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Technical Annex

Annex 3: Soil Carbon Management

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Contents

1 Introduction

This report is focussed on improving soil carbon management (SCM). The report is broken down into two sub-sections, as the type and impact of management interventions may differ markedly in **improved** and **upland** soils. This is due to fundamental differences in structure, organic matter and nutrient content. As part of the review, the Welsh Government asked that the ability or otherwise for each intervention to be reported within Greenhouse Gas (GHG) Inventories should be detailed and where possible mitigation potential as a % of the sector presented. Where SCM options deliver additional benefits for example biodiversity benefits these should be identified.

SCM in improved soils

The Welsh Government requested the review to identify opportunities to enhance the carbon stock of improved grasslands. For complete coverage, we also cover interventions on cropland. Opportunities identified had to achieve permanence.

SCM upland soils

Here, the Welsh Government requested that mechanisms for reversing the decline in soil carbon in upland soils in Wales should be identified.

Abbreviations

SCM	Soil Carbon Management
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
DOC	Dissolved Organic Carbon
POC	Particulate Organic Carbon
GMEP	Glastir Monitoring and Evaluation Programme
CS	Countryside Survey
LULUCF	Land Use, Land Use Change and Forestry
N	Nitrogen
P	Phosphorus
K	Potassium
NPK	Fertiliser, see above three rows
Mg	Magnesium
LOI	Loss On Ignition
N_2O	Nitrous Oxide
CH_4	Methane

1.1 Background

Both scientists and farmers recognise that soil organic matter (SOM) and soil organic carbon (SOC) underpin important soil functions and services. SOM plays a critical role in nutrient cycling, soil water retention and maintenance of soil structure (Bot and Benites, 2005). SOM affects water resource management, which becomes increasingly relevant with the frequency of hydrological extremes. For example, SOM affects soil hydraulic properties and soil water repellency (Hermansen et al., 2019), infiltration and storage of water, and therefore the rate of surface runoff and groundwater recharge (Robinson et al., 2010).

In the agricultural context SOM can also increase crop yield and ease of farming, with potential benefits for farmers. SOM has been directly related to increased yields of non-legume crops on organic farms (Brock et al., 2011), and can increase the range of water contents at which soil can be tilled (increasing workability, Obour et al., 2018). Concerns have been raised that there may be critical levels (2%) of SOC below which soil processes and the productivity of agriculture are compromised. However, the quantitative evidence for this is slight (Loveland and Webb, 2003).

The most highly publicised benefit of SOC relates to climate change mitigation. In recognition of the importance of SOC for climate change mitigation, the international 4 per mille, or 4 in a thousand, initiative has emerged (https://www.4p1000.org/). This initiative highlights that a marginal increase (0.4%) in C stocks for all soils globally could significantly reduce the level of CO_2 in the atmosphere. Subsequent papers have challenged the extent of the potential due to e.g. the area of land which is actively managed globally, the negative effect on food security if land is taken out of food production, or the lack of organic material available to farmers and/or the already widespread uptake of practices which can increase soil C (e.g. Poulton et al., 2018). For example, in the UK SOC gains in excess of 4 in a thousand might be achieved by way of organic and manufactured inputs, introduction of pasture leys to arable systems and afforestation of arable land (Poulton et al., 2018). However, it must be recognised that a local increase in SOC stocks (i.e. “SOC storage”) does not necessarily entail climate change mitigation. As Chenu et al. (2019) emphasise, “SOC storage” can be treated as distinct from “SOC sequestration”; the latter implies genuine removal of CO_2 from the atmosphere on an annual basis, contributing to net reductions in Wales’ greenhouse gas emissions. It should also be pointed out that for arable systems, soil carbon has been declining at a rate of 0.4% between 1978 and 2007 across Great Britain i.e. exactly the opposite trend promoted by the 4 in a thousand initiative (Reynolds et al. 2013).

As well as increasing SOC on enclosed farmland (which is typically carbon-poor), it is important to protect and manage habitats that already have high SOC stocks. Indeed, habitats with high SOC stocks are generally most threatened with SOC declines (Crowther et al., 2016). For example, Bellamy et al. (2005) reported evidence of declines in topsoil Carbon (C) across England and Wales between 1978 and 2003, especially in regions such as the Welsh uplands which have high existing C stocks (though some concurrent work did not find clear evidence of SOC declines for Great Britain as a whole or it's individual countries, including Scotland. See Chamberlain et al., 2010; Kirby et al., 2005 respectively). Similar SOC declines have been reported more recently in the Welsh uplands for ‘habitat’ land between 2007 and 2016 through the Glastir Monitoring and Evaluation Programme (Figure 1.1.1; Emmett and the GMEP team, 2017). Given the size of upland SOC stocks, such declines might overshadow potential SOC gains on enclosed farmland. Overall, improved pasture systems, due to their dominance in area, have the highest soil carbon store at 48% of soil carbon to 1m depth (Bradley et al. 2005). An update to improve estimates for organic soils indicated that mineral soils represent the greatest soil C store at 45% with organic soils representing 30% and organo-mineral soils representing 18% when calculated beyond 1m depth (Smith et al, 2007). The most recent study (Evans et al., 2015) emphasised that soil carbon density i.e. soil C on an area basis was greatest for peatlands therefore providing a very efficient potential mitigation option in terms of area.

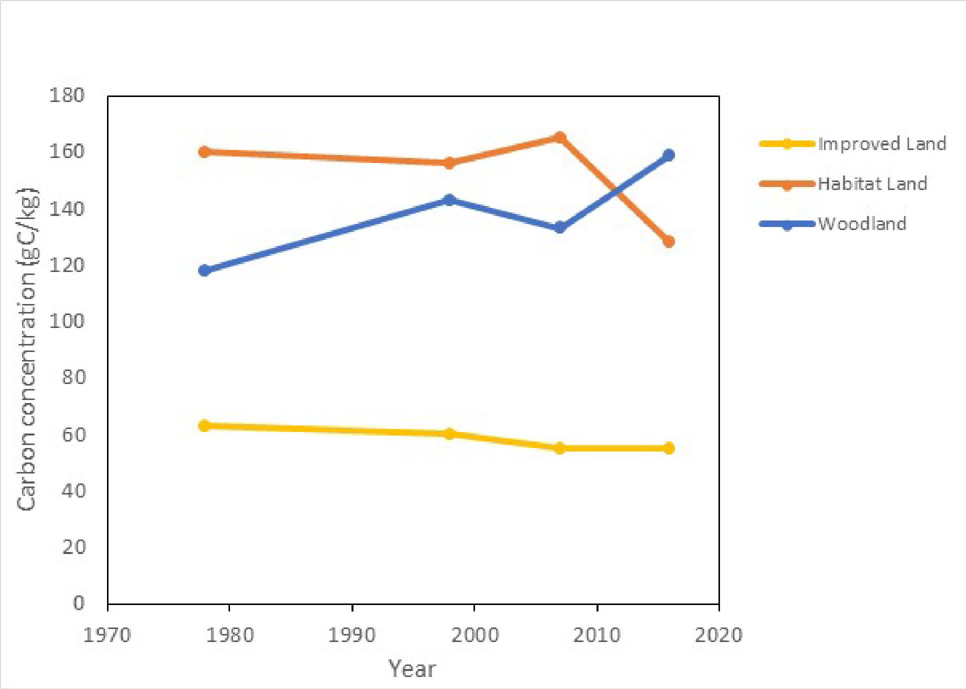

Figure 1.1.1 Change in topsoil (0-15cm) carbon concentration (g C/kg) across Wales since 1978 as assessed by GMEP (Emmett and the GMEP team 2017). 'Habitat' land is all land which is not woodland and unimproved. The only statistically significant changes are for the loss in soil C concentration from 2007 to 2016 in 'Habitat' land.

Welsh soils currently have intermediate levels of soil C content relative to England and Scotland suggesting some potential for improvement (Emmett et al., 2010). However the potential for soil C to mitigate agricultural GHG emissions on mineral and organo-mineral soils is limited for reasons outlined in Poulton et al. (2018); other measures such as peatland restoration, use of alternative energy sources and carbon capture in tree biomass show promise.

When considering interventions to increase or protect SOC stocks, it is important to consider the mechanisms through which SOC is gained and lost. SOC is primarily gained through organic matter inputs from vegetation, and lost through microbial respiration, harvest, fire, insect damage and leaching as dissolved organic carbon (DOC) (Smith, 2008). Spatio-temporal variation in these opposing processes is governed by a combination of geology and land use, but also climatic factors (Smith et al., 2008a). Both land use change and climate change are underpinned by agriculture; given that in 2015, 88% of the land area of Wales was utilised as agricultural land (Armstrong, 2016), it is critical to understand how agricultural practices affect SOC. This will aid development of a Sustainable Farming Scheme to increase the amount of carbon sequestered in soils without compromising farm businesses and their ability to provide multiple benefits to the environment and wider society.

1.2 Cross-cutting limitations to SOC increases

This report will review methods to increase SOC on enclosed farmland on an intervention-by-intervention basis, but there are general limitations to SOC storage and sequestration which are somewhat cross-cutting:

- Increases in SOC due to a given change in management or land use are finite. SOC stocks will tend to saturate as a new equilibrium state is reached (Powlson et al., 2011). Similarly, potential SOC gains may be greatest in areas which currently have low SOC stock (Minasny et al., 2017) which are undergoing land use changes, rather than minor management changes (Schlesinger and Amundson, 2019).
- Increases in SOC due to a given change in management or land use may not be permanent. Returning to original management is likely to release accrued SOC (Powlson et al., 2011). Furthermore, SOC release may occur more quickly than SOC accumulation (Soussana et al., 2004).
- Interventions to increase SOC can cause displacement effects. Firstly, they can increase agricultural CO_2 emissions. The manufacture of N fertiliser is responsible for 70% of agricultural GHG emissions (IPCC, 2007); emissions due to increased consumption of N fertiliser may entirely offset SOC gains. Secondly, they can increase emissions of other powerful Greenhouse Gasses (GHGs), particularly N_2O and CH_4 (Powlson et al., 2011). Finally, they may reduce SOC elsewhere (or prevent SOC increases), for example through agricultural intensification to meet demand (or redirecting organic inputs from other fields, Powlson et al., 2011).
- However, increasing SOM could also have synergistic effects through agricultural GHG emissions reduction; Powlson et al. (2011) argue that small increases in SOM could improve soil properties and plant productivity while reducing dependency on N fertiliser (Johnston et al., 2009).

Other limitations to achieving “4 per mille” include a lack of resources, as well as the fact that some practices are currently uneconomic and/or trade-off with food security concerns (Poulton et al., 2018). While SOM is increasingly on farmers’ agendas, nutrients which are directly related to yields tend to take priority (Farming Connect 2019, *pers. comm.*), and are sometimes applied in excess of productive needs.

1.3 Past work on SOC trends and drivers

Bellamy et al. (2005) attributed declines in topsoil C across England and Wales to effects of climate change, but it has been argued that climate change is not an adequate explanation (Hopkins et al., 2009; Smith et al., 2007a). Smith et al. (2007a) suggest that small declines on managed mineral soils may be driven by reduced manure inputs, reduced crop residue inputs and deep tillage; larger declines on organic soils were argued to be caused by drainage, recovery from acidification, nitrogen deposition, burning, fertilisation, liming and artefacts from the method Bellamy et al. (2005) used to calculate bulk density (Smith et al., 2007a). Other studies have found evidence of declines in SOC on arable land and increases on improved grassland between 1978 and 2007 (Chamberlain et al., 2010; Emmett et al., 2010), but found no evidence of overall change in soil C across all land uses and management regimes. However, in Wales, apparent declines have occurred in C concentration in upland habitats between 2007 and 2016 (Emmett and the GMEP team, 2017), and the drivers of this trend are under investigation. One consistent finding is that where SOC declines occur, they can occur disproportionately in regions with higher starting C stocks (Bellamy et al., 2005; Crowther et al., 2016; Goidts and van Wesemael, 2007; Reijneveld et al., 2009) although the consistent finding of decline in many arable / intensively managed soils goes against this trend (e.g. Reynolds et al. 2013). Nonetheless, SOC trends are generally derived based on the top 0-30cm of soil, so their representativeness of total stock of SOC is uncertain (Buckingham et al., 2013). In some systems, the C stored below 1m equates to >50% of the C stored above 1m (Jobbagy and Jackson, 2000).

It is interesting to put trends from the UK into the international context. In the Netherlands, increases were observed in mineral topsoil C on agricultural land from 1984-2004 (Reijneveld et al., 2009). In Belgium, decreases have been observed on arable soils, with increases on grassland soils from 1955-2005 (Goidts and van Wesemael, 2007; Lettens et al., 2005). Interestingly, Goidts and van Wesemael (2007) attribute declines on arable land to reduced organic inputs and cropping changes, while increases on grassland are attributed to increases in livestock density. On the other hand, Poeplau et al. (2015) argued that 7.7% increases in topsoil (0-20cm) SOC in Sweden were primarily attributable to an increase in grass ley as a proportion of total agricultural area. Declines have been detected in the lowlands of New Zealand (Schipper et al., 2007). Soil characteristics and geology in the above regions may differ greatly from soils in England and Wales.

The effects of management and pedoclimatic conditions on SOC have been reviewed elsewhere (Buckingham et al., 2013; Conant, 2010; Freibauer et al., 2004; Powlson et al., 2011; Schils et al., 2008; Smith et al., 2008b; Wiesmeier et al., 2019). As a general rule, soil C increases with precipitation (rainfall) and clay content, and decreases with temperature. Furthermore, it is fundamentally related to vegetation; it is possible that vegetation is the main mechanism by which precipitation affects soil C (Jobbagy and Jackson, 2000).

Management changes within a land use type can clearly affect SOC (Smith et al., 2008b). Some management interventions apply to both improved soils and upland soils, e.g. grazing intensity, although their outcome may differ depending on the context. However, major land use transitions are critical for SOC (Powlson et al., 2011); reports compiled with regard to GHG inventory compilation, and other reports, conclude that the impact of cropland and grassland management on SOC is likely to be small compared to land use change (Conant et al., 2001; Moxley et al., 2014). In this report land use changes are addressed as interventions, from starting points of cropland, improved grasslands and uplands. Prevention of land use changes are also crudely addressed.

Beyond SOC stocks, SOC stability is also important. An increase in stabilised SOM is ideal as this is less vulnerable to future loss. The mineral component of the soil can play a critical role in slowing decomposition of SOC: Castellano et al. (2015) summarise how some SOC is stabilized in mineral soil fractions, and this interacts with litter quality, but there are saturation effects. However, SOC in peat and surface organic layers forms important long term stores particularly in upland soils, bogs and fens. There are clear limits to SOC storage in mineral soils, in contrast to peatland soils which can continue to grow in mass and volume in the absence of perturbation.

This review sets out to summarise and build upon previous reviews, synthesising results against a set of key interventions and practices in the Welsh context. Interventions are considered in three contexts: improved grassland, cropland and upland habitats. Much of the available evidence does not originate in Wales, so there are limitations in application to a Welsh Sustainable Farming Scheme. However, where possible we highlight research occurring close to Wales or in a similar pedo-climatic context.

