 3–5 years (**Gaudig et al., 2014; Krebs et al., 2018**). *Sphagnum* farming on floating mats on flooded cut over bogs and lignite mine lakes, was found to have costs of establishment in the region of four to five times that of establishment on former bog grassland, or cut-over bogs (**Wichmann et al., 2018**).

3.1.2.2 Sphagnum Farming in the UK

Since 2018 a set of *Sphagnum* farming trials, initially funded by Innovate UK, have been running on two experimental sites in Leicestershire (a former agricultural site, currently with organo-mineral soil) and one to the west of Greater Manchester (a former extraction site on deep peat). These trials differ from the German experiments in a number of important ways. Firstly, the 'founder' material was not obtained by wild harvest but through a process of micropropagation, meaning that the founder material was free from contaminating weeds. Secondly, rather than raising the water table close to the ground surface, irrigation from above has been employed to maintain the *Sphagnum* crop in a reasonably saturated but not inundated condition. Thirdly, a varied set of cover 'mulches' typically used in conventional agriculture have been laid over the *Sphagnum* in order to maintain high humidity. Founder material has been applied as either small 'plugs', or as a gel which can be sprayed onto the prepared ground.

Under this system, *Sphagnum* growth on both sites has been as good as, or better than, the observed growth rates in German tests. For the gel, mean coverage of the plots after six months ranged from 33% to 53%, while for the plugs, growth rates gave mean coverage for the same time period ranging from 44% to 62%. After 10 months, the best performing treatments had achieved almost 100% cover on both sites. By way of comparison, the best performing plots in Germany achieved 30% cover after 6 months and a mean cover of 40% after 12 months (**Gaudig et al., 2017**). Yields obtained exceeded 500 m^3 ha^{-1} yr^{-1} in the first year, equivalent to 5.5 t ha^{-1} yr^{-1} dry matter. This lies towards the upper end of the range noted earlier for productivities observed on the German experimental *Sphagnum* farming sites (3 – 6 t ha^{-1} yr^{-1}: **Gaudig et al., 2018**). Productivity levels are expected to be higher once complete cover is achieved, and as the *Sphagnum* layer becomes increasingly self-sustaining.

3.1.2.3 Practical Challenges for Sphagnum Farming

Within the context of predicted global demand, or even simply in relation to the current volumes of growing media used in the UK, the volumes of farmed *Sphagnum* harvested so far from trials in Germany are little more than small-scale experimental quantities, while the first trial-plot harvests in the UK will not be obtained until 2021. Although upscaling the *Sphagnum* farming process to a farming/industrial scale is a priority, several key challenges must first be overcome:

Sourcing founder material: Plant material is required for the development of a *Sphagnum* farming operation, and currently only vegetative micropropagation of gathered *Sphagnum* shoots (from wild populations) is considered a viable option for the supply of juvenile plants at a commercial scale. Although availability of *Sphagnum* spores are high (one capsule can hold 18,500 to 240,000 spores), the factors which induce sporulation and germination are not yet properly understood (**Gaudig et al., 2017**). Moors for the Future are, for example, planting 1.7 million plugs of micropropagated *Sphagnum,* obtained from local stock, during the course of 2020, as part of a restoration programme for the blanket bogs of the Peak District.

Irrigation: Land levelling, irrigation and water management infrastructure are likely to be needed to support *Sphagnum* farming, particularly during the establishment phase. **Pouliot et al. (2015)** demonstrated a potential link between irrigation and yield beyond the establishment phase. There are also challenges to establishing a wetland crop within a landscape dominated by conventional agriculture. While surface irrigation is noted as a means of avoiding the need to raise local water tables, the infrastructure and control systems required to establish and maintain such irrigation require investigation and development. Surface/overhead irrigation may be an established practice within conventional agriculture, but the particular control systems required for successful *Sphagnum* farming in any given location remain to be established. Precipitation inputs will need to be balanced appropriately against water losses, and such losses will be significantly influenced by many factors including, for example, the use (or non-use) of particular cover mulches. Where irrigation is provided by raising the local water table close to the ground surface, this will require control systems that prevent surface flooding (**Gaudig et al., 2018**). Although *Sphagnum* species are tolerant of flooding the main species so far used in *Sphagnum* farming trials are naturally-terrestrial rather than aquatic species. Long periods under water will tend to reduce or even halt growth of these species.

Water quality: Most species of Sphagnum used so far in trials are intolerant of high calcium levels (**Clymo, 1983**), while high levels of nitrogen and/or phosphorus will tend to give weed species a competitive advantage. Water supplies may therefore require treatment through a constructed wetland system in order to reduce nutrient levels sufficiently to enable the *Sphagnum* crop to out-compete weed species. On the other hand, despite having low nutrient demands there may be circumstances where nutrient demands of the *Sphagnum* are not met by the underlying peat, atmospheric deposition, or irrigation water, in which case additional fertilisation of the site could be required as harvested biomass mines the existing reserves (**Gaudig et al., 2017**). Further work is necessary to establish the balance of nutrient levels in irrigation water required to encourage maximum *Sphagnum* growth without favouring weed species.

Undesirable side-effects: The effects of production methods adopted during *Sphagnum* farming have highlighted additional barriers, including the existence of secondary metabolites (which may hamper root growth and lower the yield of the cultivated plant), which may cause undesired nitrogen immobilisation in the growing medium. The biological and physical stability of *Sphagnum* in mixes also requires further investigation, as do the processing methods used, which may produce hydrophobicity (water repellency; **Gaudig et al., 2017**).

Mechanisation of planting: Hand-planting of founder *Sphagnum* material in the form of 'plugs' many not be a feasible option for some farming enterprises. Larger cropped areas will require mechanised planting systems. Rice-planting machinery has been developed to work in flooded rice fields, but in their normal configuration they churn up the soil to a considerable degree. While the planting arms themselves may be adaptable to planting *Sphagnum*, the tracks of any such vehicle would need major re-design in order to avoid significant disturbance to the ground surface – particularly where that surface is peat. The mechanics of spreading a gel may be simpler than that of planting plugs, but any such machine would still need to address the same issues of tracking across the peat surface.

Weed control: Weed growth has proved to be a significant issue for the German test sites, where wild-harvested founder material was used (**Gaudig et al., 2017**). Weeds have also required considerable maintenance effort on the UK test sites, most notably on the organo-mineral site where the existing seed bank was not sufficiently recognised and addressed during the set-up phase. Strict certification standards exist in Germany and the Netherlands regarding acceptable levels of viable seeds and other plant fragments within growing media (**Kumar, 2017**). Such contaminating material must be rendered non-viable during processing. Where plant material is used as mulch, the GHG emissions from its use must also be assessed. Appropriate methods of weed control, including treatment of any existing seed bank, methods for prevention or removal of invading weeds, and subsequent treatment and screening of the crop, all require further R&D to devise methods that are capable of operating on a farm scale.

Mechanised harvesting of crop: Harvesting of the crop by mechanical means without causing major physical damage to the cropping surface undoubtedly represents one of the most challenging and unresolved aspects of *Sphagnum* paludiculture. Any form of tracked vehicle, no matter how low the ground pressure, will result in mechanical disruption and compression of the crop surface. Harvesting in Germany so far has relied on the use of a specially designed bucket on an excavator running along a causeway adjacent to the trial plot (**Gaudig et al., 2018**). This involves the sacrifice of alternating strips of ground from which the excavator operates, thereby halving the potential cropping area. It seems likely that some degree of land sacrifice will be required for any methods of mechanised harvesting, but significant R&D investment in agricultural engineering is required to devise methods that minimise the sacrificial area and avoid disruption of the peat soil.

Processing of crop: Various aspects of postharvest processing require development to establish standardised procedures. Drying of the crop to particular moisture contents reduces the weight of material that must be moved or transported, but although certain methods have been tested, including stacking followed by air drying, use of glasshouses, or drying using warm air (**Kumar, 2017; Gaudig et al., 2018**), no single method has yet been established as the optimal, most sustainable approach. Care must also be taken to avoid drying the material to the point (<20% moisture content) where it becomes hydrophobic (**Gaudig et al., 2018**). Moisture content can also affect machine handling. Some processing machinery is unable to process material above certain moisture contents, while very low moisture contents can mean that particles readily become airborne and therefore cannot easily be conveyed through the processing machinery.

3.1.2.4 Economic Considerations

The highest market prices for cultivated *Sphagnum* biomass can be achieved through the supply to the reptile and floristry sectors, achieving £500 and £200-£250 per cubic metre, respectively (**Neal Wright, pers. comm.**). It is expected that an immediate market exists, in the range of 50,000 to

100,000 m^3 per year in the UK, which would directly substitute for the wild harvest of *Sphagnum* (currently practiced in Chile, USA, Canada and New Zealand), which is gathered to supply the floristry sector. However, the growing media sector poses the greatest market potential, even though the prices achieved by volume are much reduced. If determined to be a suitable raw material, *Sphagnum* cultivated for use in growing media is estimated to be marketed at between £25-50 per cubic metre (**Neal Wright, pers. comm.**); this value will be influenced by technical issues such as outturn and performance, either alone or in combination with other materials for containerised production systems. Current demand for peat within the UK growing media industry amounts to 2.5 million m^3.

Global average dry *Sphagnum* biomass production is in the range of 260 g m^{-2} yr^{-1}, with 1,450 g m^{-2} yr^{-1} being the highest yields achieved so far (**Gaudig et al., 2017**). It is expected that 500 g m^{-2} yr^{-1} (5 t ha^{-1} yr^{-1}) can be achieved in the UK (**Neal Wright, pers. comm.**), which would imply a sale price of between £833 and £1,667 ha^{-1} yr^{-1}; assuming a bulk density of 0.15 g cm^{-3} and a sale price of between £25-50 per cubic metre. In contrast, current prices for peat are £20-22 per cubic metre; currently with high availability and very low costs of production. If farmed *Sphagnum* is to become a raw material within a substrate blend or used as the sole ingredient, it will need to be affordable, available in sufficient volumes, managed to consistently provide the crop with a good growing environment, and adaptable to existing grower mechanised systems. In the first instance key characteristics that define the ability to work, either as a single or blended raw material for containerised plant production use, will need to be measured and evaluated, and crop specific blends may be required. Where a single raw material has replaced peat, as is the case for coir in soft fruit production, this has been as a result of the ability of coir to resist shrinkage and slumping in over-wintered crops, combined with its favourable inherent properties as a growing medium. At this stage it is unclear whether *Sphagnum* fibre will have similar, or equally valuable, properties which would bolster its use as a growing media constituent in the future.

Current production costs of *Sphagnum* biomass are approximately £50 m^{-3} based on production costs of £25,000 ha^{-1} yr^{-1} and an achieved yield of 500 m^3 ha^{-1} yr^{-1}. Assuming a value of £25-£50 m^{-3} for the growing media market, this would generate an income of between £12,500-£25,000 ha^{-1} yr^{-1}, resulting in anything from a loss of £12,500 to a break-even point. Production costs are partly associated with start-up costs, including founder material, weed control, irrigation systems and cover-materials for the crop. Consequently, costs would be expected to fall in subsequent rotations, as the scale of operations increases. Clearly, however, for this type of paludiculture to be economically viable, production costs need to be reduced and/or productivity would need to increase.

Given the multifaceted potential benefits of *Sphagnum* farming (preservation of peatland soils, sustainable after-use of peat extraction sites, replacement of a fossil resource with a renewable resource, and the understood ecosystem services ascribed to restoration), it is possible that payments for carbon benefits could help offset some of the costs associated with *Sphagnum* production. It remains unclear, even with e.g. carbon payments, whether this intervention would allow farmed *Sphagnum* to be commercially viable, particularly when compared to currently commercially available main growing media raw materials (bark, wood fibre, coir and green compost) used by the UK growing media industry.

If the identified technological and economic barriers could be overcome, and assuming farmed *Sphagnum* could match the properties of horticulture peat, then (based on the observed productivity on the UK pilot plots and a rotational harvesting interval of 3 – 5 years, and a current UK demand for horticulture peat of around 2.52 million m^3 yr^{-1}), this would require a minimum of 5,000 ha in the UK to be devoted to *Sphagnum* farming, though the harvesting cycle may potentially require as much as 25,000 ha. This would represent at most some 10% of deep and wasted peat under cropping and grassland in England and Wales, but potentially could be as little as 6% of grassland only. However, if the CCC 25% recommended targets for reinstatement of lowland peatlands are enacted **(Table 1.1; CCC, 2020)**, large scale land use change will be required. This will need to evaluate the opportunity costs of *Sphagnum* farming and the near-term gains of alternative (low-volume/high-value) uses for *Sphagnum* (see **Section 3.4.1**) or other paludiculture crops, should rewetting be deemed the preferred course of action.

3.2 Paludiculture for Bioenergy

Bioenergy, the use of cultivated biomass for the purpose of energy production, presents an opportunity to mitigate the degradation of peat soils whilst maintaining productive output through paludiculture systems. The two major end uses that are currently available include: **(1) combustion of dried biomass**, and **(2) anaerobic digestion of green material**. We discuss the characteristics of key bioenergy crop species, including productivity and biomass quality. Energy conversion values for potential paludiculture bioenergy crops are also discussed.

3.2.1 Potential Bioenergy Crops from Paludiculture

We used the '*UK Paludiculture Live List*' as the basis for species selection (a UK list of paludiculture plants, adapted from the 'Database of Potential Paludiculture Plants' and refined for UK agroclimatic zones by J. Clough[1], R. Lindsay[1] and S. Abel[2]). We filtered plant species according to their main uses, in which 'energy' must be clearly recognised; this resulted in 24 plant species, including reed grass (3 species), cattail (3 species), sedge (7 species), aquatic herb (1 species), deciduous trees (6 species) and coniferous trees (4 species). All these species are natives or considered neophytes (introduced species now naturalised), and all are perennial (see **Appendix I**). Neophytes were only considered on the understanding that they are not invasive to the UK.

Additionally, *Miscanthus* grass, although not identified as an energy crop in the '*UK Paludiculture Live List*', is a prominent bioenergy crop in the UK, with good potential for biofuel production (**van Hilst et al., 2010**). There is evidence that *Miscanthus* grass can be productive in a paludiculture system (**Silvestri et al., 2017**).

Following species selection, a review was conducted, targeting both peer-reviewed and grey literature. We used selected taxa within general search terms, and the information/data in each relevant publication was summarised to highlight the species and types of data presented (see **Appendix II**).

3.2.1.1 Suitability of Tree Species within a Paludiculture Context

Flooding tolerance varies greatly between tree species. The physiology and growth of sensitive species is negatively affected by the low oxygen conditions under long-term inundation, or when flooded during the growing season. Nevertheless, tree species grown in rewetted organic soils are capable of producing crops for bioenergy (e.g. **Silvestri et al., 2017**); woody biomass has become a key raw material for electricity generation in the UK. Tree species such as Alder, Poplar and Willow

[1] University of East London
[2] University of Greifswald

are considered to be flood tolerant, benefiting from morphological adaptations to flooding such as lenticels and aerenchyma tissues, which provide a pathway for diffusion of oxygen to root tissues (see **Lukac et al., 2011** for a review). There are, however, a number of other practical and transitional issues that indicate that the use of tree species in UK paludiculture systems is highly challenging:

- Tree species that can be grown as coniferous short-rotation forestry (*Picea abies*, *Pinus* spp.) are considered unsuitable for deployment in lowland paludiculture because they have limited ability for re-growth after cutting (**Silvestri et al., 2017**). Additionally, there are currently regulations preventing planting on deep peat soils.

- It is more typical that the deciduous Alder (*Alnus glutinosa*), Ash (*Fraxinus excelsior*), Birch (*Betula pubescens*), Poplar (*Populus* spp.), and particularly Willow (*Salix* spp.), are coppiced commercially. For short-rotation coppice (SRC), however, heavy machinery is required for harvesting and this is not suitable for peat soils, without significant development and modification of harvesting machinery. Remaining stumps from SRC at the end of the tree's economic lifecycle also present a potential problem for the type of harvesting machinery required in these systems (e.g. puncturing soft tyres, damaging tracks).

- The coppicing for Alder, Ash, and Birch tends to take place every 8-12 years, and often with longer rotation lengths, whereas Poplar and Willow are typically coppiced every 2-5 years, depending on soil and environmental conditions. Comparing biomass production of SRC species is complicated by the fact that annual yields are affected by the age of the stand and the number of years since the previous harvest. As for plantations on mineral soils, this has implications for managing a reliable supply of biomass for a bioenergy supply chain.

While the use of tree species in lowland peats is not considered a practical alternative to conventional cropping, there are a number of publications that have covered the potential for Alder as a paludiculture crop in other cases (e.g. **Wichtmann & Joosten, 2007**). As an N-fixing species, further work on potential N_2O fluxes in Alder paludiculture is needed. Recent research has also suggested that trees may also have the potential to transport methane production at depth to the atmosphere, thus by-passing zones of oxidation in surface layers (**Barba et al., 2019**).

3.2.1.2 Potential of Non-Tree Species in a Paludiculture Context

The review [as guided by the '*UK Paludiculture Live List'*] identified *Phragmites australis* and *Typha latifolia* as showing the greatest potential for bioenergy use*. Phalaris arundinacea*, *Glyceria maxima* and *Carex* sedges also have some potential, however our focus will be on *P. australis* and *T. latifolia,* as these two species are also amongst the four main crops considered in the Dream Fund's Waterworks Project. *Miscanthus x giganteus*, a typical UK bioenergy crop is also evaluated, as a useful comparison, given that it is not normally grown in paludiculture systems.

While sedge species have potential for bioenergy use (e.g. **Timmerman, 2003**) we only compare these informally to the selected species. The aquatic herb (*Hydrilla verticillata*) is not assessed further, given its typical deep water habitat and invasive nature; although several publications have covered these (**Evans & Wilkie, 2010**; **Jain & Kalamdhad, 2018**).

3.2.2 Characteristics of Key Crops

This section provides a comparative assessment of the selected species (*P. australis*, *T. latifolia*, *Miscanthus x giganteus*) focusing on characteristics important to bioenergy production, including crop productivity, biomass quality and key issues. These are summarised in **Table 3.2,** with more detailed discussion in the following sub-sections.

Table 3.2 *Ranges of potential biomass production and higher heating value, and some key issues for the potential paludiculture crops.*

	P. australis	*T. latifolia*	*Miscanthus*
Potential biomass production ($t\ ha^{-1}\ yr^{-1}$)	3.72–12.60	3.58–22.10	12.0–24.7
Higher Heating Value ($MJ\ kg^{-1}$)	16.9–17.7	17	18.0–18.8
Potential issues	Unknown impacts of management on methane fluxes. Biogas production may be economically unviable. Seedling growth in peat may be stunted.	Unknown impacts of management on methane fluxes. Biogas production may be economically unviable.	Establishing rhizomes may be sensitive to anoxic conditions. Growth suited to organo-mineral and mineral soils.

3.2.2.1 Phragmites australis (Common Reed)

P. australis is a cosmopolitan graminoid species which can grow in a range of conditions from damp ground to deep standing water (>1 m) and has an extensive system of stout rhizomes. It is common throughout the UK, and in natural conditions such as wetlands it can form extensive dense reedbeds as a dominant species (**Packer et al., 2017**). While it prefers a pH greater than 4.5, it can grow in more acidic conditions (**Packer et al., 2017**).

There appears to be a large degree of variability in biomass production in *P. australis* stands, found in both natural and cultivated environments, with reported yields as high as 40-50 t ha^{-1} yr^{-1} shown to be achievable with suitable climate and fertiliser additions (see **Table 1** in **Kuhlman et al., 2013**). However, under climatic conditions similar to the UK and without fertiliser additions, annual harvested biomass for *P. australis* generally ranges between 3.72 and 12.60 t ha^{-1} yr^{-1} (**Table 3.3**). **Kuhlman et al. (2013)** estimated that *P. australis* could yield 15 t ha^{-1} y^{-1} without adding fertiliser. Higher yields (25 t ha^{-1} y^{-1}) could be achieved when *P. australis* is fed with nutrient-rich water, a prediction supported by the mesocosm study by **Ren et al. (2019)**. In the UK, similar biomass yields have been reported from reed habitat at RSPB Ham Wall Nature Reserve, Somerset, with 6.20, 7.00 and 8.76 t ha^{-1} yr^{-1} being recorded in 1-year-old, 3-year-old and 15-year-old reedbeds, respectively (**Table 3.3**; see **Mills, 2016** for details). Indeed, cutting has been shown to increase the aboveground biomass production in *P. australis* (**Graneli, 1989**; **Ostendorp, 1999**) and regular winter harvesting of reed results in strong regrowth in stands and also reduces pests and diseases (**Köbbing, Thevs &**

Zerbe, 2013). **Philips et al. (2017)** state that there are 5000 hectares of reed habitat in the UK which require active management through annual cutting and/or burning.

It is noted that *Phalaris arundinacea* (Reed canary-grass) and *Glyceria maxima* (Reed sweet-grass) have a similar range of biomass production (see **Table 2** in **Wichtmann & Schafer, 2007**), as do *Carex* sedges (**Timmerman, 2013**). In a study in the Czech Republic, *P. arundinacea* harvested biomass was monitored over 13 years and ranged between 3.4 and 11.3 t ha^{-1} yr^{-1} at one site and between 3.4 and 11.5 t ha^{-1} yr^{-1} at a second site (**Strašil, 2012**). **Abel et al. (2013)** report productivity ranges of between 2.0-15.0 t ha^{-1} yr^{-1} and 4.0-14.9 t ha^{-1} yr^{-1} for *P. arundinacea* and *G. maxima*, respectively.

Table 3.3 *Examples of literature-derived data highlighting the range of harvested biomass values for Phragmites australis (common reed) under natural conditions.*

Harvested biomass (t ha^{-1} y^{-1})	Location	Source
4.0–12.6	South Finland	Table 7 in Mills (2016) [Komulainen, Simi, Hagelberg, Ikonen & Lyytinen 2008]
5.0	South Sweden	Table 1 in Köbbing, Thevs & Zerbe (2013)
7.0	Austria	Table 1 in Köbbing, Thevs & Zerbe (2013)
6.0–9.3	Estonia	Table 7 in Mills (2016) [Kask, 2007]
7.2	Latvia	Table 1 in Köbbing, Thevs & Zerbe (2013)
5.0	Romania	Table 1 in Köbbing, Thevs & Zerbe (2013)
5.0	Ukraine	Table 1 in Köbbing, Thevs & Zerbe (2013)
11.5	North West Italy	Silvestri et al. (2017)
7.3–11.7	Germany	Table 7 in Mills (2016) [W. Wichtmann pers. comm.]
3.72–7.02	Germany	Günther et al (2015)
6.2–8.76	UK	Table 6 in Mills (2016)

There are also significant differences in within-species productivity, with genotypic variation in biomass yields. In a mesocosm experiment, **Ren et al. (2019)** found genotypes of *P. australis* from the Netherlands and Romania to have greater aboveground yields than those from Denmark and Italy. This suggests that the source of genetic material/plants may be an important factor to consider in optimising biomass production and/or to constrain estimates of the GHG mitigation potential through bioenergy use. While there is much variability in biomass production, it is noted in **Wichtmann and Joosten (2007)** that a harvest of 15 t ha^{-1} can be sustained in combination with continuing peat accumulation under *P. australis*.

P. australis biomass can be used as an energy source for combustion and for biogas production (**Kuhlman et al., 2013**; **Köbbing et al., 2013**). Combustion of harvested biomass is affected by the moisture content, mineral concentrations and ash content of the plant material, which are in turn affected by the timing of the harvest. To maximise the calorific value of plant material for bioenergy production, the biomass must be harvested at, or dried down to less than, 20% moisture content (**Mills, 2016**). Moisture content of *P. australis* biomass decreases through the season and typically reaches 15-20% moisture content by late winter, therefore biomass harvested at this stage is most suitable for direct bioenergy generation by combustion. **Wichmann (2017)** noted that winter-harvested reed is suited to combustion in furnaces designed for straw or *Miscanthus*, and that this provides an alternative pathway for reed biomass that does not meet the quality requirement for use in thatching (e.g. heterogeneous or older stands).

Biomass mineral concentrations impact combustion efficiency. Winter harvesting also means that ash content of biomass is lower, which is beneficial to combustion efficiency; and lower content of other nutrients reduces degradation of equipment (**Köbbing et al., 2013**). **Wichtmann and Joosten (2007)** stated a 5.12% ash content dry weight for *P. australis*. Variability in energy values for *P. australis* biomass have been cited. *P. australis* with 10% water content was stated to produce average calorific or heating value of 14-15 MJ kg^{-1} (**Kobbling et al., 2013**; **Wichtmann et al., 2014**), **Wichtmann and Joosten (2007)** presented a minimum heating value of 16.9 MJ kg^{-1}, and **Kuhlman et al. (2013)** use a heating value of 17.7 MJ kg^{-1} to estimate energy yields of reeds in the Netherlands.

In the study of **Silvestri et al. (2017)**, the multi-adaptive decision framework found that perennial Reed Grasses were generally more suitable than other species when considering biogas conversion, with *P. australis* having the highest overall suitability value. Green biomass is required for biogas production through digestion and so harvesting must take place in summer or early autumn for this bioenergy pathway. **Akulain (2013)** showed that higher amounts of biogas were produced by anaerobic digestion of reed material harvested in October (107.9 l kg^{-1}) compared to August (60.6 l kg^{-1}). **Dragoni et al. (2017)** looked at timing of harvest of green material of *P. australis* and the effect on biomass yield and methane yields, and also showed that harvest in September produced greatest biomethane yield; this was largely related to quantity of biomass rather than quality. Despite *P. australis* having the highest suitability of crops for biogas production in **Silvestri et al. (2017)**, accounting for the costs associated with harvesting and transport, it was shown to render 'chaff for biogas production' economically unviable without subsidy (**Wichmann, 2017**).

The aerenchymatous tissues in *P. australis* (and also *T. latifolia*), which allow oxygen to be carried to the roots, also enable the transfer of CH_4 produced in peat to the atmosphere (**Whiting & Chaton, 1996**; **Afreen et al., 2007**). If CH_4 emissions are sufficiently high, and in particular if cultivation requires inundation, this could reduce or negate any climate mitigation benefits of reduced CO_2 emission. Harvesting is also likely to affect CH_4 emissions (**Günther et al., 2015**). There are conflicting results with regards to the cutting response of aerenchymatous plants on CH_4 emissions, some studies show that cutting can decrease CH_4 emissions (**Duan et al., 2006**), whereas, other studies suggest there is only an effect when plants are harvested below the water surface (**Ding et al., 2004**; **Juutinen et al., 2004**). **Günther et al. (2015)** found that cutting did not directly affect CH_4 emissions and that variability in CH_4 emissions was dependent on weather conditions.

Seedling growth of *P. australis* may be stunted when grown on a peat substrate (**Spence, 1964**; **Haslam, 1972**, both cited in **Packer et al., 2017**). It is not known to what extent this effect would translate with establishment using rhizome material (**Table 3.2**).

3.2.2.2 Typha latifolia (Cattail/Bulrush/Reedmace)

T. latifolia is a generally cosmopolitan species that can grow in a variety of climatic conditions, and has rhizomatous growth. It is common in shallower wetlands, and by streams, ponds and ditches. *T. latifolia* has been recorded throughout the UK. It is commonly employed as a tool for catchment nutrient management (**Grosshans and Grieger, 2013**) but its high productivity has also attracted interest because of its potential use as a feedstock for bioenergy. It is suggested that *T. latifolia* is most suited to conditions where water levels are >30 cm above the ground surface in winter and close to the surface in summer (**Timmerman, 2003**) and that optimum water levels for *Typha* reedbeds are 20 to 150 cm above the surface (**Wichtmann & Joosten, 2007**).

As with *P. australis*, the data in the literature highlight large variability in biomass production; in natural stands, annual harvested biomass for *T. latifolia* can range between 3.58 t ha^{-1} and 22.10 t ha^{-1} (**Table 3.4**). Much higher yields (equivalent to 44.5 t ha^{-1}) have been demonstrated under experimental conditions with fertilisation (**Ren et al., 2019**). In fact, **Ren et al. (2019)** found that *T. latifolia* produced the greatest quantity of aboveground biomass of all wetland species assessed in a mesocosm experiment. The same characteristics of biomass quality as discussed above for *P. australis* are relevant for *T. latifolia*.

Table 3.4 *Examples of literature-derived data highlighting the range of harvested biomass values for Typha latifolia (Cattail/Bulrush/Common reedmace)*

Harvested biomass ($t\ ha^{-1}\ y^{-1}$)	Location	Source
4.7–8.2[1]	Minnesota, US	Table 1 in Dubbe et al. (1988)
4.3–14.8	North Central US	Table 1 in Dubbe et al. (1988)
4.8–22.1	Germany	Table 7 in Mills (2016) [Timmerman, 2013]
4.02–6.64	Germany	Table 1 in Günther et al (2015)
3.58	Italy	Table 3 in Giannini et al. (2019)
7.0–44.5[2]	Denmark	Figure 3 in Ren et al. (2019)

[1] Cultivated stands; [2] Range from mesocosm experiment with no fertilisation to 300 kg N ha^{-1}.

Heating values for *T. latifolia* also appear similar to those recorded for *P. australis*, with literature reporting values around 17 MJ kg^{-1} (**Gravalos et al., 2016**). In Manitoba, **Grosshans (2014)** found values of 17 MJ kg^{-1} to 20 MJ kg^{-1} when harvested material from natural stands of *Typha* was compressed into densified fuel products, resulting in calorific values comparable to commercial wood pellets. Harvesting in the autumn meant that the harvested material was already comparatively dry, meaning that further drying down to 6 to 15 % DM was more easily achieved. As an additional benefit, the living stands of *Typha* had captured 30 to 60 kg ha^{-1} $year^{-1}$ of phosphorus, 88% of which was then recovered within the ash when the *Typha* was burnt as a biofuel.

Rebaque et al. (2017), in contrast, examined the potential for production of bioethanol from harvested *Typha* rather than as a directly combustible feedstock. They found that the plant has a high cellulose content compared with other members of the Poaceae Family, and that it also possessed a range of other features which rendered the plant particularly amenable to lignocellulose saccharification. **Rebaque et al. (2017)** therefore concluded that *Typha* had significant promise as a potential source of bioethanol production.

