Sustainable climate change mitigation in UK agriculture

A review of climate change mitigation measures in agriculture, and the impacts on biodiversity, climate change and resource protection

Report to the RSPB by Ellie Crane, March 2020

The report should be cited as follows:

Crane E, (2020) Sustainable climate change mitigation in UK agriculture. A review of evidence for the RSPB.

About the author

Ellie Crane is a freelance consultant working in the fields of UK and EU land use and environmental policy. She holds a degree in Natural Sciences from Cambridge University and a Masters in Applied Ecology and Conservation from the University of East Anglia. Her previous roles include Land Use Policy and Sustainability Advice Support Officer at the Joint Nature Conservation Committee; Land Management Specialist within the Common Agricultural Policy team at Natural England; and Land Use Policy officer at the RSPB. Since becoming an independent consultant, she has worked with Friends of the Earth and the RSPB on topics ranging from pesticide use to climate change mitigation in the land use sectors.

March 2020

Contents

Introduction

Climate change threatens health, livelihoods, food security, water supply and human security. Impacts on human and natural systems have already been observed. Future impacts depend on the rate, peak and duration of warming and include continued sea level rise, biodiversity losses and increases in regional hot extremes, heavy precipitation and the probability of drought. In a 2018 report, the Intergovernmental Panel on Climate Change recommended that to reduce the impacts, global warming should not exceed 1.5°C above pre-industrial levels. This would require global net anthropogenic carbon dioxide emissions to decline by about 45% from 2010 levels by 2030, reaching net zero around 2050 (IPCC, 2018).

All sectors must play their part in reducing net emissions of greenhouse gases. Agriculture is no exception: 11% of the UK's overall emissions come from the agriculture and land sectors (Table 1), and agriculture is likely to be one of the largest emitters by 2050. Emissions from the agricultural sector have not decreased over the past 5 years (CCC, 2018a). Compared to other sectors, farming has additional pressures to contend with and responsibilities to meet. Agriculture is arguably the activity likely to be affected most by climate change, so mitigation activities must go alongside actions to help agriculture adapt to climate change (CCC, 2018a). Accounting as it does for 75% of the UK's land area, agriculture will also play a critical role in helping biodiversity adapt to climate change, for example by buffering or providing connections between wildlife sites (Lawton, 2010). At the same time, agricultural land must continue to produce enough food, fuel and fibre to meet society's needs, while contributing to other objectives such as water management. Therefore, climate change mitigation in agriculture must be viewed as just one aspect of a shift towards more sustainable land management.

Table 1: main greenhouse gases associated with agriculture

GWP = Global Warming Potential, a relative scale that compares the gas in questions to the same mass of carbon dioxide (Defra, 2017a; National Atmospheric Emissions Inventory).[1]

Greenhouse gas	GWP over 100 years	Main sources relevant to UK agriculture
Carbon dioxide, CO_2	1	Fossil fuel use, land use change, soil management
Nitrous oxide, N_2O	310	Breakdown of nitrogen compounds in soils. Management practices that lead to increased availability of nitrogen (for example, applying manures and mineral fertilisers) increase N_2O emissions.
Methane, CH_4	21	Enteric fermentation by livestock, particularly ruminants such as cattle and sheep, and anaerobic decomposition of manure.

It is of fundamental importance to avoid a situation where measures to reduce emissions of one greenhouse gas lead to an increase in emissions of others (Monteny *et al*., 2006; Brink *et al*., 2005). Many past studies have ignored issues of land use change (for example, that associated with overseas soybean production for UK animal feed) and therefore underestimate the contribution of carbon dioxide to agricultural emissions (Garnett, 2007). In addition, when selecting and prioritising mitigation measures, the multiple objectives of land management must be taken into account in order to avoid perverse and harmful

[1] This report uses GWP_{100} to ensure consistency with national and international GHG accounting frameworks. It is not within the scope of this report to assess GWP* although we note its relevance in other discussions.

outcomes for the natural environment and to capitalise fully on the potential for synergies. At the very least, it is essential that climate change mitigation measures do not increase deleterious impacts on the natural environment, for example, by harming biodiversity or increasing the risk of pollution. If subsidies or incentives are to be linked to climate change mitigation objectives, these must be focused on measures which also move the industry in a more sustainable direction overall.

Land sparing and land sharing

Researchers have proposed two contrasting strategies to meet the need to deliver multiple objectives from land, in particular food production and biodiversity conservation. At their extremes, ***land sparing*** involves managing farmland at the highest possible sustainable yield. This maximises food production per unit area, thus potentially sparing large blocks of land for conservation. ***Land sharing*** involves promoting wildlife-friendly farming practices and on-farm habitat features, reducing the yield of farmland as far as possible while still delivering the required level of food production. Because food production per unit area is reduced, no land is available to be dedicated to conservation. In practice, a continuum of intermediate strategies exists.

In the current document we do not advocate land sparing over land sharing or vice versa. The report highlights that mitigation measures involving land use change are likely to be the most effective and deliver the most co-benefits. This is one half of a land sparing strategy. Measures that have the potential to maintain or increase yields while addressing causes of environmental harm (for example mixed farming, agroforestry, nitrogen-fixing crops, cover crops) could form the other half of this strategy. Measures that could be termed land sparing because they result in decreased yields (for example organic farming, reduced stocking density, changing to spring cultivars) may be the most appropriate in certain places, or necessary to achieve specific conservation objectives. A key challenge for farming is to increase yields within genuinely sustainable production systems (whereas the current focus tends to be on incrementally reducing the environmental harm of high-yielding systems). It is also very apparent that reducing overall demand for agricultural products (especially animal products), whether through tackling waste, changing diets or other methods, will be key to the success of any strategy (CCC, 2020).

Context

The Climate Change Act 2008 was amended in 2019 to legally bind the UK to net zero emissions by 2050. Thus far, the policy framework as it applies to agriculture involves an industry-led voluntary approach (CCC, 2018a). In January 2019 the president of the National Farmers' Union in England called for a more ambitious target of zero net emissions by 2040 (as reported in, for example, Farmers' Guardian, 8th January 2019).

The Committee on Climate Change has produced two comprehensive reports on land use and climate change (CCC, 2018a; CCC, 2020). In these, they note that progress in the agriculture sector under the current policy framework is insufficient to meet future carbon budgets. They call for more challenging measures, beyond the incremental changes seen so far. Specifically, in the most recent 2020 report they assert that the following key measures could achieve emissions reductions of 64% by 2050 while maintaining per capita food production:

- Increasing UK forestry cover from 13% to at least 17% by 2050 by planting around 30,000 hectares (90 – 120 million trees) of broadleaf and conifer woodland each year.
- Low-carbon farming practices – such as 'controlled-release' fertilisers, improving livestock health and slurry acidification.
- Restoring at least 50% of upland peat and 25% of lowland peat.
- Expand the planting of UK bioenergy crops to around 23,000 hectares each year.

To release the required area of land from its current agricultural uses, the Committee proposes improving sustainable agricultural productivity; reducing consumption and production of the most carbon-intensive foods; reducing food waste along the supply chain; and increasing forest productivity.

The current report

A large number of measures are identified in the scientific and grey literature that have the potential to reduce net emissions of greenhouse gases from agriculture. In the current report, we assess a selection of these measures for their likely potential impacts on biodiversity and resource protection. Table 2 provides a summary of the findings. In some cases, there could be considerable variation in impacts depending on details; where there is significant uncertainty, this is highlighted. We furthermore seek to identify where there is scope for synergy between different objectives, and where there is a risk of conflict. This assessment is built on an extensive review of the relevant literature, as well as consultations with a number of RSPB specialists within the field of agriculture, ecosystem services and biodiversity. It updates the 2011 RSPB report of the same name.

Much work has been done by other authors to estimate the total abatement potential and cost-effectiveness of mitigation measures proposed for the UK (see for example Moran *et al*., 2008; MacLeod *et al*., 2010; Eory, *et al*., 2015). The current document draws on these reports (among others) for technical details of mitigation measures, but makes no attempt to assess financial implications for farmers or the wider economy, or likely take-up of the measure.

Mitigation measures

Table 2: Summary of mitigation measures

M = mitigation potential score from - - (significantly increases net emissions) to ++ (very high mitigation potential). **RP** = resource protection score from - - (very harmful) to ++ (very beneficial). **B** = biodiversity impact score from - - (very harmful) to ++ (very beneficial)
0 means effects appear to be neutral. **?** means that there is insufficient evidence to assess the measure. **+/-** etc means that effects will vary depending on how the measure is applied.

Rank = overall assessment: **1** = good mitigation potential and potential benefits to biodiversity or resource protection; **2** = some mitigation potential and potential benefits/ no significant risks to biodiversity and resource protection. **3** = significant questions over mitigation potential and/ or effects on biodiversity or resource protection; **4** = likely to increase overall emissions and/or have a negative impact on biodiversity or resource protection.

Measure	M	RP	B	Rank
Land use change				
Peatland restoration and management	++	++	++	1
Forest and woodland creation	+	+/-	+/-	2
Convert non-native to native woodland	?	+/-	+	3
Conserve/ restore grassland and heathland	++	++	++	1
Conserve/ restore coastal habitats	++	++	++	1
Farming system				
Organic	+	+	++	2
Mixed farming	+	+	++	2
Integrated Pest Management	?	0/+	0/+	3
Agroforestry	++	+	0/+	2
Manure and nitrogen fertiliser management				
Make most efficient use of mineral fertilisers	+	+	+	2
Use controlled-release fertilisers	+	?	?	3
Incorporate applied manure and/or slurry into soil	+	+	+	2
Use composts and straw-based manures in preference to slurry	?	+	+	3
Anaerobic digestion and application of digested slurry	++	0/-	0/-	3
Pastoral farming				
Animal breeding and genetics	+	-	-	3
Match nutritional content of feed to animal's requirements	+	+/-	+/-	3
Replace roughage with concentrates	-	-	-	4
Dietary additives and vaccinations against methanogenesis	+	?	?	3
Fertilise pastures	-	--	--	4
Plant grazing land with a legume-grass mix	+/-	+	0/-	3
Reduce stocking density on intensively managed grassland	+	++	++	2
Extend the grazing season for cattle	+	-	-	3
Arable farming				
Change from winter to spring cultivars	+	+	+	2
Use improved crop varieties, nitrogen-efficient cultivars	+	+	?	2
Include nitrogen-fixing crops in rotation	+	0/+	0/+	2
Use cover crops	+	++	+/-	2
Manage water levels	-	--	--	4
Bioenergy				
Bioenergy from agricultural and forestry residues	+	0/-	?	3
Bioenergy from forestry	+/-	+/-	+/-	3
Bioenergy from dedicated crops	+/-	0/-	0/-	3

Land use change

Croplands tend to have lower carbon stocks than almost any other land-cover type, which means that generally, avoiding conversion into cropland, or converting croplands to another type of land cover will be beneficial from a greenhouse gas perspective (Freibauer *et al.*, 2004; Smith *et al.*, 2008). The reversion of cropland to another land cover has been identified as one of the most effective methods of reducing emissions (Smith et al., 2008). In many cases, conversion from cropland into another land-cover type also means reducing the need for external inputs such as fertiliser, further enhancing the mitigation effect and bringing potential benefits for resource protection and biodiversity.

However, converting croplands on any significant scale carries with it the risk of displacing the lost production elsewhere, which could result in no net benefits or even harm. This makes it all the more crucial to carefully select the scale and location of any land-use change.

From a biodiversity perspective, increasing the area of low input, permanent grassland, woodland or wetland will generally be beneficial, due to the character of these habitats themselves, and because of the potential for increased landscape heterogeneity (Benton *et al.*, 2003). However, design and location are important in order to achieve full biodiversity potential (Berry *et al.*, 2008), necessitating a case-by-case assessment prior to any intervention.

Peatland restoration and management

Mitigation potential	++
Resource protection	++
Biodiversity	++

Mitigation impacts

The organic soils (peat) found in peatlands store vast amounts of carbon and constitute so-called 'carbon hotspots' – top priorities in the context of land use and climate change mitigation (Commission of the European Communities, 2009). Scottish and English peatlands contain around 1600 million tonnes and 584 million tonnes of carbon, respectively (Natural England, 2010). A minimum estimate of the carbon content of peatlands across the whole UK is 3121 million tonnes (Lindsay, 2010), although any such figure must be interpreted with some caution due to the significant levels of uncertainty regarding the extent, depth and density of peat.

Degradation of peatlands leads to oxidation of carbon, which is released to the atmosphere as carbon dioxide, and the loss of dissolved and particulate organic carbon in water outflows. Many UK peatlands are currently net sources of greenhouse gases to the atmosphere. It has been estimated that English peatlands emit the equivalent of around three million tonnes of carbon dioxide every year. Only 1% of England's deep peat is considered to be undamaged (waterlogged and continuing to form peat and sequester carbon) (Natural England, 2010).

Drainage is the most widespread form of peatland degradation. Drainage lowers the water table and activates decomposition through oxidation of stored carbon (Smith *et al*., 2008). Cultivation of drained peatland leads to further net losses of greenhouse gases through mechanical soil disturbance such as deep ploughing. Overgrazing and burning may also have detrimental effects on the carbon stores in peatland, particularly where it results in increases in the area of bare peat. As well as causing carbon oxidation, a lowering of the water table can reduce emissions of methane while causing increased emissions of nitrous oxide, though the latter is probably only significant if previous land management has been associated with fertiliser application.

Given that most drained peat soils are net sources of greenhouse gases to the atmosphere, restoring them by halting and reversing processes that lead to degradation has the potential to cost-effectively reduce net emissions (Moran *et al*., 2008; Baird *et al*., 2009; Lindsay, 2010). This often consists of raising the water table (for example, through grip blocking in upland peatlands or re-wetting cultivated fen peat soils). Ceasing cultivation or adopting soil-conservation techniques can also be an important mitigation measure. Restored peatlands can be kept in agricultural use by allowing, for example, grazing, as long as the livestock type and stocking density are appropriate (Morris *et al.* 2010).