2 Outcomes

More organic matter, and thus C, in the soil contributes towards the following outcomes:

1) Climate change mitigation

   SOC is a critical store of carbon, although GHG mitigation from other agricultural sources is also very important (Powlson et al., 2011). Interventions that increase SOC may also have indirect effects on climate change mitigation, both positive and negative.

2) Improved productivity

   Loss of SOM is expected to negatively affect production (Quinton et al., 2010). Conversely, increases in SOM can increase productivity and workability of the land (Brock et al., 2011; Obour et al., 2018). However, positive effects of SOM on plant production are in some ways underpinned by processes of carbon decay; soil organic matter is fuel for microbes and animals that carry out key soil functions (Janzen, 2006). Increasing SOC to increase productivity may also involve "use" of SOC as a resource.

3) Reduced levels of financial risk

   Increased SOM may support stable production levels under extreme events, e.g. drought, due to greater moisture retention and associated plant production. There is a proven link between SOM levels and reduced erosion risk (Smith et al., 2008a). Some promising interventions to increase SOC could also reduce farmers' dependence on manufactured fertiliser, the prices of which are subject to change.

4) Reduced levels of biological and environmental risk

   SOM can reduce rapid surface runoff due to increased penetration of rainfall into the soil and/or storage in the soil (Robinson et al., 2010). This may reduce the risk of transfer of fertiliser, organic wastes, sediment and control chemicals into water bodies.

2.1 Cross-cutting metrics & verification

It is critical that interventions provide additionality (i.e. actions are taken, and outputs produced, beyond those which would have been taken in the absence of an intervention). This requires an understanding of (1) pre-intervention measurements of relevant metrics, but also (2) whether farmers or landowners were already going to carry out an intervention, or something similar, regardless of outside incentive. For example, in terms of securing environmental outcomes, it is futile to offer payments for reduced fertiliser inputs on land currently managed as rough grassland and not receiving fertiliser applications. This would be true unless the farmer or landowner were highly likely to improve that land, for example in response to changing market forces. Similarly, it is important to prevent a change in management in some regions (i.e. prevent drainage of peatland) but drive changes in management elsewhere (i.e. reduced and/or better targeted fertiliser application). The balance between the above factors is very difficult to find, and it may not be possible to get complete information.

As noted throughout the report, it is also important to ensure displacement effects do not occur. For example, if a farmer is compensated to increase SOC on one field, this could have knock on effects on other fields within their farm for no net gain, or net loss of SOC. Similarly, interventions for SOC on one farm could have knock on effects on SOC on other farms, though such effects are very difficult to predict or measure. One study even suggests that "land sparing", whereby agriculture is intensified on productive land while habitat is restored on unproductive land, could be a viable strategy for climate change mitigation (Lamb et al., 2016).

2.1.1 Soil sampling and SOC measurements

In terms of metrics and verification of SOC, measurement through soil testing represents the gold standard. Given opportunities and farmer interest in sampling critical soil nutrients, there could be synergies that make testing SOC affordable.

Under GMEP, reported SOC trends are based on loss on ignition (LOI, Emmett and the GMEP team, 2017). Other methods are also available to measure soil C, although these may be more costly and labour-intensive (Wang et al., 2012). It is also important to measure bulk density to understand changes in SOC stock.

The following are soil metrics that are suitable for national survey of SOC, and could be conducted in a structured assessment (possibly independent auditing):

- Research lab data (pH, LOI, Bulk density etc.)
- Visual assessment data (Peat condition, erosion features)
- Farmer questionnaire data (E.g. presence of field drains)

The following are further soil metrics that are suitable to educate farmers on soil condition and could be collected in a less structured way (i.e. farmer-led).

- Commercial lab data (N, P, K, Mg)
- Visual assessment data (Visual Evaluation of Soil Structure: VESS)

A suggested work flow for interventions related to soil C could be as follows:

1. Farmer conducts a basic assessment using mapping of farm with respect to potential options in scheme
2. Farmer meets with contract manager and agrees options plan
3. Farmer (and/or independent auditor) implements measurements at start of options
4. Farmer implements options
5. Farmer (and/or independent auditor) implements measurements at end of assessment period

Considerations for the scheme developers include:

- Will there be a basic payment for implementing an intervention?
- Will there be an advanced payment for achieving a specific result (i.e. increase in SOC)?

2.1.2 Proxies for SOC

In the absence of SOC measurements, there is some scope to base payments on proxies for outcomes (i.e. the actions), and limited scope for capital investments (e.g. seed for biological N fixation or diversification). Basic guidance for payments could draw on previous reviews and the Land Use, Land Use Change and Forestry (LULUCF) inventory (Buckingham et al., 2013; Freibauer et al., 2004; Smith et al., 2008b, 2007b). However, the cost of intervention will be a very important factor for farmers, and must be considered.

2.1.3 Public and private funding for SOC

Private sector funds may work in conjunction with public funds, provided the public funding mechanisms and regulations are set up to accommodate this.

Accurate measurement of soil carbon fluxes and storage is important, both to justify public funding for land managers, and as a potential way to secure alternative sources of funding from the private sector. The development of the peatland carbon code has partly been motivated to facilitate such private funding (see section below - “Protect and Restore Peatlands”). By establishing a consistent and verifiable method for measuring carbon sequestration, the peatland carbon code is designed to reduce transaction costs and give confidence to those who may want to purchase carbon credits (Evans et al., 2019a; Smyth et al., 2015). Such purchases are currently limited to a small voluntary market in the UK, although an interesting recent development is a pilot purchase of carbon credits from peatland restoration in Lancashire by Heathrow Airport (https://your.heathrow.com/heathrow-airport-unveils-plan-for-carbon-neutral-growth/).

Soil carbon trading has potential to grow as a voluntary market. Following the recent trend to achieve no net loss or net gain in infrastructure projects (e.g. by Network Rail and highways England), similar commitments to be carbon neutral or positive would stimulate significant demand for carbon credits. Unlike biodiversity, such credits could be purchased from anywhere in the world, but there is likely to be demand for UK credits from some UK businesses (as demonstrated by Heathrow). The market may also be stimulated by Government action, with the need to utilise land use management as a means of carbon storage being recently noted in the advice from the Climate Change Committee (https://www.theccc.org.uk/wp-content/uploads/2018/11/Land-use-Reducing-emissions-and-preparing-for-climate-change-CCC-2018.pdf : recommendation 1: New land use policy should promote transformational land uses and reward landowners for environmental outcomes that deliver climate mitigation and adaptation objectives).

3 Policy Relevance and Policy Outcomes

The main policy relevance for this topic with respect to the Natural Resources Policy priorities which support the outcomes of the Well-Being of Future Generations (Wales) Act and the sustainability and management of Natural Principles in the Environment (Wales) Act is:

- *Restoration of our uplands and managing them for biodiversity, carbon, water, flood risk and recreational benefits*

with potential co-benefits for:

- *Maintaining, enhancing and restoring floodplains and hydrological systems to reduce flood risk and improve water quality and supply; (including catchment management approaches, natural flood management, soil management etc.)*

4 Interventions on Improved Grassland

Improved grassland is an important and dominant land use in Wales (Armstrong, 2016). There is no single accepted definition of improved grassland. Previous work in Wales has categorised improved grassland based on coverage of key improved grassland sown species, especially *Lolium perenne* and *Trifolium repens* (Emmett and the GMEP team, 2017). Here, we define improved grassland as grassland in receipt of agro-chemical inputs (e.g. lime, manure, manufactured fertiliser), which is not a part of an arable rotation. This definition is necessarily vague; the literature on SOC of improved grassland is highly diverse, and authors do not always specify their own definition of improved grassland.

4.1 Increased Manufactured Fertiliser

4.1.1 Causality

PINK:[1] **There is evidence from outside of Wales supporting an increase in SOC following manufactured fertiliser application to improved land on mineral soils. Evidence from Wales is scarce. Furthermore there are severe trade-offs, notably nitrogen pollution and increased GHG emissions. Agronomy and targeting are crucial to secure SOC and productivity while minimising trade-offs.**

Plant growth can be limited by the availability of Nitrogen, Phosphorus or Potassium. Manufactured fertilisers are used with intent to lift nutrient limitations and increase plant productivity. Soil testing is not commonplace on improved grassland in Wales, so it is often unclear where and how plant growth is limited by nutrient availability. Application rates of N, P and K across Great Britain have fallen for more than 30 years (https://www.agindustries.org.uk/sectors/fertiliser/uk-fertiliser-consumption-trends-and-statistics/) without clear negative consequences for yield, suggesting that fertilisers are often applied in excess of crop needs.

Manufactured fertiliser can increase C inputs to the soil through plant material and root exudates (Buckingham et al., 2013). Manufactured fertilisers can also affect SOC through the microbial community. Poeplau et al. (2018) suggest that SOC increases caused by fertiliser could have been driven by increased microbial C use efficiency. This would lead to an increase in microbial necromass, contributing to stabilization of C in the fine mineral fraction of the soil.

However, N fertilisation can also increase the quality of plant litter (i.e. decrease the C:N ratio of litter), accelerating C mineralization through microbial respiration (Lu et al., 2011). Soussana et al. (2004) found that moderate N fertiliser increased SOC, as increases in carbon inputs exceeded increases in C mineralisation. They also found that intensive N fertilisation could greatly increase C mineralisation and reduce SOC.

Hassink (1994) detected no increase in soil carbon with N fertilisation in their field study. Furthermore, Hopkins et al. (2009) found mixed effects of a variety of manufactured fertiliser treatments on SOC trends in experimental grasslands. A meta-analysis by Lu et al. (2011) found that nitrogen addition increased soil C in cropland, but not forests or grasslands. Soussana et al. (2007) assessed GHG budgets of 9 grassland sites and demonstrated that C storage was positively related

[1] See colour coding defined and used in Section 8.

to N fertiliser supply, but this represented a combination of manufactured and organic fertilisers.

4.1.2 Co-benefits and trade-offs

See the separate, Soil Nutrient Management review for more information on this intervention type. There is a clear and severe trade-off between fertiliser-induced C sequestration and CO_2 emissions during fertiliser production. 80% of N manufactured through the Haber-Bosch process is used for agricultural fertiliser (Erisman et al., 2008). The Haber-Bosch process is energy-intensive (Smith, 2002) and thus consumes large quantities of fossil fuels. Poeplau et al. (2018) suggest that the full carbon footprint of fertiliser induced SOC sequestration should be considered. They argue that a reduction in CO_2 emissions could be achieved if NPK fertilisers were substituted for PK fertilisers and N fixation by legumes.

Adding reactive N to the soil can increase N_2O emissions (Freibauer et al., 2004). This can trade-off severely with C storage following fertiliser application, especially given the global warming potential of N_2O is 310 times that of CO_2 (Buckingham et al., 2013). Increases in N_2O emissions will be especially high if fertiliser applications are excessive or poorly timed, so more effective timing could alleviate this trade-off somewhat (Powlson et al., 2011).

In combination, these trade-offs mean that use of manufactured fertiliser for climate change mitigation should be treated with caution (although there are arguments in favour of manufactured fertilisers for food production). This is because the C storage benefits of N fertilisation may be outweighed by GHG emissions during fertiliser production and application (Freibauer et al., 2004). In an experiment at Broadbalk in Rothamsted, the annual GHG emissions from N fertiliser use were four times the annual increase in SOC (Powlson et al., 2011). Clearly increasing N use efficiency could play an important role in alleviating such trade-offs.

Another potential trade-off is increases in nitrate leaching and nitrogen pollution, especially where fertiliser application is excessive or poorly timed (Goulding et al., 2000). This could reduce productivity of linked freshwater and coastal systems for fish and shellfish, as well as many other benefits we derived from water resources (e.g. recreation).

A clear co-benefit of increased manufactured fertiliser application is increases in plant productivity and food production on farmland. However, there may be a conflict between SOC and production. Soussana et al. (2007) found that SOC increases due to N fertiliser could be counterbalanced by herbage use through cutting and grazing. If fertiliser use is always associated with increased herbage use, SOC storage outcomes may be diminished. Nonetheless, Soussana et al. (2007) suggest that in the absence of N supply *and* herbage use, grasslands are net C sinks.

4.1.3 Magnitude

Poeplau et al. (2018) investigated effects of mineral fertilisation (N, PK, NPK, and increased dosage NPK) at seven long term grassland fertilisation experiments in Germany and the Netherlands. Significant effects of PK (0.28 t C ha^{-1} y^{-1}) and NPK fertiliser (0.13 t C ha^{-1} y^{-1}) were identified over 34 years. Significant effects of increased dosage NPK fertiliser (0.37 t C ha^{-1} y^{-1}) were identified over 20 years. Freibauer et al. (2004) report sequestration of 0.2 t C ha^{-1} y^{-1} due to fertilisation of nutrient poor grasslands, but -0.9 – 1.1 t C ha^{-1} y^{-}1 for intensification of organic soils.

Furthermore, a recent review supporting the LULUCF GHG inventory found between -21 to 27 t C ha^{-1} effects of change in management involving application of manufactured fertiliser (Buckingham et al., 2013).

Effects of N fertiliser on SOC are often amalgamated with effects of other land management practices, such as manure application: one meta-analysis reports that fertilisation in general (manures and manufactured fertilisers) can increase SOC by 0.3 t C ha^{-1} y^{-1} (Conant et al., 2001). Ammann et al. (2007) investigated the C budget of a temperate grassland, which was newly converted from arable, for three years. They found that SOC storage was 2 t C ha^{-1} y^{-1} higher under “intensive” management, in which manures and N fertilisers were applied, than “extensive” treatment, in which no manure or fertiliser was applied. Smith et al. (2008b) report that increased grazing, fertilisation or fire on grasslands results in CO_2 mitigation of 0.11 – 1.5 t C ha^{-1} y^{-1} in cool, moist regions.

4.1.4 Timescale

Changes in SOC may occur even over short time periods. Ammann et al. (2007) saw increases in SOC storage on a newly created grassland that had manure and N fertiliser applied within three years. Poeplau et al. (2018) recorded increases in SOC over 20-37 years.

4.1.5 Spatial issues

Buckingham et al. (2013) report SOC increases as a result of manufactured fertiliser use on mineral soils. However, the authors warned that impacts of intensification on organic soils are less well understood and probably negative. For example, Freibauer et al. (2004) report sequestration of 0.2 t C ha^{-1} y^{-1} due to fertilisation of nutrient poor grasslands, but -0.9 – 1.1 t C ha^{-1} y^{-}1 for intensification of organic soils.

N_2O emissions due to N fertilisation could be more severe in areas which already have surplus N; yield-scaled N_2O emissions increase exponentially with N surplus, so adding N fertiliser in the wrong places will disproportionately increase N_2O emissions (van Groenigen et al., 2010).

4.1.6 Displacement

Local increases in SOC can be essentially displaced by GHG emissions from fossil fuels and N_2O - see trade-offs above.

4.1.7 Longevity

All else being equal (e.g. grazing pressure unchanged) increases in SOC are likely to reverse should manufactured fertilisers no longer be applied.