3.2.2.3 Miscanthus x giganteus

Miscanthus x giganteus is a hybrid rhizomatous C4 perennial grass, growing to heights of up to 4 m. It is harvested annually and can grow in a wide range of environmental conditions. It is not a wetland species, although it has the capacity to develop aerenchyma under waterlogged conditions. While it is not clear how *Miscanthus* would respond to paludicultural management in the establishment phase, it is expected to tolerate moderately low-oxygen conditions. **John Clifton-Brown** (***pers. comm.***) noted that the establishment of *Miscanthus* is difficult, expensive and, in establishing a crop, it is known that plugs and rhizomes are sensitive to waterlogging. There is, however, a misconception that this crop cannot withstand flooding and inundation. In fact, mature *Miscanthus* plants have little problem with surviving overwinter or summer flooding for up to 2 months (**John Clifton-Brown, *pers. comm.***). **Mann et al. (2013)** showed that rhizomes were all viable after being kept in experimental flood conditions and subsequent shoot biomass was similar to control conditions. Work at a *Miscanthus* plantation on mineral soils in Lincolnshire, following the 2012 floods suggests that overall growth slowed but final yields were economically viable (**Niall McNamara, *pers. comm.***).

Under paludiculture conditions, hand planting and harvesting might work for *Miscanthus* over small-scale areas with appropriate attention to agronomy (**John Clifton-Brown, *pers. comm.***). However, it is also possible that the same, or similar machines to those used to harvest Perennial Reed Grass in paludiculture systems could be used to harvest *Miscanthus* (**Rebecca Rowe, *pers. comm.***). Further trials and development of machinery are likely to be required.

Miscanthus biomass yield is affected by establishment and agronomic factors. In the UK, average annual biomass produced by *Miscanthus* is around 12 t ha^{-1} under typical growing conditions (**Hastings et al., 2017**). The study of different paludiculture crops in North-East Italy by **Silvestri et al. (2017)** included *Miscanthus* and demonstrated an average biomass production of 24.7 t ha^{-1} over two seasons. This represented a 14% reduction in *Miscanthus* yield under paludiculture conditions, compared with a controlled drained cultivation. Although the location of the paludiculture system in **Silvestri et al. (2017)** is representative of a warmer climate than typical UK conditions, it suggests that productivity of *Miscanthus* in a paludiculture system is likely to be acceptable if the same reduction was to be found in a temperate climate (e.g. an expected biomass production of around 10 t ha^{-1}, which is greater than that achieved from cultivation of *Phragmites* and *Typha*, under similar climatic conditions).

It is most common for *Miscanthus* biomass to be used for heat and power by combustion. **Silvestri et al. (2017)** reported a higher heating value of 18.8 MJ kg^{-1} (SD=0.16) for *Miscanthus* biomass, and **Hastings et al. (2017)** use a similar higher heating value of 18.0 MJ kg^{-1}. **Robertson et al. (2017)** tabled a lower heating value or realized calorific value of 14.0 MJ kg^{-1}, assuming a 20% moisture content as reported in **Lewandowski et al. (2000)**. *Miscanthus* can also be used for anaerobic digestion but its greatest biogas production potential is when green-cut in autumn as opposed to the traditional spring harvesting (**Kieser and Lewandowski, 2017**). It is suggested that green cutting of *Miscanthus* can lead to reduced yields in subsequent years because of limited recycling of nutrients back into the rhizome during the plant senescence phase; this would be true also of other perennial grass species discussed above (**Table 3.2**).

The viability of *Miscanthus* on lowland deep peat soils has not yet been tested. It is possible that it may be more suitable for planting on shallow (wasted) peats, but again there is a need for field trials to establish optimal conditions for its growth within a paludiculture system. As already noted, *Miscanthus* is not a wetland species, and any development of its cultivation as a bioenergy crop would need to consider potential biodiversity impacts, including the risk of it becoming invasive under some circumstances.

3.2.3 Paludiculture and Solar Power

Although not directly concerned with bioenergy production, it is worth noting that some forms of renewable energy production may be compatible with the high water table management of peatlands. Photovoltaic (PV) panels are increasingly being deployed over water bodies, and in principle the same approach could work over wet peat, with solar arrays supported on scaffolding or rail systems above the peat surface. While this would shade the ground surface, it will also enhance microclimate humidity and could aid the growth of certain paludiculture species. *Sphagnum* in particular grows better in partial shade than it does in full sunlight (**Clymo and Hayward, 1982**). Spacing of the panels could be wider than normal in order to allow management and/or harvesting of the paludiculture crop but consequent reductions in energy capture might be more than compensated for by the combined effects of avoided emissions from the paludiculture peat soil, plus improved efficiency of the PV panels (**Nagengast et al., 2013; Chemisana and Lamnatou, 2014**), plus the remaining carbon sequestered in the paludiculture crop residues if/when it is harvested. It may also be possible to install wind turbines within a landscape under paludiculture management, provided that this does not lead to large-scale damage of the peat.

3.3 Paludiculture for Food Production

This sub-chapter explores the production of food crops for human and animal consumption, including the production of crops or development of favourable environments for the raising of animals for human consumption.

3.3.1 Wetland Crops

As noted earlier, most of the UK's lowland peat is currently under drainage-based management for horticulture, arable and livestock production. Changing land-use to the production of fibre or bioenergy crops has the potential to negatively impact on national food security, whilst also displacing GHG emissions from food production in the UK to other countries. Consequently, the possibility of growing food crops on high water table peatland holds considerable appeal, as a means of avoiding having to trade-off GHG emissions and food production. Rice, a wetland species, produces around one fifth of human calorific requirements, so clearly wetland-based food production systems are possible. Rice has been successfully grown on restored former drained agricultural fen peatlands in California and is widely grown on shallow peat in Southeast Asia. On the other hand, attempts to cultivate rice on deep tropical peatlands in Indonesia (the Mega Rice Project) were a failure, resulting in catastrophic and ongoing deforestation, land degradation and fires.

At present, there are no varieties of rice that could be grown under existing climatic conditions in the UK, although it is conceivable that this could change as a result of climate change and/or the development of new cold-tolerant varieties. Alternative wetland food crops are currently being investigated, notably *Glyceria fluitans*, which is a perennial grass native to Europe. In the past it has held significant cultural and culinary status which lead to it being widely foraged for in the wild (**Luczaj et al., 2012**). However, its use largely faded post WW1, and it has not been subject to domestication or conventional cultivation. Researchers have investigated the plant ecology in detail in the past (**Borril, 1956**). However, as *G. fluitans* can be grown on wet or waterlogged soil it has the potential to develop into a viable crop for paludiculture farming. It is a versatile choice as it can tolerate soils ranging from weakly acid to base rich, enabling it to be grown on a variety of rewetted peat sites (**Hill et al., 2000**). Therefore, *G. fluitans* may be a useful food or fodder crop. Field trials of *G. fluitans* are currently being undertaken at the Great Fen as part of the new Water Works project.

Some wetland plants are already food staples, albeit in cultivated form, such as celery (*Apium graveolens*) and water cress (*Nasturtium officinale*), while others have a history of food usage and are in some cases making a comeback, such as nettle (*Urtica dioica*) and water pepper (*Persicaria hydropiper*). The distinction between food and medicine sometimes becomes blurred (**Abel, 2016**), with several plant species having a history of use as both food or flavouring and medicine. Meadowsweet (*Filipendula ulmaria*) is one such example, where it has been used as a flavouring for drinks but also as a herbal remedy for fever and headaches, while round leaved sundew (*Drosera rotundifolia*) was a key ingredient in a popular health tonic called *rosa solis* widely used in Renaissance times to promote youthfulness.

Berry crops are also important in some countries. In Finland, for example, almost 14% of the national edible berry crop (national production is >1 million tonnes, annually) is obtained from peatland systems; with lingonberry (*Vaccinium vitis-idea*), cloudberry (*Rubus chamaemorus*), bog bilberry (*Vaccinium uliginosum*) and crowberry (*Empetrum nigrum*) making up the bulk of this peat-based crop – worth US$60 million (£44.2 million[3]) annually (**Salo, 1996**).

While the ubiquitous cranberry juice and familiar cranberry sauce sold to accompany Christmas turkey are both products of the billion-dollar American cranberry (*Vaccinium macrocarpon*) industry, the smaller native cranberry (V. oxycoccos) is perfectly edible and once formed a significant item of market produce in certain areas of Britain. All the peatland-associated berry species contributing to the Finnish economy also occur in the UK - which also include bilberry (*Vaccinium myrtillus*) and bearberry (*Artocstaphylos uva-ursi*).

Of these several species, only cloudberry (*Rubus chamaemorus*) is gathered exclusively from peatlands in Finland, the remainder being harvested in greater quantities from the mineral forest floor. While Finnish peatlands supply 25,000 tonnes of bog bilberry (*Vaccinium uliginosum*) annually, the mineral soils of the forest floor provide 475,000 tonnes of *V. uliginosum* berries each year (**Salo, 1996**). Paludiculture cultivation for all but cloudberry is thus possible on both wet mineral soils and peat soils.

Finally, many of the reed crops discussed above may also be used as fodder for livestock grazing, although at present this largely takes place as part of the conservation management of wetland sites, rather than to support economic livestock production (see **section 3.3.2**).

3.3.2 Litter, Fodder and Grazing

In former times, written records for the Fens of East Anglia and archaeological evidence from further afield show that the wetlands of the UK provided much in the way of animal produce. There is evidence from Neolithic times for the hunting of auroch, deer, wild boar, beaver and otter (**Godwin, 1978**; **Coles and Orme, 1980**), while abundant waterfowl provided a huge variety of species that were hunted, including pelican, crane, bittern and black grouse, as shown by the evidence of remains from lake villages. Written records from the 13th Century detail royal demands for swans, herons, cranes and bitterns (**Darby, 1940**), while Elizabethan records reveal species provided for a wedding feast to have included mallard, teal, swans, cranes, heron, knot, stint, godwit, curlew, plovers and

[3] Conversion based on 5-year (2015-2019) annual average USD/GBP exchange rate (1 GBP = 1.3566 USD). Source: Bank of England Database

larks (**Godwin, 1978**). The other major source of animal produce within the Fens came from fisheries, of which eels were the most important component in pre-drainage times – important and abundant enough to be used as a form of currency or tax (**Darby, 1940**) – while in later times, as the extent of the inland fisheries dwindled in the face of widespread drainage, pike remained a major source of traded animal protein (**Darby, 1956**). Today the majority of species mentioned above are either extinct in Britain or are strictly protected by legislation, but red deer are now part of the farming livestock, as are boar. Interest is also growing in the use of more exotic livestock within farming as a whole with a view to farm diversification and penetrating niche markets, but of particular note for wetland conditions are water buffalo and Konik ponies. The former already occurs as established herds in continental Europe while for a number of years now, herds of Konik ponies have formed part of the grazing system for conservation management on a number of wetland sites in the UK. Whether this practice could be scaled up to support commercial-scale food production has yet to be established.

The use of fen litter for animal bedding was once widespread but has now all-but ceased in the UK, in part because the areas of supply have vanished. Those areas that still support fen communities are now often under conservation management. Production of fodder from fen mowing has an equally long history and, along with fen grazing, has been the driving force maintaining open fen conditions for many hundreds of years across much of Europe. Fen fodder was not necessarily the first choice but was in many cases the only choice, despite its lower nutritional quality compared with other sources of fodder. Nonetheless, some types of livestock were able to make good use of such fodder and some continue to do so today, such as the horses of the Camargue (**Müller and Sweers, 2016**). Typical productivity of fen communities is in the range of 3-12 t DM yr^{-1} for sedge and reed fen communities (**Oehmke and Abel, 2016**).

Some wetland species provide reasonably palatable fodder, particularly reed canary grass (*Glyceria fluitans)*, redtop (*Agrostis gigantea*), reed manna grass (*Glyceria maxima)*, marsh foxtail (*Alopecurus geniculatus*) and lesser pond sedge (*Carex acutiformis*), though all are of lower forage quality than conventional silage. Some livestock, notably horses, cattle breeds with low nutritional demands such as Angus or Limousin, and water buffalo can use fen fodder successfully, and in preference to modern silage, which can prove too rich (**Müller and Sweers, 2016; McBride et al., 2011**). Indeed, the fibre and nutritional composition of such fodder can be positively beneficial for horses (**Zielke, 2016**).

Rewetting of formerly drained peatlands, and other flood-prone soils, would tend naturally to encourage establishment of wetland species such as rushes, sedges and wetland grasses (**McBride et al., 2011**). This resulting plant assemblage can then be mown for use as animal litter. However, experiments to re-establish fen litter as animal bedding have encountered issues that have rendered the litter unsuitable for modern animal husbandry (**Andrews, 2000**). Furthermore, although litter-meadow areas maintained for conservation purposes may require mowing or grazing to maintain their biodiversity interest, the costs of transporting the mown material from the site and drying it sufficiently for litter sale are generally prohibitive, while arrangements for safe on-site grazing by stock can be equally challenging (**Brouwer et al., 2001**).

Fodder hygiene (i.e. freedom from poisonous weeds and other pathogens) and the need for fast drying to prevent mould development mean that fodder production faces a number of regulatory hurdles that constrain large-scale take-up until suitable systems have been developed to address and resolve these various issues. This would require investment in R&D for weeding, harvesting, drying and transport systems.

3.3.3 Fisheries

Inland fisheries as a source of animal protein from coarse fish all but disappeared in the UK with the draining of the East Anglian Fens, and the formerly abundant eel is now listed as critically endangered and a Priority Species under the UK Post-2010 Biodiversity Framework. The only inland fishery industry (i.e. for food, not sport) in the UK of note today is trout farming, introduced in the 1950s and now producing more than 17,000 tonnes of game-fish annually for the food industry from the waters of northern rivers (British Trout Association website). There is no fundamental reason why small-scale inland fisheries could not be re-established on a commercial basis as part of a paludiculture enterprise, using the established model of trout fisheries ('blackwater' fish are an important economic resource in the peatlands of Southeast Asia). Carp production within the European Union uses specially constructed fish ponds which produce 66,330 tonnes of fish, worth €140 million (£116.4 million[4]) annually (European Union Fisheries). Given the popularity of fishing as a sport, there is also the potential for co-financing any such venture through sport fishing.

3.4 Other Potential Paludiculture Products

This chapter considers niche, previously popular or under developed uses for paludiculture crops. We explore alternative uses for *Sphagnum*, the potential of some medicinal crops, and plants grown for their use as construction materials and in the production of fabrics.

3.4.1 Alternative Uses for *Sphagnum*

There are a number of other potential alternative uses for *Sphagnum*, linked to its antibacterial and absorbent qualities. During the First World War it was harvested in the UK as a material to create the 'first field dressing' - the brainchild of Lt. Col. Charles Cathcart of the Royal Army Medical Corps and a Senior Surgeon at the Royal Edinburgh Infirmary, and his associate Issac Bayley Balfour a professor of Botany. Their basic trials had confirmed that *Sphagnum* absorbed large volumes of blood, drying out the wound but with the added benefit of appearing to prevent infection (**Ayres, 2015**). Towards the end of the war, 1,000,000 dressings a month were produced (**Nichols, 1918**). In addition, the British War office ordered some 20,000,000 dressings from Canada in 1918, further highlighting its importance (**Riegler, 1989**). These field dressings were largely produced by volunteers who would hand harvest *Sphagnum* in the wild. The high cost of hand harvesting, the need for particular 'surgical' *Sphagnum* types (i.e. robust, stout branches, large leaves), and the lower cost of mass produced alternatives, all contributed to the decline in its use.

Sphagnum has also been investigated as a sanitary product due to its absorbent properties (**Johnson & Johnson 1994**), and as a method for preventing food spoilage (**Stalheim et al., 2009**). It has been shown to house compounds that are suitable as biocontrol measures for fungal and microbial pathogens in plants and humans (**Opelt et al., 2007**). With a growing interest in sustainably sourced and biodegradable products, it is possible that markets could be developed for these and other products, provided that a sufficiently large and sustainable source of material can be established. The unique hydrological and antibacterial properties of *Sphagnum* suggest that there may be further novel uses, for example in the development of medically useful compounds and extracts. This will require further research and development.

[4] Conversion based on 5-year (2015-2019) annual average EUR/GBP exchange rate (1 GBP = 1.2028 EUR). Source: Bank of England Database

Finally, *Sphagnum* has historically been used to preserve food, for example, to preserve salmon as a food store on Viking voyages. The preservative properties of *Sphagnum* come from the pectin-like exudate called sphagnan, which immobilises decomposer microorganisms and is thus of potential interest to the modern food industry (**Børsheim et al., 2001; Stalheim et al., 2009**).

3.4.2 Other Medicinal Crops

Possibly the most well-known and widely used medication obtained from a wetland plant is aspirin, derived originally from willow (*Salix* spp.). A great many other wetland plants contribute to the catalogue of species used at one time or another for medical purposes, either by conventional medicine or in herbal medicine. While coming under increasingly rigorous regulation, the herbal medicine industry is predicted to have a global value of US$129 billion (£95 billion[3]) by 2023, raising concerns that wild harvesting of many plant species will have a hugely negative impact on biodiversity as well as potentially compromising quality control of herbal medicine ingredients.

Agricultural production of species used in the herbal medicine industry is one way to reduce the burden on biodiversity resulting from wild harvesting, while also providing an important guarantee of quality control. Farm statistics in the UK are already showing a trend towards adoption of more exotic crops, in part aimed at the medical market (**Defra, 2019**).

Paludiculture offers opportunities to establish supply chains for a whole range of wetland plants considered to have medical properties. An example of which is sundew (*Drosera rotundifolia*), which has been used in herbal medicine since earliest times for bronchial ailments. It remains a popular and extensively tested herbal medicine (**Krenn et al., 2004; Paper et al., 2005**), so much so that an estimated 6 – 20 tonnes of dry mass is wild harvested every year, largely from Finland and Madagascar (**Baranyai, 2016**). Conservation legislation protecting the species in many European countries has resulted in demand outstripping supply. Experimental trials have shown that sundew can be grown as a joint crop along with farmed *Sphagnum* (**Baranyai, 2016**), although the species can also grow on highly leached sand and silt if there is little or no competition (**Crowder et al., 1990**).

3.4.3 Use of Paludiculture Crops in Construction Materials

3.4.3.1 Traditional Uses

Perennial reed grasses, particularly *Phragmites spp.*, have been used for millennia as a construction material. The best-known example is the use of reeds for thatched roofing; in Europe reed is almost exclusively used for thatching **(Wichmann & Köbbing, 2015)**. Thatching requires reed with specific qualities, so the efficiency from harvested biomass to final product varies and unsuitable material can make up to 50% of standing biomass **(Wichmann & Köbbing, 2015)**. In an expert-based assessment of coastal wetlands, the potential for direct use of reed stems as thatch or insulation material was deemed to be greater in homogenous reed habitat compared to transitional reed habitat **(Karstens, Inácio and Schernewski, 2019)**. Hence, there may be implications for the utility of less established or more variable stands of reed as a reliable source for thatching material. High quality reed most suitable for thatching is straight and thin, with low moisture content. Harvesting in winter generally ensures a sufficiently low moisture content below 18% **(Köbbing et al., 2013)**.

Reed is traded as bundles and a standard 'Euro bundle' has a circumference of 60 cm, with variable lengths (**Wichmann & Köbbing, 2015**). A standard bundle weighs approximately 4.5-6.0 kg, and a typical thatch roof with a 30 cm thickness would need 10–11 bundles per square metre (**Köbbing et al., 2013**), hence approximately 45–60 kg of reed per square metre.

The number of bundles that can be harvested from a hectare of reed will depend on the productivity of the reed, quality of reed material and the size of the bundles; the yield of bundles from reedbed can therefore vary from 250 to 1000 bundles per hectare (**Wichmann & Köbbing, 2015**). **Yates (2006)** noted that the reed management and harvesting for thatching had 'virtually disappeared in the UK' and that significant quantities of reed were imported from Hungary and Turkey. In 1989, the UK imported 1.5-1.8 million bundles (75-85% of usage) to meet shortfalls (**Köbbing et al., 2013**) and until 2013 an import share of 75% was maintained (**Wichmann & Köbbing, 2015**); while exact figures are not available, the market situation is similar today. This level of import indicates there is strong market demand for reed thatch and a lack of domestic supply. However, it has been suggested that high labour costs and nature conservation policy limit the availability of UK-grown thatching reed (**Wichmann & Köbbing, 2015**).

In a comparison of reed-importing European countries (Netherlands, Germany, Denmark and UK), the cost of a standard reed bundle in the UK was estimated at around €2.40–3.00 (£2.00-2.50[4]) in 2013 **(Köbbing et al., 2013).** Allowing for inflation and assuming average current cost of a bundle at £2.64 would yield revenue of between £659 and £2638 per hectare (based on 250–1000 bundles per hectare; **Wichmann & Köbbing, 2015**). This range is slightly higher, but similar, to that in the economic modelling study on the commercial viability of reed-based paludiculture by **Wichmann (2017),** who estimated revenue from harvesting bundles of reed at €607–2,380 (£505-1,979[4]) ha^{-1} $year^{-1}$. **Wichmann (2017)** also found harvesting reed bundles for thatching, despite having the highest harvesting costs, to be the most profitable option for reed-based paludiculture, notably even compared to production for bioenergy. This study also noted, however, that combustion for bioenergy would provide an alternative pathway for reed biomass that does not meet the quality requirement for use in thatching and can enable profitability in these circumstances. This calculation could also change as market conditions, financial incentives and carbon prices change over time.

Historically, the rush species *Juncus effusus* was used for flooring in the UK, and in Japan it is still widely used, and of great cultural significance, in the construction of *tatami* mats. In the UK, the presence of *J. effusus* generally indicates wet, low-grade agricultural land, and under these conditions the plant has a rather short and thick-stemmed growth habit. However, if given sufficient nutrients and freedom from competition, it develops the tall, thin-stemmed growth habit on which the *tatami* industry depends. Given its ubiquitous nature, *J. effusus* cultivation might prove to be a relatively uncomplicated crop to develop, especially if Japanese methods of cultivation are studied.

3.4.3.2 Novel Uses

Köbbing et al. (2013) present a comprehensive overview of the other uses of reed material and some economic information. These other potential uses include garden fencing, panelling for construction, insulation material, and paper making (or pulp production).

Reed is widely available as garden screening or fencing in the UK. Though no wholesale costs could be obtained, it retails for approximately £2-3 per square metre, with each square metre weighing approximately 0.7 kg. The use of more robust reed matting or panel products are found in construction, but with limited production in the UK, their use is also more prominent in the refurbishment sector of building than the new build sector (**Yates, 2006**). These mats or panels are compressed in a weaving loom, and usually fixed to a wall and plastered or covered with clay. A square metre of 3 cm thick reed panelling requires 13 kg of reed material and, using an example from Austria, these panels cost €6–10 (£5-8.30[4]) per square metre **(Köbbing et al., 2013)**. One advantage in the use of reed for construction panelling is that the quality of reed required for this is lower than that required for thatching.

Granulate panel or fibreboard that use processed material (e.g. chips or clippings) are further construction products suitable for use of reed. Chopped reed is mixed with glue before producing the panels and, notably, the reed material can include stems, leaves and 'waste' material from thatching **(Köbbing et al., 2013)**. The cost of reed-based granulate panels is €19 (£15.80[4]) per square metre, again using an example from Austria **(Köbbing et al., 2013)**. *Typha spp.* have been used in Germany to produce lightweight construction boards for structural support and to produce insulating materials; on a volume basis, annual *Typha* yields per hectare can exceed those of timber **(Wichtmann and Joosten, 2007)**. The board has high mechanical strength, provides good but breathable insulation, and is fireproof. It has been used in the restoration of historic buildings in both Germany and Bulgaria (**Georgiev et al., 2019**), but it is also used within the wider construction industry as a sustainable alternative to standard plasterboard. In Germany, at least two companies are also making use of *Typha spp.* for insulation materials (**Nowotny, 2016**). Similar to granulate panels, fibreboard is typically made from wood fragments and wheat straw bonded with resin, and the long fibres of reed species make them suitable for use in its production **(Croon, 2013)**.

Another potential use of wetland fibre crops is for the production of lightweight aggregate (LWA, < 1000 kg m^{-3}) to improve the thermal insulation properties of concrete and reduce structural dead load, allowing the construction of larger buildings with the same foundation size, all of which are associated with reduced CO_2 emissions. LWA is predominantly an expanded (or bloated) clay aggregate made from pulverised fuel ash (PFA) from coal-fired power stations and naturally occurring pumice. Organic matter can be mixed with LWA raw materials to provide combustion energy and produce a porous low-density structure. LWA manufacture has been estimated to emit 0.22 tonne of CO_2 per tonne of aggregate. This compares favourably with the emissions from Portland cement production, where 0.83 tonnes of CO_2 is generated per tonne of cement. Energy for LWA production can be provided from biomass combustion given the relatively low sintering temperature required. A number of studies have used organic matter such as sewage sludge in the mix as an energy source (**Ayati et al., 2018**), but sedge litter from paludiculture would also be a potential source of such organic matter.

From a climate change mitigation perspective, the use of harvested biomass in construction or building materials is particularly appealing as it can provide a long-term store for carbon removed from the atmosphere via photosynthesis (equivalent to Harvested Wood Products) and could therefore form part of a greenhouse gas removal management system. The development of processing facilities, supply chains and markets for paludiculture products for use by the building industry would require significant investment, but certain products appear to have considerable potential as an alternative productive use of lowland peat.

3.4.4 Use of Paludiculture Crops in Fabrics

In Japan, cloaks made from rice straw date back at least 1,500 years, whereas in Western Europe the nettle (*Urtica dioica*) has a record at least as long for use in clothing. The natural habitat for the plant is wet fen carr (wooded fen) where it can grow to considerable heights. It is aided in this by the strong fibres in its stem, and it is these fibres that have formed the raw material for clothing as far back as the Bronze Age (**Harwood and Edom, 2012**).

In recent years there has been an upsurge of interest in nettle as a source of sustainable fibres for luxury clothing because, despite being a wetland plant, nettle requires less water than cotton to produce its fibres, the fibres are stronger than cotton and are also softer (**Di Virgilio et al., 2015; Debnath, 2015; SwiCoFil; FashionUnited**). In 2003, Defra commissioned an investigation into the development potential of nettle fibres (Sustainable Technology In Nettle Growing - STING - LK0820).

Di Virgilio et al. (2015) state that nettle can achieve 3 – 12 t ha^{-1} dry stalk yield, even under low-input regimes. As well as fibres, the plant contains a number of potentially valuable compounds that could find uses in the medicinal and cosmetic sectors. According to **Di Virgilio et al. (2015)** the main constraints currently hindering large-scale production of nettle are the lack of sufficient crop volume and the cost-effectiveness of processes required to extract the fibres from the crop – thus explaining its present restriction to the luxury clothing market.

As a paludiculture crop, nettle is able to thrive in waters rich in nitrogen and phosphorus and thus has the potential to be both a crop in its own right, but also as part of a constructed wetland system for natural water treatment. Its tolerance of shade also means that it could form a companion crop beneath an alder (*Alnus glutinosa*) canopy, grown as a paludiculture crop for timber or bioenergy.

4 GHG EMISSIONS/REMOVALS AND ASSOCIATED IMPACTS OF PALUDICULTURE

This chapter considers the direct and indirect impacts of a transition from conventional to paludiculture management of lowland peat on greenhouse gas emissions and associated processes. **Section 4.1** considers direct GHG emissions, **Section 4.2** considers the indirect GHG emissions reductions that might be achieved via the production of paludiculture crops for bioenergy, **Section 4.3** evaluates potential impacts on land subsidence, and **Section 4.4** estimates the potential for energy and emission savings from reduced requirements for pumped drainage and surface irrigation.

4.1 Estimation of Direct GHG Emissions and Removals

4.1.1 Carbon Dioxide and Methane

The study of GHG emissions and removals from re-wetted peatlands under paludiculture management is an emerging discipline, and to date there are a limited number of studies providing empirical data. Of these, all suggest that paludiculture has the potential to reduce CO_2 emissions relative to conventional drainage-based agriculture or peat extraction, and some studies suggest that sites under paludiculture management could become net CO_2 sinks. **Shurpali et al. (2009)** found that organic soils under reed canary grass (*Phalaris arundinacea*) acted as net CO_2 sinks in wet years, but could become net sources during dry years. **Günther et al. (2015)** found that the GHG balance of a re-wetted fen used for biomass harvesting was similar to that of a nearby pristine fen, despite the biomass removal. **Karki et al. (2015)** found that the net GHG balance of a reed canary grass paludiculture site was positive (around 9 t CO_2e ha^{-1} yr^{-1}) when water table depth (WTD) was at 40 cm, but reduced to near-zero values (with net uptake of CO_2) when WTD was at 0 cm or 10 cm. **Mander et al. (2012)** also measured net CO_2 uptake at two reed canary grass sites after accounting for above-ground biomass accumulation, with slightly higher uptake at a fertilised compared with an unfertilised site. In this last case it appears that the peat itself was losing carbon, and that the site would therefore become a net CO_2 source following biomass harvest. In contrast, **Günther et al. (2015)** reported that a former bog grassland now managed for *Sphagnum* farming was acting as a strong net CO_2 (and overall GHG) sink. Initial CO_2 flux measurements at the UK *Sphagnum* farming field trials also suggest that, once complete *Sphagnum* cover is achieved, net emissions cease, and the site may even become a net sink for CO_2 (**Richard Lindsay, pers. comm**.). Since these sites are subject to surface irrigation rather than active raising of the water table, it is possible that this has not been offset by CH_4 emissions, although this has not yet been tested.

It should be noted that the studies described above only considered direct emissions from the site, and not the potential impact of displacing emissions associated with food production (e.g. fertiliser use, fuel use during farming activities and transport). These emissions may be considerable in some cases, for example if food production is displaced to other countries and air freighted to the UK. A full GHG assessment of paludiculture versus conventional agriculture thus requires a full life cycle analysis and assessment of alternative sources of food production, in addition to site measurements.