The effects on greenhouse gas fluxes of restoring peatlands are not straightforward: for example, emissions of methane tend to increase when the water table is raised (Bussel *et al*., 2010). Further research is needed to fully understand these effects, but current understanding is that restored peatlands have less of an impact on global warming than do unrestored peatlands (Baird *et al*., 2009). A study of rewetted industrial peatlands in Ireland found that bare peat was a carbon source while the vegetated plots were a carbon sink (Wilson *et al.* 2013). Natural England authors (2010) estimated emissions reductions following peatlands restoration in England of 2.40 Mt carbon dioxide equivalent per year.

Environmental impacts

Peatlands provide a number of very important ecosystem services (Bates *et al*., 2009; Defra, 2009; Frelih-Larsen et al., 2008; Weiske, 2005; Weiske, 2007). They have high soil quality and some research indicates they may be able to filter pollutants, with benefits for water quality (Berry *et al*., 2008). Furthermore, many UK peatlands are recognised as nationally or internationally important for biodiversity (Natural England, 2010). Bain *et al.* (2011) list a number of species that would be supported by peatland restoration, including rare species of Sphagnum mosses, invertebrates such as the bog hoverfly, and bird species such as golden plover, greenshank, red-throated diver, dunlin and common scoter.

[Return to contents](#)

Forest and woodland creation

Mitigation potential	+
Resource protection	+/-
Biodiversity	+/-

Mitigation impacts

Despite the substantial forest and woodland creation that has occurred since 1945, the UK has a low level of woodland cover compared with most European countries (Read *et al.*, 2009). The UK Government has declared an 'aspiration' of 12% tree cover by 2060 and promises to plant 11 million trees (plus 1 million in urban areas) over the next 25 years (HM Government, 2018a), but increased targets are expected in 2020. The Forestry Commission reports that woodlands planted since 1990, coupled to a woodland creation programme of 23 200 ha per year could, by the 2050s, be delivering annual emissions abatement equivalent to 10% of total greenhouse gas emissions at that time. Such a programme would represent a 4% change in land cover and would bring UK the forest area to 16% (Read *et al.*, 2009). A new Government tree planting strategy is anticipated in 2020 which is likely to increase these targets.

The mitigation potential of woodland creation depends on the initial land-cover type and the techniques employed. Converting, for example, permanent grasslands to woodlands using aggressive techniques such as deep ploughing or mounding could result in a large initial loss of soil carbon, a debt that may take some time to recover (Zanchi *et al*., 2010). The carbon sequestration of woodland reaches an equilibrium over time and is potentially reversible on harvesting (FCRN, 2010), although this can be partly mitigated by careful selection of harvesting techniques, such as continuous cover forestry without stump removal.

There is a need for more information about the effects of trees on soil carbon (FCRN, 2010). Research suggests that in some situations the rapid growth and repeated harvesting of biomass may degrade soils if managed improperly, potentially leading to a net loss of soil carbon (Berthrong *et al*., 2009).

Draining peatland for afforestation causes a large loss of carbon to the atmosphere (Alonso *et al.* 2012). However, forested drained peatland can vary from being a greenhouse gas source to a small sink because carbon uptake by the vegetation can balance soil emissions (Vanguelova *et al*., 2018; Hargreaves *et al.* 2003). Some evidence suggests natural bog forest is a more effective carbon dioxide sink than drained forest in the long term (Vanguelova *et al*., 2018).

Falloon *et al.* (2009) calculate that converting arable land to woodland results in a soil carbon sequestration rate of up to 0.62 tonnes carbon ha^{-1} yr^{-1} in addition to the carbon storage in above-ground biomass, although they report a high degree of uncertainty in this estimate.

In summary, the creation of woodlands has the potential to increase carbon stocks, but this needs to be done in the context of sustainable multiple benefit forestry. If productive agricultural land is afforested, the possible impacts of displacing food production elsewhere need to be taken into consideration (Powlson *et al.* 2011). For large, long-term forest carbon stocks, plant slow-growing deciduous trees and manage under a low-intensity system such as continuous cover forestry or with a long rotation time.

Environmental impacts

Where appropriately planned and managed, forest and woodland creation can reduce erosion and may attenuate flood risk and improve water quality (though this will depend on the timber extraction method if the wood is harvested). There is significant interest in increasing woodland cover in flood plains and upper catchments to ameliorate spate flows. On the other hand, planting trees in water sensitive areas may conflict with climate change adaptation objectives if stream flows are reduced. Careful assessment of potential significant site and cumulative environmental impacts is therefore required.

The UK Government and devolved administrations have commitments to carrying out forestry in a sustainable manner for multiple benefits. This requires the careful location and design of new woodlands, as well as woodland management, to prevent biodiversity loss and damage (Forestry Commission, 2017). It is important that the UK Government and devolved administrations' minimum requirements for sustainable forestry are a condition of all woodland planting, felling and management. It is also important that these requirements are rigorously compliance checked and enforced, and that the site-based and cumulative environmental impacts of such woodland creation are assessed.

New woodlands and forests may bring biodiversity benefits through increased landscape heterogeneity and the provision of habitats for many taxa (Berry *et al.,* 2008). The impacts on biodiversity will vary greatly, however, depending on the relative importance of the existing habitats and their management for open ground species and the type of afforestation project (species composition, management regime, scale etc). Certainly, there will be negative effects for a number of species if valuable open landscape habitats are replaced by young commercial forestry plantations (Freibauer *et al.*, 2004).

Return to contents

Convert non-native to native woodland

Mitigation potential	?
Resource protection	+/-
Biodiversity	+

Planted forests of non-native conifers make up around 36% of Britain's total wooded area (Barsoum *et al.* 2016). Increasing the area of native woodlands, including converting non-native conifer to native woodland where appropriate, is an aim of the UK Forestry Standard Guidelines on Biodiversity (Forestry Commission 2017).

Mitigation impacts

Non-native conifer plantations often have dense canopies, leading to low light levels and low temperatures at the forest floor. This, combined with the typically low pH in the upper soil layers under conifers, can lead to slower decomposition rates and an accumulation of organic matter, contributing towards carbon sequestration. Conversion usually opens up the canopy, cancelling these effects to some degree. It does, however, improve conditions for ground vegetation to grow, potentially adding to the forest's ability to absorb carbon (Barsoum *et al.* 2016).

Environmental impacts
As stated above, conversion will tend to open up the woodland canopy, increasing light levels and soil temperatures and leading to greater nutrient availability. This improves soil fertility but can pose a risk to water quality in nitrogen-saturated soils (Barsoum *et al.* 2016).

The UK has a legacy of relatively small, often isolated native and ancient woodland sites in poor ecological condition, with large areas of plantation forestry, much of which were established on areas of considerable importance for birds and other biodiversity. Improving the biodiversity of existing native woodlands is important as well as any scope for new native woodland planting to expand or buffer native woodland remnants.

Native woodland species abundance and diversity can increase substantially following conversion, although colonisation will take place at different rates for different species (and in some cases may never happen without intervention). Bird and bat species often benefit from the increased structural complexity following conversion. Soil fauna has also been observed to respond positively. However, closed canopy conifer plantations provide a safe haven for red squirrels in the UK, so conversion is unlikely to benefit them in areas where grey squirrels are also present (Barsoum *et al.* 2016).

[Return to contents]

Conserve/ restore grassland and heathland

Mitigation potential	++
Resource protection	++
Biodiversity	++

Mitigation impacts
Permanent grasslands, i.e. grasslands that are maintained perpetually or for a long period of time without ploughing or reseeding, have higher stocks of soil carbon than do croplands (Countryside Survey, 2009; Soussana *et al*., 2004). Grassland soils can store as much as 438 t C ha^{-1}, depending on soil type (Alonso *et al.,* 2012). Acid grassland has the highest carbon stock of any UK broad habitat (Jefferson *et al.*, 2011). The carbon stock on heathlands is estimated at around 88 t C ha^{-1} in the soil and 2 t C ha^{-1} in the vegetation (Alonso *et al.,* 2012).

Since the loss of carbon when permanent grasslands are converted into croplands or temporary grasslands is much faster than its gain upon restoration (Moran *et al*., 2008; Smith *et al*., 2000; Soussana *et al*., 2004), avoiding conversion in the first place is the most effective mitigation measure. Reverting improved grassland or arable land to semi-natural grassland can, however, increase carbon storage (Soussana *et al*., 2004; Ostle *et al*., 2009). Falloon et al. (2009) calculate that converting arable land to grassland results in a soil carbon sequestration rate of 1.2 – 1.69 tonnes C ha^{-1} yr^{-1}, although they report a high degree of uncertainty in this estimate. Alonso *et al.* (2012) report annual sequestration by restored grassland of 11.62 tCO_2-e ha^{-1}, and 3.32t CO_2-e ha^{-1} for restored heathland. Arable reversion can occur over an entire land area, or in localised spots such as buffer strips and field margins, field corners and hedgerows.

When croplands are converted into grassland, rates of carbon sequestration initially increase, and eventually reach a plateau as a new equilibrium is established (Soussana *et al*., 2007). This saturation effect means that net sequestration will not go on forever, but since these grasslands will hold vast stocks of carbon, they nevertheless contribute to mitigation by locking up this carbon in storage that is permanent (as long as the land use change is not reversed). The carbon benefits of grassland conservation and restoration can be quickly lost, however, if the grasslands are ploughed up. For heathland, the rate of carbon sequestration varies depending on the growth stage of the vegetation; whereas the bare ground stage may be a net source, the building and mature stages are net sinks and there is no significant sequestration in later stages (Alonso *et al.,* 2012).

Avoiding degradation of existing grasslands can also have significant mitigation benefits. Increased floristic diversity of grassland may help increase soil carbon sequestration rate (De Deyn *et al*., 2008; Fornara *et al.*, 2008; Steinbeiss et al., 2008). Increased species richness has also been shown to increase production, although the mechanism is unclear (Bullock *et al*., 2007; Hooper *et al*., 2005). De Deyn *et al.* (2010) theorise that the presence of key species such as red clover in the plant community can drive increased sequestration rates. Evidence suggests that in certain circumstances, grasslands with low agricultural inputs and high plant diversity have the same productivity as high input, low diversity grasslands. If this higher plant diversity increases production without a change in inputs, increased carbon sequestration rates can result (Weigelt *et al*., 2009; Tilman *et al.*, 2006; Klumpp *et al*., 2009). Furthermore, reducing the intensity of management (e.g. by reducing or stopping nitrogen fertilisation) decreases the associated greenhouse gas emissions (Alonso *et al.,* 2012).

Environmental impacts

The preservation or restoration of permanent, low-input grasslands has the potential to meet multiple objectives, including those for resource protection in the Nitrates Directive and the Water Framework Directive. Resource protection benefits include improved water quality (due to less nutrient leaching and runoff) (Jarvie *et al*., 2008, 2010; Phoenix *et al*., 2008), reduction of soil erosion through enhanced soil stability and quality, flood protection and landscape value (ADAS, 2009; Choudhury *et al*., 2004; Falloon *et al*., 2004). For many of these benefits to materialise, it is crucial that these grasslands are truly permanent, i.e. not subject to ploughing and re-sowing, are subject to low or no fertilizer or herbicide inputs, and that these conditions are met for periods of 15 years or more. There is evidence that more biodiverse grasslands provide greater benefits (Zavaleta *et al*. 2010). For example, more complex plant community composition (and to some extent species richness) reduces leaching of inorganic nitrogen from grasslands (Scherer-Lorenzen et al., 2003; Phoenix *et al*., 2008).

The preservation or restoration of permanent grassland is likely to be beneficial to biodiversity (Vickery *et al*., 2001) but the degree of benefit, especially for birds, will depend on the exact type of grassland established, whether the grassland is semi-natural, the species composition if sown, and the presence of other pockets of semi-natural grassland around the site in question. The net benefits will also depend on the previous management of the arable land. For example, the conversion of rotations with spring crops and certain agri-environmental scheme prescriptions to permanent grassland could potentially lead to a reduction in biodiversity (Berry *et al*., 2008).

Return to contents

Conserve/ restore coastal habitats

Mitigation potential	++
Resource protection	++
Biodiversity	++

The carbon stored and exchanged in coastal and marine ecosystems is known as 'blue carbon'. The Northwest European shelf seas absorb 15 – 40 million tonnes of atmospheric carbon per year. Of this, coastal habitats are estimated to take up 0.4 – 6.8 million tonnes per year, and the ocean 16 to 40 million tonnes per year (Kröger *et al.* 2018). Healthy saltmarsh can sequester almost 10 times as much carbon as forests of an equivalent area. Sand dune habitat is also potentially responsible for a significant amount of carbon storage as a result of soil organic matter accumulation (Miles, R. *et al.* 2018 and references therein).

Mitigation impacts
Protecting coastal habitats preserves uptake of carbon from the atmosphere and increases carbon stocks. On the other hand, the degradation or conversion to other land uses of coastal habitats causes the release of stored carbon (Kröger *et al.* 2018).

Between 1945 and 2010 the UK lost more than 15% of its saltmarsh habitat, 46% of its shingle and over 16,000ha of its sand dune habitat area. This has been primarily as a result of development and land reclamation for agriculture (Miles, R. *et al.* 2018). The strengthening of statutory protection in the 1980s slowed but did not halt the rate of loss (Jones *et al.* 2011). In addition, the remaining coastal habitat remains vulnerable to climate change, sea level rise and coastal squeeze (Miles, R. *et al.* 2018).

Coastal managed realignment involves removing or breaching sea walls (or installing tidal exchange structures within them) so that the land behind them floods. This allows the inter-tidal habitat to migrate inland, mitigating coastal squeeze. One study of two sites in the UK compared the ecosystem services delivered by restored inter-tidal habitat with those delivered by the land if it remained under agriculture behind a sea wall. At both sites, the net value of sequestered carbon in accreting sediments outweighed the income foregone from crops and grazing (MacDonald *et al.* 2017).

As well as the extent, the quality of remaining coastal habitats has declined. This restricts the habitats' capacity to adapt to climate change and sea level rise and may indirectly alter ecosystem service provision. Agriculture has contributed to declines in various ways, including through localised eutrophication caused by fertiliser run-off from arable land, and in some cases due to the loss of traditional low-intensity grazing (Jones *et al.* 2011).