4.1.8 Climate interactions

Lu et al. (2011) propose that under elevated CO_2, there may be increasing N limitation within ecosystems. The result might be that the positive effects of N fertilisation on SOC and C sequestration are increased under elevated CO_2.

4.1.9 Social and economic barriers

NPK fertilisers are expensive, and prices can be linked to oil prices. However, there are minimal barriers to application as this practice is already widespread across improved grasslands and croplands in Wales. Farmer consultation suggests that applications in the region of 150 kg ha^{-1} y^{-1} are not uncommon on improved grassland (Farming Connect 2019, *pers. comm.*). Farmers in Wales may be somewhat open to incentives that increase, decrease or target the application of manufactured fertiliser, though cessation of fertiliser on intensive fields is likely to be viewed as unproductive and economically unfavourable.

Contractors are increasingly used to optimize application of fertilisers using GPS. This is an important consideration related to manufactured fertiliser interventions.

4.2 Organic Fertiliser

4.2.1 Causality

BLUE[2]: There is good evidence that carefully targeted organic inputs can increase SOM on improved grassland on mineral soils. The benefits are greatest when dependence on manufactured fertiliser is reduced, along with associated GHG emissions. Care must be taken to avoid nutrient excesses and nitrogen leaching.

Similarly to manufactured fertilisers, organic fertilisers are intended to lift nutrient limitations and increase plant productivity. As well as direct organic inputs, increased C inputs to the soil originate from plant material and root exudates (Buckingham et al., 2013). The positive impacts of manure application on SOC may exceed the impacts of manufactured fertilisers: Jones et al. (2006) applied a variety of manures, slurries and mineral fertilisers to cut grasslands in southern Scotland, finding that all manure treatments increased topsoil C concentration, while mineral fertilisers did not. Interestingly, SOC increases were observed despite increases in DOC and CO_2 losses through respiration.

Soussana et al. (2007) assessed GHG budgets of nine grassland sites and demonstrated that C storage was positively related to N fertiliser supply, but this represented a combination of manufactured and organic fertilisers.

One study suggests that processing of organic fertiliser, for example by producing digestates, may increase SOC storage effects; Chenu et al. (2019) highlight a number of studies suggesting greater long-term SOC storage from labile, easily degraded compounds than recalcitrant, lignin-rich material. This may be because labile compounds are processed with higher microbial carbon use efficiency, increasing SOC storage as microbial necromass. Another explanation is that soluble compounds migrate in soil between mineral surfaces, where they can be protected. However, a large experiment compared effects of compost, manure, digestates and slurry on various soil properties and crop yields across seven UK sites (WRAP, 2015). Overall, the experiment found that composts and farmyard manures increased SOM as compared with digestates or slurry (given roughly equivalent N in dosage). As such, it remains unclear whether “fresh” or “processed” organic matter will have the greatest benefits for C sequestration, especially given possible emissions while processing organic materials.

[2] Colour coding relate to confidence levels as explained more fully in Section 8.

4.2.2 Co-benefits and trade-offs

See the separate, Soil Nutrient Management review for more information on this intervention type. Application of manures and composts as opposed to slurry, digestates or manufactured fertilisers can increase earthworm numbers, nutrient availability and crop yields (WRAP, 2015). Furthermore, with an integrated nutrient management plan, considerable savings could be made on manufactured fertiliser.

Key trade-offs of manure application identified during a review to inform LULUCF inventories were nitrate leaching and N_2O emissions (Moxley et al., 2014), especially where application is excessive or poorly timed (Goulding et al., 2000). In fact, these trade-offs may be more severe per kg of N in manure as compared with manufactured fertiliser (Bergström and Goulding, 2005). Jones et al. (2006) found that for some types of organic fertiliser (though not all) increases in SOC were outweighed by N_2O emissions, given the increased global warming potential of N_2O. As for manufactured fertiliser, targeting of organic fertilisers is critical; loss of nutrients, pollution and emissions may be particularly acute if manure is applied at the wrong time (Powlson et al., 2011). Because amounts of nutrients in organic inputs are not completely known, excesses in non-N nutrients may also occur (e.g. P).

A clear co-benefit of increased organic fertiliser application is increases in plant productivity and food production on farmland. However, Soussana et al. (2007) found that SOC increases due to N fertiliser could be counterbalanced by herbage use through cutting and grazing. If organic fertiliser use is always associated with increased herbage use, SOC storage outcomes may be diminished. Nonetheless, Soussana et al. (2007) suggest that in the absence of N supply *and* herbage use, grasslands are net C sinks.

Organic fertiliser inputs may displace inputs of manufactured fertiliser, for example in organic farming systems. This could provide significant co-benefits through reduced emissions from fertiliser manufacture. Smith et al. (2011) reviewed possible benefits of organic farming for SOC sequestration in Wales. While it was unclear whether a switch to organic farming would increase SOC on-site, potential benefits were identified in terms of reduced GHG emissions from fertiliser production. However, the effect of such emissions-reductions must still be weighed up against on-site emissions from use of manure (Powlson et al., 2011).

Freibauer et al. (2004) highlight that manure application can improve soil structure and water holding capacity. The authors also suggest that the preferential use of manure on arable land could prevent trace gas emissions that are more pronounced when manure is applied to grassland.

4.2.3 Magnitude

A review to inform LULUCF inventories (Buckingham et al., 2013) found positive changes in grassland soil C stocks brought about through slurry or manure applications (0.7 to 15 t C ha^{-1}). Jones et al. (2006) reported C storage of 15.7-48.3 t C ha^{-1} following application of manure for six years. Smith et al. (2008b) report CO_2 mitigation potential of -0.62 – 6.20 t CO_2 ha^{-1} y^{-1} for application of manure or biosolids in cool moist regions, although this includes both cropland and grassland.

Manure application effects for SOC are sometimes presented in combination with other interventions. A meta-analysis by Conant et al. (2001) reports that fertilisation in general (manures and manufactured fertilisers) can increase SOC by 0.3 t C ha^{-1} y^{-1}.

4.2.4 Timescale

Increases in SOC can occur over short timescales. Jones et al. (2006) reported positive effects of manure application for SOC on cut grasslands within six years.

4.2.5 Spatial issues

Care should be taken not to generalise the effects of manure application to organic-rich soils. While there is strong support for positive effects of manure application for SOC on mineral soils, the review of Buckingham et al. (2013) concluded that inputs to organic-rich pasture soils could lead to decreases in SOC.

N_2O emissions due to N fertilisation could be more severe in areas which already have surplus N; yield-scaled N_2O emissions increase exponentially with N surplus, so adding N fertiliser in the wrong places will disproportionately increase N_2O emissions (van Groenigen et al., 2010).

4.2.6 Displacement

Targeting organic fertiliser to build soil carbon stocks in one area could be offset by removal of carbon inputs elsewhere (Conant, 2010). Manure application could be targeted to increase SOC on arable soils (which have low organic matter as a starting point) rather than on grasslands (Buckingham et al., 2013; Powlson et al., 2012, 2011). While local changes in SOC alone do not constitute climate change mitigation (Chenu et al., 2019), such targeting could lead to a net reduction in GHG emissions. The alternative fate of other organic fertilisers, such as household green waste, should also be considered. This would ensure that local increases in SOC are not offset by decreases in SOC or emissions elsewhere.

Furthermore, redirecting organic fertilisers so that they are applied further from the point of production could increase gaseous emissions through transportation (Freibauer et al., 2004).

4.2.7 Longevity

As with manufactured fertilisers, all else being equal, cessation of manure application would be likely to reverse increases in SOC. Furthermore, positive effects of manure for SOC can saturate, as demonstrated at the Broadbalk wheat experiment at Rothamsted (Powlson et al., 2012).

One experiment suggests that lignin-rich material in composts could provide particularly stable SOM which is resistant to decomposition (WRAP, 2015).

4.2.8 Climate interactions

Organic fertiliser effects on SOC may be diminished under increased drought conditions. Smith et al. (2008b) report lower mean mitigation potential due to manure or biosolids application in dry areas as opposed to moist areas. Furthermore, Jones et al. (2006) found that increases N_2O emissions after applying organic fertilisers were greater in a particularly wet year of their six year study.

Lu et al. (2011) propose that under elevated CO_2, there may be increasing N limitation within ecosystems. The result might be that the positive effects of N fertilisation on SOC and C sequestration are increased under elevated CO_2.

4.2.9 Social and economic barriers

Farmers are increasingly aware of the value of their organic fertiliser resources, and many already have the infrastructure to distribute them on their fields. However, manures and slurries do not tend to be shared between farms under different ownership (Farming Connect 2019, pers. comm.). As such there may be social and economic barriers to targeted redistribution of organic fertilisers. As the review of Buckingham et al. (2013) indicates, "even within a single farm, manure application is often unevenly distributed with the fields nearest the livestock buildings being the preferred sites of application".

4.3 Liming

4.3.1 Causality

AMBER: More data is needed, in a Welsh context, to reach a conclusion on effects of liming on soil C. Carefully targeted lime application to reduce manufactured fertiliser inputs *could* provide emissions-reduction and increased productivity. There may be risks to liming on organic soils.

Liming (application of calcium carbonate) is carried out to increase pH on agricultural land where acidity limits plant productivity. This can affect SOC firstly by increasing plant productivity, but also by affecting C decomposition (Buckingham et al., 2013). Research carried out at Rothamsted suggests that liming can increase SOC (Fornara et al., 2011). Liming increased biological activity in the soil, and although soil respiration increased, plant soil inputs were more readily incorporated into organo-mineral pools. However, Rothamsted park grass soils are mineral soils with low starting SOC in comparison to most Welsh grasslands.

Paradelo et al. (2015) reviewed the impact of liming on SOC stocks, finding variable effects across grasslands, croplands and woodlands. They concluded that the impacts of liming are highly context dependent, and that more data are needed.

Liming may be associated with other agricultural practices, such as drainage and N fertilisers. More investigation and meta-analysis is needed into the impacts of liming on SOC in the Welsh context, taking into account wider management and effects over long timescales.

4.3.2 Co-benefits and trade-offs

pH increases following liming are associated with CO_2 emissions (Gibbons et al., 2014). For every 2 mol of acid neutralized by calcium carbonate, up to 1 mol of CO_2 could be released to the atmosphere (Whitmore et al., 2015). Snyder et al. (2009) propose that: "One possible way to avoid the emission associated with lime use is to apply oxide (e.g. quicklime or slaked lime) rather than carbonate materials, if they can be produced with CO_2 recovery". There may also be opportunities to use silicates rather than carbonates to prevent these added CO_2 emissions from agricultural lime (Whitmore et al., 2015). Furthermore, limestone extraction can be a destructive process.

Liming has the co-benefit of increased productivity, depending on the soil to which it is applied (Paradelo et al., 2015). Furthermore, liming could increase the efficiency of effects of NPK fertiliser on productivity, potentially reducing the need for NPK and its negative externalities (Gibbons et al., 2014).

Liming can have knock-on effects on biota of receiving lakes and watercourses. Those effects are not necessarily negative, e.g. the recovery of acid-sensitive taxa (Shapiera et al., 2012).

4.3.3 Magnitude

At Rothamsted, liming lead to SOC increases of 16.2 t C ha^{-1} over a period of ~100 years (Fornara et al., 2011).

4.3.4 Timescale

Effects on SOC could take a long time to occur: At Rothamsted, effects of liming became clear after ~50 years (Fornara et al., 2011).

4.3.5 Spatial issues

As with fertilisation, care should be taken when generalising results to organic soils. Rangel-Castro et al. (2004) experimented with short-term, high-dose liming on upland soils and found a decrease in SOC, suggesting plant C inputs were outweighed by increased C decomposition. Similarly, Lochon et al. (2018) carried out experimental incubations of soil from three upland grasslands in France, and found that lime addition consistently increased C mineralisation and CO_2 production over 84 days.

It is possible that liming on organic soils will shift the system from one of low productivity, low SOC decomposition under anaerobic conditions to one of high productivity, high SOC decomposition under aerobic conditions. Effects of such a shift on net CO_2 exchange are unresolved.

4.3.6 Displacement

There is not sufficient evidence to discuss displacement effects of liming.

4.3.7 Longevity

There is not sufficient evidence to discuss longevity of effects of liming on SOC.

4.3.8 Climate interactions

There is not sufficient evidence to discuss climate interactions of effects of liming on SOC.

4.3.9 Social and economic barriers

Liming has been encouraged by incentives in the past, which increased the frequency of application – investment cost represents a barrier to liming management (Gibbons et al., 2014). Liming has recently seen another surge in use across Wales (Farming Connect, *pers. comm.*). Incentives for liming could be very popular among farmers due to positive effects on productivity and nutrient utilisation by plants. Incentives to reduce lime might be challenged on the basis that productivity is reduced.

4.4 Grazing and Cutting

4.4.1 Causality

AMBER: The relationship between grazing/cutting and SOC is complex. Reductions in overgrazing could benefit SOC and productivity in tandem. Grazing and cutting also remove vegetative C from the land, and consuming livestock produce GHGs which are more potent than CO_2.

Grazing and cutting are used to extract yield from grasslands. This directly removes C from the system (Soussana et al., 2007), but also affects soil moisture and subsequent plant growth (and thereby SOC; Abdalla et al., 2018). Grazing animals also provide concentrated N inputs to the soil through manure and urea, further affecting productivity and SOC (Soussana and Lemaire, 2014). Root exudates are an important pathway for transfer of C from plants to soil (Jones et al., 2009), and some evidence suggests that grazing can enhance root exudation of carbon (Hamilton III et al., 2008). Clearly the relationship between grazing animals and GHG budgets is complicated (Garnett et al., 2017).

Soussana and Lemaire (2014) assert that at low stocking density, herbivores enhance net primary productivity and thus SOC storage. However, intensive grazing can reduce water availability, alter plant community composition and drive a decrease in productivity (and thereby SOC; Abdalla et al., 2018). Extreme grazing can strip surface vegetation and cause erosion of the topsoil. In line with the above, a meta-analysis by Zhou et al. (2017) found that while light grazing appeared to lead to slight increases in soil C, moderate to heavy grazing significantly decreased SOC, especially in the microbial biomass. Soussana et al. (2007) assessed GHG budgets of 9 grassland sites and demonstrated that net C storage was negatively related to herbage use through cutting and grazing. They put forward the hypothesis that “net carbon storage per unit ground area declines with C use by herbivores”.

Nonetheless, Smith et al. (2007b) reported mixed effects of grazing intensity on mitigation of CO_2, CH_4 and N_2O emissions; results are dependent on grazing practices, plant species, soils and climate. Another review in extensive grazing systems (little to no N fertiliser) found that the impact of grazing on SOC is highly context and climate dependent (Abdalla et al., 2018). In fact, in moist cool climates (the likely category for most grasslands in Wales) SOC decreased with grazing intensity.