At the present time, the number of studies available that report annual CO_2 and CH_4 fluxes from paludiculture sites are not sufficient to provide robust emission factor estimates or response functions, particularly for the diverse range of potential paludiculture crops. Available data do suggest that peatlands managed for paludiculture exhibit broadly the same patterns of behaviour in relation to carbon cycling, and resulting CO_2 and CH_4 fluxes, as natural and managed peatlands exposed to similar abiotic conditions. In particular, the broader evidence base suggests that WTD exerts a dominant influence on both CO_2 and CH_4 fluxes, which overrides the effects of other potential factors such as vegetation, management or peat type. This was a key conclusion of the Defra SP1210 Lowland Peat project (**Evans et al., 2017**), and is also supported by synthesis studies such as that of **Couwenberg et al. (2011)**. In work linked to the current project, we have collated a large dataset of published flux data from the UK and other high-latitude peatlands, in order to derive response functions that relate CO_2 and CH_4 emissions to annual average WTD (**Evans et al., in prep**). For CO_2, data from flux tower studies was utilised, with adjusted emissions to account for biomass offtake at sites where vegetation was harvested, assuming that all of this biomass would be returned to the atmosphere as CO_2. The dataset largely comprises either semi-natural or agriculturally managed sites, but does include a number of re-wetted sites, and one paludiculture site (the reed canary grass site of **Shurpali et al., 2009**).

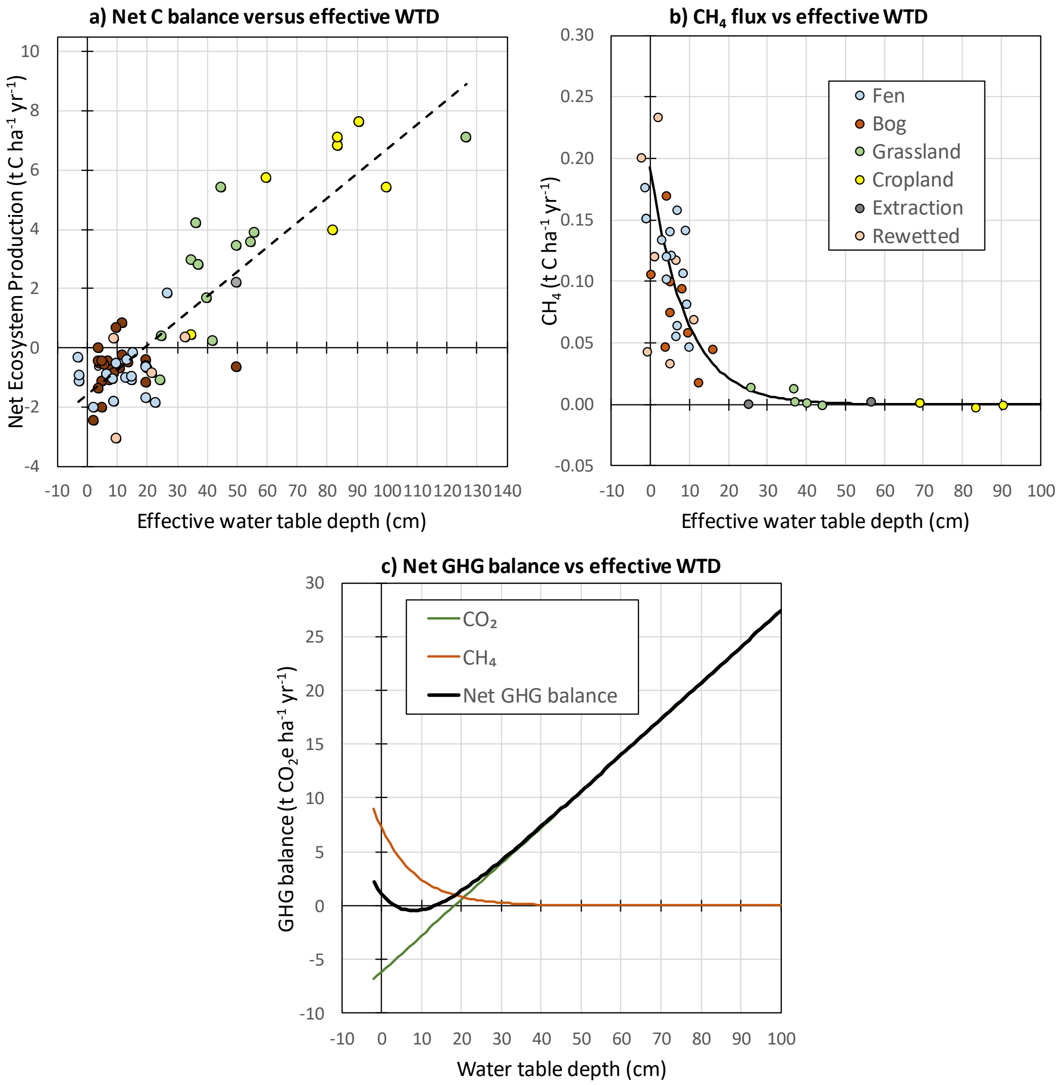


Figure 4.1 *Observed relationships between a) CO2, b) CH4 and c) combined CO2 and CH4 fluxes expressed in terms of their net climate impact based on 100 year global warming potentials, versus effective mean water table depth, for a range of peatland sites under different management.*

In the analysis, if WTD exceeded peat depth we set the 'effective' value of WTD equal to the peat depth. This assumes that once the water table drops below the base of the peat, no further increase in CO_2 emissions will occur. The results suggest that effective WTD accounts for around 75% of observed variance in site CO_2 balance across all sites, despite these spanning a wide range of peat types, climatic conditions and management activities (**Figure 4.1**). The relationship appears to be linear, with sites having WTD > 17 cm acting as net CO_2 sources, and sites with WTD < 17 cm acting as net CO_2 sinks.

The analysis of CH_4 flux data was based on static chamber measurements, and was restricted to UK-based studies. Otherwise the methods conformed to those described above for CO_2. The results for CH_4 show a similarly strong, but highly non-linear relationship, with sites having a WTD > 20 cm generally having near-zero CH_4 fluxes, and sites with WTD closer to the surface having significant emissions (**Figure 4.1**). Note that both datasets exclude sites that are permanently inundated to a depth of > 5 cm. **Figure 4.1c** shows the net climate impact of CO_2 + CH_4 emissions versus WTD based on a 100 year Global Warming Potential (GWP) of 28 (**Myhre et al., 2015**). This suggests that emissions will be increasingly positive due to CO_2 emissions when WTD > 20 cm, and positive due to CH_4 emissions when WTD approaches or inundates the peat surface. The optimal climate impact (slight net cooling effect) occurs at around WTD = 8 cm. The data suggest that paludiculture crops that require continuous inundation of the peat surface are unlikely to provide much if any climate mitigation benefit due to high CH_4 emissions. For example, a paludiculture crop requiring 5 cm of surface inundation would (based on a 100 year GWP approach) produce similar net CO_2 + CH_4 emissions to an agricultural grassland with a mean WTD of 50-60 cm. This has implications for prospective bioenergy species such as *Phragmites* and *Typha* for which optimal water levels are above the ground surface for at least part of the year (see **Section 3.2.2**). The use of 100 year GWPs is somewhat contentious in this respect, because over longer time horizons an inundated system would have a net cooling impact, because the relatively short atmospheric lifetime of CH_4 would signify a reduced warming impact over time, whereas the CO_2 removed from the atmosphere by the growing peat will have a permanent cooling effect. Since 100 year GWPs are routinely used in IPCC emissions reporting and assessments, we have adhered to this approach here.

Clearly, the relationships shown in **Figure 4.1** are generalised, and do not explicitly account for the influence of different plant species or management activities on the carbon cycle, or on CH_4 emissions in particular, which are strongly plant-mediated (**Dias et al., 2010; Couwenberg et al., 2011; Gray et al., 2013**). In practice, it is likely that plant effects are to some extent captured implicitly in the observed relationships, because wetter sites will tend to support plant functional types that are adapted to waterlogged conditions, for example species with aerenchymatous stems and roots, which provide a plant-mediated CH_4 pathway to the atmosphere (**Karki et al., 2016**). Thus, the success of peatland rewetting in providing a reduction in greenhouse gas emissions depends in part on the effect that the specific paludiculture crop has on CH_4 emissions.

In a study using mesocosms, **Karki et al. (2016)** compared the carbon balance of rewetted and drained peat under a crop of *Phalaris arundinacea* and demonstrated that the rewetted site had an annual balance of 0.68 kg C m^{-2} compared with 0.03 kg C m^{-2} for the drained site, even when considering the higher CH_4 emissions under rewetted conditions. In a later mesocosm study, **Vroom et al. (2018)** demonstrated that *Typha latifolia* grown as a paludiculture crop strongly mitigated CH_4 emissions compared to an unvegetated control. In addition, harvesting of the crop effectively removed various forms of N and P and thus provided a co-benefit of water purification, in a similar manner to wetlands constructed for wastewater treatment. The effects of site conditions on CH_4 emissions can be difficult to predict (**Carmichael et al., 2014**; **Van de Riet and Hefting, 2013**) and will depend, *inter alia*, on the interplay between site water levels, water quality (i.e. nutrient levels), substrate quality and plant species. Nevertheless, most studies indicate that there is an immediate climate benefit from rewetting drained peatlands (**Couwenberg et al., 2011**).

In managed paludicultural systems, some variation around the mean value for any given WTD may be expected depending on the particular traits of the species being cultivated. At the current time we do not have sufficient data to derive the adjustment factors that would be required to account for these species-specific effects, although it is possible that they could be added in future, for example based on plant functional traits (e.g. **Gray et al., 2013**).

4.1.1.1 Emissions Calculator

We used the relationships shown in **Figure 4.1** to develop a simple 'Paludiculture Emissions Calculator'. At present, this simply requires the mean WTD requirements for the crop, and peat depth of the site, to be specified. This estimates the CO_2 and CH_4 emissions/removals that would occur for the resulting 'effective' WTD, as described above. If the pre-paludiculture management of the site is known (i.e. the mean WTD of the prior conventional cropping system) this can be used as a counterfactual, enabling the net mitigation potential of a transition to paludiculture to be estimated. Illustrative examples (not based on specific sites or crop types) are shown in **Table 4.1**. An Excel version of the Emissions Calculator will be made available via the project website, which can be used to run specific examples based on real data as this becomes available.

Table 4.1 *Paludiculture Emissions Calculator: Estimated CO_2 and CH_4 emissions/removals, and net GHG mitigation potential, a set of theoretical transitions from conventional to paludiculture crops for an example deep peat site (upper rows) and a shallow peat site (lower rows). Bold data are user specified, other data are calculated*

CROP	SITE PROPERTIES			EMISSIONS/REMOVALS						MITIGATION		
	WTD	Peat depth	Effective WTD	CO_2	CO_2	CH_4	CH_4	C balance	GHG	CO_2	CH_4	GHG
	cm	cm	cm	$t\ C\ ha^{-1}\ yr^{-1}$	$t\ CO_2\ ha^{-1}\ yr^{-1}$	$t\ C\ ha^{-1}\ yr^{-1}$	$t\ CO_2e\ ha^{-1}\ yr^{-1}$	$t\ C\ ha^{-1}\ yr^{-1}$	$t\ CO_2e\ ha^{-1}\ yr^{-1}$	$t\ CO_2e\ ha^{-1}\ yr^{-1}$		
Deep peat example data												
Paludiculture crop 1	**0**	**100**	0	-1.58	-5.78	0.17	6.51	-1.40	0.73	-13.1	6.2	-6.9
Conventional crop 1	**40**	**100**	40	2.00	7.33	0.01	0.27	2.01	7.59			
Paludiculture crop 2	**10**	**100**	10	-0.68	-2.51	0.08	2.93	-0.61	0.42	-16.4	2.9	-13.5
Conventional crop 2	**60**	**100**	60	3.79	13.88	0.00	0.05	3.79	13.94			
Paludiculture crop 3	**20**	**100**	20	0.21	0.77	0.04	1.32	0.25	2.09	-19.7	1.3	-18.4
Conventional crop 3	**80**	**100**	80	5.57	20.44	0.00	0.01	5.57	20.45			
Paludiculture crop 4	**30**	**100**	30	1.10	4.05	0.02	0.59	1.12	4.64	-22.9	0.6	-22.4
Conventional crop 4	**100**	**100**	100	7.36	27.00	0.00	0.00	7.36	27.00			
Shallow peat example data												
Paludiculture crop 1	**0**	**30**	0	-1.58	-5.78	0.17	6.51	-1.40	0.73	-9.8	5.9	-3.9
Conventional crop 1	**40**	**30**	30	1.10	4.05	0.02	0.59	1.12	4.64			
Paludiculture crop 2	**10**	**30**	10	-0.68	-2.51	0.08	2.93	-0.61	0.42	-6.6	2.3	-4.2
Conventional crop 2	**60**	**30**	30	1.10	4.05	0.02	0.59	1.12	4.64			
Paludiculture crop 3	**20**	**30**	20	0.21	0.77	0.04	1.32	0.25	2.09	-3.3	0.7	-2.6
Conventional crop 3	**80**	**30**	30	1.10	4.05	0.02	0.59	1.12	4.64			
Paludiculture crop 4	**30**	**30**	30	1.10	4.05	0.02	0.59	1.12	4.64	0.0	0.0	0.0
Conventional crop 4	**100**	**30**	30	1.10	4.05	0.02	0.59	1.12	4.64			

The current Emissions Calculator includes both net direct and indirect CO_2 emissions/removals. In other words, it includes both the gaseous emissions from the field, and indirect emissions from harvested biomass, assuming that all of this biomass will be emitted to the atmosphere in due course. This assumption is considered valid for food crops, but if some or all of the harvested biomass is used for energy generation this could be included in the total mitigation calculation as avoided fossil fuel emissions. Similarly, the additional mitigation provided by the use of harvested biomass from paludiculture crops in long-lived products such as building materials should be accounted for using an approach similar to that used for Harvested Wood Products. This is considered below.

Current omissions from the calculator include the role of ditches as potential CO_2 and CH_4 sources. This is a potentially important issue, because many paludiculture systems rely on maintenance of the pre-existing ditch network to manage (albeit higher) water levels. For example, in a study of the GHG balance of a *Sphagnum* farming experiment on former grassland, **Günther et al. (2017)** noted that while the bog surface was acting as a net GHG sink, the ditches were acting as an approximately equal net emission source on a per unit area basis. While it is unlikely that emissions from ditches in paludiculture systems are higher than they were under previous drainage-based management, ditch fluxes are of sufficient importance to justify their inclusion in the emissions calculations. Indirect CO_2 emissions via organic carbon leaching to drainage waters could also represent an unspecified emission term, although in this case there is evidence to suggest that fluxes are likely to be lower in re-wetted versus drained systems (**Evans et al., 2016)**. Finally, as noted above, the approach also does not consider any displaced emissions that might result from the relocation of food production to other regions.

4.1.2 Nitrous Oxide

At this stage, we are not able to quantify the potential effects of a transition to paludiculture crops on N_2O emissions. In Defra project SP1210, N_2O emissions were only measured at a subset of sites, and for relatively short periods. The results suggested that emissions from arable and intensive grassland sites could be high, consistent with the default Tier 1 emission factors produced for the IPCC Wetland Supplement (**IPCC, 2014**). In a similar analysis to our assessment for CO_2 shown above, **Couwenberg et al. (2011)** found a positive relationship between mean N_2O emissions and mean WTD, which would imply that any reduction in drainage depth should generate an additional climate mitigation benefit by also reducing N_2O emissions. On the other hand, a global data synthesis by **Pärn et al. (2018)** showed that intermediate, fluctuating WTDs in N-rich organic soils tend to generate the highest N_2O emissions, which may be problematic for some paludiculture crops, especially those grown on former farmland or those in which water levels are not strongly controlled.

Several studies of fertilised reed canary grass systems have shown significant N_2O emissions, especially in the period immediately after fertilisation (**Kandel et al., 2013; Karki et al., 2015; Kandel et al., 2019**). These emissions are sufficient to significantly influence the overall GHG balance of the system; for example, **Kandel et al. (2019)** report average annual N_2O emissions of 2.5 kg N_2O-N ha^{-1} yr^{-1}, which equates to 1.2 t CO_2e ha^{-1} yr^{-1} and is slightly higher than the IPCC Tier 1 emission factor for shallow-drained grassland on organic soil. On the other hand, studies of unfertilised paludiculture sites, including reed canary grass (**Hyvönen et al., 2009; Mander et al., 2012**) and cultivated *Sphagnum* (**Günther et al., 2017**), showed negligible N_2O emissions. Even at a fertilised reed canary grass site, **Karki et al. (2015)** found that N_2O emissions reduced from values similar to those reported by **Kandel et al. (2019)** when WTD was 40 cm, to < 0.5 t CO_2e ha^{-1} yr^{-1} when WTD was within 10 cm of the surface. The authors concluded that this finding was consistent with the relationship of **Couwenberg et al. (2011)**, who suggested that N_2O emissions would only be significant when WTD > 15 cm. They also suggest the paludiculture crops with high biomass production will effectively limit the amount of mineral N available for denitrification (as will repeated harvesting) thus minimising N_2O emissions.

Overall, the balance of evidence suggests that a transition to paludiculture will reduce N_2O emissions relative to conventional agriculture, especially where water tables are high, fertilisation rates are low or zero, and crop biomass growth and harvest minimises the amount of mineral N in the soil. On the other hand, paludiculture crops that require high rates of plant growth to be economically viable, such as alternative food crops or high-yielding biomass crops, may require fertilisation or lower water levels, and in these circumstances N_2O emissions could be significant. At the present time, N_2O reductions are not included in estimates of GHG emissions from paludiculture, due to a lack of data. This effectively assumes that N_2O emissions will, at worst, remain the same as they were under drainage-based agriculture, and should thus provide a conservative estimate of the total climate mitigation benefits of paludiculture. With additional field data, it should be possible to develop response functions similar to those shown in **Figure 4.1**, in order to capture the climate mitigation benefits of reduced N_2O emissions where these occur.

4.2 Indirect GHG Emissions and Removals

Data from selected species are presented as an example of estimating GHG emissions associated with bioenergy crop cultivation, processing and energy conversion, and compared with the estimated energy derived from biomass against traditional energy sources (National Grid and Natural Gas).

4.2.1 GHG Mitigation Potential of Bioenergy

To date there have been few studies that have extensively explored the potential for paludiculture-based bioenergy, despite estimates of 500,000 tonnes of conservation biomass available for harvest from UK habitats each year that could be used as a fuel source (**Philips et al., 2017**). A review of energy conversion technology against feedstock quality and pre-processing is outside the scope of this review, but merits further attention.

The average GHG fluxes in paludicultural systems with perennial reed grasses may be close to neutral, though climatic conditions and water supply can influence annual variability. **Gunther et al. (2015)** reported that the GHG balances on a 15-year-old paludiculture site were similar to those of pristine fens which would strongly suggest paludiculture is preferable to conventional agricultural practices on drained peatlands in terms of carbon balance. **Shurpali et al. (2009)** measured microclimatic variables and net ecosystem CO_2 exchange (NEE) in a cutover peatland soil with Reed Canary Grass (*Phalaris arundinacea*); while there was high inter-annual variability with some wetter years, there was always a net uptake of carbon, even after accounting for biomass offtake.

Using harvested biomass to produce bioenergy may provide GHG mitigation with these systems by offsetting CO_2 generated from the use of fossil fuels. The net increase of CO_2 from burning fossil fuels is avoided with energy derived from biomass through combustion or biogas production, since it has been absorbed from the atmosphere during crop photosynthesis. Below, we present a simple worked example, using yield and biomass quality data for *P. australis*, and relevant information on harvesting machinery and operation, to estimate GHG emissions associated with bioenergy crop production and processing for combustion[5]. Using literature-derived values representing the harvesting of biomass and key biomass characteristics, we estimate CO_2-equivalent emissions associated with bioenergy crop production from ‘the cradle to the farm-gate’, and consider potential energy yields. Using the example of *P. australis* harvested in winter to optimise moisture content and biomass quality, we estimate the energy derived per unit of calorific material and relate this to traditional fuel comparators.

[5] A full Life Cycle Analysis may also account for GHG emissions associated with the manufacture and maintenance of plant, equipment, machinery and vehicles.

4.2.1.1 Biomass Data

We use yield data from the RSPB Nature Reserve at Ham Wall, Somerset (as presented in **Mills, 2016**) with 6.20 t ha^{-1}, 7.00 t ha^{-1} and 8.76 t ha^{-1} recorded in 1-year-old, 3-year-old and 15-year-old reedbeds, respectively (**Table 4.2**). These provide potential yields considered relevant to a newly established, young and older paludiculture system under temperate conditions. It is assumed that the harvested biomass has 15% moisture content, which is typical of winter harvested material; lower moisture contents for *P. australis* are possible (**Köbbing et al., 2013**; **Wichtmann et al., 2016**) so the values used here can be considered slightly conservative.

4.2.1.2 Harvesting Machinery and Diesel Consumption

The fuel used in establishment and harvesting operations needs to be represented in the carbon balance of biomass capture. We assume that establishment is carried out by manual collection and spreading of rhizomes without machinery and do not include GHG costs of establishment in these calculations. A large number of variables associated with the use and selection of different machinery affects the efficiency of harvesting operations and this has implications for the calculation of CO_2-equivalents through fuel use during harvesting operations. We include an estimated fuel cost for delivery and recovery of the harvesting machinery (=5l diesel; similar to **Lindegaard, 2013**), and we assume a cut and collect system for biomass harvesting.

Mills (2016) presents a comparison of different types of harvesting machines and a decision-making tree to aid selection of suitable machinery for prevailing site conditions. Depending on the size of the area to be harvested, number of access points and the sensitivity of the substrate, with a decision tree leading to the Softrak 120hp or Pisten Bully Greentech 300 harvesting machines (**Diagram 1, p30 in Mills, 2016**). **Mills (2016)** report the Softrak 120hp to have a fuel efficiency of 3.6 l h^{-1}, 14–18 l ha^{-1} (producing 42–54 kg CO_2 ha^{-1}) and the Pisten Bully Greentech 300 to have a fuel efficiency of 12.5 l h^{-1}, 75–100 l ha^{-1} (producing 225–300 kg CO_2 ha^{-1}, but can do 6–8 ha per day compared to 4–5 ha per day for the Softrak 120hp). A fuel efficiency of 15 l ha^{-1} was also reported from trials of the Softrak at Ham Wall Nature Reserve in an LCA (**Lindegaard, 2013**).

We use fuel efficiencies of 14, 16 and 18 l ha^{-1} for 1-year-old, 3-year-old and 15-year-old reedbeds, respectively, given the expectation that density of reedbeds and effort to harvest increase with age. We also add a fuel cost to account for haulage of harvested biomass, with an average fuel efficiency of 15 l ha^{-1} as reported by **Lindegaard (2013)** for the Softrak machine.

4.2.1.3 Transport and Processing

Other activities past the 'farm-gate' have emissions attached to the production of bioenergy from the harvested biomass. There may be additional fuel used in subsequent transport of material, energy used for drying, and energy used for pelleting or briquetting the biomass. The use of loose material is appropriate if the bioenergy plant is close to the harvest site but it is less economical to transport over distance (**Mills, 2016**). It is suggested that the use of reed biomass for combustion is not appropriate beyond a local scale and that transport of biomass should not exceed 50 km (**Lital et al., 2012** cited in **Köbbing et al., 2013**). The size of the harvested material has an impact on bulk density of biomass and hence the volume to be transported. The low density (20–60 kg m^{-3}) of reed biomass means that it occupies more space in storage and transport than other energy fuels. For example, the Softrak 120hp can double chop reed biomass and this bulk density is 39 kg m^{-3}, whereas the Pisten Bully precision chops the reed biomass and this bulk density is 54 kg m^{-3} (**Mills, 2016**). Briquettes and pellets can be produced to reduce volume of biomass and increase ease of transport;

they may also be more suitable for particular biomass burners. However, additional energy is required to process the loose biomass under high pressure into these forms, with more power being required for production of pellets (e.g. 152.41 kWh t^{-1}; **Röder et al., 2015**). **Mills (2016)** state that briquetting uses about 25% less power for the same calorific value. For the current example we assume that biomass is used locally, attaching a fuel cost for transportation by tractor and trailer with 30 m^3 volume per load, 10 miles roundtrip per load, and 1.7 miles l^{-1} fuel efficiency (**Lindegaard, 2013**).

Adding all fuel use from harvesting and transport, gives emissions of 0.187, 0.203 and 0.232 kg CO_2 ha^{-1} for the 1-year-old, 3-year-old and 15-year-old reedbeds, respectively (**Table 4.2**). Future calculations would benefit by including site-specific and system-specific information in order to refine the estimate of GHG emissions.

4.2.1.4 Conversion Efficiencies and Realised Energy Yields

We assume that loose chopped reed material is directly combusted. The calorific or heating values for *P. australis* biomass in the literature generally range from 15–17.7 MJ kg^{-1}. Given the assumed 15% moisture content, we use a Lower Heating Value (LHV) of 15 MJ kg^{-1}; this reduction in potential energy is similar to the LHV used by **Robertson et al. (2017)** given a 20% moisture content for *Miscanthus*. Winter harvested reed biomass is well suited for combustion in state-of the-art furnaces designed for materials such as straw or *Miscanthus* (**Wichmann, 2017**). In combined heat and power (CHP) plants the efficiency of biomass to energy can be much higher than large-scale power plants; as **Robertson et al. (2017)** used for comparison, we adopt a conversion efficiency of 70%. This gives realised energy yields of 65100, 73500 and 91980 MJ ha^{-1} for the 1-year-old, 3-year-old and 15-year-old reedbeds, respectively (**Table 4.2**).

Table 4.2 *Variables for example of P. australis stands of different ages at RSPB Ham Wall Nature Reserve and calculate of CO_2 mitigation potential*

Variable	Stand Age			Source/notes
	1-year	3-year	15-year	
ield/Biomass ($t\ ha^{-1}$)	6.20	7.0	8.76	Taken from **Mills (2016)**.
Estimated harvest volume ($m^3\ ha^{-1}$)	124	140	175	Assumes 50 kg m^{-3} density.
Number of roundtrips to transport biomass to plant (Loads ha^{-1})	4.13	4.67	5.84	Assumes 30 m^3 trailer volume.
Emission during harvesting & transport from fuel used ($t\ CO_2e\ ha^{-1}$)	0.187	0.203	0.232	Based on emission of 3205.55 g CO_2e per litre of 100% mineral diesel. [**UK Government GHG Conversion Factors for Company Reporting, 2019**].
Realised energy yield ($MJ\ ha^{-1}$)	65100	73500	91980	Assumes CHP at 70% efficiency.
Estimated CO_2 offset-I ($t\ CO_2e\ ha^{-1}$)	4.59	5.18	6.48	Based on average carbon intensity of UK Electricity Factor 2019 (253.58 g CO_2 kWh^{-1}) [**UK Government GHG Conversion Factors for Company Reporting, 2019].**
Estimated CO_2 offset-II ($t\ CO_2e\ ha^{-1}$)	3.71	4.18	5.24	Based on Natural Gas (0.20492kg CO_2e per kWh). [**UK Government GHG Conversion Factors for Company Reporting, 2019**].

4.2.1.5 Potential CO_2 Savings

We can derive carbon intensity for these scenarios and equate them to traditional fuel comparators. Using the energy intensities for two comparators, UK Electricity Factor in 2019 and typical Natural Gas, and taking into account emissions above, we estimate potential savings of 3.71 – 4.59, 4.18–5.18, and 5.24–6.48 t CO_2 ha^{-1}, for the 1-year-old, 3-year-old and 15-year-old reedbeds, respectively (i.e. estimated offset minus emissions from fuel). **Kuhlman et al. (2013)** reached slightly higher values suggesting that if combustion of reeds substituted natural gas (heating or electricity), that each gigajoule of energy from reed biomass would save 62 kg of CO_2 (and 16.4 t CO_2 ha^{-1} after accounting for the energy spent on harvesting and transporting the biomass), but they also assumed a higher heating value for the biomass.

It is worth noting that the magnitude of potential emissions reductions obtained from these estimates are at least as large as the estimated rate of CO_2 sequestration by a near-natural fen (Tier 2 emission factor -5.5 t CO_2 ha^{-1} yr^{-1}; **Evans et al., 2017**), while re-wetted fens are considered to still act as marginal net CO_2 sources (Tier 2 emission factor +0.7 t CO_2 ha^{-1} yr^{-1}). A key uncertainty for the overall climate mitigation potential of bioenergy-based paludiculture on lowland peat is the extent to which regular biomass harvesting is compatible with carbon accumulation into the peat (as suggested by the study of **Gunther et al., 2015**) or alternatively whether this biomass removal effectively reduces or halts peat formation. Given that peat formation relies on the accumulation and incomplete decomposition of plant biomass, it seems likely that regular removal of this biomass will negatively impact on CO_2 sequestration into peat biomass, but more measurements are needed to confirm this. Similarly, there is a need to quantify relative emissions of CH_4 and N_2O from bioenergy crops compared to a natural reference level. Provided that CH_4 and N_2O emissions are not higher than natural reference levels, however, our analysis suggests that management of re-wetted fens for bioenergy as part of a high water table paludiculture system could achieve greater net climate mitigation than restoration to natural conditions alone. Either option will, however, generate a large net climate mitigation benefit when compared to a counterfactual of conventional drainage-based agriculture.

4.3 Peat Subsidence

Land subsidence resulting from long-term drainage of peat is a widespread problem in many regions including the UK, Netherlands, Germany and Southeast Asia, due to a combination of peat compaction and oxidation (e.g. **Hooijer et al., 2012**). Subsidence can cause damage to buildings and linear infrastructure such as roads, increased requirements for expensive pumped drainage, and ultimately to the degradation and loss of land from agricultural production; these issues are described in detail in the WP2 report (**Page et al., 2020**).

In the context of paludiculture, it would be expected that raising water levels in former agricultural land should slow or halt oxidative peat loss, and similarly reduce compaction. It is also possible that currently compacted peat may to some extent 'rebound', as the refilling of pore spaces with water increases peat buoyancy and raises the ground surface. This effect has been observed as a short-term response to rain events in agriculturally drained peatlands using peat surface elevation monitoring systems (**CEH, unpublished data**) and as a longer-term response to rewetting in lowland raised bogs. On the other hand, rewetting of degraded peat or mineral soils could lead to adverse outcomes such as slumping, and further work may be needed to establish long-term rewetting impacts on soil structure. In the best case, if paludiculture management leads to net CO_2 sequestration after accounting for biomass removal (i.e. the re-initiation of peat formation) this should lead to a long-term increase in peat elevation, although realistically this is likely to be small (i.e. a few mm per year).