A major global concern is the expansion of 'dead zones' in the coastal oceans: areas where oxygen levels are too low to support many of the species usually found there. Hypoxia can occur naturally, but fertiliser runoff from agricultural and nitrogen deposition from fossil fuels are driving an expansion in the duration, intensity, and extent of coastal hypoxia globally (Doney 2010). However, this does not appear to be a problem in UK waters: Kröger *et al.* (2018) found that although low oxygen levels occur near the seabed around the UK, they arise naturally and are not caused by excess nutrients. Oxygen levels in UK shelf seas are in general not currently low enough to threaten marine life. Continued monitoring is recommended, however, as climate change is predicted to alter conditions in these habitats.

In summary, conserving and restoring coastal habitats could have significant mitigation impacts. In some circumstances (e.g. managed realignment) a complete loss of agricultural production will be necessary; in others a change of agricultural system to protect coastal habitats from negative impacts such as fertiliser run-off, or to restore positive management such as low-intensity grazing.

Environmental impacts
The coastal habitats around the UK are of international conservation importance, and their biodiversity value is described in detail by Miles *et al.* (2018).

Coastal habitats are associated with many more ecosystem services, including flood and coastal defence; tourism and recreation; fisheries production and water quality improvement (Miles, R. *et al.* 2018). For the 50 managed realignment projects progressed in the UK since 1991, habitat creation is the major driver, followed by flood risk management benefits (Miles, R. *et al.* 2018).

[Return to contents]

Alternative farming systems

'Conventional' arable farming as practiced in the UK is highly productive per hectare of land, but also resource intensive in terms of inputs, particularly nitrogen fertilisers, and dependent on pesticides, which bring a range of environmental risks. The high productivity of enclosed farmland in the UK has generally been at the expense of other ecosystem services and biodiversity (McCracken *et al.,* 2011). The measures outlined in this section offer opportunities for carbon mitigation and improvements in other environmental impacts, while maintaining some or all of the agricultural yields.

Organic

Mitigation potential	+
Resource protection	+
Biodiversity	++

Council Regulation (EC) No 834/2007 sets out the principles of organic farming. These include designing the farming system around biological processes and ecological systems; restricting the use of external inputs and limiting the use of synthetic pesticides to exceptional cases.

Mitigation impacts

Research suggests that some, but not all, food produced from organic systems in England and Wales has a greenhouse gas benefit compared to conventional farming, when taking into account the true greenhouse gas cost of production and transport of inorganic fertiliser in conventional systems (Williams *et al*., 2006). In comparing the relative emissions impacts of different farming systems, it is important to note that as many differences exist in farming practices within the same system as exist between systems (CCC, 2010). Thus under any given farming system there is likely to be great variation in emissions.

The production and use of inputs such as fertilisers and pesticides are associated with significant greenhouse gas emissions. Less intensive cropping systems with reduced input requirements can potentially reduce greenhouse gas emissions. A review of European case studies found that median energy use was 21% lower in organic farming systems per product unit than on conventional farms, though with wide variation between case studies (Tuomisto *et al.* 2012).

There is evidence for higher soil carbon content in organically farmed soils than conventionally farmed (FAO, 2011; Leifeld *et al.* 2010). Tuomisto *et al.* (2012) found that, among the European case studies reviewed, the median soil organic matter (SOM) content was 7% higher in organic farms compared to conventional farms. Further analysis of the data indicated that although organic farming generally leads to higher SOM content, some conventional farming systems have the potential to achieve similar or even higher SOM levels when they include the application of manures.

It is important to note that an increase in soil organic carbon content does not necessarily constitute increased carbon sequestration from the atmosphere. For example, adding organic materials such as crop residues or animal manure to soil may increase soil organic carbon but generally does not constitute an additional transfer of carbon from the atmosphere to land, depending on the alternative fate of the residue (Powlson *et al.,* 2011; Leifeld *et al.* 2013). Nevertheless, there is some evidence for increased carbon sequestration by soils under organic agriculture (Gattinger *et al.* 2012; Falloon *et al.* 2009).

Emissions of methane and nitrous oxide, as well as carbon dioxide, must be considered when assessing the overall greenhouse gas benefits of these farming systems (Powlson *et al.* 2011). The European review (Tuomisto *et al.* 2012) found that, among the case studies examined, organic olive, beef and some arable crops had lower greenhouse gas emissions whereas organic milk, cereals and pork had higher greenhouse gas emissions compared to conventional products. The reasons for higher emissions in organic milk production were higher methane and nitrous oxide emissions and lower milk production per animal; while the reason for lower emissions in organic beef were reduced industrial inputs.

Finally, while emissions from organic systems may often be less than from conventional farms on a per- hectare basis, they may be greater on a per unit of product basis due to lower yields (CCC, 2010). Lower yields also mean that a larger area of land is needed to produce the same amount of food, compared to conventional farming (Tuomisto *et al.* 2012).

Environmental impacts

Reducing the amount of inputs is in general beneficial for both resource protection and biodiversity. In the review of European case studies carried out by Tuomisto *et al.* (2012), nitrogen leaching, nitrous oxide emissions, ammonia emissions and eutrophication potential were all lower on organic farms when measured per unit area, but higher when considered per unit of product because of the lower yields of the organic farms. The lower per-area figures for organic farms were generally due to lower nitrogen inputs. The organic systems included in the study had 55% lower total phosphorus inputs compared to conventional systems – an important consideration given that phosphorus is a finite resource (Rhodes, 2013).

Pesticide use can have adverse effects on biodiversity (see for example Beketov *et al.* 2013, Goulson *et al.* 2015, van der Sluijs *et al.* 2015). Pesticides can have both direct and indirect (via the food chain) negative effects on farmland bird populations and have been among the main causes of their decline over the last 50 years (Newton, 2004). Reducing use of pesticides should therefore have multiple benefits for birds and other taxa as well as for water quality, although the evidence for this is incomplete.

Overall, a number of studies and reviews suggest that organic farming generally supports a greater diversity of arable plants, various invertebrate groups, bats and birds. However, there can be variations in responses between individual species or taxa. For example, one meta-analysis found that on average, organic farming increases species richness by about 30%. This result has been robust over the last 30 years of published studies. Plants benefited the most from organic farming, but arthropods, birds and microbes also showed substantial positive effect (Tuck *et al*., 2014). A review of 76 studies demonstrated that species abundance and/or richness tend to be higher on organic farms than conventional farms, across a wide range of taxa. The species affected are often previously declining in range or abundance from past agricultural intensification, e.g. skylark, lapwing. The three management practices found to be most beneficial to wildlife were reduced use of chemical pesticides and inorganic fertilisers; sympathetic management of non-cropped habitats; and preservation of mixed farming. These are largely intrinsic to organic farms but can also be practiced on conventional farms (Hole *et al*., 2005). Another meta-analysis found organic farms to have on average 30% higher species richness and 50% higher organism abundance than conventional farms. However, it also showed wide variation, with 16% of studies actually finding a negative effect of organic farming on species richness (Bengtsson *et al.* 2005). The question as to whether conventional farming with specific targeted practices can result in higher biodiversity than organic farming is yet to be fully answered (Tuomisto *et al.* 2012; Hole *et al.* 2005).

[Return to contents]

Mixed farming

Mitigation potential	+
Resource protection	+
Biodiversity	++

Mixed farming is where both arable crops and livestock are raised in an integrated, interdependent system. This can be achieved on an individual farm or through co-operation between different, specialised farms in a geographical area (Moraine *et al.* 2014). Once the norm in the UK, mixed farming declined in the 1950s onward, with arable farming now predominating in the east and pastoral farming in the west (Robinson *et al.* 2002).

Mitigation impacts
Where feasible, re-integrating crop and livestock production may improve the efficiency of land and resource use, in particular by enhanced nutrient cycling and by improving soil health (Moraine *et al.* 2014 and references therein). This reduces the need for external fertiliser input, thus reducing surplus nitrogen in the system and possibly reducing direct and indirect emissions of nitrous oxide as well as ammonium losses (Berry *et al*., 2008; Weiske, 2005). Incorporating livestock and fodder crops into arable rotations can also help with pest and weed control (Eady *et al.* 2012). If this results in reduced inputs of pesticides (or a reduction in other pest control interventions such as mechanical weeding), carbon costs associated with these will decrease. A life-cycle analysis of a mixed farm in Western Australia demonstrated that taking into account the benefits of a mixed farming system (for example the dung and urine from livestock grazing on crop stubble reducing the need for nitrogen fertiliser) can significantly reduce the calculated carbon footprint of its products (Eady *et al.* 2012).

In some places mixed farming may not be feasible (for example on hill farms where arable cropping is not viable) or desirable (for example where there is significant carbon locked up in permanent pasture which would be lost if the land was cultivated). It is still possible to realise some of the benefits by making links between the two systems: for example, growing fodder crops to export to livestock farms and importing manure into the arable system. However, the carbon costs of transport must be taken into account (Moraine *et al.* 2014).

Environmental impacts
In addition to the benefits of reducing inorganic nitrogen and pesticide input to the system, integration as a form of reducing farm specialisation may also lead to increased landscape heterogeneity, with benefits for biodiversity (Benton *et al*., 2003; Newton, 2004). However, environmental and biodiversity benefits will depend on the type of mixed farming established: for example, outdoor pigs may be kept as a break in an arable rotation but managing their environmental impacts can be challenging (Edwards, 2007). Systems where livestock densities are high are generally not beneficial to biodiversity or resource protection.

[Return to contents](#)

Integrated Pest Management

Mitigation potential	?
Resource protection	0/+
Biodiversity	0/+

Integrated Pest Management (IPM) is a system of crop management that aims to achieve adequate control of pests, weeds and diseases while reducing the need for pesticide use, through the strategic use of a wide range of agronomic measures (Council Directive 2009/128/EC).

Mitigation impacts
There is no agreed strict definition of IPM, and in reality the term is applied to a broad spectrum of farming approaches. It is therefore difficult to quantify the mitigation potential of adopting IPM. If we assume that adopting IPM results in a reduction in pesticide use, there will clearly be a reduction in emissions directly associated with pesticides. Audsley *et al.* (2009) estimate that pesticide manufacturing represents about 3% of the 100-year Global Warming Potential from crops. However, other pest control measures that replace pesticide use may themselves have higher carbon emissions (see for example Paveley *et al.* 2008, Wilson *et al.* 2007). There is some limited evidence that IPM can help farmers to stabilise or increase yields (Lefebvre *et al.* 2015), particularly in the face of climate change (Lamichhane *et al.* 2015). If so, adopting IPM could help to decrease carbon emissions per unit of product.

Environmental impacts
Pesticide use can have adverse effects on biodiversity (see for example Beketov *et al.* 2013, Goulson *et al.* 2015, van der Sluijs *et al.* 2015). The definition of IPM adopted by the EU requires that when a pesticide is used (which should be as a last resort), the farmer must choose the option with the least side effects for non-target organisms and the environment and must apply the minimum amount necessary for effectiveness (Council Directive 2009/128/EC). Furthermore, strict adherence to the principles of IPM can result in farmers adopting many of the same measures as practiced in organic farming, such as extended crop rotations and promoting populations of natural enemies. Therefore, IPM can potentially deliver many of the same benefits for biodiversity and resource protection as organic

farming (see the 'organic' section of the current report). It is important to note, however, that IPM is much less strictly defined than organic farming so these benefits may not be realised in practice.

Return to contents

Agroforestry

Mitigation potential	++
Resource protection	+
Biodiversity	0/+

Agroforestry is the practice of deliberately integrating woody vegetation (trees or shrubs) with crop and/or animal production systems to benefit from the resulting ecological and economic interactions (Mosquera-Losada *et al*. 2009 as cited in Torralba *et al.* 2016).

There is a large body of research on the benefits of agroforestry. Much of this focuses on tropical systems but there is increasing interest in the potential for agroforestry to offset some of the negative impacts of intensive farming in temperate zones (Smith *et al.* 2013).

Mitigation impacts
Adding trees to an arable or pastoral system can increase carbon sequestration through both the carbon incorporated into the trees themselves and enhanced soil carbon content (see for example Mosquera-Losada *et al.* 2011; Lorenz *et al.* 2014). Because they are generally at lower density, trees in agroforestry systems produce up to three times the biomass of trees of the same age in forests (Aertsens *et al.* 2013). The trees in an agroforestry system may be more efficient at carbon dioxide utilisation from the atmosphere than annual crops and may have higher carbon returns to the soil through their litter. Incorporation of roots into the soil and the greater soil volume explored by tree roots compared to crop roots also enhance organic matter build-up in the soil (Mosquera-Losada *et al.* 2011).

By contrast with forest and woodland creation, agroforestry does not necessitate sacrificing food production. In the right circumstances and with the right design, agroforestry can provide similar levels of timber as forestry, and similar levels of food production as pasture (Torralba *et al.* 2016). Integrating trees into arable systems can increase soil fertility and enhance natural pest control (Torralba *et al.* 2016; Aertsens *et al.* 2013; Smith *et al.* 2013; Mosquera-Losada *et al.* 2011). Depending on the species used, the trees themselves can provide a food crop (see for example Chesshire, 2018). Alternatively, the trees can produce timber or fuelwood. If this results in long-term carbon storage in wood products, or displacement of fossil fuels, this further increases the mitigation potential of agroforestry (Plieninger 2011; Tsonkova 2012). Agroforestry systems are expected to be more resilient to climate change than monocultures of crops and pastures (Aertsens *et al.* 2013).

Environmental impacts
Well-designed agroforestry systems can provide or enhance a wide range of ecosystem services, including but not limited to water quality; erosion regulation; pest and disease

regulation and flood regulation (see for example Smith *et al.* 2013; Torralba *et al.* 2016). Introducing trees into arable systems can reduce the need for inorganic nitrogen fertiliser and can reduce nitrate run-off by an estimated 50% (Smith *et al.* 2013).
Clearly these benefits are context dependent, but it is clear that agroforestry has the potential at least to have a large positive impact (Torralba *et al.* 2016).