Changes in timing and intensity of grazing (rotational grazing) can affect productivity, removal and C allocation of flora in grasslands (Buckingham et al., 2013). Similarly, Conant (2010) suggests SOC benefits of reducing grazing intensity while forage species are actively growing. However, there is little consensus on SOC impacts of specific types of grazing management due to the range of grazing settings and practices (Schils et al., 2008). For example, effects of “mob grazing” on SOC are unresolved (Buckingham et al., 2013). Mob grazing involves grazing land for shorter periods of time at higher stocking densities. Russel et al. (2013) investigated impacts of mob grazing on SOC in Iowa, USA and found no advantages of season-long mob grazing over rotational or strip grazing at an equal forage allowance.

Buckingham et al. (2013) reported inconsistent effects of mowing grassland instead of grazing it, while Hassink (1994) detected no difference in SOC between grazed and mowed treatments in their field study. However, Koncz et al. (2017) propose that extensive grazing is more climate-friendly than hay-cutting due to increased export of herbage under hay-cutting.

4.4.2 Co-benefits and trade-offs

Light grazing may help to increase SOC, but livestock and their waste are associated with emissions of CH_4 and N_2O (Buckingham et al., 2013) both of which have higher global warming potential than CO_2. Increased grazing intensity is generally associated with increased agro-chemical inputs, which may have negative externalities.

Grazing and cutting are how farmers make money. Where land is being over-grazed, there could be co-benefits in terms of increased productivity. Moderating grazing intensity could also lead to co-benefits of reduced soil degradation and compaction (Freibauer et al., 2004). However, reducing grazing intensity could also cause trade-offs to food production.

4.4.3 Magnitude

Liebig et al. (2005) found that in North America, despite high variability, grazing increases SOC by 0.16 t C ha^{-1} y^{-1}. A meta-analysis reported that improved grazing management, which means introducing grazers or maintaining a moderate grazing intensity, can increase SOC by 0.35 t C ha^{-1} y^{-1} (Conant et al., 2001).

A review to inform the LULUCF inventory found that mowing instead of grazing grassland has mixed effects on SOC, with effects ranging from -25 to 14 t C ha^{-1} (Buckingham et al., 2013).

4.4.4 Timescale

The timescales of effects of grazing intensity on SOC are unclear.

4.4.5 Spatial issues

As with fertilisation and liming, effects of grazing seem to vary depend on the starting point in terms of SOC. It is thought that increased grazing can have positive effects from low starting SOC, but animal urine can also mobilise C through the soil profiles by increasing pH (Buckingham et al., 2013). Derner et al. (2006) found that grazing increased soil C in grasslands with low starting SOC (shortgrass prairies), but had no effect in grasslands with high starting SOC (tallgrass prairies). The difference was thought to be mediated by changes in the plant community. Similarly, Soussana et al. (2004) concluded that while intensification of permanent grassland could lead to an increase in SOC, intensification of nutrient-poor grasslands developed on organic soils would decrease SOC.

4.4.6 Displacement

While a switch from grazing to cutting may reduce on-site emissions from livestock, emissions from off-site digestion of harvested herbage should be taken into account (Soussana et al., 2007).

4.4.7 Longevity

It is unclear whether any increases in SOC would be maintained if grazing intensity is returned to its original state. However, a return to overgrazing would be likely to reduce SOC.

4.4.8 Climate interactions

The effects of grazing on SOC may be more positive in warmer, dryer conditions. The review of Abdalla et al. (2018) highlights studies showing that grazing can increase SOC in areas with 600mm or less rainfall, depending on soil type; they also found that in dry regions low-medium grazing intensity was associated with an increase in SOC. They cite studies showing that intensive grazing can increase root C content in areas with extreme rainfall (not 400-850mm). In moist warm climates SOC increased with grazing intensity, but in moist cool climates SOC decreased with grazing intensity.

4.4.9 Social and economic barriers

Grazing and cutting are how farmers make money, so there will be economic barriers to related interventions. Farmers might be quite open to incentives for rotational grazing, for example to optimise plant productivity while also increasing SOC (Farming Connect 2019, *pers. comm.*).

4.5 Sward Management

4.5.1 Causality

AMBER: **Positive effects of biological nitrogen fixation and deep-rooted perennials on SOC are widely documented, but not in the Welsh context. Where sward management is used to reduce manufactured fertiliser inputs, this intervention has potential to secure SOC and additional co-benefits. Benefits for wildlife will be limited without considerable sward diversification.**

A more diverse sward is thought to lead to an increase in SOC. Part of the reason for this is presence of key functional groups which are critically important for SOC, and complimentary to one another – Fornara and Tilman (2008) found that legumes and C4 grasses contributed especially to the accrual of SOC. Legumes increase SOC through N fixation and corresponding increases in productivity (Mortenson et al., 2004). Chenu et al. (2019) report a consensus that belowground inputs contribute more to soil carbon than aboveground inputs, and deep-rooting perennial species may help to enhance SOC stocks at depth (Carter and Gregorich, 2010). Increased plant diversity can also increase C inputs into the microbial community (Steinbeiss et al., 2008). However, renovation and reseeding of grassland to establish species can also lead to release of SOC (Schils et al., 2005).

Some evidence of sward diversity impacts on SOC suggests that outcomes are context-specific or subject to legacy effects. Diversity tends to positively affect productivity, and the magnitude of effects may be comparable to other major management changes (Weigelt et al., 2009). However, while diversity can increase the rate of accumulation of SOC on new grasslands (Weisser et al., 2017), increases in total potential for SOC storage are not always clear. In another example, a four year experiment showed increased SOC storage in plots with higher plant diversity (Steinbeiss et al., 2008). These results were on land previously managed as arable;

an increase in SOC was expected, but plant diversity increased the rate of C accumulation.

Enhanced species diversity and deep-rooted, productive grasses are thought to increase SOC on low-productivity pastures (Buckingham et al., 2013). Smith et al. (2008b) report positive effects of species introductions, particularly deep-rooted grasses and legumes, on CO_2 mitigation potential and SOC.

4.5.2 Co-benefits and trade-offs

See the separate, Sward Management review for more details on this intervention type. Sowing improved forage species can increase production if species are better adapted to local climate, more resilient to grazing or able to fix nitrogen into the soil (Conant, 2010). Establishing high diversity grasslands on agriculturally degraded land has the potential increase productivity with minimal N inputs (Tilman et al., 2006). Naturally, an increase in sward diversity carries biodiversity co-benefits in terms of both plants and invertebrates (Alison et al., 2017), although with a focus on function this may only amount to an increase in a few key species.

Use of legumes over urea fertiliser could reduce air pollution in the form of ammonia. Sowing legumes could provide co-benefits through reduced production and application of manufactured fertilisers, meaning reduced CO_2 and N_2O emissions, and reduced nitrogen pollution (Lüscher et al., 2014). Sowing legumes rather than using N fertiliser might actually decrease local SOC (Schils et al., 2008) but also decrease net GHG emissions due to these co-benefits. While it is unclear just how much N fixed by legumes might also be lost through leaching or N_2O emissions (Henderson et al., 2015), reductions may occur relative to manufactured fertiliser.

The use of legumes instead of N fertiliser can be an organic farming intervention. Smith et al. (2011) reviewed possible benefits of organic farming for SOC sequestration in Wales. They concluded that organic farming on grasslands would have limited potential to increase SOC, but there would be co-benefits in terms of reduced emissions from fertiliser production due to reliance on biological rather than industrial N fixation.

There may also be nutritional benefits of e.g. legumes in the sward, although this may be driven by specific species. In diverse swards, deep-rooting species such as Chicory (*Cichorium intybus*) and Sheeps Parsley (*Petroselenium crispum*) can provide minerals to livestock; Chicory, Sainfoin (*Onobrychis vicifolia*) and Birdsfoot Trefoil (*Lotus corniculatus*) also act as a natural anti-helminthic, thus providing control of intestinal parasites.

4.5.3 Magnitude

A meta-analysis reports that sowing legumes on grasslands can increase SOC by 0.75 t C ha^{-1} y^{-1} (Conant et al., 2001).

4.5.4 Timescale

Steinbeiss et al. (2008) found that the rate of SOC accumulation on newly formed grasslands increased with plant diversity over 4 years.

4.5.5 Spatial issues

Establishment of high sward diversity may be more difficult to achieve in some areas, and seed mixes would probably need to be tailored to local conditions. In England, Countryside Stewardship schemes have tested for outcomes on sward diversity, and targets are often not met.

Henderson et al. (2015) suggest that C storage due to sowing of legumes will exceed N_2O emissions on only ~10% of pasturelands. However, in that analysis France, Great Britain and Ireland appeared to be hotspots for C sequestration through sowing legumes.

4.5.6 Displacement

It is unclear whether displacement effects would occur in the case of sward management.

4.5.7 Longevity

Reseeding may be required relatively frequently to re-establish the desired sward, unless management is shifted to a highly extensive grassland system. Tillage and reseeding counteract increases in SOC (Schils et al., 2005). It is unclear for how long increases in SOC would persist following cessation of sward management. This would probably depend on what type of management is introduced instead.

4.5.8 Climate interactions

Seed mixes probably need to be tailored to local climatic conditions.

4.5.9 Social and economic barriers

There may be some appetite among farmers for incentives to increase sward diversity or introduce different forage species. Such practices are being taken up independently on some Welsh pastures (Farming Connect 2019, *pers. comm.*). Reputable plant taxa include *Trifolium*, *Plantago*, *Cichorium* and, in upland regions, *Phleum*. The infrastructure for improved sward management is likely to exist on many farms.

4.6 Prevent Tillage & Conversion to Cropland

4.6.1 Causality

BLUE: The evidence base for SOC loss following conversion to arable is very well established. Benefits of reduced or zero-tillage on Welsh grasslands are less well understood. Preventing conversion to arable is very likely to protect SOC, with potential trade-offs with productivity. The difficulty is determining if and where conversion to arable will happen.

Conversion of permanent grassland to ley-arable will tend to decrease SOC (Fullen and Booth, 2006). A modelling study suggests that ploughing of grassland is associated with loss of carbon, and recommends less frequent grassland renovation to reduce CO_2 emissions (Vellinga et al., 2004). The CLIMSOIL project highlighted policies that encourage energy crops as potentially highly detrimental to soil C, because they may encourage conversion of grassland to cropland (Schils et al., 2008).

While many grasslands in Wales are ploughed and reseeded at regular intervals, studies of min-till and no-till management tend to be biased towards arable systems.

4.6.2 Co-benefits and trade-offs

Tillage of grassland is carried out on permanent grassland to re-establish a desired sward and increase productivity. Furthermore, conversion to arable will tend to occur where farmers expect an increase in profitability by introducing a crop rotation. As such, prevention of tillage could come at a cost to agricultural productivity.

Co-benefits of preventing tillage include reduced sediment runoff and P losses (Haygarth and Jarvis, 1999).

4.6.3 Magnitude

The strongest negative effect on SOC reported by Guo & Gifford (2002) came from conversion of pasture to crops. Freibauer et al. (2004) report change in the region of -1.0 to -1.7 t C ha^{-1} y^{-1}. Another study shows that converting a permanent grassland to an annual crop can decrease SOC at a rate of -0.96 t C ha^{-1} yr^{-1} over a 20-year period Soussana et al. (2004).

4.6.4 Timescale

Soussana et al. (2004) found that declines in SOC occurred for more ~20 years after conversion to cropland, and were steepest in the years immediately after ploughing.

4.6.5 Spatial issues

The amount of C lost is likely to vary depending on the starting stock of C. For example, ploughing or converting grassland on highly organic soils might be expected to lead to substantially greater decreases in SOC as compared with ploughing mineral soils.

4.6.6 Displacement

Prevention of intensification of grassland or conversion to cropland could decrease food production with potential for displacement effects.

4.6.7 Longevity

Soussana et al. (2004) found that declines in SOC occurred for more than ~20 years following conversion to cropland. It may take longer for C to accumulate following grassland creation on cropland (Powlson et al., 2012).

4.6.8 Climate interactions

It is unclear how effects of ploughing on grassland will interact with climate.

4.6.9 Social and economic barriers

Ploughing occurs on many grasslands in Wales - permanent grassland fields might be ploughed and reseeded every 10 or so years (Farming Connect, pers. comm.). Switching to min-till agriculture might be feasible for some Welsh farmers, but no-till management might reduce profitability if sward cannot be effectively managed and weeds cannot be kept under control.

Transitions to arable may be unlikely due to availability of suitable land in Wales - Freibauer et al. (2004) report no change in arable extent across Europe since 1992, for example. However, Welsh Agricultural June survey statistics show that the area of cropland in Wales fell steadily between 1945 and 2007 (from 359,000 to 64,000 ha), but have since increased slightly to 94,000 ha in 2018 (https://statswales.gov.wales/Catalogue/Agriculture/Agricultural-Survey). It is likely that the maximum extent of cropland post-WWII included a large area of what was unsuitable land.

4.7 Afforestation, Hedgerows, Agroforestry and Habitat Restoration

4.7.1 Causality

AMBER: Evidence around effects on SOC from afforestation/agroforestry on pastures is ultimately mixed. However, given potential positive effects on biodiversity and above-ground carbon storage, some such interventions warrant serious consideration.

Previous work reports mixed effects of afforestation of grassland for SOC (Soussana et al., 2004). Furthermore, Guo & Gifford (2002) report negative effects of converting pasture to plantation forest or secondary forest, although for secondary forest the effect was non-significant. Notably, conversion from forest to pasture was found to increase SOC. Accordingly, Wiesmeier et al. (2019) suggest that “the storage of SOC increases in the order cropland < forest < grassland”, noting some exceptions between forest and grassland. For example, a study of land use change from agriculture (mostly pasture and rough grazing) to Short-Rotation Forestry (SRF) using transitional chronosequence sites across Great Britain highlighted the large variability in estimated rates of change in soil C (positive to negative). Coniferous plantations tended to increase soil C at 0-30 cm depth, in addition to increasing mass of the litter layer (Keith et al., 2015). Another study from the same project demonstrated consistent decreases in soil C stocks with land use change from grassland to Short-Rotation Coppice (SRC) willow plantation (Rowe et al., 2016).

Meta-analyses looking at profile & composition suggest afforestation of grassland will generally release aggregated C from subsoil (30-80cm), but eventually (48yrs) the system (including forest floor) will have a net gain from increased POM (Poeplau et al., 2013, 2011). If only mineral soil is considered, the carbon debt takes longer (closer to 150 years) to be repaid, indicating that it takes a long time for a small proportion of the POM to become incorporated. The depth considered affects estimated rate of change: for grassland to forest- loss of soil C was faster with increasing depth (when forest floor was removed for sampling).

In terms of agroforestry on grasslands, research in Canada has shown that SOC stocks to 30cm were significantly greater in the forested areas than in adjacent herblands (Baah-Acheamfour et al., 2015).

Little evidence has been collected on the impacts of hedgerows on SOC, but one recent study based in the Conwy catchment revealed subtle positive effects of hedgerows on SOC stocks (Ford et al., 2019).

4.7.2 Co-benefits and trade-offs

The most relevant co-benefit would be an increase in above-ground carbon. Keith et al. (2015) showed that aboveground C stocks of trees in some SRF plantations were equivalent to the topsoil C stocks (0-30 cm); rates of aboveground C change were greatest for Sitka and Eucalyptus. Depending on the type of woodland restored, there may be other co-benefits such as recreation and timber production.