The reduction of oxidative peat subsidence is nevertheless important because – in addition to the reduction in CO_2 emissions – it also implies a reduced loss of soil. This will help to maintain agricultural productivity over the long-term, whereas conventional drainage-based agriculture will inevitably lead to soil depletion and a resulting loss of productivity. Since subsidence and CO_2 emissions are associated, this issue is considered alongside the assessment of GHG mitigation potential of paludiculture.

4.3.1 Subsidence Calculator

Given the links between oxidation, CO_2 emission and subsidence, if CO_2 emissions can be calculated according to the relationships shown in **Figure 4.1**, this can also be used to predict rates of oxidative subsidence. Furthermore, most studies of the relationship between subsidence and CO_2 emission (including those used to calculate emission factors for the IPCC Wetland Supplement) have assumed a constant ratio of oxidative subsidence to compaction, which for temperate peatlands is typically 1:1. If a constant ratio is assumed, then the predicted CO_2 loss can also be used to estimate total subsidence.

Table 4.3 shows examples of the implemented subsidence model for the sites and paludiculture/conventional crop pairs shown in **Table 4.1**. The calculator takes the field-level C balance estimate derived from **Table 4.1**, and requires additional user data on the carbon content and bulk density of the soil. This simple model only calculates the extent to which previous rates of subsidence are reduced by a transition to paludiculture management, with a 'best case' of zero subsidence in paludiculture systems that are no longer net CO_2 sources. We have not included estimates of the magnitude of surface elevation 'rebound' that could occur as a result of rewetting, although it may be possible to add this at a later stage, once we have more data on the relationship between peat surface movement and water table depth change. These data are being collected by CEH at a number of sites using a time-lapse camera-based measurement system. Similarly, if some paludiculture systems lead to the re-initiation of active peat formation this could be included in the model, leading to an additional net benefit.

Table 4.3 *Subsidence Calculator: Estimated oxidative and total subsidence for a set of theoretical example sites as shown in* ***Table 4.1****. Bold data are user specified, italic data are taken from the emissions calculator; other data are calculated*

CROP	SITE PROPERTIES			SUBSIDENCE		
	C content %	Bulk density g cm^{-3}	C balance g C m^{-2} yr^{-1}	Oxidative cm yr^{-1}	Total cm yr^{-1}	Mitigation cm yr^{-1}
Paludiculture crop 1			*-140*	0.00	0.00	-0.33
Conventional crop 1	**40**	**0.3**	*201*	0.17	0.33	
Paludiculture crop 2			*-61*	0.00	0.00	-0.54
Conventional crop 2	**35**	**0.4**	*379*	0.27	0.54	
Paludiculture crop 3			*25*	0.02	0.04	-0.95
Conventional crop 3	**45**	**0.25**	*557*	0.50	0.99	
Paludiculture crop 4			*112*	0.11	0.22	-1.25
Conventional crop 4	**50**	**0.2**	*736*	0.74	1.47	

The subsidence calculator has the potential to be used to project future peat surface elevation changes under different land-management options (including higher water level management of conventional arable and horticulture, as well as paludiculture), which have the potential to be linked to other environmental outcomes. For example, the combination of subsidence mitigation and changes in average water table depth could be used to predict changes in future energy costs of pumped drainage based on changes in the relative height to which water will need to be pumped (see **Section 4.4**). It could also be used to estimate changes in potential flood water storage capacity (see **Section 5**) and risks to infrastructure (**Page et al., 2020**).

4.4 Water and Energy Use

Subsidence due to both oxidation and compaction also contribute to the challenges of draining lowland peat. In many drained lowland peat areas, subsidence has caused the land surface to fall below the level of the rivers, which still flow close to their historical levels, but are now embanked, channelised and intensively managed to mitigate river flooding risks (see **Section 5**). Across large areas of the Fens, the ground surface is now below sea-level. This has required a shift from gravity drainage to pumped drainage, incurring significant financial and energy costs, and contributing indirectly to GHG emissions from energy generation.

Arable and horticulture production systems in the Fens use sub-surface drainage pipes and ditches to achieve suitable field conditions for crop growth and management. Water can only be drained out of fields under the influence of gravity and this requires ditch water levels to be below the field water table. Ditch water levels are lowered by the use of pumps to move water up into rivers (against the influence of gravity) and discharge it from agricultural land. A transition to paludiculture could reduce the energy required for pumped drainage via several mechanisms. Firstly, as paludiculture systems can be managed for shallower water tables than arable systems and cope with at least short term inundation, a larger proportion of winter inputs could be stored in the system. Similarly, by reducing or reversing subsidence, a continued long-term increase in the vertical elevation of the field surface will increase soil volume available to store winter inputs. Increased winter storage will reduce the volume of water needing to be discharged by pumping. Finally, as paludiculture systems have shallower water tables, they are likely to lose more water via evapotranspiration, so the overall requirement for pumped drainage may be smaller.

4.4.1 Water Fluxes under Conventional and Paludiculture Management

To estimate the water pumping requirements for paludiculture versus conventional agricultural management, water flux data from the final report on project SP1210 were used (**Evans *et al.*, 2017**). Data were obtained for six sites which were treated as two groups due to regional variations in precipitation. The first group includes three sites from the East Anglian Fens: one conservation-managed site, Wicken Sedge Fen (EF-LN) and a pair of arable/horticulture sites on deep peat (Rosedene, EF-DA) and shallow peat (Redmere, EF-SA). The arable sites were used to represent conventional agricultural management in the East Anglian Fens. Wicken Sedge Fen is a fragment of surviving fen under natural vegetation which is managed for a shallow water table, and offers the best current Fenland analogue for paludiculture. Updated water flux data for Wicken were obtained from **Peacock *et al*. (2019)**. The second group includes three raised bog sites from the Manchester Mosses: One rewetted site (Astley Moss, MM-RW), one extraction site (Little Woolden Moss, MM-

EX) and one arable site on deep peat (also at Little Woolden Moss, MM-DA). These sites are not subject to the same drainage pumping system found in East Anglia, but again they provide a comparison of water fluxes between extraction, conventional agriculture and high water table management of natural vegetation. These sites were included to improve the robustness of conclusions given limited data availability.

There was a consistent pattern of discharge for all six sites during the winter months (October-March) when precipitation exceeded evapotranspiration (ET). This pattern was very pronounced at Wicken and estimated annual discharge for the site calculated in **Peacock *et al*. (2019)** was 192 mm. This discharge figure was substantially elevated as the result of unusually large winter inputs in November 2013, following a dry summer. Here we used the annual discharge estimated over the period December 2013 – November 2015 (inclusive), which was 154 mm and is likely to be more representative for the site. At Wicken, ET rapidly increased in the spring (January-May) and often exceeded ET from the arable sites during this period. This was presumably attributable to a combination of wetter conditions and high transpiration rates from the permanent tall fen vegetation. Relatively high ET during the crucial winter period is the cause of low discharge estimates at Wicken as greater losses through ET reduce the volume of water needing to be discharged. Among the arable sites in East Anglia, discharge was greater from Rosedene (237 mm) than Redmere (199 mm). These two sites have very similar inputs but ET was higher at Redmere than Rosedene in 33 of the 39 months for which data was available. This effect was more pronounced in summer but remained in winter and could be due to the network of shelterbelts and hedges at Rosedene reducing air movement relative to Redmere which is much more open and exposed. The winter discharge pattern remained strong on the arable sites but there were also some (relatively infrequent) summer discharge events. These were most likely the result of managed river inputs coinciding with heavy summer rainfall events and necessitating discharge to prevent crop damage.

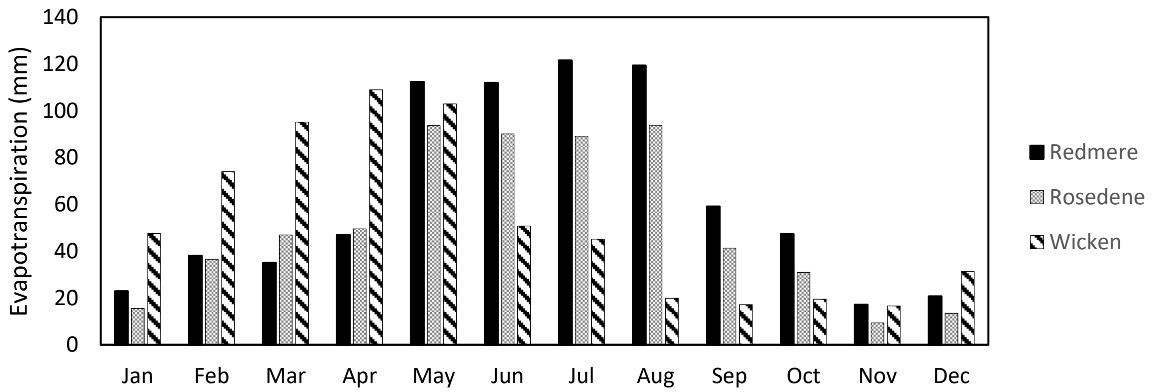


***Figure 4.2** Evapotranspiration from East Anglian fen peat sites for 2014 (Source: **Evans et al., 2017**)*

Because of these differences in ET, lateral discharge is highest at Rosedene, and lowest at Wicken. If we take the average of the two arable sites to be typical of conventional agriculture across the Fens, and Wicken to be indicative of the water balance that would occur under paludiculture, the data suggest that the pumping (and therefore energy) requirements to remove water from areas under paludiculture should be around 25% lower than those for land under arable/horticulture management. However, it is important to note that higher ET losses at Wicken (and by inference under future paludiculture crops) could – depending on the time of year when it occurs – have implications for overall water availability, particularly in water-scarce areas such as East Anglia.

For the Manchester Mosses sites, precipitation accounted for all water inputs and was similar across the sites, so discharge was approximately inversely proportional to ET. The rewetted site at Astley Moss was fully bunded and effectively hydrologically sealed, so the water table was continuously above the peat surface during the measurement period. As at Wicken Fen, the combination of high water levels and natural vegetation led to the highest rates of ET and the lowest discharge (280 mm). ET was slightly lower at the arable site possibly as a result of the absence of exposed standing water and reduced winter transpiration (less vegetation cover) and so the site had intermediate discharge rates (349 mm). The deep drained bare peat at the extraction site had the lowest ET and hence the highest discharge rates (494 mm). This site probably represents an extreme case with very dry surface peat and near-zero transpiration rates. The Manchester Mosses sites all showed the high winter discharge pattern but showed slightly more frequent discharge in summer months. This is likely due to higher rainfall in the region relative to East Anglia. The overall pattern of discharge between the Manchester Mosses sites supports the pattern observed at East Anglia, with discharge 20% lower from the rewetted site compared with the arable site.

4.4.2 Energy use and Associated GHG Emissions due to Pumping

In East Anglia, the Internal Drainage Boards (IDBs) supervise all matters related to water level management within defined drainage districts. The IDBs are responsible for management of ordinary watercourses and surface water flooding in their district. This excludes rivers and river flooding which are the responsibility of the Environment Agency. The boards operate the pumps, inlets and weirs used to manage water levels within districts. Data on the economic and environmental impacts of drainage pumping were obtained from the publicly available records of the Ely Group of Internal Drainage Boards. Data were collected for the pumping stations serving the areas including EF-SA (Lark) and EF-DA (Catsholme). Pumping data were used to estimate the mean number of hours pumped per annum for each station. The Lark dataset included the calendar years 2005-2018. The Catsholme dataset included twelve separate, continuous twelve month periods from 2005-2018. Pump technical data were used to derive power use and water pumped per hectare served, and the energy costs involved in pumping 1 m^3 of water (**Table 4.4**). Electricity costs were estimated using data for the Burnt Fen IDB for the period March 2009 – March 2019. Average annual costs were £20,176, giving costs of £2.91 ha^{-1}. Lark station drains the majority of the Burnt Fen district so we assume here that Lark station would be responsible for the proportion of costs equal to the area of the district it serves. This gives an estimate of costs per unit electrical energy used of £0.069 kWh^{-1}; this is only slightly higher than known average wholesale electrical energy prices (£0.055 kWh^{-1}).

Table 4.4 *Power use and volume pumped for pumping stations serving the arable sites*

	Unit	Lark station	Catsholme station
Site served by station		Redmere (EF-SA)	Rosedene (EF-DA)
Mean annual pump operating time	hrs yr^{-1}	2558	2165
Single pump power requirement	kW	76.5	67.5
Total annual pump power use	kWh yr^{-1}	195702	146166
Area served	ha	4623	3000
Annual power use per hectare	kWh ha^{-1} yr^{-1}	42.3	48.7
Single pump water capacity	m^3 s^{-1}	0.866	1.07
Annual volume of water pumped	m^3 yr^{-1}	7975444	8341185
Water volume pumped per hectare	m^3 ha^{-1} yr^{-1}	1725	2780
Power use per m^3 water pumped	kWh m^{-3}	0.0245	0.0175

To remove the effects of site-specific infrastructure on our estimates, we used the average value for power use per cubic metre of water pumped from the two stations (0.021 kWh m^{-3}) in our calculations. As all agricultural peatland lies below river levels, we assumed that all discharge from a site must eventually be pumped for removal from districts. A value of 0.525 kg CO_2e kWh^{-1} was used to derive upstream GHG emissions associated with high voltage UK grid electricity production (**Wernet *et al.*, 2016**).

Estimates for energy use associated with drainage pumping for the three Fenland sites considered above are shown in **Table 4.5**. Wicken Fen (taken as illustrative of paludiculture without the unusually large November 2013 inputs) had the lowest energy requirements and associated GHG emissions from pumping due to its higher winter/spring ET. This was especially pronounced when winter inputs were not excessive. The relatively sheltered Rosedene had the highest energy costs and emissions. Based on these figures we estimate that paludiculture would result in a 35% reduction in pumping energy use following conversion from arable production on deep peat, and a 23% reduction following conversion from arable production on shallow peat. However, both the financial costs and GHG emissions associated with pumping are minor under all forms of management. For example, the energy-related GHG emissions are several orders of magnitude smaller than the estimated direct CO_2 emissions resulting from peat oxidation at the arable sites.

Note that these calculations do not consider the other financial costs of pumped drainage including infrastructure and operation. However, as some pumped drainage would still be needed even in paludiculture systems as a result of the historic lowering of the land surface, these costs are unlikely to change even if pump usage declines.

Table 4.5 *Estimated economic and environmental impacts of drainage pumping for fen peat under arable and paludiculture management*

Scenario	Discharge (mm yr^{-1})	Power use (kWh ha^{-1} yr^{-1})	Cost (£ ha^{-1} yr^{-1})	GHG emissions (kg CO_2e ha^{-1} yr^{-1})
Arable on deep fen peat	237	49.84	3.44	26.2
Arable on shallow fen peat	199	41.85	2.89	22.0
Paludiculture on deep fen peat	154	32.39	2.23	17.0

4.4.3 Energy use and Associated GHG Emissions due to Surface Irrigation

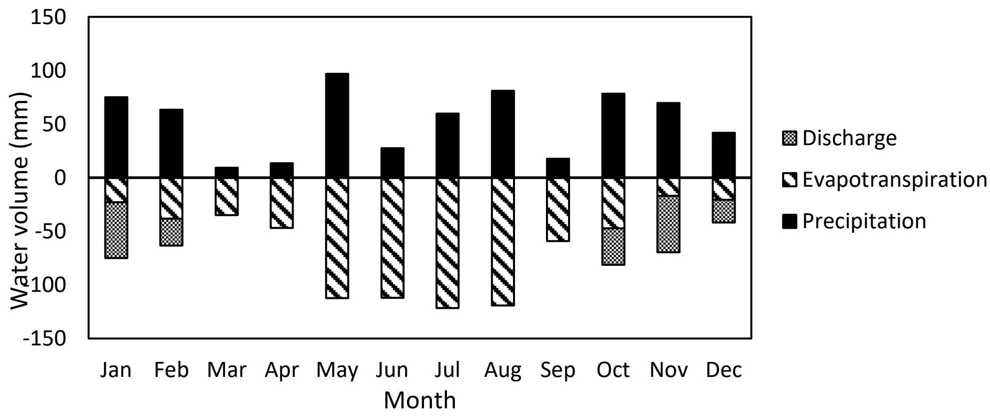
Sprinkler irrigation represents an additional source of energy use, notably on sites managed for horticulture, potatoes and cereal production without subsurface irrigation. Applying this scenario to the Redmere site under a lettuce crop, we estimate that irrigation using water that is pumped up from ditches to irrigate field surfaces would require an estimated energy use of 171 kWh ha^{-1} yr^{-1} (based on 90 mm yr^{-1} and 0.18 kWh m^{-3}; **Plapally & Lienhard, 2012**). This is several times higher than the energy cost of pumped drainage on a per-hectare basis, and thus significant where it occurs. In terms of GHG emissions, this equates to an additional 90 kg CO_2e ha^{-1} yr^{-1}. In paludiculture systems that do not require surface irrigation, this would represent an additional (although small) GHG emissions reduction resulting from conversion to paludiculture. However, there may also be situations where surface irrigation is required, potentially at higher rates than in conventional systems (for example where high near-surface soil moisture levels are required at sites that cannot be hydrologically isolated from adjacent farmland, as discussed earlier) in which case these emissions should be considered. The use of on-site renewable sources for energy generation would however negate these emissions.

4.4.4 Practical Considerations for Water Management

Whilst paludiculture adoption would likely result in reduced pumping and irrigation requirements, drainage pumping would still remain a vital part of peatland management in areas where land subsidence has already occurred, such as the East Anglian Fens. During periods when inputs exceed ET, pumping of discharge up into rivers will be necessary to manage surface water flooding risks; this is illustrated by the seasonal pattern of estimated discharge and pumping activity at Redmere Farm shown in **Figure 4.3**. This conclusion is also supported by the excellent agreement between the site discharge estimates for the period (**Table 4.5**) and the estimates of water volume pumped derived from pumping data for the same period (Rosedene = 219 mm; Redmere = 181 mm). We estimate there would be a substantial proportional reduction (~25%) in pumping requirements under management for paludiculture in comparison to arable production. The scale of any benefits is likely to be variable and will depend on factors including site physical characteristics, location in drainage district, specific energy usage of the pumping stations serving a site and elevation of the land surface relative to the river.

Whilst any benefit is valuable, it is important that the scale of the benefits is contextualised. The largest estimate of reductions in upstream GHG emissions from reduced drainage pumping following paludiculture adoption was -0.008 t CO_2e ha^{-1} yr^{-1} at Rosedene Farm. Where surface irrigation ceased with the adoption of paludiculture, this would give a reduction in upstream GHG emissions of 0.09 t CO_2e ha^{-1} yr^{-1}, which is ten times greater than that from the reduction in pumping. Nevertheless, these emissions reductions are still several orders of magnitude smaller than the estimated annual direct CO_2 emissions from oxidising peat under drainage-based agriculture, which are in the region of 25-28 t CO_2 ha^{-1} yr^{-1} (**Evans et al., 2017; Taft *et al.*, 2017**). Therefore, it is clear that reduced direct GHG emissions from the peat following rewetting would remain the primary climate abatement benefit of a transition to paludiculture.

a)



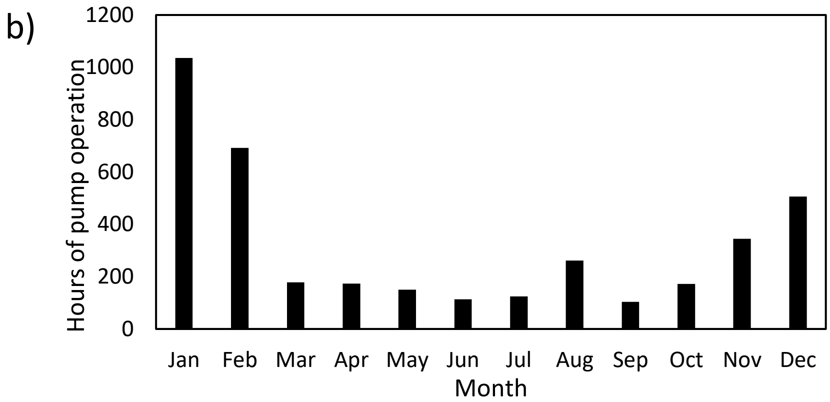


Figure 4.3 *a) Water fluxes for Redmere Farm (EF-SA) in 2014 showing how the balance of precipitation and evapotranspiration must be lost as discharge (notably in winter; Source**: Evans et al., 2017**). b) Pump operation at Lark station for 2014 showing how pumping operations are most intensive in winter periods*

5 POTENTIAL CO-BENEFITS OF PALUDICULTURE

Ecosystems play an important and wide-ranging role in the delivery of co-benefits to human populations (**Millennium Ecosystem Assessment, 2005**; **Smith et al., 2012**). These 'ecosystem services' include the supply of resources, such as food, water, timber, fuel and fibre; various regulatory functions, embracing climate regulation, air and water purification, carbon sequestration and nutrient recycling; and a range of less tangible aesthetic, cultural, educational and other non-material benefits. This section considers the potential societal benefits that could be delivered if drained lowland peatlands were to be managed for paludiculture.

5.1 Ecosystem Service Benefits

The development of ecosystem service accounting methodologies (**e.g. Costanza et al., 1997**; **Daily, 1997**) has enabled a better understanding of the true costs of the services provided by intact ecosystems and the value of services foregone if an ecosystem is degraded or lost. For wetlands, the value of global ecosystem services (environmental, social, economic) has been estimated at US$14.9 billion (£10.98 billion[3]) (based on **Costanza et al., 1997**), yet wetland 'development' activities often fail to recognise either the financial or non-valorised consequences associated with the loss of these benefits.

In many cases, the delivery of ecosystem co-benefits depends on the maintenance of ecosystem functions, i.e. ecological processes. The conservation, or restoration, of wetland biodiversity (habitats and species), for example, maintains or reinstates nutrient cycling and productivity, although the exact mechanisms by which this is achieved are often only partially understood. As a result, the linkage of scientific process understanding with effective economic valuation and policy can present difficulties (**Evans et al., 2014**).

In the UK, peatlands can be broadly classified as upland or lowland, fen or bog. To date, most of the literature on peatland ecosystem service provision has focused on upland systems (predominantly blanket bogs), rather than lowland systems, i.e. lowland fens and raised bogs (e.g. **Bonn et al., 2009a, 2014**; **Martin-Otega et al., 2014**; **Moxey & Moran, 2014**). And, in this regard, it is worth noting that not all peatlands provide the same services. For example, while degraded upland peatlands may function as sources of flood water, lowland peatlands more often act as recipients of flood water and can in some cases offer flood storage potential (**Bonn et al., 2009b**).

The range of ecosystem services provided by fen peatlands in Europe and North America have been summarised by **Lamers et al. (2015)**, along with an indication of the modifications that arise following land use change (**Table 5.1**).

Table 5.1 *An Overview of Fen Ecosystem Services, and their Modifications as a Result of Land-use Change (Source:* ***Lamers et al., 2015****)*

Original Services	Modifications
Provisioning Services	
Natural area	Agricultural area, urban area
Freshwater supply	Loss of water, bad water quality
Food (including fish, waterfowl) production	Crop production
Crafts & building supplies (reed, rushes)	Loss of this material
Peat for fuel, horticulture, pollutant sorption, medication (sorbent)	Peat removed, no new production
Supporting Services	
Biodiversity (landscape, ecosystem, and species scale)	Biodiversity loss at all scales
Peat soil formation	Peat loss, land loss, land subsidence
Peatland biogeochemistry	Strongly altered biogeochemical cycling of elements and water
Regulating Services	
Surface water and groundwater regulation (also water retention and storage with respect to climate change)	Hydrological disturbances, water losses, desiccation, acidification
Nutrient sink and water purification	Nutrient source, no purification
Carbon sink for atmospheric CO_2	Carbon source
Flood protection	Increased flooding risks due to loss of wetland buffer, land subsidence, land loss
Cultural Services	
Recreation in nature (also provisioning health)	Loss of recreational and health value
Mosaic, cultural-historic landscape	Uniformity, loss of cultural-historic values
Nature and environmental education	Education about peatland degradation
Archaeological and palaeontological record in peat	Loss of this record
Inspirational values (cultural communities, art, etc.)	Loss of these values

In the UK, many of the ecosystem benefits identified for minerotrophic fen peatlands (summarised in **Table 5.1**) will apply equally to lowland ombrotrophic peatlands (raised bogs); these include, for example, climate regulation through carbon sequestration and storage, the provision of habitats for wildlife, preservation of archaeological and palaeo-environmental records within the peat, recreation opportunities, cultural heritage and a range of other non-material benefits (**Bonn et al. 2009a**,**b**; https://www.humberheadpeatlands.org.uk/index.php?page=ourWork).

Ecosystem services not only vary with topographic setting, but also according to peatland size and condition, and to the perceptions of different stakeholders. This is illustrated (**Table 5.2**) by the results of a stakeholder group ranking of peatland ecosystem services for three ombrotrophic peatland locations (two upland, one lowland) in the UK (**Bonn et al., 2009b**).

***Table 5.2** Ranking (highest to lowest) of peatland ecosystem services by local stakeholder groups for three locations in the UK (**Bonn et al., 2009b**)*

Migneint (upland peatland)	Peak District (upland peatland)	Thorne & Hatfield Moors (lowland peatland)
Biodiversity	Freshwater provision	Carbon storage
Carbon storage	Carbon storage	Wildlife watching
Freshwater provision	Water quality	Landscape aesthetics and recreation
Landscape	Fire risk mitigation	Flooding
Water quality	Recreation	
Recreation	Education	
Pollination	Aesthetics	
Hydro-power	Biodiversity	
Fire risk mitigation		
Timber		
Wind power		
Peat extraction		

When peatlands are utilised to increase their provisioning services (e.g. through drainage to enable cultivation, forestry, livestock grazing, peat extraction) there will be trade-offs with other co-benefits (**Howe et al., 2014**). Whilst drainage and modification of the original vegetation cover may increase direct provisioning services (e.g. in terms of food production, employment and business revenues) (**NFU, 2019**) there will be an inevitable and concomitant reduction in other services. For UK lowland peatlands, these will likely include a loss of carbon storage function and an increase in greenhouse gas emissions (**Evans et al., 2017**), gradual loss of the peat soil (peat subsidence) (**Hutchinson, 1980**), an altered microclimate (**Acreman et al., 2011**), reduced biodiversity support (**Acreman et al., 2011**), degradation and loss of peat archaeological and palaeo-archives (**Godwin, 1981**), and reduced provision or loss of recreational and other cultural services (**Joosten & Clarke, 2002**). In other words, there are trade-offs between provisioning ecosystem services, which may have a direct and present economic value, and regulating and other services, which decline as the integrity of peatland ecological processes are compromised by the acquisition of the provisioning services (e.g. **Bennett et al., 2009**; **Mouchet et al., 2014**).

There have only been a limited number of studies specific to the range of ecosystem services provided by lowland peatlands in the UK, and the potential benefits or trade-offs of different management regimes. **Acreman et al. (2011)** reviewed the ecosystem services provided by the peat-dominated Somerset Levels and Moors. These include food provision, mainly through grazing of dairy and beef cattle, water supply, willow withies for basket making, carbon sequestration, flood water storage, regulation of microclimate, and a range of cultural services, including recreation and archaeology. Some services were independent of other co-benefits, e.g. there was little relationship between fishing and archaeology. Others were synergistic and reinforcing, e.g. the presence of archaeological artefacts increased educational services whilst the diversity of bird species increased tourism. But some services were deemed to be potentially in conflict, indicating that management decisions could necessitate trade-offs. For example, raising water levels would reduce peat CO_2 emissions and increase biodiversity interest, but would reduce the potential for flood water storage and increase the emissions of CH_4. Higher water levels could also increase the habitat for disease vectors and nuisance insects (biting flies etc.). In an earlier study of the Somerset Levels, **Lloyd (2006)** exemplified some of these trade-offs: local environmental groups wanted to increase the managed flooding time to benefit bird populations and plant communities, whilst simultaneously conserving soil carbon and minimising GHG emissions; local farmers, on the other hand, wanted the flooding regime reduced so that the meadow crop was predominantly grasses rather than reeds and they could extend their cattle grazing period.

The rehabilitation and restoration of degraded lowland peatlands, involving rewetting and re-establishment of wetland plant communities, may offer opportunities to re-establish or increase the range of beneficial ecosystem co-benefits. **Peh et al. (2014)** evaluated the changes in the delivery of ecosystem services resulting from the conversion of drained, intensively farmed arable land to wetland at Wicken Fen NNR in Cambridgeshire. Their results suggested that restoration led to a net gain to society at an estimated benefit of US$199 (£147[3]) ha^{-1} yr^{-1}, for a one-off investment in restoration of US$2,320 (£1,710[3]) ha^{-1} (this GHG emission cost was based on the US Government CO_2 value of US$22.78 (£16.79[3]) t CO_2, adjusted to 2011). While restoration resulted in loss of arable production, there were gains in cultural and regulating services, including nature-based recreation, grazing, flood protection, and a reduction in greenhouse gas emissions, as well as lower management costs. The ecosystem service beneficiaries also changed – under arable production, they were local arable farmers, but, under restoration, the beneficiaries shifted to graziers, countryside users from towns and villages, and the wider community.

Studies on the conversion (rewetting) of degraded, drained peatlands to various forms of paludiculture (the cultivation of biomass on wet peatlands) in Europe have shown the potential to reduce greenhouse gas emissions, to restore carbon and nitrogen retention, to restore habitats for rare and threatened species, to revitalise traditional types of land use and combine these with new ones, and to produce biomass with a versatile range of uses (**Wichtmann & Tanneberger, 2011**). Most of this work has been undertaken at sites in NW Europe (Germany, in particular), although with a few exceptions (e.g. northern Italy, **Giannini et al., 2017**).

Wichmann (2012) assessed the changes in ecosystem service provision at a 4 ha peatland site in NW Germany. A formerly drained and fertilised 'bog grassland' was rewetted to support *Sphagnum* farming. Compared to the former bog grassland, there was an improvement in provisioning services (through production of renewable *Sphagnum* biomass for growing media), regulating services (avoided greenhouse gas emissions, estimated at 10 t CO_2e ha^{-1} yr^{-1} compared to the bog grassland, and an increase in area of rare wetland habitat) and cultural services (preservation of the peat palaeo-archive).