The potential benefits to biodiversity of allowing crops and trees to grow together depend very much on the system and species used: habitat is provided for wildlife only where the species planted and the stand structure achieved provide opportunities for biodiversity. New agroforestry plantings that utilise fast growing non-native species may have little biodiversity benefit. However, broad-scale analysis shows that in Europe, when compared to conventional land uses such as grassland and arable land, agroforestry supports higher levels of biodiversity (Torralba *et al.* 2016). Research in Ireland has shown that planting wide-spaced trees into both upland and lowland pastures can increase some invertebrates, birds and flora, in addition to reducing nutrient leakage (McAdam *et al.* 2009).

Return to contents

Manure and nitrogen fertiliser management

The basic concept behind improved nitrogen management is to maximise the efficiency of nitrogen use by ensuring that crop nutrient requirements are always met while minimising losses of nitrogen from the system.

Fertilisers can contain nitrogen as nitrate or ammonium. As organic matter in soil decomposes, nitrification takes place: first ammonium, then nitrogen dioxide and finally nitrate ions are formed. Nitrate is highly water soluble and mobile and is the primary form of nitrogen leached to groundwater. The opposite process, denitrification, involves nitrate being reduced to nitrous oxide and finally to nitrogen. This process is the main source of the greenhouse gas nitrous oxide (Follett, 1995). Different soil management practices will result in different balances of these processes and therefore affect the forms in which nitrogen exists in and is lost from the soil, with implications both for greenhouse gas emissions and water quality.

Reducing the total amount of excess nitrogen in soils leads to lower emissions of nitrous oxide and ammonia (Snyder *et al.*, 2009). It also reduces leaching of nitrate, limiting indirect nitrous oxide emissions, eutrophication and groundwater pollution, and it may also reduce soil acidification (Crews *et al.*, 2004; Kirchman et al., 2002). Reducing the use of mineral fertilisers also reduces emissions associated with their manufacture. Measures aimed at avoiding nitrogen excess, as opposed to an across-the- board reduction in fertiliser use, need not lead to reductions in yield.

Make most efficient use of mineral fertilisers

Mitigation potential	+
Resource protection	+
Biodiversity	+

Mitigation impacts

A reduction in mineral fertiliser use reduces associated nitrous oxide emissions from soil and carbon dioxide emissions from energy-intensive fertiliser manufacture (Schulte *et al.*, 2012).

Using organic fertilisers (manures and slurries) as far as possible, and fully accounting for them in nutrient budgets, has the potential to reduce the need for mineral fertilisers. Organic fertilisers have additional benefits compared to mineral fertilisers, including increasing soil organic carbon (Vanden Nest *et al.* 2014) and improving soil structure. Clearly it is essential that farmers actually do take account of the extra nitrogen delivered by manures and slurries when determining fertiliser rates. The British Survey of Fertiliser Practice has shown that manure nitrogen is not always fully allowed for in nutrient budgets in the UK (IGER & ADAS, 2007).

When mineral fertilisers are used, best practice should be followed to maximise the efficiency of nitrogen use, thereby minimising both the amount of fertiliser needed and nitrogen losses from the system.

It is important to match the type of fertiliser to seasonal conditions. Emissions of nitrous oxide following fertilisation depend on the type of fertiliser (nitrate- or ammonium-based) (Monteny *et al*., 2006) and weather conditions (mainly humidity). A flexible nitrogen fertiliser strategy, where ammonium-based fertilisers are applied under wet conditions, and nitrate-based fertilisers under dry conditions, could therefore reduce greenhouse emissions (Bates *et al*., 2009). In practice however, this may be unlikely to happen as farmers will have brought their fertiliser in advance.

Fertilisers should be applied based on precise estimation of crop needs and placed precisely into the soil so as to maximise accessibility to crop roots (Bellarby *et al.,* 2008; Newell Price *et al.,* 2011). For mineral nitrogen fertilisers, the main technique consists of placing the fertiliser in a concentrated band in the soil at seeding. Proper maintenance, setting and calibration of spreading machinery helps to ensure that fertilisers are accurately and uniformly applied (IGER & ADAS, 2007; Bates *et al*., 2009). Both under- and over-application of fertiliser can result in reduced yields, and over-application can lead to increased nitrate leaching (Newell Price *et al.,* 2011).

The timing of application is also important: fertilisers should be applied when least susceptible to loss, often just prior to plant uptake (Bellarby *et al*., 2008; Defra, 2010). This exact timing will depend on the soil and crop type, but it generally means not applying during autumn and winter. It is also important to avoid applying fertilisers when weather and soil conditions increase the risk of runoff (Newell Price *et al.* 2011). Improved timing of manure application may involve longer storage, with attendant risks of emissions of ammonia, nitrous oxide and methane, which may partly offset gains.

There are a number of recognised 'fertiliser recommendation systems' available to farmers. These take account of the following factors: soil nutrient supply (based on soil analysis), winter rainfall, previous cropping and soil type, crop nutrient requirements for a given soil and climate, crop requirement for nutrients at various growth stages, the amount of nutrients supplied to the crop by added organic manures and by previous manure applications, soil pH and the need for lime (Newell Price *et al.* 2011).

Precision farming is an integrated agricultural management system that incorporates many of the measures described above. It is based on the principle of matching nutrient supply and demand and makes use of high technology in the form of e.g. global positioning system (GPS), geographical information system (GIS), and remote sensing. These technologies allow for the management of areas of land or crop within a field with different levels of input (including pesticides as well as fertilisers), depending upon the yield potential of the crop in that particular area of land. Precision farming reduces greenhouse gas emissions per unit product both by reducing fertiliser applications and by increasing yield. It has been estimated that nitrogen use in arable systems could be reduced by 20% with no effect on yield (Eory *et al.* 2015).

Environmental impacts
Excess nutrient load - the amount of reactive nitrogen in the environment, particularly from fertiliser use - has more than doubled compared to pre-industrial times and is considered

one of the five principle pressures on biodiversity, with adverse impacts on terrestrial, inland water and coastal ecosystems (CBD, 2010). There is evidence that anthropogenic inputs of nutrients have impacted on biodiversity in the UK (MacDonald, 2006). The impact of nitrogen on European terrestrial biodiversity has been thoroughly reviewed in the European Nitrogen Assessment (2011). A key finding was that biodiversity has likely been in decline in Europe for many decades due to elevated nitrogen deposition. Therefore, insofar as the above measures reduce nitrogen use without impacting on agricultural yields (or even positively benefiting them), they can be expected to be entirely positive for resource protection and biodiversity.

Return to contents

Use controlled-release fertilisers or nitrification inhibitors

Mitigation potential	+
Resource protection	?
Biodiversity	?

Mitigation impacts
Controlled-release fertilisers are intended to match nutrient release with crop demand, reducing the pool of nitrogen in soil available for breakdown by bacteria. This aims to reduce nitrous oxide emissions. The controlled release is achieved by coating the fertiliser particles with a material that breaks down over the course of 2-6 months. Research shows some uncertainty over the effectiveness of the measure, with some individual studies showing zero effect, but a meta-review estimated that the mean emissions reduction is 35% compared to non-coated fertiliser (Eory *et al.,* 2015).

Nitrification inhibitors aim to achieve the same effect by inhibiting the activity of the bacteria that oxidise ammonium ions to nitrate. This is intended to provide better synchrony between nitrate supply and crop uptake, reducing direct nitrous oxide emissions. The measure may also reduce nitrate leaching and increase grass or crop yields but there is no experimental data on this as yet. Field trials have proved nitrification inhibitors to produce significant (average 48%) reductions in direct nitrous oxide emissions. They are not currently used commercially in the UK, and it is worth noting that they have been withdrawn in New Zealand due to concerns over contamination of milk (Eory *et al.,* 2015).

Environmental impacts
Using controlled release fertilisers does not guarantee an improvement for groundwaters, but if excess amounts of nitrogen in the system can be reduced, this would be beneficial from a resource protection perspective, with knock-on benefits for aquatic biodiversity.

Nitrate leaching loss reductions of up to 35% have been measured with nitrification inhibitors. However, ammonia emissions to air and ammonium/nitrite losses to water may be increased by a small amount (Newell Price *et al.,* 2011).

Return to contents

Incorporate applied manure/ slurry into soil

Mitigation potential	+
Resource protection	+
Biodiversity	+

Mitigation impacts
Reducing the exposure of organic fertilisers to the air can significantly reduce emissions of ammonia. The magnitude of the reduction depends on the time frame between application and incorporation and is greatest when incorporation takes place immediately after (or at the same time as) application (Weiske, 2005; Newell Price *et al.,* 2011).

Slurry may be injected into the soil or placed onto the soil surface with a trailing hose or trailing shoe (Weiske et al., 2006). Deep injection would typically achieve >90% reduction and shallow injection around a 70% reduction in ammonia emissions compared with surface broadcast application. Shallow injection is most suited to grassland and on arable land prior to crop establishment. Deep injection is most suited to arable land prior to crop establishment. Both result in a small increase in carbon dioxide emissions compared to broadcast applications (Newell Price *et al.,* 2011).

Manure can be incorporated into the soil using a plough, discs or tines. Ammonia emissions would be reduced by around 60% where soil incorporation by ploughing occurred six hours after slurry application, and around 40% where manure was incorporated by ploughing after 24 hours (Newell Price *et al.,* 2011).

To increase infiltration rate into the soil, manure or slurry can also be diluted with water before application, or water can be applied after fertiliser application.

Emissions of nitrous oxide can increase after slurry incorporation, partly offsetting greenhouse gas savings (Weiske, 2005; Newell Price *et al.,* 2011). However, reducing ammonia emissions increases nutrient use efficiency overall and therefore reduces the amount of additional fertiliser needed (see 'use best practice in fertiliser management' in the current document).

Environmental impacts
Slurry injection requires large machinery and there is a risk of damage to grassland sward if conditions are hot and dry (Weiske, 2005; Newell Price *et al.,* 2011). Umbilical application systems which pump slurry directly through a pipe system to the tractor avoid the need for heavy tankers. Improved tyre specification on tankers can also reduce compaction. Assessing ground condition is essential for all such field operations to prevent compaction.

Shallow injection (particularly of dilute slurries) on sloping land can result in runoff along the injection slots (Newell Price *et al.,* 2011). With manure incorporation, if the procedure damages soil structure, this may compromise crop yields and result in applied fertiliser and organic manure N being poorly utilised by crops, and increase the risks of nitrate leaching over the next winter drainage period (Newell Price *et al.,* 2011; Cuttle *et al.* 2007). Incorporating manure and slurry increases the potential for nitrate, ammonium and nitrite leaching losses and nitrous oxide emissions.

However, overall nitrogen use efficiency is increased, so synthetic fertiliser inputs can be reduced. Incorporating manure also reduces losses in phosphorus surface runoff and faecal indicator organisms (Newell Price *et al.,* 2011), reducing the risk of pollution of waterways (Weiske, 2005). It is worth noting, however, that faecal indicator organisms are protected from UV radiation after incorporation and can survive for longer than on the surface (Cuttle *et al*., 2007).

Reduced risk of waterway pollution will benefit aquatic biodiversity.

Return to contents

Use composts and straw-based manures in preference to slurry

Mitigation potential	?
Resource protection	+
Biodiversity	+

This measure entails changing from a system where the manure from housed animals is collected as a liquid (i.e. slurry) to one where animals are kept on bedding (e.g. straw) to produce solid manure (Newell Price *et al.*, 2011).

Mitigation impacts
Compared to slurry systems, manure-based systems are expected to generate lower emissions of methane and ammonia but probably higher emissions of nitrous oxide. Carbon dioxide emissions may also be increased by additional manure handling activities (Newell Price *et al.,* 2011; ADAS, 2009)

Environmental impacts
Solid manures are more easily stored than slurries and present less risk of pollutant loss during and following land spreading. Nitrate, ammonium and nitrite leaching losses could be reduced by up to 50%, while faecal indicator organism and phosphorus runoff would also be reduced (Newell Price *et al.*, 2011). Reduced risk of water course pollution benefits aquatic biodiversity.

Return to contents

Anaerobic digestion and application of digested slurry

Mitigation potential	++
Resource protection	0/+
Biodiversity	0/-

Mitigation impacts
Anaerobic digestion (AD) of organic material can provide a substitute for both fossil fuels and manufactured fertiliser and thereby has the potential to reduce total greenhouse gas

emissions. There are three main types of feedstock: energy crops (see 'purpose grown energy crops' in the current document), animal manures and other organic wastes (Amon *et al.*, 2007). Within the context of the EU goal to supply 20% of European energy demands from renewable sources by 2020, it has been estimated that over 25% of bioenergy demands could be met from biogas sourced from animal manure, crop silages, feed residues etc. (Holm-Nielsen *et al.*, 2009).

Stored manures emit methane. When manure is digested in an AD facility, the emitted methane (biogas) can be collected and burnt to produce electricity and heat. This saves greenhouse emissions in two ways: by capturing methane emissions from manure storage and by substituting fossil fuels with the energy produced (Weiske *et al*., 2006). It has been concluded that 'biogas from manure stands out as having very high reduction in greenhouse gas emissions and very high fossil fuel savings compared to the conventional storage and soil application of the manure.' (Thyø *et al.*, 2007).

Fermented slurry is a by-product of AD which can be used for fertilisation, reducing the need for fossil-fuel based fertilisers. Greenhouse emissions after field application appear similar to those from untreated slurry (Wulf *et al*., 2002). However, digested slurry has a higher pH, increasing the risk of emissions of ammonia from this form of fertiliser, with consequent potential for indirect emissions of nitrous oxide (Weiske, 2005). Application methods that minimise the surface area exposed to air and give contact with the ground (see 'Incorporate applied manure/ slurry into soil' in the current document) should therefore be used to minimise ammonia loss.

A study of pig farms concluded that AD of slurry in a 1:1 ratio with grass silage would generate greenhouse gas savings by displacing fossil fuels in electricity production (Schulte *et al.* 2012). A case study of Italian dairy farming resulted in an estimated 36.5% reduction in greenhouse gas emissions if manure is processed in an AD plant for biogas, with digestate stored in a gas tight tank (Battini *et al.* 2014). A Spanish dairy farm case study found a reduction of 978-1776kg CO_2 eq yr^{-1} per livestock unit, with the main savings coming from avoided methane emissions from slurry. From dairy, sow and pig farms studied in Finland, the greenhouse emissions savings from anaerobic digestion of manure were due to replaced fossil fuel consumption followed by reduced fertiliser use and production, and from manure management (Kaparaju *et al.*, 2011).