There is evidence that hedgerows in the British landscape provide regulatory services by improving water quality, reducing flood risk, reducing soil losses through water and wind erosion, improving crop pollination by providing pollinator habitat and climate change mitigation through the storage and accumulation of carbon above and below ground (Wolton et al., 2014).

Silvopastoral and silvoarable agroforestry in the UK can provide shelter and shade for livestock and crops, improve nutrient cycling, improve air quality through pollutant capture, provide habitat for pollinators and other wildlife and improve water retention (Jose, 2009; Smith, 2010). There is also potential for silvopastoral agroforestry to act as riparian buffer strips. Riparian buffer strips have interactions with terrestrial and aquatic environments, and are often characterised by high primary productivity and plant and animal biodiversity. They provide benefits for water quality downstream i.e. via uptake and assimilation of nutrients from groundwater and surface water, promote stream bank stability and erosion control, forage and habitat for wildlife and space for flood water storage resulting in improved flood defence downstream (Naiman and Décamps, 1997; Sabater et al., 2003; Wharton and Gilvear, 2007).

In terms of runoff reduction, Chandler et al. (2018) compared soil saturated hydraulic conductivity between ungrazed farm woodland under contrasting tree species (Scots pine and sycamore), grazed silvopasture and upland pasture; this study showed that the coniferous farm woodland had the greatest saturated hydraulic conductivity but also that grazing negated beneficial effects of trees on water regulation.

Notable trade-offs following habitat restoration and afforestation include reductions in agricultural productivity, and the ‘locking up’ of land in forestry for decades. Furthermore, if large conifer plantations are established, this is perceived by some to negatively affect the landscape and the community (Farming Connect, *pers. comm.*).

4.7.3 Magnitude

Soussana et al. (2004) propose a small increase in C following afforestation of grassland of 0.1 t C ha^{-1} y^{-1}, with a high degree of uncertainty.

A meta-analysis found that conversion from pasture/grassland to agroforestry increased SOC stocks by 9-10% (De Stefano and Jacobson, 2018).

4.7.4 Timescale

Timescales for effects of afforestation on SOC are unclear.

4.7.5 Spatial issues

Soussana et al. (2004) highlight positive SOC impacts of afforestation on clay or calcareous soils in a mountain climate, but negative effects in warmer climates on sandy or acidic soils. Poeplau et al. (2011) found soils with higher clay content lost SOC more slowly on transition from grass to forest.

For afforestation, forest type and management are also important, for example Guo and Gifford (2002) found decreases for coniferous plantation in wet regions, but no change for broadleaved woodland. Pérez-Cruzado et al. (2012) found topsoil losses in the first 10 years after afforestation, which were dependent on tree species (comparison of 2 eucalyptus species). In monoculture plantations and 'common garden' experiments, clear tree species effects on SOC and other soil attributes have been demonstrated (Reich et al., 2005; Vesterdal et al., 2012).

4.7.6 Displacement

Reduction in agricultural productivity locally could contribute to intensification elsewhere, for example through effects on prices of agricultural goods.

4.7.7 Longevity

Afforestation is likely to achieve permanence due to the mechanical difficulties of reversing this process, and legal protection of woodlands.

4.7.8 Climate interactions

Local climate may affect the type of forest that is suitable to restore. Higher temperature increased rate of SOC loss for this transition in a meta-analysis (Poeplau et al., 2011).

4.7.9 Social and economic barriers

Large coniferous plantations may negatively affect farming communities. Farmers are unlikely to want to afforest large areas of land because of the permanence of this decision. However, smaller areas of land providing wood chip and/or shelter could be of interest - wood chip prices have recently boomed in relation to biomass burners. Farmers may also be amenable to creation of hedgerows, although effects of hedgerows on SOC are not well quantified (Farming Connect, *pers. comm*.).

5 Interventions on Arable Land

In this review, arable land or cropland includes all land in a crop rotation that is not managed permanently as grassland. Globally speaking, cropland management methods may have greater biophysical GHG mitigation potential than either grazing land management or restoration of cultivated organic soils (Smith et al., 2008b). However, in Wales cropland is a clear minority – 75% of land use in Wales is grassland pasture (Armstrong, 2016).

While crop type may affect SOC in arable soils (Moxley et al., 2014), there was not sufficient evidence to represent an intervention in this review. Similarly, intercropping of field crops might improve soil C storage relative to monocultures, but has not been widely investigated (Whitmore et al., 2015). There are also possible benefits of irrigation and water management for SOC, though these effects are mixed and sparsely reported (Smith et al., 2008b).

5.1 Increased Manufactured Fertiliser

5.1.1 Causality

PINK: There is evidence from outside of Wales supporting an increase in SOC following manufactured fertiliser application to cropland. Evidence from Wales is scarce. Furthermore there are severe trade-offs, notably nitrogen pollution and increased GHG emissions. Agronomy and targeting are crucial to secure SOC and productivity while minimising trade-offs.

N fertilisation of crops increases SOC over time according to many studies (Snyder et al., 2009). The primary mechanism for this is supposed to be increased biomass production. Furthermore, SOM stabilises with a C:N ratio of 10:1, so N inputs may be critical to assist C stabilisation (Snyder et al., 2009). However, the review of Alvarez (2005) found that N fertiliser only increased SOC if crop residues were incorporated into the soil.

5.1.2 Co-benefits and trade-offs

See the separate, Soil Nutrient Management review for more information on this intervention type. A key trade-off is increased N_2O emissions and leaching of nitrogen following application of manufactured fertiliser (Buckingham et al., 2013), as well as high CO_2 emissions from production of N fertiliser (Freibauer et al., 2004).

Co-benefits include an increase in productivity where nutrients limit plant growth.

5.1.3 Magnitude

Estimates of change in soil C stocks caused by manufactured fertiliser application on croplands ranged from 11 to 23 t C ha^{-1} in a review to inform the LULUCF inventory (Buckingham et al., 2013). Powlson et al. (2012) present evidence that NPK fertiliser increases SOC slightly, but not nearly as much as farmyard manure.

5.1.4 Timescale

Effects of manufactured fertiliser on SOC on croplands are slight, so the timescales of these effects are difficult to pinpoint.

5.1.5 Spatial issues

N fertiliser effects on SOC tend to be greater in coarse textured soils and at low temperatures (Alvarez, 2005). Furthermore, SOC increases would not occur if N is applied in excess of crop needs, or if SOC is at equilibrium (Alvarez, 2005). Buckingham et al. (2013) note that while N addition is expected to drive increases in SOC in temperate more than tropical zones, UK arable soils generally have high N inputs which may already match crop requirements, so scope for intervention may be limited.

Clearly spatial and temporal targeting is critical if N fertiliser is used to store SOC, especially in light of non-linear effects of nitrogen excess (Snyder et al., 2009). Indeed, improved agronomy and nutrient management consistently increased CO_2 mitigation potential in the review of Smith et al. (2008b).

5.1.6 Displacement

Cessation of fertiliser in some regions could reduce production, possibly displacing intensive agriculture to other regions.

5.1.7 Longevity

Effects on SOC are slight and it is unclear whether they would persist after cessation of manufactured fertiliser. This would depend on the alternative management.

5.1.8 Climate interactions

It is unclear how effects of manufactured fertiliser on SOC interact with climate.

5.1.9 Social and economic barriers

Prices of manufactured fertilisers may be subject to change depending on oil prices.

5.2 Organic Inputs

5.2.1 Causality

BLUE: There is good evidence that carefully targeted organic inputs can increase SOM on cropland. The benefits are greatest when dependence on manufactured fertiliser is reduced, along with associated GHG emissions. Care must be taken to avoid nutrient excesses and nitrogen leaching. When applying manures to crop fields, transportation costs and displacement effects must be considered.

Organic inputs, including farmyard manure, compost, biosolids and incorporation of crop residues, have been demonstrated to have positive effects on SOC on cropland (Powlson et al., 2012; Wuest and Gollany, 2012). The mechanisms for this are direct C inputs as well as increased productivity and C inputs from plant matter. However, organic inputs could have a priming effect on microbial activity, which could result in SOC mineralisation and CO_2 efflux (Buckingham et al., 2013).

Processing of organic fertiliser may increase SOC storage. Chenu et al. (2019) highlight a number of studies suggesting greater long term SOC storage from labile, easily degraded compounds than recalcitrant, lignin-rich material. This may be because labile compounds are processed with higher microbial carbon use efficiency, increasing SOC storage as microbial necromass. Another explanation is that soluble compounds migrate in soil between mineral surfaces, where they can be protected. However, a large experiment compared effects of compost, manure, digestates and slurry on various soil properties and crop yields across seven UK sites found contrasting results (WRAP, 2015). Overall, the experiment found that composts and farmyard manures increased SOM as compared with digestates or slurry (given roughly equivalent N in dosage). As such, it remains unclear whether "fresh" or "processed" organic matter will have the greatest benefits for C sequestration, especially given possible emissions while processing organic materials.

"Biochar" is another form of organic material that can be added to soil, representing products obtained by thermal treatment of organic material in low oxygen conditions. Biochar can be a side-product of liquid biofuel production. It constitutes a stable form of C in itself, but there is also inconclusive evidence that it might confer stability to existing fractions of organic matter in soil (Powlson et al., 2011). Most of the evidence base for biochar originates outside Europe.

5.2.2 Co-benefits and trade-offs

See the separate, Soil Nutrient Management review for more information on this intervention type. Application of manures and composts as opposed to slurry, digestates or manufactured fertilisers can increase earthworm numbers and nutrient availability (WRAP, 2015). Furthermore, with an integrated nutrient management plan, considerable savings could be made on manufactured fertiliser.

N_2O emissions are a possible trade-off from manure, sewage sludge and urban compost, although emissions from production of manufactured fertilisers could be reduced by using organic fertilisers (Freibauer et al., 2004). Crop residues could increase N_2O emissions by placing a source of mineralisable N into the soil (Freibauer et al., 2004), especially when residues have a low C:N ratio (Baggs et al., 2000). Nitrogen leaching could also be an issue when applying manure (Buckingham et al., 2013), and excesses of non-N nutrients may occur (e.g. P).

Freibauer et al. (2004) identified that, if poorly regulated, sewage sludge could lead to build up of heavy metals and organic pollutants in the soil. On the other hand, urban compost could increase the availability of trace minerals in the soil. However, sewage sludge is well regulated in the UK, and one UK experiment identified "no effect of compost or digestate additions on soil total metal and organic compound contaminant concentrations or crop metal concentrations" (WRAP, 2015).

For biochar, a full life-cycle assessment is needed to understand the trade-offs and co-benefits. It originates from a variety of organic source materials with various economic and environmental consequences. However, biochar may improve nutrient and water retention in soil, as well as crop growth (Powlson et al., 2011). This could reduce demand for N fertiliser, resulting in co-benefits in terms of GHG mitigation.

5.2.3 Magnitude

Evidence suggests that manure additions cause greater C storage per unit N than manufactured fertiliser (Buckingham et al., 2013). Furthermore, C storage efficiency from biosolids, including manure, may exceed that from cereal residues (Powlson et al., 2012; Wuest and Gollany, 2012).

Estimates of change in soil C stocks caused by manure application on croplands ranged from 5 to 18 t C ha^{-1} in a review to inform the LULUCF inventory (Buckingham et al., 2013).

Powlson et al. (2012) reviewed SOC impacts of various biosolid inputs to arable soils, finding variable positive effects of farm manures in the region of 0.63 t C ha^{-1} y^{-1} (at the maximum permitted application rate in UK nitrate vulnerable zones). Other inputs such as sewage sludge and green compost had greater positive effects.

Smith et al. (1997) collated experimental data on SOC from EU countries and found that addition of animal manure, sewage sludge or straw had a lower C sequestration potential than extensification (a switch to ley-arable farming).

A recent meta-analysis highlighted that cover crops increase SOC by ~6%, while biochar application increases SOC by ~39% (Bai et al., 2019).

Several studies suggest that C storage efficiency from cereal residues tends to be lower than other biosolids, including manure (Powlson et al., 2012; Wuest and Gollany, 2012).

5.2.4 Timescale

Getahun et al. (2018) found that soil loosening and straw slurry incorporation into arable soil increased SOC by >20g/kg after just one year, with positive effects on grain yield. Powlson et al. (2012) demonstrated that increases in SOC due to organic inputs were greatest within the first 20 years of application, after which they diminished. One UK experiment found effects of farmyard manure and green compost after 9 and 20 years of application respectively (WRAP, 2015).

5.2.5 Spatial issues

Manure application to arable soils comes with limitations to do with immediate availability of manure in the surrounding area (Powlson et al., 2011).

One study suggests that SOC inputs from cereal residues or biosolids are contingent on suitable amounts of N, P and S (Kirkby et al., 2013). The amount of nutrients needed may be quite predictable, and if not already present might be supplemented using fertilisers. However, the implication in the case of incorporation of cereal residues may be to fertilize stubble, which is not be recommended due to likely nitrate leaching (Buckingham et al., 2013).

N addition needs to be carefully targeted to minimise trade-offs. Indeed, improved agronomy and nutrient management consistently increased CO_2 mitigation potential in the review of Smith et al. (2008b).

5.2.6 Displacement

Manure application on arable land must be linked with GHG emissions from livestock production elsewhere (Buckingham et al., 2013). It could also be linked to an increase in NPK application elsewhere.

Powlson et al. (2011) highlight the importance of the alternative fate of organic inputs, e.g. cereal straw. If the straw would have been burnt, it may be preferable to incorporate the carbon into the soil. However, burning cereal straw could also reduce fossil fuel combustion and help mitigate climate change (Powlson et al., 2008). Another pathway is for straw to be used as animal bedding, in which case it would largely end up incorporated in SOM elsewhere (Powlson et al., 2011).

5.2.7 Longevity

It is unclear whether increased SOC would persist after cessation of organic inputs. Effects on SOC tend to be saturating with time (Powlson et al., 2012).

5.2.8 Climate interactions

Although most of the evidence on biochar comes from outside Europe, positive SOC impacts of biochar are reported to be greater in cool regions (Bai et al., 2019).

5.2.9 Social and economic barriers

Redistribution of manure towards arable regions comes with severe barriers in terms of storage and transportation.

5.3 Reduced and Zero Tillage

5.3.1 Causality

AMBER: **SOC impacts of reduced tillage are mixed, and confused by effects of sampling depth and bulk density. Some SOC gains and co-benefits are possible, particularly in association with cover cropping, but there are risks of N_2O emissions and yield losses.**

Tillage of soil has been considered a major driver of reduction in SOC on agricultural land, so no-till management has been recommended for C sequestration (Lal, 2004). Physical disturbance during tillage disaggregates and aerates the soil, accelerating SOC decomposition (Mikha and Rice, 2014), and reduced tillage is thought to prevent this.