Future scenarios for the management of lowland peatlands in England and Wales to reduce greenhouse gas emissions will require a range of different water and land management practices, that will likely result in various trade-offs in ecosystem services. In the case of lowland fen peatlands these management scenarios could span from full restoration (i.e. conversion of arable land back to rewetted fen peatland, as is happening at Great Fen and at Wicken Fen), through to high water table cultivation (i.e. paludiculture, requiring modification of current crops and land management practices, as is starting to be explored through the Water Works Project at Great Fen - https://www.wildlifebcn.org/news/water-works), and modification of current agronomic practices (e.g. maintaining a high water table during the winter months when there is less requirement to access the land with machinery and there are no economic crops in the field). Taking a range of future scenarios, alongside a Business as Usual (BAU) comparison, it is possible to indicate the likely direction of change in ecosystem services using the Fens of eastern England as an exemplar location (**Table 5.3**).

Table 5.3 *Change in the delivery of selected ecosystem services across a range of possible scenarios for agricultural fen peatlands in eastern England under current climatic conditions: current practice (Business as Usual – BAU – intensive crop production), modified BAU (e.g. high winter water table), high water table agriculture (e.g. paludiculture) requiring a change in land management practices and crops, and restoration to fen wetland. Information sources:* ***Peh et al. (2014)****,* ***Great Fen Project (n.d.)****,* ***Evans et al. (2017)****. Arrow directions and colours indicate likely trend in provision of ecosystem service (ES):*

	Land & water management scenarios			
Ecosystem co-benefits	BAU	Modified BAU (e.g. high winter WT)	High WT cultivation (high WT all year; modification of crops & land management)	Restoration to fen wetland
Food production	↔ / ↙ [a]	↗	↓	↓
Fibre/biomass production	↔	↔	↑	↔
Carbon storage	↓	↙	↔	↗
Climate benefit [b]	↓	↙	↗	↑
Flood storage/water retention	**UNKNOWN**	**UNKNOWN**	**UNKNOWN** [c]	↑
Water quality	↓	↙	↗ [d]	↑ [d]
Biodiversity	↙	↙	↔ [e]	↑
Recreation/tourism	↓	↓	↔	↑
Education	↓	↓	↔	↑
Landscape aesthetics	↔	↔	↗	↑

Notes:

a*: food production could decline as peat is lost*
b*: where climate benefit is assessed as a reduction in greenhouse gas emissions (green highlight)*
c*: see further discussion of flood storage under* ***section 5.1.3*** *below*
d*: this benefit assumes restoration of peatland function as a sink in the nutrient cycle but, at least initially, the introduction of higher water tables could lead to flushing of soil nutrients and organic carbon into adjacent waterways, as has also been observed at some re-wetted sites in the Netherlands (****Fenner et al., 2001****;* ***G. Erkens pers. comm.****;* ***Smolders et al., 2013****). Over time this would be off-set by biofiltration and uptake of nutrients by the vegetation and subsequent harvest (biomass off-take)*
e*: see further discussion of biodiversity under* ***sections 5.1.1*** *and* ***5.1.2*** *below*

Key:

Reduced provision	↓
Some reduction	↙
Neutral	↔
Some increase	↗
Increase	↑

N.B. Assessment of landscape aesthetics is subjective.

In terms of the range of ecosystem services that are examined here under current climatic conditions, the greatest benefits arise from full wetland restoration but with the trade-off of lost food and fibre production. 'Paludiculture' ranks second in terms of improved delivery of ecosystem services associated with climate mitigation and biodiversity, but again there is a trade-off with food production (assuming that paludiculture crops would be mainly grown for fibre or biomass); there may also be limited gains for the provision of cultural services, such as education, tourism and landscape aesthetics, and the potential dis-benefit of flushing of soil nutrients into waterways with higher water tables, although this may be transient. For the BAU and modified BAU scenarios, the primary focus on food production limits the provision of most other ecosystem services, in particular ecosystem carbon storage, biodiversity support, recreation/tourism and so on. It is important to note however, that, at present, farmers and landowners receive a direct economic return from food and fibre production, whereas there are few (or no) financial returns associated with the other ecosystem services. Thus, at a farm level there may be little economic incentive to sacrifice food and fibre production for possible environmental benefits, although the authors are not aware of any quantitative cost benefit analysis on this topic or any assessment of trade-offs among alternative scenarios. It is also worth noting that whilst there are some published studies to support the proposed direction of change for supporting and regulating services, there is much less available information on provisioning and cultural services and how these may change in response to a shift from drained to wet agriculture or to full wetland restoration. This represents a knowledge gap for lowland peatland ecosystems under different land management practices. In sum, paludiculture could offer a form of land use that supports provisioning services (fibre, biomass production) without greatly compromising regulatory services such as carbon storage, flood and climate regulation (**Wichtmann et al. 2016**). The impact on cultural services remains undefined.

The effects of future climatic change are likely to affect the delivery of ecosystem services under all scenarios portrayed in **Table 5.3**. The **UKCIP (2010)** climate projections for the East of England indicate trends towards warmer and drier summers by 2050 under a medium GHG emissions scenario. This may ultimately prove restrictive for restoration to fen wetland and possibly also for maintaining high summer water tables for paludiculture (i.e. owing to higher evapotranspiration rates and insufficient water supply). It could also result in larger releases of C as CO_2 from the surface of agricultural peatlands (BAU and modified BAU scenarios), and even from restoration fen and paludiculture if high summer water levels cannot be maintained. Sea level rise will likely impact on coastal wetlands, e.g. via salt-water flooding and saline intrusion. This could reduce or alter the nature or direction of ecosystem service provision in these locations, e.g. by bringing about unfavourable changes in water quality and altering the character of wetland biodiversity through replacement of freshwater species by those more typical of brackish or saline environments. The impact of a changing climate on other ecosystem services is less clear.

5.1.1 Direct Biodiversity Benefits from Paludiculture

When areas of former wetland are returned to a wetland or semi-wetland state these areas have the potential to encourage the return of wetland species, even when such rewetting is carried out with a view to growing monoculture crops capable of tolerating or thriving in such conditions. At the very least, areas of unused 'headland' (in the manner of conventional fields) can provide opportunities for wetland species to colonise both these wet marginal areas and the systems of water supply, while the area of the crop itself may provide at least temporary refuge for such species. This underpins the biodiversity benefits arising from Japanese *satoyama* landscapes, rather than through any active efforts on the part of the farmers to promote biodiversity through their agricultural activities.

It is nevertheless possible to adjust paludiculture farming techniques, even where they involve intensive monoculture production, in ways that offer the potential to enhance biodiversity benefits without significantly constraining agricultural practices or profitability. A small number of general principles, if applied, can help to maximise such biodiversity benefits within the agricultural production process:

- Maintenance of relatively high soil-water levels, except perhaps at harvesting and planting time, will tend to encourage increased wetland biodiversity;
- Harvesting in a rotational mosaic provides opportunities for wetland species to migrate between compartments as they are harvested;
- Choice of harvesting season will have more potential impact on some elements of biodiversity (*e.g.* birds, invertebrates) than on others and so should be considered carefully in light of both crop type and local biodiversity context;
- Identifying opportunities to expand crop type and product range arising spontaneously from increased wetland biodiversity *e.g*. sundew (*Drosera* spp.) for the high-value herbal medicine market (**Baranyai et al., 2016** and **Durechová et al., 2016**) when farming *Sphagnum* as the main crop.

Although currently only a few studies of actual paludiculture operations and the effects of these on biodiversity exist, it is possible to draw on evidence obtained from wetland management practices more generally in order to provide case studies with which to underpin these general principles. **Muster *et al.* (2015)** chose spider communities as a proxy for species richness within a *Sphagnum* farming operation. As the dominant predator within microfauna, spiders would broadly represent the development of lower trophic levels. Spider communities are also effective indicators of early successional habitat due to their high dispersal ability. In this study spider species richness increased from the initial set up over a three-year period on areas farmed for *Sphagnum*, with the authors recommending a harvesting mosaic to enhance the colonisation of recently harvested patches from mature ones. **Poulin *et al.* (2009)** investigated bittern populations relative to the return-time for reed cutting in natural reedbeds in France. They found that bittern populations grew when mowing was instigated every other year rather than annually. They also acknowledged, however, that this would impact on the reed quality demanded by the reed cutters, so suggested a mosaic approach to harvesting – leaving a minimum area of 20% unharvested in a patch to 80% harvested if annual cutting was required. **Hardman *et al.* (2012)** investigated Diptera and moth populations in three reedbed National Nature Reserves and found that species richness and invertebrate conservation value was maximised by “keeping the ecosystem dynamic, maintaining the transitions between different habitats and avoiding permanent long-term flooding.”

Wichtmann, Schröder & Joosten (2016) provide a valuable summary of several different studies looking at the relationship between paludiculture-style land management and biodiversity. Thus, **Görn & Fischer (2016)** summarise the relationships observed between different mowing practices and bird, butterfly, Orthoptera and ground beetle populations. They found that the various species groups responded differently to different management regimes. While ‘summer harvest’ and ‘moist meadow’ generally benefitted most groups, ‘winter harvest’ was positively detrimental to all except ground beetles, which benefitted maximally from such a regime. **Raabe & Manthey (2016)** found that areas where the water table had been raised to about 10 cm below the ground surface consistently had greater biodiversity than intensively managed meadows with low water tables. Within the re-wetted areas, newly-fallow areas in an early stage of colonisation had the greatest diversity of all, but there was an expectation that as succession progressed this diversity would decrease sharply as dense reedbeds developed. Summer mowing and winter harvest resulted in similar levels of biodiversity though the former was somewhat more favourable than the latter.

Horn, Sweers & Frase (2016) found that grazing by water buffalo (rather than traditional cow breeds) suppressed dense reedbeds and encouraged greater biodiversity.

Within the UK there is currently only a very small number of paludiculture projects to draw on, and little has currently been published from these that would shed light on the implications for biodiversity. The current list can be seen in **Table 5.4**, below.

Specific biodiversity benefits are likely to be influenced by crop type, and to this end a catalogue of potential crop species tailored to the UK environment has been drawn up. The list is derived from a global Database of Potential Paludiculture Plants (DPPP) assembled by Susanne Abel (University of Greifswald) to which certain species have been added and a substantial number removed in order to create a list of species that is largely based on UK native species and that offer some potential for adoption as crop species within a paludiculture land-management system. This catalogue of potential crop species is explored in more detail in **Appendix IV**.

5.1.2 Indirect Biodiversity Gains from Paludiculture

Conservation management of key peatland ecosystems (both bogs and fens) has been a core activity for nature conservation in the UK from the earliest days of the nature conservation movement at the start of the last century. Wicken Fen and Woodwalton Fen, in the East Anglian Fenlands, represent some of the first nature reserves established during those times, but both epitomise the ongoing challenges faced by such islands of biodiversity lying within a more intensively farmed landscape. Drainage of the surrounding agricultural landscape generates continual and ongoing challenges to the maintenance of high water tables within these two reserves, and the same is true for almost every peatland nature reserve in the UK lowlands. It is even true for some peatland sites in the uplands where agricultural or forestry activities have surrounded and isolated areas of peatland habitat. Indeed, it is true for many peatland sites across Europe, and a significant proportion of the current EU LIFE+ budget allocated to peatland sites is, in one form or another, devoted to countering the effects of this surrounding drainage regime.

The problem arises because established farming techniques require low water tables for much of the year, but if these requirements were to change, as they could do if paludiculture were to be increasingly adopted as a viable form of agricultural land management, local or regional water tables might be permitted, indeed positively encouraged, to rise. Any elevation of the local water table surrounding a wetland conservation area will reduce the hydrological gradient drawing water from the peatland into the adjacent land-drainage system. Reduced water losses would mean that water tables could be maintained at a more appropriate level for the support of, and re-colonisation and recovery by, peatland species within the conservation area. As a result, direct funding for conservation efforts within the conservation area is likely to be more cost-effective in terms of achieving sustained success and more successful in terms of sustaining and enhancing biodiversity targets for the site.

Table 5.4 *List of UK Specific Paludiculture Projects*

Project Name	Dates	Funder	Budget	Main UK Partners	Areas Investigated	Project Link	Publications
Sphagnum Farming UK - A Sustainable Alternative to Peat in Growing Media	**2018 To 2019**	Innovate UK - AGRI-TECH CATALYST	**£300,000**	Micropropagation Services Ltd.; Manchester Metropolitan University; University of East London; Lancashire Wildlife Trust	Sphagnum cultivation.	https://gtr.ukri.org/projects?ref=BB%2FR021686%2F1	In progress
Water Works Project	**2019 To 2021**	The People's Postcode Lottery Dream Fund	**£1M**	The Wildlife Trusts of Bedfordshire, Cambridgeshire and Northamptonshire; Cambridgeshire Acre; University of East London	Typha, Reed, Sphagnum, Glyceria and other cultivation	https://www.wildlifebcn.org/news/water-works	In progress
Wetland Conservation Biomass to Bioenergy	**2013 To 2015**	The Department for Energy and Climate Change (DECC)	**<£2M**	RSPB	Biomass production for energy – Reed, Typha, Fen biomass. Machinery and Harvest methods	https://decc.blog.gov.uk/author/sally-mills-rspb-reserves-bioenergy-project-manager/	Final report: https://tinyurl.com/yyhuvowz
CANAPE project	**2014 To 2020**	INTERREG European Regional Development Fund	**€5.5M (£4.57M[4])**	The Broads Authority	Converting surplus materials from conservation management into commercial products – reed and sedge.	https://www.broads-authority.gov.uk/looking-after/projects/canape	In progress

Although there are significant challenges in changing current land uses adjacent to conservation areas there are also substantial opportunities, particularly given the present state of peat subsidence and the associated costs and risks (see **Page et al., 2020**), as well as the emergence of potential new crops and farming techniques offered by paludiculture. Such benefits to nature conservation should be viewed as added-value benefit rather than as the driving force for establishment of paludiculture in any given area. Nevertheless, there is an opportunity under such circumstances for a mutually beneficial outcome whereby the paludiculture regime provides multiple benefits to the farmer while also generating significant benefits for the adjacent peatland conservation area without additional cost or effort.

These multiple-sector benefits can be maximised if the opportunities for such collaborative actions can be identified based on the distribution of ground having the potential for adoption of paludiculture and the presence of existing conservation areas. Based on a variety of landform and land-use characteristics, a decision-making tool has been put together for Cumbria to illustrate the potential for both identifying land having the potential for adoption of paludiculture, and such land that also has the potential to contribute to the positive management of peatland conservation areas. This decision-making tool is presented in **Appendix III**.

5.1.3 Flood Risk

At present, fluvial and coastal flood defences together protect agricultural land from estimated annual losses of £116 million (**Roca et al., 2011**). However, the **Association of British Insurers (2013)** calculated that costs arising from the UK floods of 2007 alone amounted to £3 billion in terms of damaged properties, disrupted infrastructure and business losses. At the same time, the **Council of Mortgage Lenders (2013)** identified that 2.7 million properties were at risk from riverine or marine flooding, 550,000 of these properties being at significant risk, while a further 2.8 million properties were at risk from surface flooding. Although trends in flood occurrence are complex, there is a tendency towards more extended and more extreme rain events throughout the year, with long-duration winter rainfall in particular becoming more frequent (**Stevens et al., 2016)**. Three of the most expensive flood events across Europe in recent times have occurred since 2000 (**Gremli et al., 2013**). With a growing interest in natural flood management, this raises new opportunities for farming and paludiculture.

Under natural conditions, the landscape contains headwater catchments dominated by natural vegetation types (e.g. peatlands, woodlands) that provide varying degrees of surface 'roughness' which assist in spreading out flood peaks and thereby lowering maximum peak heights (**Gao et al., 2016**). Once in the floodplain, floodwaters can spread and dissipate their energies over a wider surface area than the river channel into which they have until this point been confined. This natural system, however, no longer applies because of extensive deforestation and degradation of uplands, and conversion of floodplains to agricultural, industrial or urban uses.

The implications of paludiculture management for flood risk are complex, and difficult to quantify. Maintenance of low ditch levels in conventional farmland outside the growing season provides some capacity for flood storage, which could be used to reduce water levels in the river network and thereby protect inhabited areas from flooding. It is unclear, however, whether this storage capacity is ever used. In theory, the lowering of the ground surface due to subsidence also increases the amount of flood storage that could occur on field surfaces during major flood events. As the 2014 flooding of the Somerset Levels showed, widespread flooding of former wetlands is not in general viewed favourably by farmers, the media or the public, and where houses have been built within these low-lying areas the damage to property can be high. Furthermore, pumping floodwater back out of inundated fields would be highly energy-intensive, especially in areas such as the Fens where subsidence has led to field levels being a metre or more below river level (and as subsidence continues, this cost would increase further).

In principle, transitioning to paludiculture will reduce the volume of flood storage available, because both ditch water levels and the field surface will be higher. On the other hand, paludiculture crops are inherently better adapted to periodic flooding, so the economic implications of using these areas for flood storage are likely to be smaller, and indeed flooding could even have a positive impact on crop production if it adds nutrients to the system. Furthermore, because paludiculture will in most cases require areas to be hydrologically isolated from adjacent farmland by bunding, there is potential to actively manage these areas as floodwater storage zones, allowing water to be held temporarily within the landscape without flooding adjacent farmland. This is essentially the function of existing 'washland' areas, such as the Ouse Washes between the Bedford Old and New Rivers, which were part of the original 17th century drainage of the Fens. Expanding the washlands of the Fens, and of other flood-prone lowland areas, could provide valuable additional flood storage capacity, and thereby increase resilience to potentially increasing flood frequency and severity as a result of climate change. If such areas are to be managed for paludiculture, it is likely that they will require active control of water transfers onto and off the site, rather than simply providing passive 'overflow' storage. The much lower current elevation of former farmland converted to paludiculture, compared with areas such as the Ouse Washes that were never drained, means that it may be harder to remove floodwater following site inundation, although there is a possibility that water held within paludiculture-managed areas could be used to irrigate adjacent farmland. The impacts of sustained inundation for both crop productivity and CH_4 emissions (see **Section 4.1**) would need to be considered. More generally, the implication of changing management of lowland peat for flood risk requires further investigation.

6 BARRIERS AND INCENTIVES FOR PALUDICULTURE

This chapter considers some of the current constraints of paludiculture production, identifies key areas for future research, industry collaboration and policy considerations. **Section 6.1** explores the limitations imposed by water management, weed control practices, mechanisation and food security issues, and **Section 6.2** identifies current mechanisms which may prove to be suitable foundations for incentivising change.

6.1 Barriers to Transition

6.1.1 Water Management

For most agricultural areas lying within an Internal Drainage Board (IDB) district, the water table is broadly maintained at levels suitable for production of conventional agricultural crops. In all, some 1.2 million hectares of agricultural grade land in England is managed by IDBs and about 50% of this area relies on pumped drainage (**Roca et al., 2011**).

When seeking to establish paludiculture on an area within such a managed system, conflicting requirements for water management present major challenges. Wetland sites managed for nature conservation often occur as isolated islands of wetland habitat sitting within an agricultural landscape devoted to the production of 'dryland' crops. This makes it difficult for wetland and farmland to be good neighbours. The agricultural land acts as a constant drain on the wetland while the wetland constantly threatens to cause waterlogging of the adjoining farmland. In order to maintain elevated water tables within a paludiculture field it would be necessary to raise base levels in the drainage system immediately surrounding the field. Such actions would have direct consequences for neighbouring fields reliant upon conventional agricultural drainage and are therefore likely to meet with resistance from adjacent land managers. The ability to isolate areas of paludiculture hydrologically from adjacent areas of conventional agriculture may thus be an important consideration, while the inability to do so may be a significant constraint to adoption of paludiculture. Establishment of the first paludiculture venture within a floodplain is likely to be the most challenging step if drainage levels for all surrounding land are managed for conventional crops, whereas increased uptake of paludiculture in a drainage system would enable the areas of paludiculture to become increasingly mutually supporting. Implementing paludiculture in areas surrounding existing conservation areas would have the benefit of reducing the hydrological gradient drawing water away from the peatland into the local land-drainage system, and would provide a summer water source for surrounding farmland (as is the case in some tropical peatlands, where naturally forested 'peat domes' retain water that is subsequently utilised by down-gradient plantations and communities).

Although it is natural to assume that wetland plant species require waterlogged conditions, this is not always the case, and indeed many wetland plants can grow well under normal soil conditions provided that they are watered regularly. An alternative approach to paludiculture, at least for certain paludiculture crops, is therefore the use of surface irrigation rather than seeking to raise water levels in the soil. This approach has the advantage of not requiring changes to the existing drainage system and poses little or no threat to adjacent agricultural land. Surface irrigation is a well-established practice in conventional agriculture and has also proved successful in recent UK *Sphagnum* growing trials both in terms of plant growth and avoided carbon emissions, as discussed earlier. As also noted earlier, the energy cost and GHG emissions associated with surface irrigation are relatively small compared with the savings in GHG emissions resulting from maintaining the peat in a water-saturated condition.

As discussed in Section 4.4, evapotranspiration rates from wet paludiculture systems are likely to exceed those from drained croplands, and both subsurface and surface irrigated paludiculture systems will therefore require additional water during summer. This could represent a significant barrier to the adoption of paludiculture in water-scarce regions such as East Anglia, where competition for water with conventional agriculture could lead to conflict. On the other hand, excess water is pumped off the land during winter, so if more of this water could be stored until summer (for example by pumping excess water into elevated reservoirs, or by gravity-feeding water into overflow reservoirs when river levels are high) this could provide sufficient capacity to meet the higher water demand of paludiculture.

6.1.2 Weed Control

Paludiculture for monoculture crops is likely to have a competitive advantage over non-wetland plants if high moisture levels are maintained in the soil. Weed infestation may therefore arise mainly in the form of other wetland species, and methods of control may thus be less familiar, or less well developed, than for more typical agricultural weeds. That said, standard methods such as weed-control matting are likely to work on wetland weeds as effectively as they do on dryland weeds. Herbicide use, on the other hand, is likely to be more challenging for paludiculture because of the direct linkages to watercourses and other water infrastructure. Ongoing experiments with high-technology solutions such as image recognition linked to targeted laser heating to kill weed species may provide solutions in due course (**Mathiassen et al., 2006**; **Science Daily**; **The Farming Forum**), but even then, are unlikely to be capable of operating where there is standing water.

Some wetland species (for example, common reed, reedmace, reed canary grass, tall sedges and *Sphagnum* moss) grow naturally at high densities, and can in some cases alter the chemical nature of the growing surface. As a result, once the crop is established, opportunities for weed infestation are very much reduced (**Kotowski and van Diggelen, 2004**; **Gaudig et al., 2017**). In such cases the key action is probably sterilization of the pre-existing seedbank, thereby reducing the main source of weed infestation. Production of pure weed-free stock or seeds is also desirable, although the production of pure seed stocks requires a sufficiently large market, creating a ‘chicken and egg’ challenge for the development of paludiculture.

6.1.3 Mechanisation and Scale

A key part of agricultural production is scale. If there is no commercial or practical incentive to develop the machinery, infrastructure and markets necessary to make the enterprise of growing the crop worthwhile, those adopting paludiculture will be restricted to low volume/high value products. This may be sufficient for individual, isolated enterprises; however, this may limit the area which may be covered on a national scale. The larger the area taken up by a crop the greater the incentive and need for associated machinery, infrastructure and markets to be developed. Scale is thus an important consideration for adoption and development of paludiculture. This, in turn, means that identification of the UK land stock having the potential to support paludiculture represents a critical step in determining whether it is worth investing time and effort in developing this form of agriculture beyond small trial plots.

6.1.4 Trafficability

Productive use of peatlands requires effective vehicle access. Trafficability of soil for vehicle access can be described using two key parameters: bearing capacity and shear strength. Bearing capacity defines the capacity of a soil to resist load forces perpendicular to its surface. Peat soils in their pristine state are highly compressible due to their high porosity and the hollow nature of the peat fibres themselves and thus have a low bearing capacity (**Jorat *et al*., 2013**). Deep-drained peatland soils increase in bulk density and are therefore more conducive to conventional farm vehicle access, whereas seasonal or long-term rewetting will reduce bearing capacity. Evidence from 25 wet and rewetted peatland sites under various wetland vegetation covers (Cat-tail, Reed, Sedge, Reed Canary Grass) suggest that whilst the shear strength of these sites is generally amenable to conventional traffic (> 30 kPa), the bearing capacity is much too low (< 200 kPa; **Wichtmann *et al*., 2016**). This study found that vegetation type best explained shear strength at the surface whilst degradation depth predicted shear strength at 50 cm depth. They observed that mean water level was the best predictor of the bearing capacity of surface soil. Overall the evidence suggests that shallow groundwater management will require substantial adaptation from current agricultural methods. Even with vegetation cover, the bearing capacity of soils is unlikely to support conventional agricultural vehicles. Vegetation cover and other protective measures may be necessary to maintain high surface shear strength and avoid formation of highly impassable areas. Low ground-pressure vehicles and the use of dry 'ridges' within wetland production systems could mitigate trafficability and access issues, but will require capital investment and also reduce the land area available for production.

6.1.5 Crop Substitution

Substituting land used for food production, for alternative crops or restoring peatlands to their natural condition will (unless similarly productive food crops could be identified, which currently appears unlikely) either require a compensating increase in crop productivity and/or area on UK mineral soils, or increase the UK's reliance on food imports. How growers adapt to these changes has a number of potential outcomes (**Morris *et al*., 2010**):

(1) Horticulture crops will relocate, in part, to mineral soils, and therefore displace cereal crops, most likely wheat – potentially increasing our reliance on imports of our staples (this may however not be the case if livestock production declines, because a smaller proportion of the UK's cropland area would be required for producing animal feeds)
(2) Cereals on mineral soils will remain on mineral soils, but we will increase our imports of high value horticulture crops (the highest proportion of which is currently supplied by the EU)
(3) Cereals on organic soils won't be replaced, and any shortfall will be imported
(4) Growers will relocate to other countries, where they may regain access to the EU market and/or cheaper labour/land

According to **ONS (2019)**, the current profit margin of an average producer of horticulture crops in the lowlands is £556 ha^{-1} yr^{-1}, whilst arable holdings make an average loss of £12.8 ha^{-1} yr^{-1} (UK 5-year average, 2013 to 2018). To maintain business as usual, arable/cereal holdings will still require subsidies, post-EU exit; not only to maintain current stocks of staples, but also to safeguard the inter-reliance of the arable and livestock sectors (e.g. the outputs and wastes of one, being used as inputs for the other). However, if the UK's consumption of meat and livestock products decreases, then it is expected that production of cereals as feed will also decrease; although this will have a knock-on effect on the organic farming sector which depends on manures and crop residues, to replace nutrients mined through crop production. Where cropland is taken out of production, crops are displaced to other areas, or where there is reduced productivity, a need exists to evaluate the impact on jobs losses in the food supply chain, as well as on carbon emissions resulting from transport of greater volumes of food stuffs into the UK.

Paludiculture could offer a solution for growers and producers of horticulture crops, where they are grown on peat soils. This would however require that marketable yields can be maintained, or where yield reduction is slight, growers be compensated to maintain profitability of holdings; either through direct government subsidies, or other payment schemes, including but not limited to carbon credits (see **Section 6.2**).

6.2 Incentives for Transition

There is now a strong national and international policy drive towards expanded use of paludiculture. In its recent report, the Committee on Climate Change identified paludiculture as part of the possible mix for a net zero carbon strategy (**CCC, 2020**). The UN FAO is also actively promoting uptake of paludiculture as a means of achieving climate-responsible peatland management (**Joosten et al., 2012; Biancalani and Avagyan, 2014**). The EU currently support agri-environment-climate measures with annual per hectare payments of €600 (£499[4]) for annual crops, €900 (£748[4]) for specialised perennial crops and €450 (£374[4]) for other land uses (see Annex 2 of regulation (EU) No 1305/2013 for further information). This may change however in the post-Brexit era and the UK Government commitment to net zero emissions by 2050.

The Kyoto Protocol allows for the development of emissions trading schemes, of which the EU emissions trading scheme (http://ec.europa.eu/clima/policies/ets/registry/index_en.htm) is the largest in operation. An international transaction log ensures secure transfer of emission reduction units between registry systems and countries. Voluntary carbon markets such as the Verified Carbon Standard (VCS) have also been developed, allowing corporations and consumers to offset their emissions in efforts to become 'carbon neutral' (**Joosten et al., 2016**). Carbon offsetting may however only be appropriate in a paludiculture context where an intervention leads to active GHG removal, rather than simply a reduction in current emissions.

Carbon credits only take account of GHG emissions, and place little to no value on other ecosystem services. Initiatives which place value on the services provided by peatlands should act as a guide to future government schemes which could provide support to future initiatives. This approach, it is envisaged, will incentivise landowners to invest in ecologically sound and 'climate smart' solutions. However the premiums achieved by individual project owners will depend not only on the schemes which they are qualified to access, but also the mitigation level achieved, as stipulated under the schemes' guidelines. It may be possible to cover multiple aspects of a single project by different schemes, as long as there is no overlap in payments for the same mitigation measures.

Carbon credits (units) are traded commodities, and as such experience price volatility and change over time. Where carbon credits are used as a vehicle to directly fund projects, it appears that greater restrictions on carbon emissions in the future would increase the prices paid per unit, and those purchasing the units (i.e. businesses through Corporate Social Responsibility schemes) are incentivised to make changes sooner rather than later. In February 2020, carbon units are valued at approximately €25 (£20.8[4]), but some projections suggest that the UK price of carbon may need to rise to £75 by 2050 to meet the UK government's net zero target (Burke et al., 2019). Clearly if this projected increase in carbon prices occurs, it could significantly enhance the economic basis for paludiculture.

7 POTENTIAL IMPACTS OF CLIMATE CHANGE ON PALUDICULTURE

Although many forms of wetland agriculture have been practiced for millennia, the formal concept of paludiculture is relatively new. Research, field trials and reviews of paludiculture to date have, as discussed in the preceding sections, focused mainly on developing practical and economic forms of paludiculture, and evaluating their potential to contribute to climate change mitigation. Few if any studies have examined how paludiculture might be impacted by future climate change. This section therefore briefly evaluates the potential challenges and opportunities of climate change for paludiculture based on previous assessments of climate change impacts on natural and managed peatlands, and on an initial evaluation of how current climate change projections might affect the issues addressed in the preceding sections.