AD plants can be either on- farm or centralised (Defra, 2005). On-farm plants will typically require co-digestion of manures with high gas-yielding feedstocks such as maize in order to be economic, which reduces the overall greenhouse gas savings and introduces environmental problems associated with growing the crop.

For AD approaches to offer genuine mitigation, they must not drive unsustainable production systems – for example, high input livestock systems –associated with wider negative environmental impacts.

Environmental impacts

Fermented slurry from AD typically has a high readily available nitrogen content, therefore carrying a risk of nitrate leaching (though nitrogen content will vary depending on the feed stocks and the type of digested product). One set of field trials showed no significant increase in nitrogen leaching from digestate compared to that from raw slurry following spring application, but autumn application of digestate almost doubled the amount of nitrogen leached into drainage waters compared with raw slurry (Lukehurst *et al*., 2010). It is essential to accurately account for the composition of the digestate, as well as using the appropriate timing and method to ensure optimum application to crops (see 'use best practice in fertiliser management' in the current document).

Digested slurry has a higher pH than raw slurry, increasing the potential for ammonia emissions (Weiske, 2005).

Biogas plants have additional environmental impacts, for example particulate matter emissions and photochemical ozone formation potential. These impacts can be tackled by improving biogas combustion technologies (Battini *et al.* 2014)

Return to contents

Measures specific to pastoral farming

Direct emissions from livestock farming come from the respiratory process of all animals in the form of carbon dioxide. Ruminants, and to a minor extent also monogastrics, emit methane as part of their digestive process. Animal manure also emits gases such as methane, nitrous oxides, ammonia and carbon dioxide. Livestock also affect the carbon balance of land used for pasture or feed crops (FAO, 2006).

When evaluating greenhouse gas emissions from livestock, it is important to take a holistic perspective. Although this is true of all agricultural systems, the complexity of livestock production and the reliance of some sectors on bought-in feed makes capturing the whole footprint particularly important when considering potential mitigation measures relating to livestock management. An individual measure may appear to contribute to mitigation if considered in isolation but could have negative indirect effects as part of the wider system. As Garnett (2009) has argued, emissions from livestock management must be considered within a broad conceptual framework that incorporates three key perspectives: a) indirect second order effects of livestock production on land use change and associated greenhouse gas emissions; b) a comparison of the opportunity cost of using land and resources to rear animals with their use for other food or non- food purposes; and c) how far people need livestock products at all.

Livestock systems can be broadly classified along a scale from intensive to extensive, where intensive systems are associated with relatively high inputs and outputs (and will typically have higher stocking densities and be more reliant on concentrates), while extensive systems will have lower inputs and outputs.

Intensive systems are often reported to have lower emissions per unit product (although this will depend heavily on where the boundaries of the Life Cycle Assessment are drawn). However, intensive systems have a variety of direct and indirect costs. The former include local soil and water pollution (Naylor *et al*., 2005; FAO, 2006) and the latter include land-use change associated with the production of protein feeds (Garnett, 2009).

Conversely, well-managed extensive systems contribute to a host of other objectives, including climate change adaptation, soil quality and structure, biodiversity protection and enhancement, as well as the preservation of cultural landscapes (see for example Freibauer *et al*., 2004). Within the UK, High Nature Value Farming is associated with extensive livestock systems based on semi-natural grazing and low intensity grassland systems.

Many of the mitigation measures proposed for the livestock industry are based on increasing productivity to minimise land requirements and greenhouse gas emissions per unit of product (see for example CCC, 2018a). However, a drawback of this approach is that it is based on the assumption that anticipated trends in livestock demand are inevitable and cannot be challenged (Garnett, 2009). If this assumption is relaxed it becomes possible to look at how extensive livestock production could meet the (reduced) demand for livestock products while delivering other objectives.

Some other considerations for livestock farming include animal welfare, antibiotic resistance and disease control. Although important, these factors are outside the scope of the current document.

Return to contents

Animal breeding and genetics

Mitigation potential	+
Resource protection	-
Biodiversity	-

Mitigation impacts
Improving the efficiency of production through livestock breeding is considered by some to have the potential for significant cost-effective mitigation impacts, if wide uptake of breeding tools could be achieved (Eory *et al.* 2015).

Economic Breeding Index (EBI) is a profit index aimed at helping farmers identify the most profitable animals for breeding dairy herd replacements (Schulte *et al.* 2012). One study showed that increasing EBI reduced greenhouse gas emissions per unit of product (O'Brien *et al.* 2011, as reported in Schulte *et al.* 2012). This was because higher EBI cows had better fertility, which reduced emissions from non-milk producing animals. EBI is predicted to improve the efficiency of production in a number of additional ways, for example reducing mortality (Schulte *et al.* 2012).

One Irish study examined the effect of improving beef cattle indices on greenhouse gas emissions. It also found that reductions in emissions per unit product can be achieved, driven by improvements in cow survival, reduced mature cow maintenance feed requirements, shorter calving interval, reduced offspring mortality, and by increased meat production from improvements in carcass weight, conformation and fat (Quinton *et al.* 2018).

Similarly, models using data from Irish farms indicate that increasing the lifetime average daily weight gain of beef cattle can decrease greenhouse gas emissions per unit of product. This is because the greater quantities of beef produced more than offset the increase in absolute emissions that results from the greater amounts of feed given and manure produced (Foley *et al.* 2011, as reported in Lanigan *et al.* 2018).

It is important to recognise that livestock breeding programmes have focused almost exclusively on increasing production and have gone hand in hand with a move to more specialised, higher input systems. For example, the metabolic requirements of high yielding dairy cows cannot be met by a forage-based diet and as genetic potential has increased so has reliance on dietary supplementation. High genetic merit animals are generally unsuited to extensive systems and genetic selection approaches are more suited to intensive sectors. The carbon abatement achieved by these measures is per unit product so, unless some policy measure is applied to hold total production constant, there would not be an absolute reduction in greenhouse gas emissions.

Environmental impacts

The environmental effects will depend on the focus of genetic improvement programmes. If these are based mainly on production parameters, they are likely to be associated with further intensification which increases the risk of negative environmental impacts. A focus on genetic improvement could further marginalise those extensive livestock producers who traditionally use less commercial, hardy breeds for grazing on low input vegetation. For example, a number of studies have highlighted changing farming practices in upland areas with a tendency to take stock off the hill and focus on more intensive production with commercial breeds in the 'in – bye' area (see for example Silcock *et al*. 2012). This can potentially lead to problems accessing suitable grazing livestock to manage semi-natural habitats as well as increasing pressure on the 'in-bye' environment.

[Return to contents]

Match nutritional content of feed to the animal's requirements

Mitigation potential	+
Resource protection	+/-
Biodiversity	+/-

Mitigation impacts

Nitrogen use efficiency varies between species and systems, being higher in pig and poultry units than for ruminants (Rotz, 2004). Livestock (especially dairy cows) are often fed diets with higher than recommended amounts of protein to minimise the risk of low production (AHDB Dairy, 2010). Large amounts of nitrogen are then excreted in urine and manure, leading to volatile emissions of nitrous oxide and ammonia and losses through nitrate leaching (Rotz, 2004). An optimised diet can improve the efficiency with which livestock use nutrients and therefore reduce nitrogen losses and consequent greenhouse gas emissions.

Methods include avoiding excess protein in the diet, matching the amino acid levels in rations to requirements and optimising the balance of proteins to carbohydrate. The appropriate measures depend on the system. For example, supplementation with synthetic amino acids is used in pig and poultry systems so dietary crude protein can be reduced, while characterising the quality of conserved forages accurately and reducing the proportion of rumen degradable protein in diets may be particularly relevant in dairy systems. It is generally harder to improve nitrogen use efficiency for forage-based diets.

Eory *et al*. (2015) consider that there is significant mitigation potential in improving the digestibility of beef and sheep feed, thus improving yield and reducing enteric methane emissions. This involves getting advice from an animal nutritionist to improve the composition of the diet, complemented with forage analysis and improved grazing management.

Environmental impacts

Optimising nitrogen content in feed to avoid excess excretion will be beneficial for resource protection and biodiversity in so far as the potential for nitrogen loss will be reduced. However, there is considerable potential for this measure to have indirect negative impacts due to associated changes in management. For example, use of maize silage and high sugar ryegrass are recommended for improving feed nitrogen utilisation in the dairy industry (AHDB Dairy, 2010), but these crops are likely to have negative impacts on biodiversity and resource protection.

[Return to contents](#)

Replace roughage with concentrates

Mitigation potential	-
Resource protection	-
Biodiversity	-

Mitigation impacts
In general, the fraction of feed converted to methane decreases as feed quality (that is, available nutrient content) increases (Garnett, 2009). Therefore, diets higher in starchy lower-fibre concentrates, including cereals and oilseeds, generally lead to higher outputs of milk or meat relative to methane emissions (Weiske, 2005). An expert workshop commissioned to review and update the UK Agriculture Marginal Abatement Cost Curve for the 5th carbon budget period worked from the assumption that each 1% additional starch in the diet of ruminants reduces enteric methane emission by 0.78%, while yield increases by 5% (Eory *et al.* 2015). The same workshop noted that improving forage quality can have the same effect as increasing concentrates.

Increased use of maize silage is frequently listed as a mitigation measure for ruminant systems. However, production of concentrates is energy-intensive, leading to emissions of greenhouse gases. Furthermore, growing the feedstock for concentrates requires land, which may cause significant second order impacts relating to land-use change through displacement of crops for human consumption or the ploughing of land for feed crops (Garnett, 2009; Eory *et al.* 2015).

Increasing the fat content of ruminant diets, for example by adding rapeseed or linseed as a partial replacement for concentrates, reduces enteric methane emissions. Land use change effects are assumed to be negligible if using oil seeds grown in the UK replacing forages and concentrates mostly comprised of UK-grown cereal products (Eory *et al.* 2015). This measure may therefore be preferable to increasing use of concentrates.

Environmental impacts
This measure comes with many external costs (EC DG Enterprise and Industry, 2006). The direct and indirect land use change involved in growing the feedstock may have significant negative impacts on biodiversity and resource protection, depending on the land use that is displaced. Maize is a particularly challenging crop from an environmental perspective (Natural England, 2007). Garnett (2009) writes that: “In short, mitigation approaches that

advocate feeding greater levels of concentrates may be damaging when viewed from a broader perspective".

Return to contents

Specific agents, dietary additives and vaccinations against methanogenesis

Mitigation potential	+
Resource protection	?
Biodiversity	?

Mitigation impacts
Agents and additives include propionate precursors, probiotics, nitrates, antibiotics (e.g. ionophores) and bovine somatotropin (bST, a growth hormone). These agents and additives tend to improve productivity and therefore decrease emissions of methane and other greenhouse gases measured per unit product. Some of them, such as propionate precursors and probiotics, also reduce the production of methane by enteric fermentation, thus giving a double effect. Vaccination, to invoke an immune response to rumen protozoa, is another possible measure designed to reduce methane emissions and an active area of research (Wedlock *et al.* 2013). Considerable uncertainty surrounds all these measures in terms of their net effects on emissions.

Probiotics are not commonly used in the UK but industry experts predict their use might increase in the next decade. The practicalities of administering them mean they are only feasible on farms where the animals receive daily concentrates in their feed, which would exclude some extensive farms. At the other end of the spectrum, the methane abatement potential decreases as amount of forage in the diet decreases. Although there is little practical evidence yet, a recent meta-analysis concluded pro-and prebiotics reduce enteric methane emissions by 3% on average across ruminant livestock, as well as increasing milk yield (Eory *et al.* 2015 and references therein).

Environmental impacts
The effects of these substances on animal health and the environment are currently unclear (Garnett, 2009). Several, including ionophores and bST, are banned in the EU. The RSPB advocates the systematic assessment of the direct and indirect impacts of any new technology on all environmental and social objectives at all scales, before it is adopted.

Return to contents

Fertilise pastures

Mitigation potential	-
Resource protection	- -
Biodiversity	- -

Mitigation impacts

Applying either inorganic or organic fertiliser on nutrient-deficient land can in theory increase plant productivity and thus carbon sequestration (Smith *et al*., 2008). However, this is very likely to be more than offset by increases in nitrous oxide emissions from unutilised applied fertiliser in the soil, as well as carbon dioxide emissions from the production of inorganic fertiliser (Schlesinger, 2000), and the abatement potential of this measure is therefore highly uncertain. Emissions from fertilised grassland are higher than from arable crops: it has been calculated that grassland accounts for about three-quarters of all agricultural soil nitrous oxide emissions. However, background emissions are lower from unfertilised grasslands than from arable crops (quoted in Garnett , 2009). Nutrient management plans are important to reduce nutrient loss from grasslands, as are best practice techniques relating to fertiliser application.

In one long-term biodiversity restoration experiment, the *cessation* of fertilizer use was found to increase the rate of soil carbon (and N) accumulation, despite a reduction in plant biomass aboveground and belowground (De Deyn *et al.* (2010). This effect was greatest in pastures where floral seed mixes had been applied over a period of years and where red clover had recently been established. The authors postulate that the responses observed are probably due to a combined effect of clover on soil nutrient cycling and soil physical properties which together enhance the retention of carbon and nitrogen in soil.

Environmental impacts

Nutrient enrichment can have a range of detrimental impacts on water and other resources, as well as for biodiversity.

Return to contents

Plant grazing land with a legume-grass mix

Mitigation potential	+/-
Resource protection	+
Biodiversity	0/-

Mitigation impacts

Where pastures are planted with legume-grass mixtures, the legume (e.g. white clover) can provide a substantial part of the grass's nitrogen requirements, reducing the need for nitrogen fertilisation (Eory *et al.* 2015). Reducing nitrogen applications reduces greenhouse gas emissions, but the overall mitigation achieved will depend largely on what the planted mixture is replacing. For example, replacing permanent grassland with introduced grass or legume species (through ploughing and re-sowing) would lead to significant carbon losses from soils, particularly if these are high in organic matter. Legume-grass mixes need to be re-drilled around every 4 years to maintain productivity (Eory *et al.* 2015).