The effects of no-till and reduced-tillage management on SOC have been scrutinised in recent years. After considering bulk density and SOC distribution with depth, the evidence for SOC increases in no-till systems is reduced (Angers and Eriksen-Hamel, 2008; Baker et al., 2007). Specifically, reducing tillage is thought to lead to changes in the distribution of carbon with depth, with an increase in C concentration at the soil surface. For the above reasons a review to inform the LULUCF inventory concluded that tillage reduction is not a reliable option to increase the SOC of UK soils (Buckingham et al., 2013). A recent meta-analysis found that the impact of conservation tillage on SOC is small, but positive (Bai et al., 2019).

However, if tillage increases the distribution of C with depth, there may be positive effects in terms of reduced C decomposition. Deeper soil horizons tend to contain less carbon (Jobbagy and Jackson, 2000). Furthermore, organic matter that is incorporated to deeper parts of the soil may be degraded more slowly, or readily adsorbed onto fine mineral particles which may be less saturated than at the surface (Buckingham et al., 2013). For this reason deep ploughing to bury SOC-rich topsoil has been considered as a SOC storage intervention (despite energy costs), while vertical redistribution of carbon by anecic earthworms is also of interest (Chenu et al., 2019).

The positive effect of no-till management on SOC may have been overstated. Some researchers claim that factors other than tillage have been the key drivers of C declines on arable land – e.g. conversion to annual crops, periods of bare soil and drainage (Baker et al., 2007).

5.3.2 Co-benefits and trade-offs

Trade-offs include a risk of increase N_2O emissions in poorly aerated soils, often found in NW Europe, which is concerning due to the increased global warming potential of N_2O (Freibauer et al., 2004; Rochette, 2008); mitigation potential of reduced tillage could be reduced by 50-60% after consideration of increased N_2O emissions (Freibauer et al., 2004). A model by Li, et al. (2005) suggested that the GWP of N_2O emissions following reduction in tillage offset the benefits of increased soil C storage by 75-310%. However, no-till could also reduce N_2O emissions due to reduced availability of N, through protection in aggregates or concentration in surface residues. Direction of change in N_2O emissions may be affected by: previous land use and associated C and N accumulation, amount and type of fertiliser inputs (Hellebrand et al., 2008; Novoa and Tejeda, 2006), soil type (Rochette, 2008), humidity (Regina and Alakukku, 2010), soil moisture, climate, soil physical properties and topography (Li et al., 2005).

Furthermore, increased herbicide usage may be required under reduced tillage, which may be expensive and have negative environmental consequences (Buckingham et al., 2013). Furthermore, there are possibilities of crop failure or reduced yield in reduced tillage systems (Freibauer et al., 2004).

Co-benefits of reduced tillage include reduced costs and GHG emissions associated with fuel consumption (Buckingham et al., 2013). Buckingham et al. (2013) report variable effects of zero-till management on crop production, but there may be positive effects on soil moisture (Freibauer et al., 2004). There are also positive effects of reduced tillage on the soil biota, particularly earthworms and fungi, with associated benefits for soil structure and function (Spurgeon et al., 2013). Preventing tillage could also reduce sediment runoff and P losses (Haygarth and Jarvis, 1999). There are possible benefits of reduced tillage for soil aggregation and improved infiltration.

Conservation tillage is often associated with cover cropping (Chenu et al., 2019) which has also been shown to increase SOC (Bai et al., 2019).

5.3.3 Magnitude

A review to inform LULUCF inventories suggested changes of -2.2 to 8.1 t C ha^{-1} when changing to reduced tillage (Buckingham et al., 2013). A recent review in the UK suggested increases -0.23-1.37 t C ha^{-1} y^{-1}, but studies considered only sampled soil to 30cm depth, assuming consistent bulk density (Powlson et al., 2012). Freibauer et al. (2004) reported a range of 0.3-0.4 t C ha^{-1} yr^{-1} for zero- or reduced-tillage interventions in Europe.

5.3.4 Timescale

Powlson et al. (2012) report mixed to positive effects of zero-tillage on SOC (to 30cm depth) within 5-23 years.

5.3.5 Spatial issues

Trade-offs include a risk of increased N_2O emissions in poorly aerated soils, often found in NW Europe (Rochette, 2008).

Snyder et al. (2009) suggest that SOC increases may only occur if crop productivity can be maintained or increased. The effects of conservation tillage are also dependent on soil type and temperature (Luo et al., 2010).

5.3.6 Displacement

If reduced tillage corresponds to a reduction in yield, it could be argued that more intensive farming is required elsewhere to meet demands for food.

5.3.7 Longevity

A return to full tillage could reverse any positive SOC effects of reduced tillage.

5.3.8 Climate interactions

A recent meta-analysis highlighted that conservation tillage increases SOC by ~5%, with more positive effects of no-tillage management in regions characterised by warm climates (Bai et al., 2019). Trade-offs include a risk of increased N_2O emissions in poorly aerated soils, often found in NW Europe (Rochette, 2008).

5.4 Cover Crops

5.4.1 Causality

BLUE: **Cover crops are likely to prevent SOC loss through erosion, and may increase SOC through residue returns. There are risks of N_2O emissions, but possible decreases in nutrient leaching and increases in productivity.**

Cover crops can be used to eliminate bare ground, potentially increasing SOC by either increasing productivity (especially if plant residues are returned to the soil) or preventing erosion (Buckingham et al., 2013; Desjardins et al., 2005). Bare soil tends to lose carbon, at least in part due to erosion (although erosion may represent a considerable C sink in the UK; Quinton et al., 2006). Bai et al. (2019) found that leguminous cover crops were associated with greater SOC sequestration than non-leguminous crops.

5.4.2 Co-benefits and trade-offs

A reduction in bare soil could also prevent leaching of nutrients from the soil. For example, cover crops increase crop P nutrition (Hallama et al., 2019). This could prevent pollution but also increase yield.

A model produced by Lugato et al. (2018) suggests that over time, N_2O emissions from N-fixing cover crops could be sufficient to offset climate change mitigation effects of SOC sequestration.

5.4.3 Magnitude

A meta-analysis has found that cover crops could increase SOC by 0.32 t C ha^{-1} yr^{-1} (Poeplau et al., 2018). Another meta-analysis highlighted that cover crops increase SOC by ~6% (Bai et al., 2019).

5.4.4 Timescale

The timescales over which cover crops increase SOC are not clear.

5.4.5 Spatial issues

Cover crops can increase crop P nutrition, and may be more effective in areas with low available P (Hallama et al., 2019).

5.4.6 Displacement

It is unclear whether there are any displacement effects of cover cropping.

5.4.7 Longevity

It is unclear how long SOC increases due to cover cropping will be maintained, particularly if cover crops are discontinued.

5.4.8 Climate interactions

Positive SOC impacts of cover crops are reported to be greater in regions characterised by warm climates (Bai et al., 2019).

5.4.9 Social and economic barriers

Cover crops come at an immediate expense to the farmer in terms of seed and machinery costs.

5.5 Conversion to Grassland or Perennial Crops

5.5.1 Causality

BLUE: There is clear evidence that converting arable land to grassland increases SOC. This could benefit biodiversity and reduce nutrient leaching, but there may be trade-offs with profitability. Displacement effects may occur if grassland elsewhere is converted to arable.

A report compiled to inform LULUCF inventories concluded that the impact of cropland management on SOC is likely to be small compare to e.g. land use change (Moxley et al., 2014). A key land use change which could increase SOC is the transition from cropland to grassland. Integrating grasses into crop rotations can enhance carbon inputs to soil and reduce loss of carbon due to decomposition, resulting in carbon sequestration (Conant, 2010). Increasing length of grass rotations could also be effective (Moxley et al., 2014). Even small-scale grassland creation can profoundly affect SOC sequestration: 0.1 to 2.4% of 1990 UK CO_2-C emissions could be sequestered using grass margins on arable fields (Falloon et al., 2010).

Rowe et al. (2016) demonstrated significant increases in soil C stocks following land use change from arable to perennial SRC willow coppice.

5.5.2 Co-benefits and trade-offs

Freibauer et al. (2004) highlight that increasing duration of leys could reduce erosion and nutrient leaching. Furthermore, set-aside could lead to biodiversity benefits for some plant and invertebrate taxa (Alanen et al., 2011). Conversion from arable to grassland is expected to benefit earthworms (Spurgeon et al., 2013). However, Freibauer et al. (2004) also note that set-aside can cause issues with weeds in future years, and that switching to perennial crops leaves less room to respond to short-term market changes.

5.5.3 Magnitude

Evidence from Rothamsted for transition from arable to grassland revealed SOC increases of 20 t C ha^{-1} over ~50 years (Johnston et al., 2009).

Freibauer et al. (2004) report that permanent revegetation of arable land e.g. introducing perennial components, short rotation coppicing or perennial grasses, could increase SOC by 0.5–1.9 t C ha^{-1} yr^{-1}. They indicate that a transition from arable to grassland would increase SOC by 1.4 t C ha^{-1} y^{-1}.

A meta-analysis reports that converting croplands to grasslands can increase SOC by 1.01 t C ha^{-1} y^{-1} (Conant et al., 2001). CO_2 mitigation potential of set-aside and land-use change was amongst the highest reported by Smith et al. (2008b) for croplands in cool, moist regions such as Wales.

Smith et al. (1997) collated experimental data on SOC from EU countries and found that extensification (ley-arable farming) or afforestation of arable land could sequester more C than addition of animal manure, sewage sludge or straw.

0.1 to 2.4% of 1990 UK CO2-C emissions could be sequestered using grass margins on arable fields (Falloon et al., 2010).

Poeplau et al. (2015) argued that 7.7% increases in topsoil (0-20cm) SOC in Sweden were primarily attributable to an increase in grass ley as a proportion of total agricultural area.

5.5.4 Timescale

Conversion into a grass ley can increase SOC within five years (Fullen and Booth, 2006).

5.5.5 Spatial issues

Powlson et al. (2011) suggest that potential for C accumulation may be greatest in regions with a low existing C stock; e.g. Rowe et al. (2016) found the greatest increases in soil C stocks following conversion to short rotation willow coppice were found at sites with lowest arable soil C stock.

5.5.6 Displacement

While lots of C could be stored in soils by converting arable land to set-aside or perennial crops, there may be major trade-offs with food production (Moxley et al., 2014). As such, food production may simply occur elsewhere with emissions that offset local increases in SOC.

5.5.7 Longevity

Initial land-use change from cropland to grassland increases SOC stocks, but eventually grassland C gains and losses are expected to be balanced (Buckingham et al., 2013); older grasslands are likely to sequester less C. However, a fraction of SOM that accumulates following a transition becomes stable, and is expected to have a half-life of decades to centuries (Powlson et al., 2011).

The inclusion of 3-year grass or grass-clover leys in a 5-year arable rotation may also have positive effects that saturate after ~30 years (Johnston et al., 2017).

5.5.8 Climate interactions

Smith et al. (2008b) report generally more positive effects of set-aside and land use change in moist climates than in dry climates.

5.5.9 Social and economic barriers

Conversion of arable land to permanent grassland may reduce profitability of that land, and thus might be unfavourable to farmers.

5.6 Afforestation and Habitat Restoration

5.6.1 Causality

AMBER: **Afforestation of arable land will clearly increase SOC as well as above ground C and biodiversity. The issue is the trade-off with productivity – this intervention shows most promise through afforesting unproductive or degraded arable land, or through agroforestry.**

Bossuyt et al. (2010) surveyed soils on sites afforested between 29 and 131 years prior, and found the depth of the organic horizon was negatively correlated with the duration of agricultural use prior to reforestation, and positively correlated with time since reforestation. However, negative impacts could occur at depth (35-60cm) after afforestation of arable land (Richter et al., 1999). This could occur where increased transpiration dries the soil, increasing C decomposition.

In terms of agroforestry, Smith et al. (2008b) reported positive effects of agroforestry in terms of CO_2 mitigation in cool, moist regions such as Wales. Research in France and the Mediterranean has revealed accumulation of SOC under alley-cropping systems, albeit in coarse organic fractions which may be rather labile (Cardinael et al., 2017, 2015).

5.6.2 Co-benefits and trade-offs

Co-benefits of habitat restoration and afforestation on arable land include more C being stored above ground, as well as in the soil. Increases would be expected in earthworms (Spurgeon et al., 2013) and wider biodiversity (Alison et al., 2017). However, there could be trade-offs in terms of N_2O emissions, especially from woodland (Powlson et al., 2011). Effects for flood mitigation are mixed (see the separate Flood Mitigation review).

Freibauer et al. (2004) highlight that afforestation could have landscape and biodiversity benefits, improving leisure/amenity value of the land. However, this may be less the case if a commercial monoculture is established. Afforestation may also increase C sequestration in wood and wood products, although there may be a short period of increased emissions immediately after tree planting (Freibauer et al., 2004). Afforestation also reduces flexibility to respond to market changes.

Depending on the crops, silvoarable agroforestry can also increase total yields and profitability. Silvoarable systems require fewer nitrogen inputs, both because the area of crop is reduced and because the greater litter input and more extensive root systems fix nitrogen in the soil.

Some types of wetland restoration can occur in lowland areas, and provide opportunities to for C storage alongside production of raw material for industry – for example, *Typha* constructed wetlands, which are also used for water purification (Wild et al., 2001).

5.6.3 Magnitude

Evidence from Rothamsted shows that transitions from arable to woodland can store 44-64 T C ha^{-1} over 120 years (Poulton et al., 2003). Freibauer et al. (2004) report SOC increases of 0.5 – 1.4 T C ha^{-1} y^{-1} after converting arable land to woodland.

Smith et al. (1997) collated experimental data on SOC from EU countries and found that extensification (ley-arable farming) or afforestation of arable land could sequester more C than addition of animal manure, sewage sludge or straw.

A recent meta-analysis revealed SOC stock increases of 26-40% at various depths on conversion from agriculture to agroforestry (De Stefano and Jacobson, 2018).

5.6.4 Timescale

Poulton et al. (2003) found increases in SOC stocks on afforested arable land within 25 years, and increases continued for ~100 years.

5.6.5 Spatial issues

Powlson et al. (2011) suggest that potential for C accumulation may be greatest in regions with a low existing C stock. While there may be limits due to soil type or climatic factors, with transition from arable land use an increase in SOC is generally expected.

5.6.6 Displacement

Negative effects of afforestation could occur if food production needs to be displaced (Searchinger et al., 2008). However, displacement effects may be alleviated by planting on degraded cropland which is unproductive (Powlson et al., 2011).

5.6.7 Longevity

Poulton et al. (2003) found increases in SOC stocks on afforested arable land within 25 years, and increases continued for ~100 years.

A fraction of SOM that accumulates following a transition becomes stable, and is expected to have a half-life of decades to centuries (Powlson et al., 2011). Due to difficulty of clearing woodlands, as well as legal protection, afforestation may be quite permanent in the context of Wales and the UK.

5.6.8 Climate interactions

Some types of woodland may only be suitable in specific climates.

5.6.9 Social and economic barriers

Due to difficulty clearing woodland and legal protection, farmers may be reluctant to plant large areas of woodland. Coniferous plantations are viewed by some to negatively affect landscape and community (Farming Connect, *pers. comm.*).