7.1 Climate change threats to paludiculture

Climate change in the UK is generally expected to lead to higher year-round temperatures and a change in seasonal precipitation patterns towards wetter winters and drier summers with fewer but more intense rain events. Several previous studies have applied bioclimatic envelope models to predict the impacts of these projected changes on the spatial extent of peatlands in the UK and Ireland (Clark et al., 2010; Gallego-Sala et al., 2010; Coll et al., 2014; Ferreto et al., 2019). All studies have focused on natural peatland systems, primarily blanket bogs, reflecting their greater spatial extent and the availability of data to parameterise models. In general these models predict that blanket bogs may no longer be able to form in climatically marginal (warmer, drier) regions such as Eastern Scotland, Northeast England and Southwest England, with potentially severe consequences for the stability of existing carbon stocks (e.g. Ferreto et al., 2019). To some extent, these pessimistic projections of the future stability of the UK's peatlands have been used to argue that investment in restoration may be unjustified, because many peatlands may be effectively 'doomed'. However, climate envelope models are also open to criticism because they are parameterised using current peatland distribution, often for a limited area; for example, UK models are parameterised with UK data, but blanket bogs exist in other regions (including Northern Spain) that may lie outside the apparent climate envelope. Furthermore, the existence and persistence of blanket bogs over many thousands of years strongly suggests that these systems are highly resilient to climate fluctuation when in their undisturbed state, for example through their capacity to 'breathe' (whereby the bog surface follows the water table down during drought periods, maintaining moist conditions). In contrast, heavily modified peatlands lack this resilience, and are therefore far more susceptible to degradation in response to climate stress. In other words, restoration of the ecological and hydrological integrity of peatlands will, in all cases, reduce their vulnerability to climate change.

With regard to the lowland fen and raised bog systems that are the main target for paludiculture, their current limited extent, severe modification by land-use and (in the case of fens) hydrological complexity has so far precluded the application of a bioclimatic envelope modelling approach. It is therefore unclear to what extent the results of modelling studies based on upland blanket bogs may be transferrable. In general, we would expect that hotter, drier summers will accelerate the decomposition of drained peatlands while also making the conservation of semi-natural wetlands more challenging due to potential limitations on water availability. This is particularly the case for the small, hydrologically isolated fragments of natural fen in East Anglia, such as Wicken Fen, which rely on irrigation and pumping of river water to maintain water levels during the summer. As discussed in Section 4.4, these areas provide the best available analogue for water use and management in future paludiculture systems. Our analysis of water balance data indicates that evapotranspiration rates from fen wetlands are higher than those from drained croplands, and it is therefore likely that paludiculture systems will have a higher water demand, particularly during summer. Under conditions of lower and more episodic summer rainfall this could lead to problems of competition for declining water resources, particularly in already water-scarce regions of Eastern England. Apart from water scarcity, however, there is little evidence that climate change will have a detrimental impact on the growth or viability of most paludiculture crops, and in some cases it may even be beneficial (see below). Besides, there are extensive regions of the UK lowlands outside of the East Anglian Fens which have potential for adoption of paludiculture, including, for example, the Lancashire lowland plain, the Somerset Levels, the Forth Valley and the estuarine valleys of Wales. In some of these areas, increased annual rainfall could increase water supply. Indeed, evidence for increased precipitation and flood risk across much of the UK outside of the East Anglian Fens has already been identified (Blöschl et al., 2019). Designing paludiculture systems that are resilient to climate change is therefore vital. In the south and east of England, this is likely to be primarily on maintaining a secure water supply during dry periods, whereas in the north and west a focus on excess water management, as part of wider ecosystem services arising from paludiculture, may also play an important part. Constraints on water availability already apply to the development of paludiculture under current climate conditions (see Section 6.1.1.) and therefore future climate change risks could be mitigated by building sufficient capacity into local water storage systems.

7.2 Climate change threats to conventional peatland agriculture

When considering potential climate change risks to paludiculture, it is important also to consider the 'counterfactual' of continuing conventional arable and horticulture on peat. Crop yields are vulnerable to both winter flooding and summer drought, both of which are expected to become more frequent and severe in future, particularly in the south and east of the UK. While the effects of drought on crop yields may be lower in deep peat than for equivalent crops on mineral soils (Martin Hammond, G's Fresh pers. comm.), increasing severity of either drought or flooding could impact on the economic viability of current agricultural practices in future. Furthermore, current rates of soil loss in drained peatlands could accelerate in future, which would both increase already-high rates of CO_2 emission, and lead to a progressive reduction in agricultural productivity. These soil losses occur through a combination of peat oxidation and wind erosion, both of which are likely to accelerate under climate change. The rate of aerobic decomposition of peat is temperature dependent, so unless the peat becomes so dry that microbial processes become constrained (which is unlikely, in irrigated agricultural land) warmer temperatures would be expected to increase oxidative peat loss. As discussed in the accompanying report by Page et al. (2020), wind erosion events ('fen blows') occur when dry peat is exposed to high wind speeds. Risks are greatest in spring when the peat surface is unvegetated or newly planted, and particularly during and after farm operations that disturb the peat surface and break down larger soil aggregates into finer, more erodible material. Fen blows are episodic but a single event can lead to substantial soil loss. Increased incidence of dry conditions and high wind speeds at critical times of year could therefore lead to significantly increased rates of peat erosion under climate change.

When compared with conventional agriculture, lowland peatlands managed for paludiculture may be relatively resilient to climate change. Oxidation rates of water-saturated anaerobic peat are greatly reduced compared to aerobic peat under all temperature conditions (hence why peat is able to form under waterlogged conditions in the tropics) and wind erosion risk in permanently vegetated paludiculture will be negligible. Meanwhile, in regions of lowland UK that may experience more frequent flooding, conventional crop production may be adversely affected by prolonged flooding, with even existing climate extremes can result in losses running into millions of pounds (ADAS, 2014).Paludiculture crops are likely to be more resilient to such flood events, and if the challenge of maintaining and managing water supplies in summer can be overcome, may be less vulnerable to climate change than conventional cropland agriculture.

7.3 Potential opportunities for, and benefits of, paludiculture under climate change

Whilst the overall impacts of climate change are likely to be negative, it is also possible that there could be some benefits for paludiculture. Provided that sufficiently high water levels and/or surface irrigation rates can be maintained, higher temperatures and longer growing seasons could increase the yields of many paludiculture crops, thereby increasing their economic benefit. It is also possible that opportunities could arise for the cultivation of new wetland crops that were previously unsuited to the UK climate. Of particular interest is the possible cultivation of rice as a food crop. As noted earlier, varieties of rice are grown in other temperate regions such as Japan, including the northern island of Hokkaido, as well as in Southern Europe and California. In China, the centroid of the rice production area in China moved North by 370 km between 1949 and 2010 (Li et al., 2015). The northern limit of rice cultivation in China sat at ~52 degrees North by 2007 (Piao et al. 2010); Ely sits at 52.4 degrees North. Northward expansion of the Chinese rice growing area is correlated with increases in the climatically suitable region being driven by climate change, although there may be a lag as farmers take time to adopt new practices even after climate suitability increases (Liu et al., 2015). Further advances in crop breeding and improved varieties have the potential to accentuate the increases in length of the suitable growing season produced by climate change at high-latitudes (Liu et al., 2016). If milder winters and longer growing seasons, combined with the development of more cold-tolerant varieties, were to make rice production on UK lowland peats viable, this could dramatically alter the current economics of paludiculture, as well as overcoming a key concern about the displacement of food production and associated GHG emissions to other regions.

Finally, paludiculture could also provide benefits for climate change adaptation. As discussed earlier, positioning paludiculture between elevated nature reserves and low-lying agricultural land could help to reduce hydrological gradients and slow water flows within the landscape, making it easier to maintain the wet conditions required to conserve natural wetlands. This may also provide a source of water to down-gradient farmland, promote greater overall resilience to drought at a landscape scale. Paludiculture areas could even be designed as water storage 'reservoirs' within farmed landscapes, in which winter flood water can be held and released to adjacent farmland during severe dry periods. While this would require the use of paludiculture crops that could withstand temporary drying, this approach might provide a more productive and therefore economic means of farm-scale water storage than conventional reservoirs. Similarly, if areas of paludiculture can be designed to perform the function of 'washlands' by storing excess river water during winter; this could provide flood protection to farmland and urban areas, but release water to farmland as required during summer months.

8 CONCLUSIONS

There is a growing body of knowledge to suggest that raising water levels in lowland peat crop production areas to reduce environmental impacts such as GHG emissions and subsidence will not necessarily require the cultivation of economically important fresh produce to cease. There remain significant knowledge gaps on how these adaptations will fully impact upon food supplies and regional economies. Much of the research undertaken to date has involved the replacement of drainage-based food crops with (native) plant species better suited to wetter conditions, primarily for fibre production. This may well represent an appropriate use of peatland, provided that markets for these products exist; that paludiculture can be made financially viable (or supported via subsidies that reflect the wider societal value of peatland protection); and that food crops displaced from organic soils can be grown instead on mineral soils where their environmental impact will be lower.

The UK currently however, imports a significant proportion of its foodstuffs, and there are clear costs and risks to food security in transferring further food production to other countries. Where emissions from UK food production on organic soils are replaced by emissions from the production and transportation of food produced elsewhere, the transformed system must be fully evaluated, to ensure that this is not simply displacing or even increasing total emissions. It is also important to emphasise that lowland peatlands are among the UK's highest grade agricultural land, so careful consideration of the potential economic costs as well as environmental benefits of reducing food production on organic soils is required. On the other hand, several studies including the recent assessment by **ONS (2019)** suggest that reduced drainage-based cultivation of organic soils could have a net economic benefit once the climatic impact of associated greenhouse gas emissions is taken into account.

Where rewetting is practiced, the condition and location of the land will dictate the plant/crop and the management practice chosen. Ultimately, niche crops will have small market penetration without policy adjustments or export facilitation. A number of paludiculture crops have potential for upscaling following targeted research, the development of supply chains and markets, and minimisation of costs. Finally, there may be novel food crops that could be grown on high water table peatlands that have not yet been identified. Many of the most important food crops grown on peatlands globally are not native to the regions in which they are now grown, and although many of these still require drainage there are also examples of wetland-adapted food crops being grown on formerly drained peatlands. While existing varieties of rice are unsuited for current UK conditions, this could change under a future climate or as a result of crop development. This will require further research and field trials, as well as market development, but again has the potential to be scaled up if successful. In general, peatlands managed for paludiculture are likely to be more resilient to climate change than those under conventional drainage-based agriculture, but are likely to require the development of increased water storage capacity to maintain water levels during dry summers. This storage capacity could however help to provide protection to adjacent farmland and urban areas at risk of increased flooding as a result of climate change, and provide a source of water to adjacent farmland during dry periods as part of integrated farm- and landscape-scale water management systems.

Overall, we conclude that, although there is considerable potential, paludiculture does not yet offer an economically viable, large-scale or immediately implementable solution to the challenge of high GHG emissions from cultivated lowland peats. This should not preclude continued research and development into the potential of high-water table crops, or to the development and expansion of paludiculture trials with the aim of scaling these up where successful. If these products can be developed, it will be important to ensure that they do not lead to displacement of emissions from food production to other regions or countries, to regional loss of employment, or to a reduction in national food security. If these challenges can be overcome, there is the potential for paludiculture to make a valuable contribution to UK climate change mitigation, whilst also maintaining the economic output and extending the lifetime of agriculturally productive lowland peat regions. Until and unless paludiculture becomes a viable large-scale proposition, however, it remains essential that efforts are made to mitigate emissions from UK peatlands remaining under drainage-based arable and horticulture cultivation.

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References

Abel, S., Couwenberg, J., Dahms, T. & Joosten, H. (2013) The Database of Potential Paludiculture Plants (DPPP) and results for Western Pomerania. *Plant Diversity and Evolution*, 130, 219–228. [DOI: http://doi.org/10.1127/1869-6155/2013/0130-0070]

Acreman, M.C., Harding, R.J., Lloyd, C., McNamara, N.P., Mountford, J.O., Mould, D.J., Purse, B.V., Heard, M.S., Stratford, C.J. & Dury, S.J. (**2011**) Trade-off in ecosystem services of the Somerset Levels and Moors wetlands. Hydrological Sciences Journal, **56** (8), 1543–1565. [DOI: 10.1080/02626667.2011.629783]

ADAS (**2014**) *Impact of 2014 Winter Floods on Agriculture in England*. Helsby: ADAS UK Ltd. http://randd.defra.gov.uk/Document.aspx?Document=12442_TheEconomicImpactofthe2014WinterFloodsonAgricultureinEngland_Final14-07-15.pdf

Afreen, F., Zobayed, S., Armstrong, J. & Armstrong, W. (**2007**) Pressure gradients along whole culms and leaf sheaths, and other aspects of humidity-induced gas transport in *Phragmites australis*. Journal of Experimental Botany, **58** (7), 1651-1662. [DOI: 10.1093/jxb/erm017]

AHDB (**2016**) Tracking peat usage in growing media production (Project CP 100), Final report (HDC/Defra/HTA; 2012-2015). Available from: http://tiny.cc/CP100

Akulain, V.R. (**2013**) Wetland biomass - suitable for biogas production?, MSc Thesis, Halmstad University, Halmstad, Sweden. Available from: http://tiny.cc/Akulain

Andrews, R. (**2000**) *LIFE 97 ENV/UK/000511 New Wetland Harvests – Final Technical Report*. Norwich: Broads Authority.

Association of Drainage Authorities (ADA) (**2013**) Written Evidence to House of Commons Environment, Food and Rural Affairs Committee: Flood Funding. Available from: https://www.parliament.uk/documents/commons-committees/environment-food-rural-affairs/FFconsolidatedfinal.pdf

Ayati, B., Ferrándiz-Mas, V., Newport, D. and Cheeseman, C. (**2018**) Use of clay in the manufacture of lightweight aggregate. *Construction and Building Materials*, **162**, 124-131. [DOI: 10.1016/j.conbuildmat.2017.12.018]

Ayres. P. (**2013**) Wound dressing in World War I - The kindly Sphagnum Moss. *Field Bryology*, No.110, 27-34. Available from: http://tiny.cc/Ayres2013

Ayres, P.G. (**2015**) Isaac Bayley Balfour, Sphagnum moss, and the Great War (1914–1918). Archives of Natural History, **42** (1), 1-9 [DOI: 10.3366/anh.2015.0274]

Baranyai, B., Bäcker, C., Reich, C. and Lindequist, U. (**2016**) The production of 7-methyljuglone, plumbagin and quercetin in wild and cultivated Drosera rotundifolia and Drosera intermedia. Mires and Peat, **18** (19), 1-8 [DOI: 10.19189/MaP.2016.OMB.228]

Barba, J., Bradford, M.E., Brewer, P.E., Bruhn, D., Covey, K., van Haren, J., Megonigal, J.P., Mikkelson, T.N., Pangala, S.R., Pihlatie, M., Poulter, B., Rivas-Ubach, A., Schadt, C.W., Terazawa, K., Warner, D.L., Zhang, Z., Vargas, R. (**2019**) Methane emissions from tree stems: a new frontier in the global carbon cycle. New Phytologist, **222**, 18-28 [DOI: 10.1111/nph.15582]

Barrett, G.E., Alexander, P.D., Robinson, J.S., Bragg, N.C. (**2016**). Achieving environmentally sustainable growing media for soilless plant cultivation systems – A review. Sci. Hort. ***212***, 220-234 [DOI: 10.1016/j.scienta.2016.09.030]

Bennett, E.M., Peterson, G.D. & Gordon, L.J. (**2009**) Understanding relationships among multiple ecosystem services. *Ecology Letters*, **12** (12), 1394-1404. [DOI: 10.1111/j.1461-0248.2009.01387.x]

Bevan, J. (**2018**) “Surface water: The biggest flood risk of all.” Speech by Sir James Bevan KCMG, Chief Executive, Environment Agency, CIWEM Surface Water Management Conference, 17 October 2018. Available from: https://www.gov.uk/government/news/surface-water-the-biggest-flood-risk-of-all

Biancalani, R. and Avagyan, A. (eds.) (**2014**) *Towards climate-responsible peatlands management. Mitigation of Climate Change in Agriculture Series 9.* Rome: Food and Agriculture Organization of the United Nations. 107pp. Available from: http://www.fao.org/3/a-i4029e.pdf

Blok, C. (**2018**) Worldwide Peat Demand, Presentation for IPS Global Peatlands and Peat Industry Summit 2018. Rotterdam, the Netherlands

Blöschl, G., Hall, J., Viglione, A. *et al.* (**2019**) Changing climate both increases and decreases European river floods. *Nature* **573,** 108–111. doi.org/10.1038/s41586-019-1495-6

Bonn, A., Rebane, M. & Reid, C. (**2009a**) Ecosystem services: a new rationale for conservation of upland environments. *In:* Drivers of Environmental Change in Uplands, Routledge, pp. 448-474. (ISBN: 9780415564083)

Bonn, A., Holden, J., Parnell, M., Worrall, F., Chapman, P.J., Evans. C.D., Termansen, M., Beharry-Borg, N., Acreman, M.C., Rowe, E., Emmett, B., Tsuchiya, A., Davison, S., Maskill, R., Flynn, M., Hall, J., Harding, R.J., Kirkby, M.J., Lloyd, C., McNamara, N., Mountford, J.O., Mould, D., Norris, D., Purse, E. & Stratford, C.J. (**2009b**) Ecosystem services of peat – Phase 1. Project code: Defra SP0572. Available from: http://tiny.cc/SP0572

Bonn, A., Reed, M.S., Evans, C., Joosten, H., Bain, C., Farmer, J., Emmer, I., Couwenberg, J., Moxey, A., Artz, R., Tanneberger, F., von Unger, M., Smyth, M.A. & Birnie, R. (**2014**) Investing in nature: developing ecosystem services markets for peatland restoration. Ecosystem Services, **9**, 54-65. [DOI: 10.1016/j.ecoser.2014.06.011]

Borrill, M. (**1956**) A biosystematic study of some Glyceria species in Britain. Taxonomy. Watsoinia, 3, pp.291-298. Available from: http://archive.bsbi.org.uk/Wats4p89.pdf

Børsheim, Y., Christensen, B.E. and Painter, T.J. (**2001**) Preservation of fish by embedment in Sphagnum moss, peat or holocellulose: Experimental proof of the oxopolysaccharidic nature of the preservative substance and of its antimicrobial and tanning action. *Innovative Food Science & Emerging Technologies*, **2** (1), 63-74. [DOI: 10.1016/S1466-8564(00)00029-1]

Bragg, O.M. and Lindsay, R.A. (eds.) (**2003**) *Strategy and Action Plan for Mire and Peatland Conservation in Central Europe*. Wageningen: Wetlands International. 93pp. Available from: https://gfmc.online/wp-content/uploads/Peatland-Conservation-Europe.pdf

Bragg N (**2018**). AHDB Growing Media Review. 19 pp. Available from: http://tiny.cc/GMReview2018

Brouwer, R., Bateman, I.J., Turner, R.K., Adger, W.N., Boar, R., Crooks, S., Dockerty, T., Georgiou, S., Jones, A., Langford, I.H., Ledoux, L., Nishikawa, N, Powe, N., Wright, J. and Wright, S. (**2001**) Management of a multi-purpose coastal wetland: The Norfolk and Suffolk Broads, England. In: R.K. Turner, I.J. Bateman and W.N. Adger (eds.) *Economics of Coastal and Water Resources: Valuing Environmental Functions.* pp. 159-214. New York: Springer. (ISBN: 9780792365044)

Burke, J., Byrnes, R., Fankhauser, S. (**2019**). How to price carbon to reach net-zero emissions in the UK. Grantham Research Institute on Climate Change and the Environment, www.cccep.ac.uk

Carlile, B., and Coules, A. (**2013**). Towards sustainability in growing media. Acta Hortic. ***1013***, 341-349. [DOI: 10.17660/ActaHortic.2013.1013.42]

Carmichael, M.J., Bernhardt, E.S., Bräuer, S.L. & Smith, W.K. (**2014**) The role of vegetation in methane flux to the atmosphere: should vegetation be included as a distinct category in the global methane budget? Biogeochemistry, **119** (1-3), 1–24. [DOI: 10.1007/s10533-014-9974-1]

Chemisana, D. and Lamnatou, C. (**2014**) Photovoltaic-green roofs: an experimental evaluation of system performance. *Applied Energy*, **119**, 246 256. [DOI: 10.1016/j.apenergy.2013.12.027]

Coll, J., Bourke, D., Skeffington, M.S., Gormally, M., Sweeney, J. (**2014**). Projected loss of climate space for active blanket bog in Ireland. In In the bog: the ecology, landscape, archaeology and heritage of peatlands conference, Sheffield, UK, 3-5.

Clark, J.M., Gallego-Sala, A.V., Allott, T.E., Chapman, S.J., Farewell, T., Freeman, C., House, J.I., Orr, H.G., Prentice, I.C., Smith, P. (**2010**). Assessing the vulnerability of blanket peat to climate change using an ensemble of statistical bioclimatic envelope models. *Climate Research*, **45**, 131-150 [https://www.int-res.com/articles/cr_oa/c045p131.pdf]

Clymo, R.A. (**1983**) Peat. In: A.J.P. Gore (ed.) *Mires, Swamp, Fen and Moor. General Studies. Ecosystems of the World 4a.* pp.159-224. Amsterdam : Elsevier Scientific.

Clymo, R.S. and Hayward, P.M. (**1982**) The Ecology of Sphagnum. In: A.J.E. Smith (Ed.) *Bryophyte Ecology*. London : Chapman and Hall, pp. 229-289. (ISBN: 9789400958937)

Coles, J.M. and Orme, B.J. (**1980**) *Prehistory of the Somerset Levels*. Hertford: Somerset Levels Project. 64pp. (ISBN: 9780950712208)

Committee on Climate Change (**2020**) *Land use: Policies for a Net Zero UK.* Available from: https://www.theccc.org.uk/publication/net-zero-the-uks-contribution-to-stopping-global-warming/

Costanza, R., d'Arge, R., De Groot, R., Farberk, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P. & Van den Belt, M. (**1997**) The value of the world's ecosystem services and natural capital. Nature, **387**, 253–260. [DOI: 10.1038/387253a0]

Council of Mortgage Lenders (**2013**) Written Evidence to House of Commons Environment, Food and Rural Affairs Committee : Flood Funding. Available from: https://www.parliament.uk/documents/commons-committees/environment-food-rural-affairs/FFconsolidatedfinal.pdf

Couwenberg, J., Thiele, A., Tanneberger, F., Augustin, J., Bärisch, S., Dubovik, D., Liashchynskaya, N., Michaelis, D., Minke, M., Skuratovich, A. & Joosten, H. (**2011**) Assessing greenhouse gas emissions from peatlands using vegetation as a proxy. Hydrobiologia, **674** (1), 67-89. [DOI: 10.1007/s10750-011-0729-x]

Croon, F.W. (**2013**) Saving reed lands by giving economic value to reed. Mires and Peat, **13** (2013/14), Article 10, 1–10. Available from: http://mires-and-peat.net/pages/volumes/map13/map1310.php

Crowder, A.A., Pearson, M.C., Grubb, P.J. and Langlois, P.H. (**1990**) Biological flora of the British Isles – *Drosera* L., *Journal of Ecology*, **78**, 233-267. [DOI: 10.2307/2261048]

Daily, G.C. (**1997**) Nature's Services: Societal Dependence on Natural Ecosystems. Island Press. (ISBN: 9781559634762)

Darby, H.C. (**1940**) The Medieval Fenland. Cambridge University Press, Cambridge, 200 p. (ISBN: 9781107614987)

Darby, H.C. (**1956**) The draining of the Fens (2nd ed.), Cambridge University Press, Cambridge, 314 p. (ISBN: 9781107402980)

Davidson, N.C. (**2014**) How much wetland has the world lost? Long-term and recent trends in global wetland area. Marine and Freshwater Research, **65** (10), 934-941. [DOI: 10.1071/MF14173]

Debnath, S. (**2015**) Great Potential of Stinging Nettle for Sustainable Textile and Fashion. In: M.A. Gardetti and S.S. Muthu (eds.) *Handbook of Sustainable Luxury Textiles and Fashion*. pp 43-57. Zurich: Springer Nature. (ISBN: 9789812876324)

Defra (**2009a**) A preliminary assessment of the greenhouse gases associated with growing media materials, Defra Project IF0154, Prepared by Warwick-HRI. Available from: https://tiny.cc/IF0154

Defra (**2009b**) Costs to the horticultural sector of meeting the UKBAP target on peat use in horticulture, Defra Project SP0577, Prepared by ADAS UK Ltd. Available from: http://tiny.cc/SP0577

Defra (**2011a**) The costs of phasing out peat in the hardy nursery stock sector, Defra Project SP1211, Prepared by Brook Lyndhurst. Available from: http://tiny.cc/SP1211

Defra (**2011b**) The natural choice: securing the value of nature. Available from: http://tiny.cc/DefraTNC

Defra (**2013a**) Government response to the sustainable growing media task force. Available from: http://tiny.cc/ResSGMTF

Defra (**2013b**) Towards sustainable growing media. Available from: http://tiny.cc/DefraTSGM

Defra (**2018**). A green future: Our 25 year plan to improve the environment. 151 pp. Available from: https://www.gov.uk/government/publications/25-year-environment-plan

Defra (**2019**) Farming Statistics: Final Land Use, Livestock Populations and Agricultural Workforce At 1 June 2018 – England. Available from: http://tiny.cc/defra2019, Accessed 25th January 2019.

Dias, A.T.C., Hoorens, B., Van Logtestijn, R.S.P., Vermaat, J.E., Aerts, R. (**2010**) Plant species composition can be used as a proxy to predict methane emissions in Peatland ecosystems after land-use changes. Ecosystems, **13**, 526-528. [DOI: 10.1007/s10021-010-9338-1]

Ding, W., Cai, Z. & Tsuruta H. (**2004**) Diel variation in methane emissions from the stands of Carex lasiocarpa and Deyeuxia angustifolia in a cool temperate freshwater marsh. Atmospheric Environment, **38** (2), 181–188. [DOI: 10.1016/j.atmosenv.2003.09.066]

Di Virgilio, N., Papazoglou, E.G., Jankauskiene, Z., Di Lonardo, S., Praczyk, M. and Wielgusz, K. (**2015**) The potential of stinging nettle (*Urtica dioica* L.) as a crop with multiple uses. *Industrial Crops and Products*, **68**, 42-49. [DOI: 10.1016/j.indcrop.2014.08.012]

Dommain, R. (**2016**) Indonesia – Paludiculture as sustainable land use. *In:* W. Wichtmann, C. Schröder and H. Joosten (eds.) *Paludiculture - productive use of wet peatlands: Climate protection – biodiversity – regional economic benefits*. Schweitzerbart, Stuttgart. pp. 217-223. (ISBN: 9783510652839)

Dragoni, F., Giannini, V., Ragaglini, G., Bonari, E. & Silvestri, N. (**2017**) Effect of harvest time and frequency on biomass quality and biomethane potential of common reed (Phragmites australis) under paludiculture conditions. Bioenergy Research, **10** (4), 1066–78. [DOI: 10.1007/s12155-017-9866-z]

Duan, X., Wang, X. & Ouyang, Z. (**2006**) Plant-mediated CH4 emission from a Phragmites dominated wetland in an arid Region, China. Journal of Freshwater Ecology, **21** (1), 139–145. [DOI: 10.1080/02705060.2006.9664106]

Dubbe, D.R., Garver, E.G. & Pratt, D.C. (**1988**) Production of cattail (Typha spp.) biomass in Minnesota, USA. Biomass, **17** (2), 79-104. [DOI: 10.1016/0144-4565(88)90073-X]

Durechová, D., Kacániová, M., Terentjeva, M., Petrová, J., Hleba, L. and Kata, I. (**2016**) Antibacterial activity of Drosera rotundifolia L. against gram-positive and gram-negative bacteria. The Journal of Microbiology, Biotechnology and Food Sciences, **5**, p.20 [DOI: 10.15414/jmbfs.2016.5.special1.20-22]

Ely Group of Internal Drainage Boards. (**2016**). Internal Drainage Boards. Available: http://www.elydrainageboards.co.uk/. Last accessed 16/10/2019.