As an additional benefit, feeding ruminants legume forages causes the animals to emit less methane per unit of feed intake than those that are grass-fed (Waghorn et al, 2006), and increases livestock productivity (Rochon *et al.* 2003). A study comparing different dairy systems in the Netherlands (swards of ryegrass or grass-clover) found greenhouse gas emissions per unit production were 20% lower for grass-clover pasture-based systems, driven by the effects of legumes both on pasture productivity and management and livestock nutrition (Lüscher *et al*., 2014).

Environmental impacts
Planting legumes alongside grasses decreases the need for nitrogen fertiliser, which would be expected to reduce nitrate leaching. However, legumes themselves can add very high levels of nitrogen to the soil, which can result in leaching (this has mainly been studied in white clover/ grass mixtures). To reduce this risk, the proportion of grass in mixed swards needs to be sufficient to take up the nitrogen the legumes are producing. In general, studies have found that nitrogen leaching is lower under mixed swards than under highly fertilised grass swards. This is partly due to the fact that stocking density tends to be lower on mixed pastures (Lüscher *et al*., 2014). Newell Price *et al.* (2011) estimate that using clover instead of nitrogen fertiliser on grasslands reduces nitrate (plus ammonium and nitrite) leaching losses by up to 20%.

Flowering legumes provide a resource for pollinators but the capacity of simple clover and grass mixes to attract pollinators is limited (Organic Research Centre, 2015). If sown pastures were to replace existing permanent and semi-natural grasslands, there could be significant biodiversity losses.

Return to contents

Reduce stocking density on intensively managed grassland

Mitigation potential	+
Resource protection	++
Biodiversity	++

Mitigation impacts
Clearly, reducing the density of ruminant livestock will reduce emissions per unit area simply because there are fewer animals present emitting greenhouse gases. However, intensity of grazing can also influence the carbon balances of the grasslands themselves (Smith *et al*., 2008; Bell *et al*., 2009; Soussana *et al*., 2004). A certain amount of grazing stimulates root and vegetative growth, increasing productivity and sequestration rates (Marriott *et al*., 2010). Both overgrazed and ungrazed lands have both been reported as having reduced carbon accrual (Smith *et al*., 2008). Grazing also leads to carbon removal, but this can be decreased if grazing intensities are reduced (Dawson *et al*., 2007), or if grazing is deferred while forage species are most actively growing (Kemp *et al*., 2007).

According to some research, extensively managed grasslands may store more carbon than intensively managed grasslands (Soussana *et al*., 2007). However, modelling is complex because of the diversity of sward composition, soils, climate and grazing practices employed (Smith *et al*., 2008), and because of the wide range of mechanisms by which herbivores can influence soil carbon dynamics (Bardgett *et al*., 2010). In one study in the English uplands, total cessation of grazing slowed down rates of carbon cycling but after seven years had no detectable impacts on total carbon stocks in surface soil (Medina-Roldán *et al.* 2012). Grazing effects on soil organic carbon appear to be highly context-specific, varying with precipitation amounts and soil and grass types (McSherry *et al.* 2013). Given the prevalence of grasslands in the UK, further research on the impacts of different grazing practices on soil carbon sequestration has been identified as a priority (FCRN, 2010).

The influence of grazing on non- carbon dioxide gases is also not well established (Smith *et al*., 2008). Intensive grazing can lead to soil compaction, which can in turn lead to increased emissions of nitrous oxide and reduced productivity, as well as reducing the ability of soils to act as a sink for methane (Sitaula *et al*., 2000). Compaction is especially likely when conditions are wet and stocking densities are high (Newell Price *et al.* 2011). Rather than decreasing the total stock on the farm, stock can be brought inside when fields are wet to reduce soil compaction. However, ammonia and methane emissions would be increased through greater housing, storage and land spreading emissions; and carbon dioxide emissions would also increase as a result of greater forage production and manure management activities (Newell Price *et al.* 2011).

Newell Price *et al.* (2011) assert that reducing stock numbers is likely to encourage farmers to become more reliant on clover- based swards to reduce manufactured fertiliser nitrogen costs. This has the potential to bring mitigation and environmental benefits (see 'Plant grazing land with a legume-grass mix' in the current document).

The Committee on Climate Change (CCC, 2018a) has recommended *increasing* average stocking density in the UK by taking livestock off of the least productive rough grazing land and optimising grass utilisation on the most productive pastures. The mitigation potential of this measure rests in the proposal that the 'released' land will be converted to uses that sequester more carbon (for example restored natural habitats, bioenergy crops or new woodlands). However, it is important to note that the 'increased' stocking density proposed by the Committee is still within the range considered 'low' by industry.

If reducing stocking density simply results in displacing food production elsewhere, there may be no overall reduction in greenhouse gas emissions. This measure would therefore need to be pursued alongside a policy of reduced consumption of livestock products (which is also recommended by the CCC (2020). There is a growing body of evidence that current consumption trends are unsustainable from the perspectives of both environment and human health (Pelletier *et al*., 2010; Foresight, 2011; CCC, 2018).

Environmental impacts
A review for Defra (Anon., 2007) showed that soil compaction can decrease water infiltration and increase run-off, increase emissions of nitrous oxide and ammonia, decrease uptake of methane, reduce the abundance of soil fauna, decrease plant growth and yield,

and limit food availability for some birds. Avoiding poaching and soil compaction reduces leaching of nitrate, ammonium and nitrite; nitrous oxide emissions; P and associated sediment losses (Newell Price *et al.* 2011). Preventing overgrazing can also reduce wind and water- induced erosion (FAO, 2010).

Variation in grazing intensity, duration and timing, and type of stock all have a major impact on the composition of grassland vegetation, but the most appropriate grazing regimes will vary greatly depending on many factors including the habitat type, site condition and (where relevant) the conservation objectives. The concept of which grazing intensity is optimal will vary greatly depending on the objectives. For example, within a UK context, stocking densities which maximize production are often very poor from a biodiversity viewpoint. Many semi-natural habitats including grasslands actively benefit from low intensity grazing (English Nature, 2005), through maintenance and enhancement of sward structural heterogeneity, by selective defoliation due to dietary choices, trampling, nutrient cycling and propagule dispersal (Rook *et al.*, 2003). However, intensively managed grasslands are generally poor in terms of their biodiversity (Firbank, 2005). Grazing can impact bird populations through a number of mechanisms. Intensification of grassland management in the UK has reduced the suitability of grassland as feeding and breeding habitat for birds (Vickery *et al*., 2001) and increased stocking densities have been identified as one of the main causes of recent declines in populations of farmland bird species (Newton, 2004). A systematic review of the effects of grazing intensity on meadow pasture concluded that intermediate levels of grazing are usually optimal for plants, invertebrates and birds but that trade-offs are likely to exist between the requirements of different taxa (Stewart *et al.* 2008).

In a recent meta-review, out of 27 individual studies from north-west Europe, 15 found benefits to birds, plants or invertebrates in response to reducing grazing intensity on permanent grassland. The other 12 studies found mixed or zero impacts (Dicks *et al.,* 2018).

[Return to contents](#)

Extend the grazing season for cattle

Mitigation potential	+
Resource protection	-
Biodiversity	-

On farms where cows are grazed outside for part of the year and housed over the winter, the grazing season can be extended by putting animals outside earlier in spring and/or taking them in later in autumn. The limiting factor is likely to be soil condition, as grazing in winter conditions can lead to very damaging levels of poaching (Newell Price *et al.* 2011).

Mitigation impacts

When cattle are grazing, they deposit their urine directly onto the ground. It rapidly infiltrates into the soil, where physical and chemical conditions inhibit the release of ammonia. Urine deposited in the field is therefore associated with lower ammonia emissions compared with urine deposited on concrete floors within cattle housing (and associated emissions during storage and following manure spreading) (Newell Price *et al.* 2011).

Extending the grazing season increases the proportion of grazed grass in the animals' diet (with an equivalent reduction in silage). This reduces methane emissions because of improvements in animal productivity as well as a reduction in the proportion of dietary energy lost as methane. Keeping the animals outdoors for longer also reduces emissions associated with storing and spreading manure. A reduction in silage requirements reduces emissions associated with harvesting and feeding the silage to the animals (Schulte *et al.*, 2012).

Environmental impacts

Nitrate, ammonium and nitrite leaching losses and nitrous oxide emissions would all be expected to increase as a result of extending the grazing season. Phosphorus and associated sediment losses would increase where there was greater poaching damage (Newell Price *et al.* 2011). Later grazing in autumn increases the risk of compaction and nitrate leaching (Schulte *et al.*, 2012).

There is some evidence that delaying the start of mowing or grazing until later in the spring can be beneficial to plants, invertebrates or birds in some circumstances (Dicks *et al.* 2018). It seems probable that any impact on biodiversity of extending the grazing season would be negative.

Return to contents

Measures specific to arable farming

The main sources of emissions from conventional arable farming are associated with nitrogen fertilisers, with about 50% of total arable emissions coming from nitrous oxide emitted as fertilisers break down in the field, and a further 33% from the manufacture of fertilisers. Smaller amounts of emissions are associated with pesticides, field energy use, grain drying and crop residue management (AHDB, 2012). Mitigation strategies for arable farming therefore usually focus on increasing nitrogen use efficiency.

Change from winter to spring cultivars

Mitigation potential	+
Resource protection	+
Biodiversity	+

Since the 1950s, agricultural practice has shifted from spring-sown to predominately autumn/ winter-sown crops (Robinson *et al*., 2002; Clarke *et al*., 2007). This trend has partially reversed in recent years as more farmers turn to spring crops to help with weed control, and as improved spring varieties become available (Rooney, 2018). In general, individual farmers make decisions on spring or autumn cropping depending on weather and market conditions, as well as pest control considerations.

Mitigation impacts
Although winter-sown crops are perceived as more profitable and higher-yielding, spring crops may have other benefits. For example, spring-sown crops have lower nitrogen requirements than winter-sown crops (DEFRA, 2010) and may therefore lead to reduced emissions of nitrous oxide from soil and of carbon dioxide from fertiliser production and transport. However, these reductions may be outweighed by increased nitrate leaching and nitrous oxide emissions from crop residues and soil nitrogen during autumn and winter if the land is left bare (Weiske, 2005) (see also 'use cover crops' in the current document). The generally lower yields of spring crops may further offset the beneficial savings if considered in terms of greenhouse gas emissions per tonne of food.

Environmental impacts
Since spring crops results in a reduction in the use of fertilisers and pesticides during autumn and winter – the time of greatest groundwater recharge – groundwater pollution may be reduced. Newell Price *et al.* (2011) found that cultivating land in spring rather than autumn can reduce nitrate leaching losses by 20-50%; particulate phosphorus and sediment losses by 20-50% and indirect nitrous oxide emissions by a small amount. This is because autumn cultivation of land stimulates the mineralisation of nitrogen from organic matter reserves at a time when there is little nitrogen uptake by the crop. However ADAS (2007) found no evidence that spring cropping would change the impacts of diffuse nutrient pollution.

Spring cropping systems increase landscape heterogeneity and provide bare tillage and sparse swards in the nesting season, benefiting ground nesting birds such as lapwings and stone curlew. Skylark nesting densities are greater in spring, than winter, cereals (Donald, 2004). One study in England found that spring wheat held more skylarks, seed-eating songbirds and insect-eating birds than winter wheat (Henderson *et al.* 2009). Spring-sown crops are harvested later, which can be crucial for late nesting species such as corn buntings (Perkins *et al*., 2011), and enable the retention of overwinter stubbles, which can be an important seed source in modern arable systems (although effective weed control and modern machinery that leaves little spilt grain reduce biodiversity value) (Winspear *et al.*, 2005). The denser, taller vegetation of an autumn-sown crop reduces access in spring for foraging birds (Eggers *et al*., 2011). Spring crops enable the germination of more spring-germinating arable weeds, which can in turn encourage invertebrate biodiversity (Clarke *et al*., 2007). ADAS (2007) found a link between spring cropping and biodiversity, largely based on the availability of seeds left after harvest, the amount of weeds growing and associated seeds, and/or the number of invertebrates. Nevertheless, Dicks *et al.* (2018), having reviewed all the published evidence for switching to spring crops as an intervention to benefit biodiversity, concluded that the effectiveness could only be categorised as 'unknown' due to the limited evidence base.

[Return to contents](#)

Use improved crop varieties including nitrogen-efficient cultivars

Mitigation potential	+
Resource protection	+
Biodiversity	?

Mitigation impacts

Traditional crop breeding and biotechnology are used to develop crop varieties with particular desired traits, for example improved nitrogen efficiency or pest resistance. Such improvements can reduce the need for fertilisers and agrochemicals and thus reduce greenhouse emissions associated with their production, while maintaining yields. Improving nitrogen-use efficiency in plants could also reduce excess nitrogen in the system by increasing the crop's ability to compete for soil nitrogen, decreasing associated nitrous oxide emissions (ADAS, 2009; Newell Price *et al.,* 2011). Over the past 30 years, crop breeding has achieved increases in maximum potential yield and in nitrogen use efficiency (note, however, that for some crops *absolute* nitrogen requirement has still increased to achieve the maximum yields) (Eory *et al.,* 2015).

Environmental impacts

Assuming that this measure results in a decrease in excess nitrogen in the system, reductions in leaching losses would be expected, with benefits for water quality and aquatic biodiversity.

Any reduced use of fertilisers and agrochemicals could benefit biodiversity. However, in terms of the full range of possible future developments in crop varieties, much uncertainty remains in terms of potential implications for biodiversity and ecosystems. Research into the full ecological impacts of any improved varieties, from both traditional and biotechnology approaches, will be needed on a case by case basis.

[Return to contents](#)

Include nitrogen-fixing crops in rotation

Mitigation potential	+
Resource protection	0/+
Biodiversity	0/+

Growing various crops on the same piece of land in a planned sequence – rotation – is a widely used farming technique as it helps avoid pest build-up and maintain soil fertility (Crews *et al.* 2004). The agronomic and environmental benefits depend to an extent on the specific crops included in the rotation.