6 Interventions for Upland Soils

Beyond the interventions covered here, bracken control may affect SOC but few studies have been conducted on this. Interestingly, bracken cover fell drastically between 1998 and 2007 across Wales (Smart et al., 2009). Mitchell et al. (1999) report no clear trend in SOC following the control of different types of vegetation succession, including *P aquilinum*, and Marrs et al. (2007) report no decrease in soil C seven years after commencing *P. aquilinum* control in Derbyshire (though aboveground C was lost).

Shooting to control wild grazers is another potential impactful action in the uplands, as this could feed back on vegetation dynamics and SOC. However, we did not find adequate evidence to report on this.

6.1 Prevent Drainage & Restore Peatlands

BLUE: Existing peatlands, even those that represent C sinks, store huge amounts of carbon that needs protecting. Peatland restoration efforts to date have had variable results, and can increase CH_4 emissions. However, peatland restoration also has potential to reduce water treatment costs.

It is clear that peatlands store huge amounts of carbon – it is estimated that they contain a third to a half of global C (Holden, 2005). However, it is less clear which peatlands currently represent C sinks. There is a lot of variability in the evidence, and fluvial carbon fluxes are an important consideration despite being very difficult to measure (Worrall et al., 2011a). Interannual variation may contribute to this variability, as shifts in climate and hydrology may change a site from source to sink or vice versa (Clay et al., 2010). Indeed, two studies of Auchencorth Moss in Southern Scotland, separated by 10 years, yielded quite different results about the C budget (Billett et al., 2010).

One indicator of carbon sequestration is whether a peatland is "active" i.e. peat-forming, with the necessary vegetation and conditions to form peat. Peatland stores carbon largely because enzymes that decompose SOC are slowed by phenolic compounds. Furthermore, enzymes which degrade those phenolic compounds are impaired under typical peatland conditions (Whitmore et al., 2015).

6.1.1 Causality

The main pressures on peatland are drainage, harvesting, grazing, burning and forestry, although lowland peats also come under pressure from arable farming.

Drainage of peatland happens in association with agriculture, forestry and peat harvesting (Laine et al., 2009). Furthermore, government subsidies following the Second World War encouraged the cutting of drainage ditches in peatlands (Holden et al., 2004). Drainage is thought to accelerate the decay of litter and peat by increasing the availability of oxygen. Meersmans et al. (2009) attributed declines in SOC of wet grasslands in Belgium from 1960-2006 to intensive soil drainage. A meta-analysis demonstrates that drainage tends to increase soil respiration, with negative consequences for C sequestration (Worrall et al., 2011a). The drainage ditches themselves are also subject to erosion, resulting in the loss of particulate organic carbon. While most studies report increased DOC losses after drainage, there are exceptions (Buckingham et al., 2013); Worrall et al. (2011a) report mixed effects of drainage on water colour and DOC concentrations, and propose that increases in DOC may be due to a separate widespread driver.

There are two main methods to restore peatland: drainage blocking to raise the water table using a dam or sluice, and re-establishing vegetation such as *Sphagnum* (or even commercial grassy turf) on bare peat (JNCC 2011). The effects of drainage blocking on carbon budgets of peatland are largely unresolved, although recent work has set out to investigate this (Buckingham et al., 2013; Evans et al., 2011). Effects on wetland vegetation such as *Sphagnum* are also inconsistent (Bellamy et al., 2012). Several studies show decreased DOC flux as a result of drain-blocking (Worrall et al., 2011a). One study has analysed the carbon budget of a drained peatland, with some drainage-blocking, in Northern England indicating losses of ~0.64-1.07 t C ha^{-1} y^{-1}. However, the study only took place one year after blocking and there was no appropriate control site for comparison (Rowson et al., 2010; Worrall et al., 2011a).

Erosion of bare peat can result in loss of POC, and efforts to prevent this include peat stabilisation (using geotextiles or brash) or revegetation (Worrall et al., 2011a). Revegetation reduces POC fluxes, but effects on the C budget are variable, which may be related to e.g. use of lime to aid establishment of vegetation. Bare peats may be greater sources of C than revegetated peats, which was partly attributable to greater loss of POC (Worrall et al., 2011b). There is little information on the effects of different revegetation practices, or on the effects of restoration of cutover peatlands.

6.1.2 Co-benefits and trade-offs

Restored wetlands can be carbon dioxide sinks, but also sources of methane (Knox et al., 2015). Accordingly, drained peatlands are not always net GHG sources provided CH_4 emissions are sufficiently low (Worrall et al., 2011a).

Peatland restoration can have positive effects on water quality, to the extent that water companies already have a commercial incentive to restore peatland to minimise treatment costs (Smyth et al., 2015). Other potential co-benefits include flood prevention and wetland biodiversity.

6.1.3 Magnitude

0.4-19 t C ha^{-1} y^{-1} may be lost when converting wetlands to agriculture in boreal and temperate zones outside Europe; within Europe, losses of up to 4.6 t C ha^{-1} y^{-1} could be prevented by protecting peatland (Freibauer et al., 2004). Furthermore, 0.1-1 t C ha^{-1} y^{-1} could be accumulated through wetland restoration (Freibauer et al., 2004).

Many peatland studies focus on C/GHG budgets and whether sites are a C source/sink rather than just SOC. This is partly because of the difficulty of sampling and measuring peat to depth (Buckingham et al., 2013).

A study on Auchencorth Moss in Southern Scotland found that this peatland was a net C sink in the region of 0.7 t C $ha^{-1}y^{-1}$ (Dinsmore et al., 2010), while a study at Hard Hill plots in Moor House found that the unmanaged peatland was a net source in the region of 1.6 t C $ha^{-1}y^{-1}$ (Clay et al., 2010). Both studies demonstrate that fluvial export of C via DOC is an important unknown in the carbon budget, much of which could be returned to the atmosphere (Worrall et al., 2011a, 2006).

Restoration of organic soils, particularly re-establishing a high water table, could mitigate 3.67-69.67 t CO_2 ha^{-1} yr^{-1} in cool, moist regions (Smith et al., 2008b), although this would be incompletely offset by increased CH_4 emissions.

6.1.4 Timescale

Even after six years, drainage blocking may not fully restore the water table of a peatland. One study found that after six years the behaviour of the water table of a drain-blocked peatland was intermediate between that of a drained and an intact peatland (Holden et al., 2011).

Grand-Clement et al. (2015) summarise a range of studies of peatland restoration, finding mostly reduced CO_2 emissions and increased CH_4 emissions within 10 years; increases in depth to the water table and reduced or buffered runoff within one to three years and natural recolonization of *Sphagnum* and/or *Eriophorum* within six years. Effects on DOC concentrations and export were mixed.

6.1.5 Spatial issues

The practical potential of peatland restoration for C sequestration may be restricted because drained lands provide livelihoods and habitation for many people (Powlson et al., 2011).

6.1.6 Displacement

It is unclear whether protection of peatland or rewetting will have displacement effects.

6.1.7 Longevity

Drainage blacks can generally be removed at a later date, which could reverse any increases in SOC.

6.1.8 Climate interactions

DOC concentrations in UK upland streams have increased in recent decades, possibly due to falling acid deposition and rising temperatures (Evans et al., 2005). It is possible that an increasing quantity of Welsh peatlands become C sources over time.

6.1.9 Social and economic barriers

Upland farmers have quite readily taken up drainage blocking incentives in the past. Many upland regions are quite unproductive, so economic barriers to uptake may be small.

6.1.10 Metrics and verification

The peatland code has developed a carbon metric for peatlands in a range of conditions, aiming to provide a mechanism for businesses to directly sponsor peatland restoration for carbon benefits (Smyth et al., 2015). A set of updated, IPCC tier 2 emissions factors for a variety of peatland condition categories were produced very recently (Evans et al., 2019a).

6.2 Prevent Improvement & Reduce Grazing

6.2.1 Causality

AMBER: **Preventing improvement in the uplands is likely to prevent SOC losses, but reducing grazing may not automatically lead to recovery of SOC.**

More research is needed to understand the drivers of SOC decline in the uplands, and how to combat them.

European Common Agricultural Policy (CAP) subsidies for livestock farming based upon headage are said to have resulted in a 30% increase in sheep numbers on UK moorlands between the 1970s to 1990s (Worrall et al., 2011a). Intensification of upland grassland *could* positively impact SOC, for example by increasing root exudation (Hamilton III et al., 2008), but expert opinion states that improving rough grassland on organo-mineral soils will negatively affect SOC (Buckingham et al., 2013). It is possible that maintaining extensive management positively impacts SOC where nutrients are not limiting. Some evidence suggests that reduced grazing would increase SOC in the uplands. On upland *Molinia* grassland in Scotland, SOC would likely decrease under commercial sheep stocking rates but increase with low sheep grazing or no grazing (Smith et al., 2014). However, Marrs et al. (2018) found that removal of sheep grazing alone made little difference to soil chemistry and vegetation biomass at Moor House nature reserve.

Grazing is a key limiting factor for vegetation succession, which can in turn affect SOC. Overgrazing by livestock can encourage the dominance of graminoids (JNCC 2011). A wider trend on UK peatlands from *Calluna* to *Molinia* and *Nardus* has been partly attributed to overstocking of sheep and deer (Pakeman et al., 2003; Worrall et al., 2011a). Grazers also contribute through trampling and changing the nutrient status of the soil (Worrall et al., 2011a). A decline in heather has also been seen on moorlands more widely across England and Wales (Bardgett et al., 1995). One study reported no clear effect of vegetation succession on SOC on heaths in Dorset (Mitchell et al., 1997). However, analysis of GMEP and Countryside Survey data has shown positive associations between SOC concentration and cover of both *Sphagnum* and Ericoid shrubs (mainly heather) across upland Wales (J. Alison, unpublished data).

One possible mechanism for C loss on upland soils following intensification is a priming effect, whereby fresh C inputs to a system facilitate microbial decomposition of recalcitrant SOC (Soussana and Lemaire, 2014). Fresh C inputs could arise from manure of introduced grazers, or following plant community shifts due to application of fertiliser and lime. Furthermore, decompositions rates in the uplands are thought to be constrained by acidity, so liming might increase not only plant productivity but C decomposition and turnover (Buckingham et al., 2013).

Worrall et al. (2011a) suggest that agricultural management effects on peatland C and GHG fluxes may be the most severe, as cultivated peatlands and peatland managed as improved grassland have disproportionately high emissions. However, they note that their inference is based on non-UK, tier 1 emissions factors (Penman et al., 2003). One review states that grazing of some lowland peatlands has ceased, leading to the encroachment of scrub (JNCC 2011).

Experiments at Moor House have compared the carbon budget of grazed peatland with unmanaged peatland, finding that grazing increased primary productivity and reduced respiration (Clay et al., 2010; Ward et al., 2007). The result was a decrease in the size of the C source of grazed sites of 0.36 t C ha^{-1} yr^{-1}. Nonetheless, the grazing in that study was very light, and the meta-analysis of Worrall et al. (2011a) suggests that grazing removal, rather than addition of grazing, generally lead to an improved carbon and GHG budget.

6.2.2 Co-benefits and trade-offs

A possible co-benefit of grazing removal would be increased C stored in biomass above ground, and increased recalcitrance of SOC produced by plant litter as opposed to grazers (Worrall et al., 2011a). There would be a clear trade-off in terms of agricultural production, although many upland areas are not very productive.

6.2.3 Magnitude

One review reports SOC change between -0.9 – 1.1 t C ha^{-1} y^{-1} for intensification of organic soils (Freibauer et al., 2004).

6.2.4 Displacement

Reduced grazing in the uplands could lead to displacement effects with increased grazing elsewhere.

6.2.5 Climate interactions

Reductions in atmospheric sulphur deposition could also lift constraints on decomposition of C.

6.2.6 Social and economic barriers

Economically speaking, upland farms struggle to profit without subsidies. While incentives to reduce stocking rates in upland areas might be readily taken up by some farmers, reducing stocking may run against some farmers' ideologies (Farming Connect, *pers. comm.*). Anecdotally, hefting of sheep on common land may become increasingly difficult as sheep numbers diminish, which could result in a positive feedback and abandonment of some upland areas.

6.3 Controlled Burning

6.3.1 Causality

AMBER: **Evidence of controlled burning effects on SOC in the Welsh uplands is not sufficient to reach a conclusion. Reported impacts will vary depending on the stage of the burning rotation, soil type, and any associated drying of the soil. It is unclear if and where the alternative to controlled burning would be wildfire, which could lead to even greater SOC loss.**

Controlled burning may be carried out during management for heather or grouse, to prevent wildfire or simply to clear shrubs. Moorland burns for game bird management have increased across Scotland, England and Wales since the year 2000 (Douglas et al., 2015). Looking further back in time, Yallop et al. (2006) reported an increase in the extent of some types of managed burn between 1970 and 2000 in the English uplands. However, burning extent and trends across Wales specifically are unclear. Burning involves loss of carbon to the atmosphere above ground, but effects on carbon below ground are context dependent, and widely debated. Harper et al. (2018) report generally negative effects of burning management on SOC, stipulating that the weight of evidence in the UK is low.

Effects of burning on SOC may be mediated by changes in the vegetation community. Burning is expected to favour *Calluna* over *Sphagnum* and *Eriophorum*, depending on the length of the burn cycle (Hobbs, 1984). For example, muir burning for grouse encourages heather over other blanket bog species (Worrall et al., 2011a). Furthermore, late winter burning can encourage the dominance of graminoids, especially *Molinia caerulea* (JNCC, 2011). Given that *Sphagnum* is peat-forming vegetation, this is likely to negatively affect SOC accumulation on blanket bog habitats. This reduction in C accumulation rates been clearly demonstrated in the only long-term managed burning manipulation experiment in the UK, at Moor House, based on peat core analyses (Garnett et al., 2000; Marrs et al., 2018). However, one study suggests that more frequent burning rotations might favour *Sphagnum* as compared with less frequent burning rotations (Lee et al., 2013). Work by Grau-Andrés et al. (2019) suggests some resilience of *Sphagnum* to low intensity fires, while Noble et al. (2019) show that high intensity fires often result in 100% sphagnum cell damage.

Vegetation change could also drive C loss through DOC exports. For example *Calluna* is associated with increased DOC concentrations compared with *Sphagnum* and *Molinia* (Armstrong et al., 2012). However, the effects of burning on DOC have been highly variable; most studies are not experimental in nature (Evans et al., 2017), and operate at a variety of different spatial and temporal scales (Worrall et al., 2011a). Furthermore, it is difficult to understand what the steady state is because burning management usually forms a patchwork at sites at different stages of the burn cycle (Worrall et al., 2011a).

Positive effects of burn management on SOC have also been reported. Two studies have compared the carbon budget of burned peatland with unmanaged peatland, finding that burning increased primary productivity and reduced respiration (Clay et al., 2010; Ward et al., 2007). Clay et al. (2010) found that while the investigated peatlands were C sources, burning management reduced the C source by in the region of 0.4 t C $ha^{-1}y^{-1}$. However, the results of this short-term study are at odds with two core-based studies of carbon accumulation at the same experimental site, which were undertaken before (Garnett et al., 2000) and afterwards (Marrs et al., 2019), both of which clearly show that burning reduced long-term C accumulation rates compared to a no-burn control. It is worth noting that the unmanaged peatlands in these studies were dominated by mature and degenerate *Calluna vulgaris*, which is not typical peat-forming vegetation (Worrall et al., 2011a).