Emmel, M. (**2008**) Growing ornamental plants in Sphagnum biomass. *Acta Horticulturae*, **779**, 173-178. [DOI: 10.17660/ActaHortic.2008.779.20]

Evans, D.E. (**2003**) Aerenchyma formation. *New Phytologist*, **161**, 35-49. [DOI: 10.1046/j.1469-8137.2003.00907.x]

Evans, J.M. & Wilkie, A.C. (**2010**) Life cycle assessment of nutrient remediation and bioenergy production potential from the harvest of hydrilla (Hydrilla verticillata). Journal of Environmental Management, **91** (12), 2626-2631. [DOI: 10.1016/j.jenvman.2010.07.040]

Evans, C.D., Bonn, A., Holden, J., Reed, M.S., Evans, M.G., Worrall, F., Couwenberg, J. & Parnell, M. (**2014**) Relationships between anthropogenic pressures and ecosystem functions in UK blanket bogs: Linking process understanding to ecosystem service valuation. Ecosystem Services, **9**, 5-19. [DOI: 10.1016/j.ecoser.2014.06.013]

Evans, C.D., Renou-Wilson, F., Strack, M. (**2016**). The role of waterborne carbon in the greenhouse gas balance of drained and re-wetted peatlands. Aquatic Sciences, **78**, 573-590. [DOI: 10.1007/s00027-015-0447-y]

Evans, C., Morrison, R., Burden, A., Williamson, J., Baird, A., Brown, E., Callaghan, N., Chapman, P., Cumming, A., Dean, H., Dixon, S., Dooling, G., Evans, J., Gauci, V., Grayson, R., Haddaway, N., He, Y., Heppell, K., Holden, J., Hughes, S., Kaduk, J., Jones, D., Matthews, R., Menichino, N., Misselbrook, T., Page, S., Pan, G., Peacock, M., Rayment, M., Ridley, L., Robinson, I., Rylett, D., Scowen, M., Stanley, K. & Worrall, F. (**2017**) Lowland peatland systems in England and Wales – evaluating greenhouse gas fluxes and carbon balances. Project code: Defra SP1210. Available from: http://tiny.cc/SP1210

Fenner, N., Freeman, C., Hughes, S. and Reynolds, B. (**2001**) Molecular weight spectra of dissolved organic carbon in a rewetted Welsh peatland and possible implications for water quality. Soil Use and Management, **17**, 106–12. [DOI: 10.1111/j.1475-2743.2001.tb00015.x]

Ferretto, A., Brooker, R., Aitkenhead, M., Matthews, R., Smith, P. (**2019**). Potential carbon loss from Scottish peatlands under climate change. *Regional Environmental Change*, **19**, 2101-2111. [DOI: 10.1007/s10113-019-01550-3]

Gao, J., Holden, J. and Kirkby, M. (**2016**) The impact of land-cover change on flood peaks in peatland basins. Water Resources Research, **52**, 3477–3492. [DOI: 10.1002/2015WR017667]

Gates, D.M. (**1980**) *Biophysical Ecology*. New York: Springer-Verlag. 611pp. (ISBN: 9781461260264)

Gaudig, G. , Krebs, M., Prager, A., Wichmann, S., Barney, M., Caporn, S.J.M., Emmel, M., Fritz, C., Graf, M., Grobe, A., Gutierrez Pacheco, S., Hogue-Hugron, S., Holzträger, S., Irrgang, S., Kämäräinen, A., Karofeld, E., Koch G., Koebbing, J.F., Kumar, S., Matchutadze, I., Oberpaur, C., Oestmann, J., Raabe, P., Rammes, D., Rochefort, L., Schmilewksi, G., Sendžikaitė, J., Smolders, A., St-Hilaire, B., van de Riet, B., Wright, B., Wright, N., Zoch, L. and Joosten, H. (**2018**) Sphagnum farming from species selection to the production of growing media: a review. *Mires and Peat*, **20** (2017/18), Article 13, 1–30. [DOI: 10.19189/MaP.2018.OMB.340]

Gallego-Sala, A.V., Clark, J.M., House, J.I., Orr, H.G., Prentice, I.C., Smith, P., Farewell, T., Chapman, S.J. (**2010**). Bioclimatic envelope model of climate change impacts on blanket peatland distribution in Great Britain. *Climate Research*, **45**, 151-162. [www.int-res.com/articles/cr_oa/c045p151.pdf]

Gaudig, G., Joosten, H. and Kamermann, D. (**2008**) Growing growing media: Promises of *Sphagnum* biomass. *Acta Horticulturae*, **779**, 165-171. [DOI: 10.17660/ActaHortic.2008.779.19]

Gaudig, G., Fengler, F., Krebs, M., Prager, A., Schulz, J., Wichmann, S., Joosten, H. (**2014**) Sphagnum farming in Germany – a review of progress. Mires & Peat, **13**, Article 08, 1-11. Available from: http://mires-and-peat.net/media/map13/map_13_08.pdf

Gaudig, G. et al. (**2017**) Sphagnum farming from species selection to the production of growing media. Mires and Peat, Volume 20 (2017/18), Article 13, 1–30. [DOI: 10.19189/MaP.2018.OMB.340]

Georgiev, G., Krus, M., Loretz, C. and Theuerkorn, W. (**2019**) TYPHABOARD in the Restoration of Historic Black Sea Houses in Bulgaria. *Sustainability, MDPI, Open Access Journal*, **11** (4), 1-13. [DOI: 10.3390/su11041000]

Giannini, V., Silvestri, N., Dragoni, F., Pistocchi, C., Sabbatini, T. & Bonari, E. (**2017**) Growth and nutrient uptake of perennial crops in a paludicultural approach in a drained Mediterranean peatland. Ecological Engineering, **103** (Part B), 478-487. [DOI: 10.1016/j.ecoleng.2015.11.049]

Giannini, V., Bertacchi, A., Bonari, E., & Silvestri, N. (**2019**) Recolonisation by spontaneous vegetation of a rewetted peatland after topsoil removal: a focus on biomass production and nutrient uptake. Wetlands (*Published in cooperation with Society of Wetland Scientists*). [DOI: 10.1007/s13157-019-01141-2]

Godwin, H. (**1978**) *Fenland: its ancient past and uncertain future*. Cambridge: Cambridge University press. 196pp. (ISBN: 9780521217682)

Godwin H. (**1981**) The archives of peat bogs. Cambridge University Press, Cambridge, UK. (ISBN: 9780521107129)

Görn, S. & Fischer, K. (**2016**) The effect of mowing on animals. *In:* W. Wichtmann, C. Schröder and H. Joosten (eds.) *Paludiculture - productive use of wet peatlands: Climate protection – biodiversity – regional economic benefits.* Schweitzerbart, Stuttgart. pp. 94-96. (ISBN: 9783510652839)

Graneli, W. (**1989**) Influence of standing litter on shoot production in reed, Phragmites australis (Cav.) Trin. ex Steudel. Aquatic Botany, **35** (1), 99–109. [DOI: 10.1016/0304-3770(89)90070-3]

Gravalos, I., Xyradakis, P., Kateris, D., Gialamas, T., Bartzialis, D. & Giannoulis, K. (**2016**) An Experimental Determination of Gross Calorific Value of Different Agroforestry Species and Bio-Based Industry Residues. Natural Resources, **7** (1), 57-68. [DOI: 10.4236/nr.2016.71006]

Gray, A., Levy, P.E., Cooper, M.D.A., Jones, T., Gaiawyn, J., Leeson, S.R., Ward, S.E., Dinsmore, K.J., Drewer, J., Sheppard, L.J., Ostle, N.J., Evans, C.D., Burden, A., Zieliński, P. (**2013**). Methane indicator values for peatlands: a comparison of species and functional groups. Global Change Biology, **19**, 1141-1150. [DOI: 10.1111/gcb.12120]

Gremli, R., Keller, B., Sepp, T. and Szönyi, M. (**2013**) European floods: using lessons learned to reduce risks. Zurich: Zurich Insurance Group. Available from:

http://www.zurichservices.com/ZSC/REEL.nsf/3c1c9da9492c3e18c1257305004e0e3c/5449e9cd67f9cf7cc1257bcc001b2fb7/$FILE/european-floods-using-lessons-learned-en.pdf

Grosshans, R.E. (**2014**) Cattail (*Typha* spp.) Biomass Harvesting for Nutrient Capture and Sustainable Bioenergy for Integrated Watershed Management. PhD thesis, The University of Manitoba. 274pp.

Grosshans, R.E. and Grieger, L. (**2013**) *Cattail Biomass to Energy: Commercial-scale harvesting of cattail biomass for biocarbon and solid fuel*. Winnipeg: International Institute for Sustainable Development. 32pp. Available from: http://tiny.cc/CattailBioenergy

Gunn, S. (**2020**) The New Model for Media Choice. The Commercial Greenhouse Grower Magazine, March 2020, pp.13-15.

Gunnarsson, U. (**2005**) Global patterns of Sphagnum productivity. Journal of Bryology, **27**, 269–279. [DOI: 10.1179/174328205X70029]

Günther, A., Huth, V., Jurasinski, G. & Glatzel, S. (**2015**) The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen. GCB Bioenergy, **7**, 1092-1106. [DOI: 10.1111/gcbb.12214]

Günther A., Jurasinski, G., Albrecht, K., Gaudig, G., Krebs, M., Glatzel, S. (**2017**) Greenhouse gas balance of an establishing *Sphagnum* culture on a former bog grassland in Germany. Mires and Peat, **20**, Article 02, 1–16. [DOI: 10.19189/MaP.2015.OMB.210]

Hardman, C.J., Harris, D.B., Sears, J. & Droy, N. (**2012**) Habitat associations of invertebrates in reedbeds, with implications for management. Aquatic Conservation: Marine and Freshwater Ecosystems, **22** (6), 813-826. [DOI: 10.1002/aqc.2282]

Harwood, J. and Edom, G. (**2012**) Nettle Fibre: Its Prospects, Uses and Problems in Historical Perspective. *Textile History*, **43**, 107-119. [DOI: 10.1179/174329512X13284471321244]

Hasegawa, K. (**2010**) Japan looks to ancient wisdom to save biodiversity. Available from: http://tiny.cc/JapanWisdom [Accessed 10th August 2019]

Hastings, A., Mos, M., Yesufu, J.A., McCalmont, J., Schwarz, K., Shafei, R., Ashman, C., Nunn, C., Schuele, H., Cosentino, S., Scalici, G., Scordia, D., Wagner, M. & Clifton-Brown, J. (**2017**) Economic and environmental assessment of seed and rhizome propagated *Miscanthus* in the UK. Frontiers in Plant Science, **8**, Article 1058. [DOI: 10.3389/fpls.2017.01058]

Hill, M.O., Roy, D.B., Mountford, J.O. and Bunce, R.G. (**2000**) Extending Ellenberg's indicator values to a new area: an algorithmic approach. Journal of Applied Ecology, **37** (1), pp. 3-15 [DOI: 10.1046/j.1365-2664.2000.00466.x]

Hooijer, A., Page, S., Jauhiainen, J., Lee, W.A., Lu, X.X., Idris, A., Anshari, G. (**2012**) Subsidence and carbon loss in drained tropical peatlands. Biogeosciences, **9**, 1053–1071. [DOI: 10.5194/bg-9-1053-2012]

Horn, S., Sweers, W. & Frase, T. (**2016**) Suppressing reed by grazing with Water Buffalo. *In:* W. Wichtmann, C. Schröder and H. Joosten (eds.) *Paludiculture – productive use of wet peatlands: Climate protection – biodiversity – regional economic benefits*. Schweitzerbart, Stuttgart. pp. 100-101. (ISBN: 9783510652839)

Howe, C., Suich, H., Vira, B. & Mace, G. (**2014**) Creating win-wins from trade-offs? Ecosystem services for human well-being: A meta-analysis of ecosystem service trade-offs and synergies in the real world. Global Environmental Change, **28**, 263-275. [DOI: 10.1016/j.gloenvcha.2014.07.005]

Huffman, T., Qian, B., DeJong, R., Liu, J., Wang, H., Mcconkey, B.G., Brierley, T. and Yang, J. (**2015**) Upscaling modelled crop yields to regional scale: A case study using DSSAT for spring wheat on the Canadian Prairies. *Canadian Journal of Soil Science*, **95**, 49-61. [DOI: 10.4141/CJSS-2014-076]

Humphries, S and Hopwood, B. (**1999**) *Green and Pleasant Land*. London: Channel 4 Books, Macmillan Publishers Ltd. 176pp. (ISBN: 9780752272504)

Hutchinson, J.N. (**1980**). The record of peat wastage in the East Anglian Fenlands at Hole Post, 1848-1978 AD. Journal of Ecology, **68** (1), 229-249. [DOI: 10.2307/2259253]

Institute of Civil Engineers (*ICE*) (**2018**). *Draining the Fens.* Available: https://www.ice.org.uk/what-is-civil-engineering/what-do-civil-engineers-do/draining-the-fens Last accessed 16/10/2019.

IPCC (2014). 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. Methodological Guidance on Lands with Wet and Drained Soils, and Constructed Wetlands for Wastewater Treatment. IPCC Task Force on National Greenhouse Gas Inventories.

Jain, M.S. & Kalamdhad, A.S. (**2018**) A review on management of *Hydrilla verticilatta* and its utilization as potential nitrogen-rich biomass for compost or biogas production. Bioresource Technology Reports, **1**, 69-78. [DOI: 10.1016/j.biteb.2018.03.001]

Jobin, P., Caron, J. and Rochefort, L. (**2014**) Developing new potting mixes with Sphagnum fibers. Canadian Journal of Soil Science, 2014, **94** (5), 585-593. [DOI: 10.4141/cjss2013-103]

Johnson & Johnson INC. (**1994**) Sphagnum moss composition and method for the production of sheeted absorbent, EP 0 643 955 B1, European Patent Office, Munich, Germany

Joint Nature Conservation Committee (**2011**) *Towards an assessment of the state of UK Peatlands, JNCC report No. 445*. Peterborough: Joint Nature Conservation Committee. Available from: http://tiny.cc/JNCC445

Joosten, H. & Clarke D. (**2002**) Wise use of mires and peatlands – Background and principles including a framework for decision-making. IMCG / International Peat Society. (ISBN: 9519774483) Available from: http://tiny.cc/wump

Joosten, H., Tapio-Biström, M.-L. and Tol, S. (eds.) (**2012**) *Peatlands - guidance for climate change mitigation through conservation, rehabilitation and sustainable use, 2nd edition. Mitigation of Climate Change in Agriculture Series 5.* Rome: Food and Agriculture Organization of the United Nations and Wetlands International. 100pp. Available from: http://www.fao.org/3/a-an762e.pdf

Joosten, H., Couwenberg, J., von Unger, M. (**2016**) International Carbon Policies as a new driver for peatland restoration. IN: Aletta Bonn, Tim Allott, Martin Evans, Hans Joosten and Rob Stoneman. Peatland Restoration and Ecosystem Services – Science, Policy and Practice, Cambridge University Press, Cambridge, United Kingdom (ISBN: 9781107619708)

Jorat, M.E., Kreiter, S., Mörz, T., Moon, V., de Lange, W. (2013) Strength and compressibility characteristics of peat stabilized with sand columns. Journal of Geomechanics and Engineering, 5, 575-94. [DOI: 10.12989/gae.2013.5.6.575]

Juutinen, S., Alm, J., Larmola, T., Saarnio, S., Martikainen, P.J. & Silvola, J. (**2004**) Stand-specific diurnal dynamics of CH4 fluxes in boreal lakes: patterns and controls. Journal of Geophysical Research, **109**, 1–11. [DOI: 10.1029/2004JD004782]

Kämäräinen, A., Simojoki, A., Lindén, L., Jokinen, K. and Silvan, N. (**2018**). Physical growing media characteristics of *Sphagnum* biomass dominated by *Sphagnum fuscum* (Schimp.) Klinggr. *Mires and Peat*, **21** (2018), Article 17, 1-16. [DOI: 10.19189/MaP.2017.OMB.278]

Kandel, T.P., Elsgaard, L., Karki, S., Laerke, P.E. (**2013**). Biomass yield and greenhouse gas emissions from a drained fen peatland cultivated with reed canary grass under different harvest and fertilizer regimes. Bioenergy Research, **6**, 883-895. [DOI: 10.1007/s12155-013-9316-5]

Kandel, T.P., Karki, S., Elsgaard, L., Laerke, P.E. (**2013**). Fertilizer-induced fluxes dominate annual N_2O emissions from a nitrogen-rich temperate fen rewetted for paludiculture. Nutrient Cycling in Agroecosystems, **115**, 57–67. [DOI: www.10.1007/s11104-014-2164-z]

Karki, S., Elsgaard, L. & Laerke, P.E. (**2015**) Effect of reed canary grass cultivation on greenhouse gas emission from peat soil at controlled rewetting. Biogeosciences, **12**, 595-606 [DOI: 10.5194/bg-12-595-2015]

Karki, S., Elsgaard, L., Kandel, T. & Lærke, P.E. (**2016**) Carbon balance of rewetted and drained peat soils used for biomass production: a mesocosm study. GCB Bioenergy, **8** (5), 969–980. [DOI: 10.1111/gcbb.12334]

Karstens, S., Inácio, M. and Schernewski, G. (**2019**) Expert-based evaluation of ecosystem service provision in coastal reed wetlands under different management regimes. Frontiers in Environmental Science **7**, 63. [DOI: 10.3389/fenvs.2019.00063]

Kiesel, A. & Lewandowski, I. (**2017**) Miscanthus as biogas substrate - cutting tolerance and potential for anaerobic digestion. GCB Bioenergy, **9**, 153-167. [DOI: 10.1111/gcbb.12330]

Köbbing, J.F., Thevs, N. & Zerbe, S. (**2013**) The utilisation of reed (*Phragmites australis*): a review, Mires and Peat, **13**, Article 01, 1–14. ISSN 1819-754X. Available from: http://tiny.cc/map_13_01

Kotowski, W., van Andel, J., van Diggelen, R. and Hogendorf, J. (**2001**) Responses of fen plant species to groundwater level and light intensity. *Plant Ecology*, **155**, 147-156. [DOI: 10.1023/A:1013214716842]

Kotowski, W. and van Diggelen, R. (**2004**) Light as an environmental filter in fen vegetation. *Journal of Vegetation Science*, **15**, 583-594. [DOI: 10.1111/j.1654-1103.2004.tb02300.x]

Krebs, M., Gaudig, G. & Joosten, H. (**2016**) Record growth of Sphagnum papillosum in Georgia (Transcaucasus): rain frequency, temperature and microhabitat as key drivers in natural bogs. Mires and Peat, **18** (4), 1–16. [DOI: 10.19189/MaP.2015.OMB.190]

Krebs, M., Gaudig, G., Matchutadze, I. & Joosten, H. (**2018**) Sphagnum regrowth after cutting. Mires and Peat, **20** (12), 1–20. [DOI: 10.19189/MaP.2017.OMB.298]

Krenn, L., Beyer, G., Pertz, H.H., Karall, E., Kresmer, M., Galambosi, B. and Melzig, M.F. (**2004**) In vitro antispasmodic and anti-inflammatory effects of *Drosera rotundifolia*. *Arzneimittel-Forschung/Drug Research*, **54** (7), 402-405. [DOI: 10.1055/s-0031-1296991]

Kuhlman, T., Diogo, V. & Koomen, E. (**2013**) Exploring the potential of reed as a bioenergy crop in the Netherlands. Biomass and Bioenergy, **55**, 41-52. [DOI: 10.1016/j.biombioe.2012.06.024]

Kumar, S. (**2017**) *Sphagnum* moss as a growing media constituent: some effects of harvesting, processing and storage. *Mires and Peat*, **20** (2017/18), Article 07, 1-11. [DOI: 10.19189/MaP.2016.OMB.232]

Lamers, L.P.M., Vile, M.A., Grootjans, A.P., Acreman, M.C., van Diggelen, R., Evans, M.G., Richardson, C.J., Rochefort, L., Kooijman, A.M., Roelofs, J.G.M. & Smolders, A.J.P. (**2015**) Ecological restoration of rich fens in Europe and North America: from trial and error to an evidence-based approach. Biological Reviews of the Cambridge Philosophical Society, **90** (1), 182–203. [DOI: 10.1111/brv.12102]

Lewandowski, I., Clifton-Brown, J.C., Scurlock, J.M.O. and Huisman, W. (**2000**) Miscanthus: European experience with a novel energy crop. Biomass and Bioenergy, **19**, 209–227 [DOI: 10.1016/S0961-9534(00)00032-5]

Li, Z., Liu, Z., Anderson, W., Yang, P., Wu, W., Tang, H. and You, L. (**2015**). Chinese rice production area adaptations to climate changes, 1949–2010. Environmental Science & Technology, **49**, 2032-2037. [DOI: 10.1021/es505624x]

Liu, Z., Yang, P., Tang, H., Wu, W., Zhang, L., Yu, Q. and Li, Z. (**2015**). Shifts in the extent and location of rice cropping areas match the climate change pattern in China during 1980–2010. Regional Environmental Change, **15**, 919-929. [DOI 10.1007/s10113-014-0677-x]

Liu, Z., Zhang, G. and Yang, P. (**2016**). Geographical variation of climate change impact on rice yield in the rice-cropping areas of Northeast China during 1980–2008. Sustainability, **8**, 670 [DOI: 10.3390/su8070670]

Lindegaard, K. (**2013**) Wetland Biomass to Bioenergy: Efficient harvesting, processing and conversion of wetland biomass. Phase 1 Report for Department of Energy & Climate Change (DECC) by Crops for Energy Ltd. Available from: http://tiny.cc/CfE_WBB

Lindsay, R.A and Immirzi, C.P. (**1996**) *An inventory of lowland raised bogs in Great Britain. Scottish Natural Heritage Research, Survey and Monitoring Report, No.78*. Perth: Scottish Natural Heritage. 84pp. Available from: https://repository.uel.ac.uk/item/86q77

Lloyd, C.R. (**2006**) Annual carbon balance of a managed wetland meadow in the Somerset Levels, UK. Agricultural and Forest Meteorology, **138** (1-4), 168-179. [DOI: 10.1016/j.agrformet.2006.04.005]

Łuczaj, Ł.J., Dumanowski, J., Köhler, P. and Mueller-Bieniek, A. (**2012**) The use and economic value of manna grass (Glyceria) in Poland from the middle ages to the twentieth century. Human Ecology, **40** (5), pp.721-733. [DOI: 10.1007/s10745-012-9513-4]

Lukac, M., Pensa, M. and Schiller, G. (**2011**) Tree species' tolerance to water stress, salinity and fire. In: Bredemeier, M., Cohen, S., Godbold, D. L., Lode, E., Pichler, V. and Schleppi, P. (eds.) Forest Management and the Water Cycle: An Ecosystem-Based Approach. Ecological Studies (212). Springer, pp. 247-262. (ISBN: 9789048198337)

Mander, Ü., Järveoja, J., Maddison, M., Soosaar, K., Aavola, R., Ostonen, I., Salm, J.O. (2012). Reed canary grass cultivation mitigates greenhouse gas emissions from abandoned peat extraction areas. GCB Bioenergy, 4, 462-74. [DOI: 10.1111/j.1757-1707.2011.01138.x]

Mann, J.J., Barney, J.N., Kyser, G.B. & Di Tomaso, J.M. (**2013**) *Miscanthus x giganteus* and *Arundo donax* shoot and rhizome tolerance of extreme moisture stress. GCB Bioenergy, **5**, 693-700. [DOI: 10.1111/gcbb.12039]

Martin-Ortega, J., Allott, T.E.H., Glenk, K. & Schaafsma, M. (**2014**) Valuing water quality improvements from peatland restoration: Evidence and challenges. Ecosystem Services, **9**, 34-43. [DOI: 10.1016/j.ecoser.2014.06.007]

Mathiassen, S.K., Bak, T., Christensen, S. and Kudsk, P. (**2006**) The Effect of Laser Treatment as a Weed Control Method. *Biosystems Engineering*, **95** (4), 497–505. [DOI: 10.1016/j.biosystemseng.2006.08.010]

McBride, A., Diack, I., Droy, N., Hamill, B., Jones, P., Schutten, J., Skinner, A. and M. Street, M. (eds.) (**2011**) *The Fen Management Handbook*. Perth: Scottish Natural Heritage. Available from: http://tiny.cc/FMHandbook

Millennium Ecosystem Assessment (**2005**) Ecosystems and Human Well-Being: Wetlands and Water Synthesis. World Resources Institute, Washington. Available from: http://tiny.cc/MEA_EHWB

Mills, S. (**2016**) Wetland Conservation Biomass to Bioenergy End User Report. RSPB (Funded by Department of Energy & Climate Change, DECC). Available from: http://tiny.cc/RSPB_WCBB

Morris J., Graves, A., Angus, A., Hess, T., Lawson, C., Camino, M., Truckell, I. and Holman, I. (**2010**). *Restoration of Lowland Peatland in England and Impacts on Food Production and Security.* Report to Natural England. Bedford: Cranfield University. 153pp.

Mouchet, M.A., Lamarque, P., Martín-López, B., Crouzat, E., Gos, P., Byczek, C. & Lavorel, S. (**2014**) An interdisciplinary methodological guide for quantifying associations between ecosystem services. Global Environmental Change, **28**, 298-308. [DOI: 10.1016/j.gloenvcha.2014.07.012]

Moxey, A. & Moran, D. (**2014**) UK peatland restoration: Some economic arithmetic. Science of the Total Environment, **484**, 114-120. [DOI: 10.1016/j.scitotenv.2014.03.033]

Mulholland, B.J., Watson, A. and Carter, S. (**2019**) Transition to responsibly sourced growing media use within UK horticulture (Project CP 138). Interim project report, AHDB. Available from: http://tiny.cc/CP138

Müller, J. and Sweers, W. (**2016**) The production of fodder in paludiculture. In: W. Wichtmann, C. Schröder and H. Joosten (eds.) *Paludiculture – productive use of wet peatlands. Climate protection – biodiversity – regional economic benefits*. pp. 39-42. Stuttgart: Schweitzerbart Science Publishers.

Muster, C., Gaudig, G., Krebs, M. & Joosten, H. (**2015**) Sphagnum farming: the promised land for peat bog species? Biodiversity and Conservation, **24** (8), 1989-2009. [DOI: 10.1007/s10531-015-0922-8]

Myhre, G., Shindell, D. Bréon, Collins, W. JFuglestvedt, J. Huang, J., Koch, D., Lamarque, J-F., Lee, D. Mendoza, B. Nakajima, T., Robock, Stephens, A. G., Takemura, T., Zhang, H. (2013): Anthropogenic and Natural Radiative Forcing. In: Climate Change 2013: The Physical Science Basis. Contribution of

Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Nagengast, A., Hendrickson, C. and Matthews, H.S. (**2013**) Variations in photovoltaic performance due to climate and low-slope roof choice. *Energy Build*, **64**, 493 502. [DOI: 10.1016/j.enbuild.2013.05.009]

NFU (**2019**) Delivering for Britain: Food and Farming in the Fens. NFU, East Anglia. Available from: http://tiny.cc/NFU_DfB

Nichols, G.E. (**1918**) The American Red Cross Wants Information. Regarding supplies of surgical Sphagnum. The Bryologist, **21**, 81-83. Available from: https://www.jstor.org/stable/pdf/3238750.pdf

Nowotny, R. (**2016**) Blown-in insulation material from Cattail. In: W. Wichtmann, C. Schröder and H. Joosten (eds.) *Paludiculture – productive use of wet peatlands. Climate protection – biodiversity – regional economic benefits*. p.36. Stuttgart: Schweitzerbart Science Publishers. (ISBN: 978-3-510-65283-9)

ONS (**2019**) UK natural capital: peatlands. Office for National Statistics. Available from: http://tiny.cc/ONSPeatNC

Oehmke, C. and Abel, S. (**2016**) Promising plants for paludiculture. In: W. Wichtmann, C. Schröder and H. Joosten (eds.) *Paludiculture – productive use of wet peatlands. Climate protection – biodiversity – regional economic benefits*. pp. 22-38. Stuttgart: Schweitzerbart Science Publishers. (ISBN: 978-3-510-65283-9)

Opelt, K., Berg, C. and Berg, G. (**2007**) The bryophyte genus Sphagnum is a reservoir for powerful and extraordinary antagonists and potentially facultative human pathogens. FEMS Microbiology Ecology, **61** (1), pp.38-53. [DOI: 10.1111/j.1574-6941.2007.00323.x]

Ostendorp, W. (**1999**) Management impacts on stand structure of lakeshore *Phragmites* reeds. International Review of Hydrobiology, **84** (1), 33–47. [DOI: 10.1002/iroh.199900006]

Overbeck, F. & Happach, H. (**1957**) Über das Wachstum und den Wasserhaushalt einiger Hochmoorsphagnen (On the growth and water budget of some raised bog Sphagnum species). Flora oder Allgemeine Botanische Zeitung, **144**, 335–402 (*in German*). [DOI: 10.1016/S0367-1615(17)31396-4]

Packer, J.G., Meyerson, L.A., Skálová, H., Pysek, P. & Kueffer, C. (**2017**) Biological Flora of the British Isles: *Phragmites australis*. Journal of Ecology, **105** (4), 1123-1162. [DOI: 10.1111/1365-2745.12797]

Paper, D.H., Karall, E., Kremser, M. and Krenn, L. (**2005**) Comparison of the anti-inflammatory effects of *Drosera rotundifolia* and *Drosera madagascariensis* in the HET-CAM assay. *Phytotherapy Research*, **19**, 323-326. [DOI: 10.1002/ptr.1666]

Peacock, M., Gauci, V., Baird, A. J., Burden, A., Chapman, P. J., Cumming, A., Evans, J.G., Grayson, R.P., Holden, J., Kaduk, J., Morrison, R., Page, S., Pan, G., Ridley, L.M., Williamson, J., Worrall, F. and Evans, C. D. (**2019**). The full carbon balance of a rewetted cropland fen and a conservation-managed fen. *Agriculture, Ecosystems and Environment*, **269**, 1–12. [DOI: 10.1016/j.agee.2018.09.020]

Page, S., Baird, A., Cumming, A., High, K., Kaduk, J. and Evans, C. (**2020**) Scoping study: Assessment of the current societal impacts of water level management on lowland peatlands in England and Wales. Project code: Defra SP1218.

Pärn, J., Verhoeven, J.T., Butterbach-Bahl, K., Dise, N., Ullah, S., Aasa, A., Egorov, S., Espenberg, M., Järveoja, J., Jauhiainen, J., Kasak, K. (**2018**). Nitrogen-rich organic soils under warm well drained conditions are global nitrous oxide emission hotspots, Nature Communications, **9**, 1135. [DOI: 10.1038/s41467-018-03540-1]

Peh, K.S-H., Balmford, A., Field, R.H., Lamb, A., Birch, J.C., Bradbury, R.B., Brown, C., Butchart, S.H.M., Lester, M., Morrison, R., Sedgwick, I., Soans, C., Stattersfield, A.J., Stroh, P.A., Swetnam, R.D., Thomas, D.H.L., Walpole, M., Warrington, S. & Hughes, F.M.R. (**2014**) Benefits and costs of ecological restoration: Rapid assessment of changing ecosystem service values at a U.K. wetland. Ecology and Evolution, **4** (20), 3875– 3886. [DOI: 10.1002/ece3.1248]

Philips, D., Mitchell, E.J.S., Lea-Langton, A.R., Parmar, K.R., Jones, J.M. & Williams, A. (**2017**) The use of conservation biomass feedstock as potential bioenergy resources in the United Kingdom. Bioresource Technology, **212**, 271-279. [DOI: 10.1016/j.biortech.2016.04.057]

Piao, S., Ciais, P., Huang, Y., Shen, Z., Peng, S., Li, J., Zhou, L., Liu, H., Ma, Y., Ding, Y., Friedlingstein, P., (**2010**). The impacts of climate change on water resources and agriculture in China. Nature, **467**, 43-51.