Mitigation impacts
Including leguminous crops like beans or peas in rotations provides biological nitrogen fixation. Legume crops do not require nitrogen fertilisers and can also have a fertilising effect on the crop that comes after them in the rotation (the latter mainly applies to green manures, as with harvested crops such as beans most of the nitrogen is removed). Thus including legumes in rotations reduces the need for external fertiliser input and associated greenhouse gas emissions (Eory *et al.,* 2015). It may also reduce the total amount of nitrogen in the system, thus lowering nitrous oxide emissions. However, legume-derived nitrogen can also be a source of nitrous oxide (Moran *et al*., 2008; Bellarby *et al.* 2008).

Environmental impacts
The reduction in added nitrogen fertiliser across the rotation should reduce leaching risks. However, if legumes are harvested during period of high rainfall, there may be significant nitrate leaching from the soil (Underwood *et al.,* 2016).

As with any crop, the impacts on biodiversity of legumes will depend largely on how the crop is managed as well as the specific crop grown (Underwood *et al.* 2016). Mass-flowering legume crops are attractive to bee species but will not necessarily benefit them at a population level (Underwood *et al.* 2016). A recent research project concluded that, while legume crops as part of sustainable farming systems have the potential to benefit biodiversity under certain conditions, it is misleading to state that legumes benefit biodiversity in general (Legume Futures, 2014).

Return to contents

Use cover crops

Mitigation potential	+
Resource protection	++
Biodiversity	+/-

A cover crop is a non-cash crop grown primarily for the purpose of protecting or improving the land between periods of regular crop production. When used to provide soil cover at key times (usually over winter) they are also known as catch crops. The choice of crop or seed mixture will depend on the particular aims of the farmer (AHDB Cereals and Oilseeds, 2015).

Mitigation impacts

Using cover crops to avoid bare ground over winter reduces nitrate leaching (by an estimated 45% on average) and may reduce emissions of nitrous oxide and enhance soil carbon sequestration (Eory *et al.* 2015). Leguminous cover crops can also provide so-called 'green manure' if ploughed into the soil, thus reducing the level of nitrogen fertiliser needed for the next crop (Kirchman *et al*., 2002; Kirk *et al.* 2012) (see 'include nitrogen-fixing crops in rotation' in the current document). Furthermore, the growth of the cover crop in early autumn recovers some of the nitrogen lost from the recently-harvested crop (Eory *et al.,* 2012). There is a small carbon cost in the form of carbon dioxide emissions associated with establishing the cover crop (Newell Price *et al.,* 2011).

Environmental impacts

There are significant soil and water benefits to be gained by using cover crops (Eory *et al.* 2012). Cover crops can reduce nitrate leaching as described above. In one study in Sweden, a conventional arable system (using mineral fertilisers) with cover cropping was found to have better nitrogen use efficiency and reduced leaching compared to an organic system (with green manures and no cover cropping) (Torstensson *et al.* 2006). Cover crops can also reduce phosphorus and sediment run-off (straw or residues can fulfil this role as well as a living crop) (Newell Price *et al.,* 2011).

Where cover crops are used as 'green manure' there can be long term improvements in soil physical and biological characteristics, increasing fertility (Eory *et al.,* 2012; AHDB Cereals and Oilseeds, 2015). Cover crops can perturb pest cycles, potentially helping reduce pesticide use as part of an Integrated Pest Management strategy (AHDB Cereals and Oilseeds, 2015).

Depending on the species planted and their management, there may be opportunities to benefit wildlife (AHDB Cereals and Oilseeds, 2015). In the UK, catch crops are generally planted too late for seeding, but may provide other biodiversity benefits such as cover from predators. If, however, catch crops replace over-winter stubbles, there can be negative effects on biodiversity, since stubbles provide important habitat for seed-eating birds (Evans *et al*., 2004; Gillings *et al*., 2010).

[Return to contents](#)

Manage water levels

Mitigation potential	-
Resource protection	- -
Biodiversity	- -

Mitigation impacts

Managing water through irrigation or drainage is sometimes proposed as a mitigation measure, as in theory the resultant higher productivity could lead to increased carbon sequestration in soils and reduced greenhouse gas emissions per unit product (Smith *et al*., 2008). However, the interactions of water level with emissions of greenhouse gases, particularly nitrous oxide, are complex (MacLeod *et al*., 2010).

Drainage on organic soils leads to large losses of carbon from the soil, overwhelming any other potential gain. Irrigation leads to increased emissions of carbon from the use of fossil fuels in operations. Hence, both irrigation and drainage have very low or even negative abatement potential in the UK (Moran *et al*., 2008; ADAS, 2009). As Weiske (2005) concludes, “a sustainable option to optimise the water management may be to match crop-growing patterns better to available water, rather than attempt to irrigate”.

Environmental impacts
Irrigation and drainage are both associated with a range of negative environmental consequences. Irrigation worsens water scarcity by diverting water from rivers and flood flows and increases the risk of salinisation (see for example Berry *et al*., 2008); leads to nutrient leaching and surface runoff, and may result in soil erosion (Wichelns *et al.* 2006).

Drainage destroys important habitats such as wetlands and wet meadows. The loss of wetlands and wet meadows are likely to have contributed to the decline of some farmland bird species (especially waders) during the past century (Bradbury *et al.*, 2006).

Return to contents

Bioenergy

Bioenergy is a general term for energy generated from biomass (dead or living organic matter, excluding fossil fuels). Biomass growth removes carbon dioxide from the atmosphere and stores it in organic matter. Using biomass for energy can reduce emissions by directly displacing fossil fuels (biomass can also indirectly reduce emissions by replacing high-carbon materials such as steel and cement). An important aspect of sustainable biomass is ensuring a hierarchy of best use is adhered to, particularly as potential demand in the future is likely to exceed sustainable supply (CCC, 2018b). A Committee on Climate Change report (CCC, 2018b) concluded that "*biomass can be produced and used in ways that are both low-carbon and sustainable. However, improved governance will be essential to ensure this happens in practice. If this is achieved, biomass can make a significant contribution to tackling climate change. If this is not achieved, there are risks that biomass production and use could in some circumstances be worse for the climate than using fossil fuels*."

Bioenergy is an extremely large topic and the current document only offers a very brief overview of some of the measures considered most relevant. The mitigation and environmental impacts of bioenergy depend very much on the detail of the feedstocks and the processes by which they are converted into bioenergy. Biomass feedstocks can either be combusted directly to produce heat and power, or processed into gases or liquid biofuels (CCC, 2018b). One specific process worth mentioning is the production of biochar. Any source of biomass, such as agricultural residues and biomass crops, can be converted by pyrolysis to yield bio-oil syngas and process heat, together with a solid product, biochar. Biochar appears to have potential both as a direct form of long-term carbon storage and as a soil improver (Woolf *et al*., 2010) and is an active area of research and development.

Anaerobic digestion of manure and slurry is dealt with in the "Manure and fertiliser management" section of the current report.

Bioenergy from agricultural and forestry residues

Mitigation potential	+
Resource protection	0/-
Biodiversity	?

Residues are the 'left overs' from commercial production of food and forestry products, such as straw or brash.

Mitigation impacts
The benefits of this measure depend on the alternative uses of the residues. Straw, for example, has many uses in agriculture, some of which may themselves have mitigation potential (for example uses that improve soil sequestration potential) (Kretschmer *et al.,* 2011).

Repo *et al*. (2011) found that using forest residues for energy production can increase emissions in the short term. This is because combustion of forest residues releases the carbon into the atmosphere much faster than natural decomposition of the residues. Therefore, climate benefits from using forest residues for energy are only achieved over a period of decades.

Biomass that would otherwise be burned without energy recovery, wastes and sustainably harvested residues have the potential to generate greenhouse gas savings without the risk of indirect land use change (Creutzig *et al*., 2015). A proposed hierarchy of biomass feedstocks places 'Agricultural and forestry co-products and residues' above both biomass from woodlands and dedicated energy crops (Kretschmer *et al.,* 2011).

Environmental impacts
The greatest potential for impacts arises from diverting the residues from alternative uses. The most obvious example is straw, which can be left on the field surface or ploughed in to improve soil structure and reduce erosion. The environmental benefits of this will vary considerably between soils. As a by-product, the supply of straw does not necessarily match demand at any given time or place, so in some cases there will be surplus which can be used for bioenergy (Kretschmer *et al.,* 2011).

Return to contents

Bioenergy from forestry

Mitigation potential	+/-
Resource protection	+/-
Biodiversity	+/-

Mitigation impacts
Serious concerns have been raised about the net greenhouse gas impacts of using wood biomass for bioenergy (see for example EEA Scientific Committee, 2011; Chatham House, 2017; RSPB *et al.* 2012). According to one calculation, burning wood pellets produced emissions comparable to natural gas (NRDC, 2015). A European research project concluded that use of stemwood from dedicated harvest for bioenergy would cause an increase in greenhouse gas emissions compared to fossil fuels over a period of decades (although it could lead to greenhouse gas savings in the long term, i.e. decades to centuries) (Joint Research Centre of European Commission, 2014). By contrast, using wood as a construction material both sequesters carbon from the atmosphere and displaces high-carbon materials such as steel (CCC, 2018b).

The Committee on Climate Change (CCC, 2018b) concludes that depending on how it is produced, forest biomass can lead to higher or lower net greenhouse gas emissions than fossil fuel equivalents. They also note work by Forest Research to develop criteria to ensure forest bioenergy is low-carbon. These include the need for forests to be managed for a range of co-products (including long-lived construction products), not harvesting biomass from forests with slow growth rates, and focussing on feedstocks such as waste wood, industrial residues and fast-decaying forest residues, which are not needed for maintaining carbon stocks.

Environmental impacts
If carried out sensitively, planting new forests and bringing existing forests into active management could bring benefits for biodiversity and resource protection (see 'forest and woodland creation' in the current document). Any harvesting from forests needs to be done in a sustainable way in order to prevent negative biodiversity impacts (Kretschmer *et al.,* 2011).

[Return to contents](#)

Bioenergy from dedicated crops

Mitigation potential	+/-
Resource protection	0/-
Biodiversity	0/-

Mitigation impacts
Full life cycle analysis is necessary to calculate the net greenhouse gas balance of bioenergy (Koh *et al.*, 2008). As well as during combustion, greenhouse gases are emitted during the operations of growing, harvesting, transporting and processing of bioenergy crops. Furthermore, growing energy crops competes with other land uses, such as food production, and may not lead to any significant net greenhouse savings due to indirect land-use change (Gove *et al*., 2010). Direct land use change to grow bioenergy crops can increase net greenhouse gas emissions if carbon rich ecosystems like wetlands, forests and grasslands are destroyed in the process (Chum *et al*., 2011). Theoretically, net sequestration from land use change could be achieved if bioenergy crops are planted on lands with carbon-poor soils where cultivation of food is not economically viable. (Creutzig et al, 2015). A proposed hierarchy of biomass feedstocks places dedicated energy crops at the bottom (i.e. least overall benefit), after residues and arisings from agriculture and forestry and biomass harvested from woodlands (Kretschmer *et al.,* 2011).

The Committee on Climate Change (CCC, 2018a; CCC, 2018b; CCC, 2020) proposes that perennial energy crops (miscanthus, short-rotation coppice willow and short rotation forestry) could play a key role in decreasing overall greenhouse gas emissions from the land sectors. Importantly, they propose that these crops could be grown on land that has been 'freed up' from current agricultural uses by changes elsewhere in the food system, thus avoiding negative impacts from indirect land use change.

Environmental impacts
Growing energy crops can bring all the biodiversity and resource protection impacts attendant on any intensive arable system. Specific impacts will depend on what the crop is and what land use it is replacing. Maize, for example, is a particularly challenging crop from an environmental perspective (Natural England, 2007). Direct or indirect land use change may cause significant environmental harm if areas of high natural value are encroached upon.

[Return to contents](#)

Bibliography

ADAS (2007) The potential for increasing the area of spring cropping to enhance biodiversity (IFO130). http://randd.defra.gov.uk/Document.aspx?Document=IF0130_7250_FRA.pdf

ADAS (2009) Analysis of Policy Instruments for Reducing Greenhouse Gas Emissions from Agriculture, Forestry and Land Management. https://fcrn.org.uk/sites/default/files/climate-ag-instruments.pdf

Aertsens, J., De Nocker, L., Gobin, A. (2013) Valuing the carbon sequestration potential for European agriculture. Land Use Policy, 31: 584-594. https://doi.org/10.1016/j.landusepol.2012.09.003

AHDB (2012) "Understanding carbon footprinting for cereals and oilseeds". [online] https://projectblue.blob.core.windows.net/media/Default/Imported%20Publication%20Docs/Understanding%20carbon%20footprinting%20for%20cereals%20and%20oilseeds.pdf

AHDB Cereals and Oilseeds (2015). "Opportunities for cover crops in conventional arable rotations." Information sheet 41. [online] https://projectblue.blob.core.windows.net/media/Default/Imported%20Publication%20Docs/IS41%20Opportunities%20for%20cover%20crops%20in%20conventional%20arable-rotations%20(2018).pdf

AHDB Dairy (2010) "What you can do to reduce nitrous oxide emissions". Factsheet 4. [online] https://dairy.ahdb.org.uk/resources-library/technical-information/climate-change/nitrous-oxide-emissions/#.XCeHqKecaRs

Alonso, I., Weston, K., Gregg, R. & Morecroft, M. 2012. Carbon storage by habitat. Review of the evidence of the impacts of management decisions and condition on carbon stores and sources. Natural England Research Reports, Number NERR043. http://publications.naturalengland.org.uk/publication/1412347

Amon, T. *et al*. (2007) Methane production through anaerobic digestion of various energy crops grown in sustainable crop rotations. Bioresource Technology 98: 3204-3212. https://doi.org/10.1016/j.biortech.2006.07.007

Angell, B. and Phillips, K. (2006) An analysis of the role of the beef and sheep industries in achieving Defra's strategic objectives. Report DLB2406

Anon. (2007) Scoping study to assess soil compaction affecting upland and lowland grassland in England and Wales. Defra research project BD2304. http://sciencesearch.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=14699

Audsley, E. *et al.* (2009) Estimation of the greenhouse gas emissions from agricultural pesticide manufacture and use. Report by Cranfield University for the Crop Protection Association, available at https://dspace.lib.cranfield.ac.uk/bitstream/handle/1826/3913/Estimation_of_the_greenhouse_gas_emissions_from_agricultural_pesticide_manufacture_and_use%E2%80%902009.pdf?sequence=1

Baird, A. *et al* (2009) A Literature Review of Evidence on Emissions of Methane in Peatlands. Defra Project SP0574. http://randd.defra.gov.uk/Default.aspx?Module=More&Location=None&ProjectID=15992

Bain, C.G. *et al.* (2011) IUCN UK Commission of Inquiry on Peatlands. IUCN UK Peatland Programme, Edinburgh. http://www.iucn-uk-peatlandprogramme.org/sites/www.iucn-uk-peatlandprogramme.org/files/IUCN%20UK%20Commission%20of%20Inquiry%20on%20Peatlands%20Full%20Report%20spv%20web_1.pdf

Bardgett, R.D. and Wardle, D.A. (2010) Aboveground-Belowground Linkages: Biotic interactions, Ecosystem Processes and Global Change. Oxford University Press.