Heinemeyer et al. (2018) used spheroidal carbonaceous particle (SCP) dating to assess carbon accumulation in relation to burning frequency, and found a positive association between burning and carbon accumulation. However, issues with the study design, methods, and interpretation within that paper have been brought to light, including a lack of appropriate control sites and possible errors in the dating methods that would invalidate the estimated C accumulation rates (Evans et al., 2019b). At the current time this study appears to be an outlier in showing apparent C benefits from managed burning, although there is a lack of evidence more generally, making it difficult to draw firm conclusions in this area.

Some possible mechanisms of increases in SOC under burning management would be increases in plant productivity, or the production of pyrogenic carbon which is more resistant to degradation than original biomass (Harper et al., 2018). However, more research is clearly needed into impacts of burning on SOC.

6.3.2 Co-benefits and trade-offs

Controlled burning is often argued to reduce the risk and intensity of wildfires, although there is still uncertainty on how wildfires and controlled burns interact

(Davies et al., 2016). Other measures may be used to reduce the accumulation of above-ground woody biomass, for example, cutting or mowing and then leaving residues on-site to form a mulch (Worrall et al., 2011a).

Harper et al. (2018) highlight a range of evidence of effects of burning on water quality, such as increases in DOC and particulate organic carbon (POC). This is important given the amount of water coming from catchments in which burning takes place; Harper et al. (2018) specifically cite the Brecon Beacons national park as providing 90% of the drinking water to the wider urban area of Cardiff (https://www.beacons-npa.gov.uk/the-authority/who-we-are/npmp/). A study of effects of burning on water chemistry in Northern Ireland has called into question effects of burning management on DOC concentrations. However, increases in nitrates, acidity and aluminium concentrations were observed (Evans et al., 2017).

Burning is also important for certain aspects of biodiversity, as addressed in the Building ecosystem resilience section of the Sustainable Farming Scheme evidence review. It is worth bearing in mind that burning is carried out with the goal of game management or grazing improvement. Any reduction in burn frequency may thus involve a reduction in profitability or agricultural productivity.

Fires can have negative impacts in terms of air quality – a recent upland fire near Manchester had devastating effects on air quality through production of particulate matter (Longlands and Hunter, 2018).

6.3.3 Magnitude

Clay et al. (2010) found that while investigated peatlands were C sources, burning management reduced the C source by in the region of 0.4 t C $ha^{-1}y^{-1}$. However, this contradicted results recorded several years earlier, and later, from the same site (Garnett et al., 2000; Marrs et al., 2019). Garnett et al. (2000) observed a 74% reduction in C accumulation rate with managed burning, resulting in a mean annual CO_2 flux under managed burning of -1.09 t CO_2 ha^{-1} yr^{-1} as opposed to -3.81 t CO_2 ha^{-1} yr^{-1} without burning (Evans et al., 2014).

6.3.4 Spatial issues

Most of the research and discussion on managed burning has been focused on areas such as the English Pennines which have been subject to over a century of grouse moor management, along with other human pressures such as air pollution, which have increased the cover of woody, fire-prone *Calluna* at the expense of moisture-retaining, C-accumulating *Sphagnum*. In natural peatlands, the rapid vertical growth of *Sphagnum* effectively limits the amount of woody biomass that is able to accumulate above the moss layer, making such systems intrinsically less fire-prone. In addition, many areas of blanket bog that have been managed for grouse have also been subject to drying, either intentionally (via drains) or unintentionally (as the result of gully erosion), which increases the risk that wildfires will burn down into the organic soil. These highly modified systems may therefore require a level of management intervention and protection that other, less impacted blanket bogs (such as those of Wales) do not. For example, the recent study by Marrs et al. (2019) uses results from the Moor House experiment as the basis for a recommendation that 'boreal moorlands' as a whole should be burn-managed to reduced wildfire risk. The validity of this extrapolation from a single site has been strongly challenged (Baird et al., in review). In all cases where water tables have been lowered through management, re-wetting will reduce the risk of fires burning into the soil.

6.4 Afforestation & Deforestation

6.4.1 Causality

AMBER: Afforestation and deforestation are very risky for SOC in the uplands. Following afforestation, increases in C stored above ground are likely - but this may not balance C losses from peat.

Afforestation on peatland, like agriculture and peat harvesting, is associated with drainage (Laine et al., 2009), which in turn can drive loss of SOC due to increased respiration. Interestingly, deforestation on peatland can also drive carbon loss in the short term; a meta-analysis suggests that deforestation on peatland is generally not associated with benefits to the C budget (although there were very few studies; Worrall et al., 2011a). This could result from trees no longer sequestering C, and tree remains being decomposed. Even if CO_2 emissions are reduced through deforestation, for example through a reduction in root respiration, CH_4 and N_2O emissions may increase (Worrall et al., 2011a). Deforestation was considered in isolation here, not in conjunction with subsequent peatland restoration.

One meta-analysis found benefits of afforestation for the C budget, but largely from studies in Nordic and Baltic countries which often use silvicultural methods based on the management of natural tree stands (Worrall et al., 2011a). In the Scottish uplands, however, both natural regeneration (Miles and Young, 1980) and experimental planting (Mitchell et al., 2007) of birch (*Betula* spp.) in heather moorland were shown to reduce the depth of the litter layer and soil carbon content.

6.4.2 Co-benefits and trade-offs

Afforestation is likely to increase C stored above ground, and could increase woodland biodiversity (possibly at a cost to biodiversity associated with key upland habitats). Effects for flood mitigation are mixed (see the Flood Mitigation review).

6.4.3 Magnitude

One study found that undisturbed peat in Scotland accumulates ~0.25 t C $ha^{-1}y^{-1}$, but 2-4 years after afforestation between 2 and 4 t C $ha^{-1}y^{-1}$ are emitted (Hargreaves et al., 2003). In the long-term the cost to the GHG budget could be huge.

6.4.4 Timescale

Planting forest on peatland is expected to immediately lead to carbon loss. However, after recolonization of ground flora, the afforested system could become a carbon sink (Hargreaves et al., 2003). However, the C budget is driven by above ground biomass and not SOC. Tree growth might only compensate for decomposition of underlying peat SOC for 90-190 years. In the long-term, there are also potential effects of forestry rotation on organo-mineral soils associated with disturbance due to felling and replanting (Vanguelova et al., 2018).

6.4.5 Spatial issues

Acidity is a key constraint on woodland planting in Wales, and suitability for different species will vary based on soils and climate. Effects are highly dependent on the conditions before tree-planting. Over the last century, substantial areas of peatland have been planted with fast-growing conifers (usually Sitka spruce *Picea sitchensis*); this practice has declined in association with the EU Habitats Directive and guidelines from the Forestry Commission against planting trees on “active” or restorable bogs (Worrall et al., 2011a).

6.4.6 Displacement

It is also important to consider the fate of the C in felled trees, as this may return to the atmosphere upon e.g. combustion. Alternatively it could be stored for long periods in e.g. furniture.

6.4.7 Social and economic barriers

Large commercial conifer plantations may be perceived negatively by the local farming community (Farming Connect, *pers. comm.*).

7 Evidence Gaps

Some key evidence gaps highlighted during this review are as follows:

i. What is the best policy or scheme structure to provide additionality of interventions for SOC? What actions are farmers likely to take in the absence of a Sustainable Farming Scheme? How can we avoid displacement effects on SOC, both within and between farms?

ii. While fertilisation methods may have positive effects on SOC in intensive grasslands, many studies hint that on organic-rich, upland soil the opposite is true. It is critical to determine if this is the case. Furthermore, how organic (i.e. SOC rich) does a soil need to be before the effects of improvement are strictly detrimental?

iii. Depth continues to be an issue when considering SOC stocks. The vast majority of studies are in the top 30cm of the soil. Ward et al. (2016) highlight the quantity of carbon in grasslands (~60%) that occurs below 30cm. This is even more critical for peatlands, which can be extremely deep.

iv. Under what conditions does controlled burning decrease SOC? Studies must operate to depth, over long timescales, and look at effects in a wide variety of contexts; sites with a history of managed burning may respond differently to sites without a history of managed burning. In what circumstances is wildfire the alternative to managed burning?

v. How does grazing affect SOC in Wales, both in upland and improved contexts? Studies should separate effects of grazing from associated management practices where possible. How has grazing affected heather and SOC in the uplands?

vi. What is the fate of soil C following erosion? Is this C stabilised in ocean sediments, or oxidised to CO_2 (Buckingham et al., 2013)?

vii. Reduced tillage is quite well studied on cropland, but less well studied on grassland, especially in the Welsh context. Further meta-analyses of reduced tillage and SOC are needed for croplands, while primary research is needed for grasslands.

viii. When it comes to biological N fixation, under what conditions can the desired abundance of legumes be maintained (Lüscher et al., 2014)? This is critical to avoid repeated ploughing and reseeding, and to ensure SOC storage.

ix. What other technologies exist that might assist with SOC sequestration? Whitmore et al. (2015) note that genetic variation in plant traits underpinning C sequestration is beginning to be characterised, highlighting a prospect of selection or genetic engineering for such traits in agricultural species. Furthermore, perennial crops tend to sequester more C than annual crops, and commercial perennial varieties of crops may emerge in coming decades (Royal Society, 2009).

8 Summary

Colour Key:

- Blue = well tested at multiple sites with outcomes consistent with accepted logic chain. No reasonable dis-benefits or practical limitations relating to successful implementation.
- Amber = agreement in the expert community there is an intervention logic chain which can be supported but either evidence is currently limited and/or there are some trade-offs or dis-benefits which WG need to consider.
- Pink = either expert judgement does not support logic chain and/or whilst logic chain would suggest it should work there is evidence of one or more of the following:
  - its practical potential is limited due to a range of issues (e.g. beyond reasonable expectation of advisory support which can be supplied and/or highly variable outcome beyond current understanding or ability to target),
  - the outcome/benefit is so small in magnitude with few co-benefits that it may not be worth the administration costs,
  - there are significant trade-offs.

Table 8.1 *Improved Grassland*

Confidence	Intervention name	Key Outcomes	Key Benefits	Critical concerns
Blue	Organic fertiliser	GHG sequestration, Reduced levels of financial risk, Improved productivity	Climate change mitigation, Effective management of reasonable risks, Improved competitiveness	Careful targeting imperative. Benefits greatest when dependence on manufactured fertiliser is reduced. Care must be taken to avoid nutrient excesses and nitrogen leaching.
	Prevent conversion to arable/reduce tillage	GHG sequestration, Reduced levels of biological and environmental risk	Climate change mitigation, Effective management of reasonable risks	Sound evidence base, but difficult to know if and where conversion is a risk in Wales. Tillage interventions on grassland are not well understood. Possible trade-offs with productivity
Amber	Liming	GHG sequestration, Reduced levels of financial risk, Improved productivity	Climate change mitigation, Effective management of reasonable risks, Improved competitiveness	Direct carbon emissions on application, but may decrease NPK use and associated emissions. There may be risks of liming on organic soils
	Grazing & cutting	GHG sequestration, Reduced levels of financial risk, Improved productivity	Climate change mitigation, Effective management of reasonable risks, Improved competitiveness	Management to reduce overgrazing can benefit SOC and productivity. However, grazing & cutting remove vegetative C and facilitate GHG production by livestock
	Sward management	GHG sequestration, Reduced levels of financial risk,	Climate change mitigation, Effective	Used in the right context, and to reduce manufactured fertiliser, biological N fixation and deep

		Improved productivity	management of reasonable risks, Improved competitiveness	rooted grasses might secure SOC and co-benefits. Biodiversity benefits may be limited without additional diversification
	Afforestation, hedgerows, agroforestry and habitat restoration	GHG sequestration, Improved provisioning of functioning habitats, Flood risk mitigation	Climate change mitigation, Resilient ecosystems, High water quality	Evidence for SOC mixed, but above ground C storage significant. Possible production trade-offs, but biodiversity benefits
Pink	Manufactured fertiliser application	Improved productivity	Improved competitiveness	GHG emissions displaced to point of N manufacture; nitrogen leaching and nitrous oxide emissions

Table 8.2 Cropland

Confidence	Intervention name	Key Outcomes	Key Benefits	Critical concerns
Blue	Cover cropping	GHG sequestration, Reduction of water pollutants, Reduced levels of biological and environmental risk	Climate change mitigation, High water quality, Effective management of reasonable risks	Likely to prevent SOC erosion. May increase SOC through residue returns. Risks of N_2O emissions, but less nutrient leaching and possible productivity gains
	Convert to grassland/include grass leys	GHG sequestration, Reduced levels of biological and environmental risk	Climate change mitigation, Effective management of reasonable risks	Clear evidence of SOC gains. Could benefit biodiversity and reduce nutrient leaching, but trade-offs with profitability. Displacement effects possible
Amber	Afforestation and agroforestry	GHG sequestration, Improved provisioning of functioning habitats	Climate change mitigation, Resilient ecosystems	Clearly increases SOC, above ground C and biodiversity but trades-off with production. Shows promise if afforesting unproductive land, or through agroforestry
	Organic inputs & biochar	GHG sequestration	Climate change mitigation	Might be impractical to move manure to arable land. Life cycle issues for other organic inputs
	Reduced and Zero Tillage	GHG sequestration	Climate change mitigation	SOC impacts are mixed, with increases at the surface and decreases at depth. Co-benefits are possible, linked to cover cropping, but risks of N_2O emissions and yield losses
Pink	Increased Manufactured fertiliser	Improved productivity	Improved competitiveness	GHG emissions displaced to point of N manufacture; nitrogen leaching and nitrous oxide emissions

Table 8.3 Upland habitats

Confidence	Intervention name	Key Outcomes	Key Benefits	Critical concerns
Blue	Prevent drainage and restore peatland	GHG sequestration, Reduction of water pollutants, Improved provisioning of functioning habitats	Climate change mitigation, High water quality, Resilient ecosystems	Peat stores huge amounts of carbon that needs protecting. Restoration efforts have had mixed results, and release CH4. Restored peat can reduce water treatment costs
Amber	Prevent improvement and reduce grazing intensity	GHG sequestration, Improved provisioning of functioning habitats	Climate change mitigation, Resilient ecosystems	Preventing improvement is likely to prevent SOC loss, but reduced grazing may not lead to SOC recovery. More research is needed on SOC declines and how to combat them
	Burning	Improved provisioning of functioning habitats, Greater ability to respond to market conditions through diversified income	Resilient ecosystems, New business income stream(s) from non-agricultural business sources	The evidence for effects of controlled burning on SOC is scarce and mixed. Largely points to decreased SOC, but the alternative could be wildfire. Water quality concerns
	Afforestation/deforestation	Greater ability to respond to market conditions through diversified income	New business income stream(s) from non-agricultural business sources	Afforestation and deforestation are very risky for SOC in the uplands. Following afforestation, increases in C stored above ground may not balance C losses from peat

9 References

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