Plappally, A. K., & Lienhard V, J. H. (**2012**). Energy requirements for water production, treatment, end use, reclamation, and disposal. Renewable and Sustainable Energy Reviews, **16**, 4818–4848. [DOI: 10.1016/j.rser.2012.05.022]

Poulin, B., Lefebvre, G., Allard, S. & Mathevet, R. (**2009**) Reed harvest and summer drawdown enhance bittern habitat in the Camargue. Biological Conservation, **142** (3), 689-695. [DOI: 10.1016/j.biocon.2008.11.020]

Pouliot, R., Hugron, S. & Rochefort, L. (**2015**) Sphagnum farming: A long-term study on producing peat moss biomass sustainably. Ecological Engineering, **74**, 135-147. [DOI: 10.1016/j.ecoleng.2014.10.007]

Qi, A., Holland, R.A., Taylor, G. and Richter, G.M. (**2018**) Grassland futures in Great Britain – Productivity assessment and scenarios for land use change opportunities. *Science of the Total Environment*, **634**, 1108-1118. [DOI: 10.1016/j.scitotenv.2018.03.395]

Rackham, O. (**1986**) *The History of the Countryside – the classic history of Britain's landscape, flora and fauna*. London: J M Dent & Sons. 445pp. (ISBN: 978-1842124406)

Ramsar (**1994**) Convention on Wetlands of International Importance especially as Waterfowl Habitat. The Ramsar Convention Secretariat. Available from: http://tiny.cc/Ramsar94

Raabe, S. & Manthey, M. (**2016**) The effect of mowing on plants. *In:* W. Wichtmann, C. Schröder and H. Joosten (eds.) *Paludiculture - productive use of wet peatlands: Climate protection – biodiversity – regional economic benefits.* Schweitzerbart, Stuttgart. pp. 96-98. (ISBN: 9783510652839)

Rebaque, D., Martínez-Rubio, R., Fornalé, S., García-Angulo, P., Alonso-Simón, A., Álvarez, J.M., Caparros-Ruiz, D., Acebes, J.L. and Encina, A. (**2017**) Characterization of structural cell wall polysaccharides in cattail (Typha latifolia): Evaluation as potential biofuel feedstock. *Carbohydrate Polymers*, **175**, 679-688. [DOI: 10.1016/j.carbpol.2017.08.021]

Ren, L., Eller, F., Lambertini, C., Guo, W-Y., Brix, H., Sorrell, B.K. (2019) Assessing nutrient responses and biomass quality for selection of appropriate paludiculture crops. *Science of the Total Environment* **664**, 1150-1161. [DOI: http://doi.org/10.1016/j.scitotenv.2019.01.419]

Riegler, N.N. (**1989**) Sphagnum moss in World War I: The making of surgical dressings by volunteers in Toronto, Canada, 1917-1918. Canadian Bulletin of Medical History, **6** (1), pp.27-43. [DOI: 10.3138/cbmh.6.1.27]

Ringler, C. and Cai, X. (**2006**) Valuing fisheries and wetlands using integrated economic-hydrologic modeling: Mekong River basin. Journal of Water Resources Planning and Management-ASCE, **132** (6), 399-525. [DOI: 10.1061/(ASCE)0733-9496(2006)132:6(480)]

Roca, M., Sayers, P., Bast, H., Flikweert, J., Panzeri, M., Ogunyoye, F., Hess, T. and Young, R. (**2011**) *Developing the evidence base to describe the flood risk to agricultural land in England and Wales. Joint Defra/EA Flood and Coastal Erosion Risk Management R&D Programme, R&D Technical Report FD2634/TR*. London: Defra. 105pp. Available from: http://tiny.cc/FD2634

Robertson, A.D., Whitaker, J., Morrison, R., Davies, C.A., Smith, P. & McNamara, N.P. (**2017**) A *Miscanthus* plantation can be carbon neutral without increasing soil carbon stocks. GCB Bioenergy, **9** (3), 645-661. [DOI: 10.1111/gcbb.12397]

Röder, M., Whitaker, C. & Thornley, P. (**2015**) How certain are greenhouse gas reductions from bioenergy? Life cycle assessment and uncertainty analysis of wood pellet-to-electricity supply chains from forest residues. Biomass and Bioenergy, **79**, 50-63. [DOI: 10.1016/j.biombioe.2015.03.030]

Rotherham, I.D. (**2013**) *The Lost Fens – England's greatest ecological disaster*. Stroud: The History Press. 207pp. (ISBN: 9780752486994)

Salo, K. (**1996**) Peatland berries – a valuable nourishing resource. In: H. Vasander (ed.) *Peatlands in Finland*. Helsinki: Finnish Peatland Society, 39-44. (ISBN: 952-90-7971-0)

Sawada, H., Matsuyama, H., Matsunaka,, H., Fujita, M., Okamura, N., Seki, M., Kojima, H., Kiribuchi-Otobe, C., Takayama, T., Oda, S., Nakamura, K., Sakai, T., Matsuzaki, M. and Kato, K. (**2019**) Evaluation of dry matter production and yield in early-sown wheat using near-isogenic lines for the vernalization locus Vrn-D1. *Plant Production Science*, **22** (2), 275-284. [DOI: 10.1080/1343943X.2018.1563495]

Schröder, C. and Schoeder, P. (**2016**) Potential areas in Mecklenburg-West Pommerania. In: W. Wichtmann, C. Schröder and H. Joosten (eds.) *Paludiculture – productive use of wet peatlands: climate protection – biodiversity – regional economic benefits.* Stuttgart: Schweitzerbart Science Publishers, 180-182. (ISBN: 978-3-510-65283-9)

Setiadi, B. (**2014**) Aquaculture and tropical peatland fishery. In: R. Biancalani and A. Avagyan (eds.) *Towards climate-responsible peatlands management. Mitigation of Climate Change in Agriculture Series 9.* Rome: Food and Agriculture Organization of the United Nations, 58-60. Available from: http://www.fao.org/3/a-i4029e.pdf

Sheail, J. & Wells, T.C.E. (**1983**) The fenlands of Huntingdonshire, England. A case study in catastrophic change. *In:* Mires: swamp, bog, fen and moor. Ecosystems of the world 4B (ed. by. A.J.P. Gore), Amsterdam: Elsevier, pp. 375-393

Sheenan, S. (**2018**) *The First Farmers of Europe – an evolutionary perspective*. Cambridge: Cambridge University Press. 266pp. [DOI: 10.1017/9781108386029]

Shiono, K., Ejiri, M., Shimizu, K. and Yamada, S. (**2019**) Improved waterlogging tolerance of barley (*Hordeum vulgare*) by pretreatment with ethephon. *Plant Production Science*, **22** (2), 285-295. [DOI: 10.1080/1343943X.2019.1581579]

Shurpali, N., Hyvönen, N., Huttunen, J.T., Clement, R.T., Reichstein, M., Nykänen, H., Biasi, C., Martikainen, P.J. (**2009**). Cultivation of a perennial grass for bioenergy on a boreal organic soil – carbon sink or source? GCB Bioenergy, 1, 35-50. [DOI: 10.1111/j.1757-1707.2009.01003.x]

Silvestri, N., Giannini, V., Dragoni, F. & Bonari, E. (**2017**) A multi-adaptive framework for the crop choice in paludiculture cropping systems. Italian Journal of Agronomy, **12** (1), 69-76. [DOI: 10.4081/ija.2016.734]

Skoglund, P., Malmström, H., Raghavan, M., Storå, J., Hall, P., Willerslev, E., Gilbert, M.T.P., Götherström, A. and Jakobsson, M. (**2012**) Origins and Genetic Legacy of Neolithic Farmers and Hunter-Gatherers in Europe. *Science*, **336** (6080), pp. 466-469. [DOI: 10.1126/science.1216304]

Smith, P., Ashmore, M., Black, H., Burgess, P.J., Evans, C., Quine, T., Thomson, A.M. & Hicks, K. (**2012**) REVIEW: The role of ecosystems in regulating climate, and soil, water and air quality. Journal of Applied Ecology, **50** (4), 812–829. [DOI: 10.1111/1365-2664.12016]

Smolders, A.J.P., van Diggelen, J.M.H., Geurts, J.J.M., Poelen, M.D.M., Roelofs, J.G.M., Lucassen, E.C.H.E.T. and Lamers, L.P.M. (**2013**) Waterkwaliteit in het veenweidegebied. Landschap, **30** (3), 145-153 (*in German*). Available from: https://www.landschap.nl/wp-content/uploads/2013-3_145-153.pdf

Stalheim, T., Ballance, S., Christensen, B.E. and Granum, P.E. (**2009**) Sphagnan–a pectin-like polymer isolated from Sphagnum moss can inhibit the growth of some typical food spoilage and food poisoning bacteria by lowering the pH. Journal of applied microbiology, **106** (3), pp.967-976 [DOI: 10.1111/j.1365-2672.2008.04057.x]

Stevens, A.J., Clarke, D. and Nicholls, R.J. (**2016**) Trends in reported flooding in the UK: 1884–2013, *Hydrological Sciences Journal*, **61** (1), 50–63. [DOI: 10.1080/02626667.2014.950581]

Stokes, J.R., Alspach, P.A. & Stanley, J. (**1999**) Effect of water table on growth of three New Zealand Sphagnum species: implications for S. cristatum management. Journal of Bryology, **21** (1), 25–29. [DOI: 10.1179/jbr.1999.21.1.25]

Strašil, Z. (**2012**) Evaluation of reed canary grass (*Phalaris arundinacea L.*) grown for energy use. Res. Agr. Eng., **58**, 119–130. [DOI: 10.17221/35/2011-RAE]

Taft, H. E., Cross, P. A., Edwards-Jones, G., Moorhouse, E. R., & Jones, D. L. (**2017**) Greenhouse gas emissions from intensively managed peat soils in an arable production system. Agriculture, Ecosystems and Environment, **237**, 162–172. https://doi.org/10.1016/j.agee.2016.11.015

Tanneberger, F., Gatkowski, D. & Krogulec, J. (**2016**) Poland – Paludiculture for biodiversity and peatland protection. *In:* W. Wichtmann, C. Schröder and H. Joosten (eds.) *Paludiculture - productive use of wet peatlands: Climate protection – biodiversity – regional economic benefits*. Schweitzerbart, Stuttgart. pp. 207-208. (ISBN: 9783510652839)

Te-Tzu Chang (**1987**) The Impact of Rice on Human Civilization and Population Expansion, Interdisciplinary Science Reviews, **12** (1), 63-69. [DOI: 10.1179/isr.1987.12.1.63]

Thomson, A., Misselbrook, T., Moxley, J., Buys, G., Evans, C., Malcolm, H., Whitaker, J., McNamara, N. and Reinsch, S. (**2018**) Quantifying the impact of future land use scenarios to 2050 and beyond – Final Report (CEH and Rothamsted Research). Available from: http://tiny.cc/Quant2050

Timmermann, T. (**2003**) Nutzungsmöglichkeiten der Röhrichte und Riede nährstoffreicher Moore Mecklenburg-Vorpommerns, Greifswalder Geographische Arbeiten. (in German)

UKCIP (**2010**) UK Climate Impacts Programme. Available from: http://www.ukcip.org.uk

van der Hilst, F., Dornburg, V., Sanders, J.P.M., Elbersen, B., Graves, A., Turkenburg, W.C., Elbersen, H.W., van Dam, J.M.C. & Faaji, A.P.C. (**2010**) Potential, spatial distribution and economic performance of regional biomass chains: The North of the Netherlands as example. Agricultural Systems, **103** (7), 403-417. [DOI: 10.1016/j.agsy.2010.03.010]

van de Riet, B. & Hefting, M. (**2013**) Rewetting drained peat meadows: risks and benefits in terms of nutrient release and greenhouse gas exchange. Water, Air & Soil Pollution, **224**, 1440. [DOI: 10.1007/s11270-013-1440-5]

Verdonck, O., Penninck, R and De Boot, M. (**1983**) Physical properties of different horticultural substrates. Acta Horticulturae, 150 – pp. 155-160. [DOI: 10.17660/ActaHortic.1984.150.16]

Vroom, R.J.E., Xiea, F., Geurtsa, J.J.M., Chojnowskaa, A., Smolders, A.J.P., Lamersa, L.P.M. & Fritza, C. (**2018**) *Typha latifolia* paludiculture effectively improves water quality and reduces greenhouse gas emissions in rewetted peatlands. Ecological Engineering, **124**, 88-98. [DOI: 10.1016/j.ecoleng.2018.09.008]

Watanabe, T., Okuyama, M. & Fukamachi, K. (**2012**) A Review of Japan's Environmental Policies for *Satoyama* and *Satoumi* Landscape Restoration. Global Environmental Research, **16**, 125-135. Available from: http://tiny.cc/GER_125

Welsh Government (**2019**) Farming Facts and Figures, Wales 2019. Cardiff: Ystadegau ar gyfer Cymru/Statistics for Wales. Available from: http://tiny.cc/WalesFFF2019

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E. and Weidema, B. (**2016**). The ecoinvent database version 3 (part 1): overview and methodology. *The International Journal of Life Cycle Assessment*, **21,** 1218-1230. [DOI: 10.1007/s11367-016-1087-8]

Whiting, G.J. & Chanton, J.P. (**1996**) Control of the diurnal pattern of methane emission from emergent aquatic macrophytes by gas transport mechanisms, Aquatic Botany, **54** (2-3), 237–253. [DOI: 10.1016/0304-3770(96)01048-0]

Wichmann, S. (**2012**) Paludiculture - ecosystem services of Sphagnum farming on rewetted bogs in NW Germany. 14th International Peat Congress, Stockholm, Sweden, extended abstract No. 369. Available from: http://tiny.cc/IPC14_369

Wichmann, S. (**2017**) Commercial viability of paludiculture: A comparison of harvesting reeds for biogas production, direct combustion, and thatching. Ecological Engineering, **103** (Part B), 497-505. [DOI: 10.1016/j.ecoleng.2016.03.018]

Wichmann, S. and Köbbing, J.F. (**2015**) Common reed for thatching–A first review of the European market. Industrial Crops and Products, **77**, 1063–1073. [DOI: 10.1016/j.indcrop.2015.09.027]

Wichmann, S. (**2018**) Economic incentives for climate smart agriculture on peatlands in the EU. Greifswald Mire Centre Report

Wichtmann, W. & Joosten, H. (**2007**). Paludiculture: peat formation and renewable resources from rewetted peatlands. IMCG Newsletter, 24-28. Available from: http://www.imcg.net/media/newsletter/nl0703.pdf

Wichtmann, W. & Schafer, A. (**2007**) Alternative management options for degraded fens - Utilisation of biomass from rewetted peatlands. *In:* Wetlands: Monitoring, Modelling and Management, Taylor & Francis Group, London. (ISBN: 9780429082108) [DOI: 10.13140/2.1.3635.4565]

Wichtmann, W. & Tanneberger, F. (**2011**) Land use options for rewetted peatlands. *In:* Carbon Credits from Peatland Rewetting. Climate – biodiversity – land use. Schweizerbart Science Publishers, Stuttgart, Germany. pp. 107–132. (ISBN: 9783510652716)

Wichtmann, W., Schroder, C. & Joosten, H. (eds) (**2016**) Paludiculture – productive use of wet peatlands: Climate protection – biodiversity – regional economic benefits. Schweizerbart Science Publishers, Stuttgart, Germany. (ISBN: 9783510652839)

Williams, M. (**1990a**) Understanding Wetlands. In: M. Williams (ed.) *Wetlands – A Threatened Landscape*. Oxford: Basil Blackwell Ltd, 1-41. (ISBN: 978-0-631-19199-5)

Williams, M. (**1990b**) Agricultural impacts on temperate wetlands. In: M. Williams (ed.) *Wetlands – A Threatened Landscape*. Oxford: Basil Blackwell Ltd, 181-216. (ISBN: 978-0-631-19199-5)

Yates, T. (**2006**) The use of non-food crops in the UK construction industry. Journal of the Science of Food and Agriculture, **86**, 1790–1796. [DOI: 10.1002/jsfa.2608]

Yavas, I., Unay, A. and Aydin, M. (**2012**) The waterlogging tolerance of wheat varieties in western of Turkey. *Scientific World Journal*, **2012**, Article 529128. [DOI: 10.1100/2012/529128]

Zielke, L. (**2016**) Reed canary grass from re-wetted peatland as fodder for horses. In: W. Wichtmann, C. Schröder and H. Joosten (eds.) *Paludiculture – productive use of wet peatlands. Climate protection – biodiversity – regional economic benefits*. p. 34. Stuttgart: Schweitzerbart Science Publishers. (ISBN: 978-3-510-65283-9)

Appendix I Plant Species with 'energy' in 'main use categories' in the UK Paludiculture Live List.

Scientific Name	Plant Type	Vernacular Name	UK Native
Alnus glutinosa	Deciduous Tree	black alder, european alder, common alder	yes
Betula pubescens	Deciduous Tree	downy birch, white birch	yes

Carex acuta	Sedge	slender tufted sedge, slim sedge, acute sedge	yes
Carex aquatilis	Sedge	water sedge, leafy tussock sedge	yes
Carex riparia	Sedge	greater pond sedge	yes
Carex rostrata	Sedge	beaked sedge, blue sedge, bottle sedge	yes
Cladium mariscus	Sedge	great fen-sedge, saw-sedge, swamp sawgrass	yes
Fraxinus excelsior	Deciduous Tree	ash, European ash, common ash	yes
Glyceria maxima	Reed Grass	reed sweet grass, giant manna grass	yes
Hydrilla verticillata	Aquatic Herb	waterthyme, Indian star-vine	yes
Phalaris arundinacea	Reed Grass	reed canary grass	yes
Phragmites australis	Reed Grass	common reed	yes
Picea abies	Coniferous Tree	Norway spruce	Neophyte*
Pinus contorta	Coniferous Tree	beach pine	Neophyte*
Pinus strobus	Coniferous Tree	eastern white pine, northern white pine, soft pine, Weymouth pine	Neophyte*
Pinus sylvestris	Coniferous Tree	Scot's Pine	yes
Populus spp.	Deciduous Tree	poplar, aspen, cottonwood	yes
Populus tremula	Deciduous Tree	aspen, common aspen, Eurasian aspen, European aspen, trembling poplar, quaking aspen	yes
Salix spp.	Deciduous Tree	willow, sallow, osier	yes
Schoenoplectus lacustris	Sedge	common bulrush, tule, kouna	yes
Schoenoplectus tabernaemontani	Sedge	soft stem bulrush, river club rush	yes
Typha angustifolia	Cattails	lesser bulrush, narrow leaf cattail, lesser reedmace	yes
Typha latifolia	Cattails	bulrush	yes
Typha x glauca	Cattails	hybrid cattail	yes

*Neophyte refers to a non-native plant species which has been introduced in recent history.

Appendix II Literature Identified on Selected Taxa, Including Available Data

	Tree										Sedge								Cattail			Reed grass				Grass	Herb	Moss		Data		
PUBLICATION	Alnus glutinosa	Betula pubescens	Fraxinus excelsior	Picea abies	Pinus contorta	Pinus strobus	Pinus sylvestris	Populus spp.	Populus tremula	Salix spp.	Carex acuta	Carex acutaformis	Carex aquatilis	Carex riparia	Carex rostrata	Cladium mariscus	Schoenoplectus lacustris	Schoenoplectus tabernaemontani	Typha angustifolia	Typha latifolia	Typha x glauca	Arundo donax	Glyceria maxima	Phalaris arundinacea	Phragmites australis	Miscanthus X Giganteus	Hydrilla verticillata	Sphagnum spp.		Species growth & productivity	Biomass quality & energy conversion	Direct GHG emissions
Abel & Joosten (2013)	*										*	*	*	*	*	*			*	*	*			*	*					*	*	
Barz et al (2007)																									*					*	*	
Carson et al (2018)																			*		*			*						*	*	
Ciceck et al. (2006)																		*	*		*				*						*	
Dragoni et al. (2017)																									*					*	*	
Dubbe et al. (1988)																			*	*	*									*	*	
Evans & Wilkie (2010)																											*			*	*	*
Gaudig et al. (2014)																												*		*		
Giannini et al (2016)																									*					*		
Giannini et al. (2019)																				*					*					*		
Gravalos et al. (2016)							*	*											*						*						*	
Grosshans (2014)								*		*									*	*	*									*	*	
Günther et al. (2014)												*								*					*							*
Günther et al. (2015)												*								*					*					*		*
Günther et al. (2018)																												*				*
Jain & Kalamdhad (2018)																											*				*	
Karki et al. (2015)																								*						*		*
Kiesel & Lewandowski (2017)																										*				*	*	

	Tree										Sedge								Cattail			Reed grass				Grass	Herb	Moss		Data			
PUBLICATION	Alnus glutinosa	Betula pubescens	Fraxinus excelsior	Picea abies	Pinus contorta	Pinus strobus	Pinus sylvestris	Populus spp.	Populus tremula	Salix spp.	Carex acuta	Carex acutaformis	Carex aquatilis	Carex riparia	Carex rostrata	Cladium mariscus	Schoenoplectus lacustris	Schoenoplectus tabernaemontani	Typha angustifolia	Typha latifolia	Typha x glauca	Arundo donax	Glyceria maxima	Phalaris arundinacea	Phragmites australis	Miscanthus X Giganteus	Hydrilla verticillata	Sphagnum spp.		Species growth & productivity	Biomass quality & energy conversion	Direct GHG emissions	
Köbbing, Thevs & Zerbe (2013)																									*					*	*		
Kuhlman et al. (2013)																									*					*	*		
Lord (2015)										*														*		*				*	*		
Melts et al. (2019)											*	*	*	*	*			*	*	*					*					*	*		
Pouliot, Hugron & Rochefort (2015)																												*		*			
Ren et al. (2019)																				*		*			*					*	*		
Robertson et al. (2017)																										*					*	*	
Roy, Dutta & Gallant (2018)																										*		*			*		
Silvestri et al. (2015)								*		*															*	*				*	*		
Sim et al. (2011)																			*											*	*		
Strašil (2012)																								*						*			
Tho et al. (2016)																									*					*	*		
Timmermann (2003)	*											*		*						*			*	*	*					*			
Vaičekonytė et al. (2013)																									*					*	*		
Wichmann (2017)																									*					*			
Wichtmann & Joosten (2007)	*											*		*					*	*					*					*	*		
Wichtmarm and Schafer (2007)	*																			*					*					*			

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.

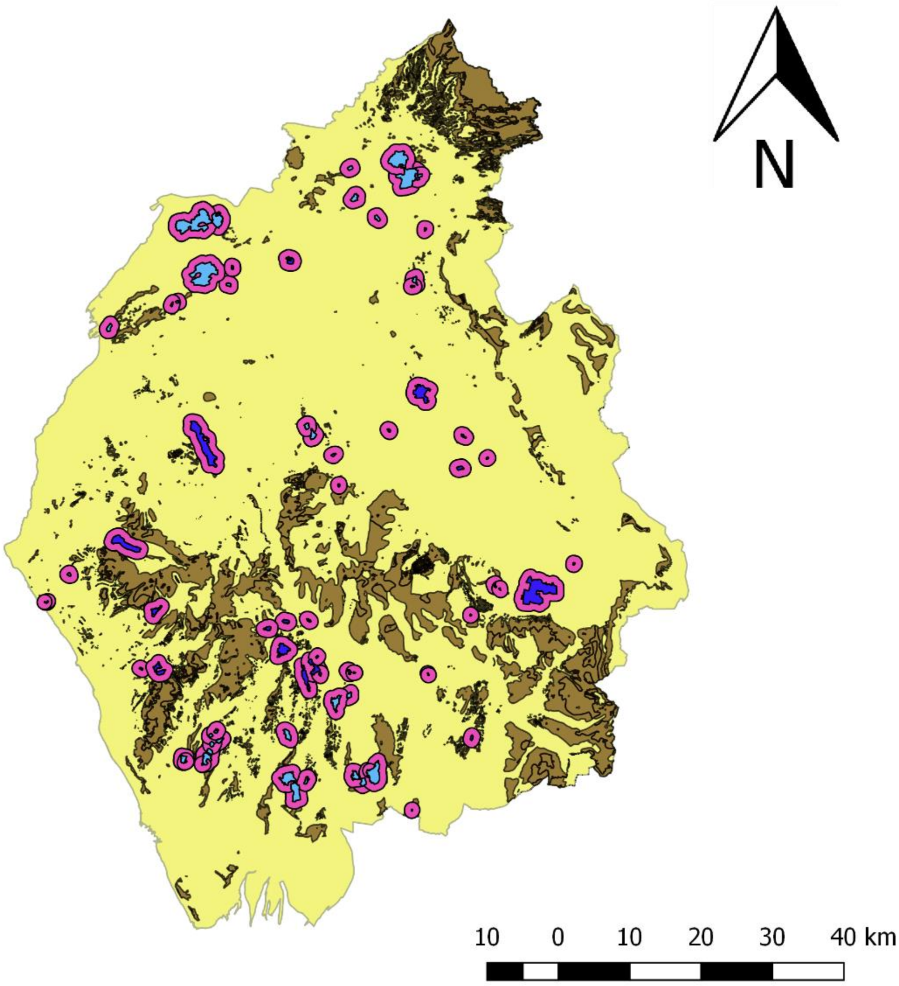

Figure 4: Basic paludiculture potential map for Cumbria – the light brown areas are peaty soils with a slope angle of 2 degrees or less – selected as suitable for paludiculture in this study. Areas of light blue are Lowland Raised Bog sites and dark blue are Lowland Fen sites. Pink shading illustrates 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).

Appendix IV Abridged Version of Current UK Paludiculture Live List

PLANT TYPE & SCIENTIFIC NAME	MAIN USE CATEGORIES					
	CONDITIONER	ENERGY	FODDER	FOOD	MEDICINE	RAW MATERIAL
ANNUAL						
Echinochloa colona **[2]** *(AgB), E. crus-galli* **[3]** *(AgB), E. frumentacea* **[2]** *(AgB); Stellaria media (PWP)*			✓			
Galium aparine (L,Se,St)			✓	✓		
Linum usitatissimum (Se,St)				✓		✓
Persicaria hydropiper (L,Se,St)				✓	✓	
AQUATIC						
Aponogeton distachyos **[3]** *(Fl,Sh,R)*				✓		✓
Lemna minor (P)			✓		✓	
Nasturtium officinale **[3]** *(L,PGP)*			✓	✓	✓	
Nymphaea alba (Fl,P,R)				✓	✓	✓
BIENNIAL						
Apium graveolens (L,Se,St)				✓	✓	
Barbarea vulgaris (L)				✓		
Digitalis purpurea (Fl,L,St)					✓	
DECIDUOUS FERN						
Pteridium aquilinum (L,St)			✓			✓
DECIDUOUS HERB						
Equisetum arvense (St), E. palustre (St)						✓
SHRUB						
Calluna vulgaris (L); Ribes rubrum (Fr); Vaccinium corymbosum **[3]** *(Fr), V. macrocarpon* **[3]** *(Fr), V. myrtillus (Fr), V. oxycoccos (Fr), V. uliginosum (Fr), V. vitis-idaea (Fr)*				✓		
TREE						
Betula pubescens (Sa,W)		✓		✓		✓
Salix spp. (AgB,Ba,St)		✓			✓	✓
Sambucus canadensis **[3]** *(Fr), S. nigra (Fr)*				✓		
Juniperus communis (Fr,W)				✓	✓	✓
Rhamnus cathartica (Ba,Fr)					✓	
Frangula alnus (Ba,W)					✓	✓
Alnus glutinosa (AgB,W); Fraxinus excelsior (W); Picea abies **[3]** *(W); Pinus contorta* **[3]** *(W), P. strobus* **[3]** *(W), P. sylvestris (W); Populus spp. (W), P. tremula (W)*		✓				✓

PLANT TYPE & SCIENTIFIC NAME	MAIN USE CATEGORIES					
	CONDITIONER	ENERGY	FODDER	FOOD	MEDICINE	RAW MATERIAL
PERENNIAL						
Acorus calamus **[3]** *(L,R); Agrimonia eupatoria (L); Althaea officinalis (L,R); Caltha palustris (L); Chamaemelum nobile; Eupatorium cannabinum (L,R); Filipendula ulmaria (Fl,L,R); Hypericum elodes (L); Ledum palustre* **[3]** *(L); Lycopus europaeus (PPGP); Mentha pulegium (L); Menyanthes trifoliata (PPGP); Oenanthe aquatica (Fr); Petasites hybridus (L,R); Pulicaria dysenterica (AgB); Stachys palustris (AgB)*					√	
Agrostis canina (AgB), A. gigantea **[1]** *(AgB), A. stolonifera (AgB); Alopecurus geniculatus (AgB); Carex acutiformis (L), C. nigra (L); Cynodon dactylon (AgB); Juncus gerardii (AgB); Lotus pedunculatus (AgB); Plantgao sp. (AgB); Trifolium fragiferum (AgB); Tripolium pannonicum subsp. tripolium (AgB)*			√			
Allium ursinum (Fl,L); Armoracia rusticana **[1]** *(R); Butomus umbellatus (R); Empetrum nigrum (Fr); Mentha sp. (L); Pesicaria bistorta (L); Potentilla anserina (R); Rorippa nasturtium-aquaticum (L); Rumex acetosa (L); Veronica beccabunga (L)*				√		
Angelica archangelica **[3]** *(L,R,Se); Ledum palustre subsp. groenlandicum* **[3]** *(L); Mentha aquatica (L); Ribes nigrum* **[3]** *(Fr,L)*				√	√	
Carex acuta (AgB); Cladium mariscus (AgB); Phragmites australis (AgB); Schoenoplectus lacustris (St), S. tabernaemontani (AgB,St); Typha angustifolia (AgB,Se), T. latifolia (AgB)		√				√
Carex aquatilis (AgB), C. riparia (AgB)		√				
Carex rostrata (AgB); Glyceria maxima (AgB); Phalaris arundinacea (AgB)		√	√			
Drosera rotundifolia (L,St); Myrica gale (Fr,L); Symphytum officinale (L,R)					√	√
Eriophorum angustifolium (Fl,P,St), E. latifolium (Fl,P,St), E. vaginatum (Fl,P,St); Juncus acutus (Fl,St), J. effusus (St); Saponaria officinalis **[1]** *(L)*						√
Glyceria fluitans (L,Se)			√	√		
Hierochloe odorata (AgB,L,St); Iris pseudacorus (Fl,P,R,Se)				√	√	√
Hydrilla verticillata (Bi)	√	√	√			
Sphagnum spp. (Bi)	√					
Typha x glauca (AgB,Se)	√	√				√
Urtica dioca (AgB)				√		√
All Native to UK except: ***[1]*** *= Archaeophyt;* ***[2]*** *= Casual;* ***[3]*** *= Neophyte*						
Part(s) Used: AgB = Aboveground Biomass; Ba = Bark; Bi = Biomass; Fl = Flower; Fr = Fruit; L = Leaves; P = Plant; PGP = Plant (Green Part); PPGP = Plant (Probably Green Parts); PWP = Plant (Whole Plant); R = Root; Sa = Sap; Se = Seed; Sh = Shoots; St = Stem; W = Wood						