Barsoum, N and Henderson, L. (2016). Converting planted non-native conifer to native woodlands: a review of the benefits, drawbacks and experience in Britain. Forestry Commission. https://www.forestry.gov.uk/PDF/FCRN024.pdf/$FILE/FCRN024.pdf

Bates, J. *et al*. (2009) Agriculture: methane and nitrous oxide, SERPEC-CC. https://www.ecofys.com/files/files/serpec_agriculture_report.pdf

Battini, F. *et al*. (2014). Mitigating the environmental impacts of milk production via anaerobic digestion of manure: Case study of a dairy farm in the Po Valley. Science of The Total Environment, 481, pp.196-208. https://doi.org/10.1016/j.scitotenv.2014.02.038

Beketov, M.A. *et al.* (2013) Pesticides reduce regional biodiversity of stream invertebrates. Proceedings of the National Academy of Sciences 110: 11039-11043; https://doi.org/10.1073/pnas.1305618110

Bell, M.J. and Worral, F. (2009) Estimating a region's soil organic carbon baseline: The undervalued role of land-management. Geoderma 152: 74-84. https://doi.org/10.1016/j.geoderma.2009.05.020

Bellarby, J. *et al.* (2008) Cool Farming: Climate Impacts of Agriculture and Mitigation Potential. Greenpeace, pp.1-43. http://eprints.lancs.ac.uk/68831/1/1111.pdf

Bengtsson, J., Ahnstrom, J., Weibull, A.C. (2005). The effects of organic agriculture on biodiversity and abundance: a meta-analysis. Journal of Applied Ecology 42: 261-269. https://doi.org/10.1111/j.1365-2664.2005.01005.x

Benton, T.G., Vickery, J.A. & Wilson, J.D. (2003) Farmland biodiversity: is habitat heterogeneity the key? Trends in Ecology & Evolution, 18, 182-188. https://doi.org/10.1016/S0169-5347(03)00011-9

Berry, P. M. *et al*. (2008) Deliverables 2.2 and 2.3: Meta-analysis of adaptation and mitigation measures across the EU25 and their impacts and recommendations how negative impacts can be avoided. https://climate-adapt.eea.europa.eu/metadata/projects/minimisation-of-and-adaptation-to-climate-change-impacts-on-biodiversity

Berthrong, S.T., Jobbagy, E.G., and Jackson, R.B. (2009) A global meta-analysis of soil exchangeable cations, pH, carbon, and nitrogen with afforestation. Ecological Applications, 19: 2228-2241. https://doi.org/10.1890/08-1730.1

Bradbury, R.B. and Kirby, W.B. (2006) Farmland birds and resource protection in the UK: Cross-cutting solutions for multi-functional farming? Biological Conservation 129: 530-542. https://doi.org/10.1016/j.biocon.2005.11.020

Brink, C. *et al*. (2005) Cost-effective emission abatement in agriculture in the presence of interrelations: cases for the Netherlands and Europe. Ecological Economics 53: 59-74. http://doi.org/10.1016/j.ecolecon.2004.05.008

Bullock, J.M., Pywell, R.F. & Walker, K.J. (2007) Long-term enhancement of agricultural production by restoration of biodiversity. Journal of Applied Ecology, 44, 6-12. https://doi.org/10.1111/j.1365-2664.2006.01252.x

Bussel et al (2010) How do draining and re-wetting affect carbon stores and greenhouse gas fluxes in peatland soils? Collaboration for Environmental Evidence, Systematic Review CEE 08-012 (SR 49) http://www.environmentalevidence.org/completed-reviews/how-do-draining-and-re-wetting-affect-carbon-stores-and-greenhouse-gas-fluxes-in-peat-soils

CBD (2010) Global Biodiversity Outlook 3. https://www.cbd.int/gbo3/

CCC (2010) Committee on Climate Change. The Fourth Carbon Budget - Reducing emissions through the 2020s. 7 December 2010. https://www.theccc.org.uk/publication/the-fourth-carbon-budget-reducing-emissions-through-the-2020s-2/

CCC (2018a) Committee on Climate Change. Land use: Reducing emissions and preparing for climate change. November 2018. https://www.theccc.org.uk/publication/land-use-reducing-emissions-and-preparing-for-climate-change/

CCC (2018b) Committee on Climate Change. Biomass in a low-carbon economy. November 2018. https://www.theccc.org.uk/publication/biomass-in-a-low-carbon-economy/

CCC (2020) Committee on Climate Change. Land use: Policies for a Net Zero UK. January 2020. https://www.theccc.org.uk/publication/land-use-policies-for-a-net-zero-uk/

Chatham House (2017). Woody Biomass for Power and Heat. [online] Available at: https://reader.chathamhouse.org/woody-biomass-power-and-heat-impacts-global-climate#

Chesshire, H. (2018) "Agroforestry delivers for nature and farming" [online] https://www.woodlandtrust.org.uk/blog/2018/04/agroforestry/

Choudhury, K. *et al*. (2004) Integration of Biodiversity Concerns in Climate Change Mitigation Activities, German Federal Environmental Agency. https://www.umweltbundesamt.de/sites/default/files/medien/publikation/long/2793.pdf

Chum H. *et al.* (2011). Bioenergy. In IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation, (eds Edenhofer O, Pichs-Madruga R, Sokona Y, Seyboth K, Matschoss P, Kadner S, Zwickel T, Eickemeier P, Hansen G, Schl€omer S, Von Stechow C), pp.209–332. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. https://www.ipcc.ch/report/renewable-energy-sources-and-climate-change-mitigation/

Clarke, J.H. *et al.* (2007). The SAFFIE Project Report. ADAS, Boxworth, UK. http://nora.nerc.ac.uk/id/eprint/6926/

Commission of the European Communities (2009) The role of European agriculture in climate change mitigation. Commission staff working document. https://ec.europa.eu/agriculture/climate-change/pdf/sec2009_1093_en.pdf

Council Directive 2009/128/EC of the European Parliament and of the Council of 21 October 2009 establishing a framework for Community action to achieve the sustainable use of pesticides. Available at https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32009L0128

Council Regulation (EC) No 834/2007 of 28 June 2007 on organic production and labelling of organic products and repealing Regulation (EEC) No 2092/91. Available at https://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2007:189:0001:0023:EN:PDF

Countryside Survey (2009). England Results from 2007. NERC/Centre for Ecology & Hydrology, DEFRA, Natural England, 119pp. (CEH Project Number: C03259). http://nora.nerc.ac.uk/id/eprint/8283/

Creutzig, F., *et al.* (2015). Bioenergy and climate change mitigation: an assessment. GCB Bioenergy, 7, pp.916–944. https://doi.org/10.1111/gcbb.12205

Crews, T. E. and Peoples, M. B. (2004) Legume versus fertilizer sources of nitrogen: ecological tradeoffs and human needs. Agriculture, Ecosystems and Environment 102: 279-297. https://doi.org/10.1016/j.agee.2003.09.018

Cuttle, S. P. et al. (2007) An Inventory of Methods to Control Diffuse Water Pollution from Agriculture (DWPA). User Manual, prepared as part of DEFRA Project ES0203

Dawson, J.J.C. & Smith, P. (2007) Carbon losses from soil and its consequences for land-use management. Science of the Total Environment, 382, 165-190. https://doi.org/10.1016/j.scitotenv.2007.03.023

De Deyn, G.B., Cornelissen, H.C. & Bardgett, R.D. (2008) Plant functional traits and soil carbon sequestration in contrasting biomes. Ecology Letters, 11, 516–531. http://doi.org/10.1111/j.1461-0248.2008.01164.x

De Deyn, *et al.* (2011), Additional carbon sequestration benefits of grassland diversity restoration. Journal of Applied Ecology, 48: 600-608. http://doi.org/10.1111/j.1365-2664.2010.01925.x

DEFRA (2005) Assessment of Methane Management and Recovery Options for Livestock Manures and Slurries. Final report on project AC0402. http://sciencesearch.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=14564

DEFRA (2009) Ecosystem services of peat – Phase 1 Final Report. Project SPO572. http://sciencesearch.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=15990

DEFRA (2010) Fertiliser manual (RB209) 8th edition. TSO Norwich. https://www.gov.uk/government/publications/fertiliser-manual-rb209--2

DEFRA (2017a) Agricultural Statistics and Climate Change, 8th Edition. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/666073/agriclimate-8edition-8dec17.pdf

DEFRA (2017b) The Greenhouse Gas Action Plan for Agriculture Review 2016. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/599129/ghgap-2016-review.pdf

Dicks, L.V., *et al.* (2018) Farmland Conservation Pages 245-284 in: W.J. Sutherland, L.V. Dicks, N. Ockendon, S.O. Petrovan & R.K. Smith (eds) *What Works in Conservation 2018.* Open Book Publishers, Cambridge, UK. https://www.conservationevidence.com/actions/137

Donald, P.F. (2004) The skylark. Poyser, London

Doney, S. (2010) The Growing Human Footprint on Coastal and Open-Ocean Biogeochemistry. Science 328: 1512. http://doi.org/10.1126/science.1185198

Eady, S., Carre, A. and Grant, T. (2012) Life cycle assessment modelling of complex agricultural systems with multiple food and fibre co-products. Journal of Cleaner Production, 28: 143-149. https://doi.org/10.1016/j.jclepro.2011.10.005

EC – DG Enterprise and Industry (2006) Cost-Effectiveness of Greenhouse Gases Emission Reductions in Various Sectors. Final Report. https://circabc.europa.eu/sd/a/13385584-e99a-4134-8481-fe31ae6710ba/report_greenhouse_gases_en.pdf

Edwards, 2007. Nutrition and the Environmental Impact of Outdoor Pigs. At http://www.thepigsite.com/articles/2108/nutrition-and-the-environmental-impact-of-outdoor-pigs/

EEA Scientific Committee (2011). Opinion of the EEA Scientific Committee on Greenhouse Gas Accounting in Relation to Bioenergy. [online] Available at: http://www.eea.europa.eu/about-us/governance/scientific-committee/sc-opinions/opinions-on-scientific-issues/sc-opinion-on-greenhouse-gas

Eggers, S., Unella, M. & Pärta, T. (2011) Autumn-sowing of cereals reduces breeding bird numbers in a heterogeneous agricultural landscape. Biological Conservation 144: 1137-1144. https://doi.org/10.1016/j.biocon.2010.12.033

English Nature (2005) The importance of livestock grazing for wildlife conservation. http://publications.naturalengland.org.uk/publication/68026

Eory, V. *et al.* (2015) "Review and update the UK Agriculture Marginal Abatement Cost Curve to assess the greenhouse gas abatement potential for the 5th carbon budget period and to 2050." Report prepared for the Committee on Climate Change. https://www.theccc.org.uk/wp-content/uploads/2015/11/Scotland's-Rural-Collage-SRUC-Ricardo-Energy-and-Environment-2015-Review-and-update-of-the-UK-agriculture-MACC-to-assess-abatement-potential-for-the-fifth-carbon-budget-period-and-to-2050.pdf

European Nitrogen Assessment, ed. Mark A. Sutton, Clare M. Howard, Jan Willem Erisman, Gilles Billen, Albert Bleeker, Peringe Grennfelt, Hans van Grinsven and Burna Grizzetti. Published by Cambridge University Press, 2011.

Evans A, Vickery J, Shrubb M. (2004). Importance of overwintered stubble for farmland bird recovery: a reply to Potts. BIRD STUDY 51: 94-95. http://doi.org/10.1080/00063650409461339

Falloon, P. *et al*. (2004) Managing field margins for biodiversity and carbon sequestration: a Great Britain case study. Soil Use and Management 20: 240-247. https://doi.org/10.1111/j.1475-2743.2004.tb00364.x

Falloon P. *et al*. (2009) Carbon Sequestration and Greenhouse Gas Fluxes from Cropland Soils – Climate Opportunities and Threats. In: Singh S.N. (eds) Climate Change and Crops. Environmental Science and Engineering. Springer, Berlin, Heidelberg https://www.researchgate.net/profile/Mateja_Germ/publication/251158492_Crop_Responses_to_Enhanced_UV-B_Radiation/links/56333c2d08ae5848780a9e05.pdf#page=93

FAO (2006) Livestock's long shadow. Rome. http://www.fao.org/docrep/010/a0701e/a0701e00.HTM

FAO (2010) Challenges and opportunities for carbon sequestration in grassland systems. A technical report on grassland management and climate change mitigation. Integrated Crop Management, Vol 9. http://www.fao.org/fileadmin/templates/agphome/documents/climate/AGPC_grassland_webversion_19.pdf

FAO (2011) Organic agriculture and climate change mitigation. A report of the Round Table on Organic Agriculture and Climate Change. December 2011, Rome, Italy. http://www.fao.org/fileadmin/templates/organicag/pdf/11_12_2_RTOACC_23_webfiles.pdf

FCRN (2010) Soil carbon workshop: summary of discussions. Food Climate Research Network. https://fcrn.org.uk/fcrn/seminars/soil-carbon

Finch, T. *et al.* (in review). Evaluating the land sharing-sparing continuum for bird conservation in temperate, farmland-dominated landscapes.

Firbank, L.G. (2005) Striking a new balance between agricultural production and biodiversity. Annals of Applied Biology 146 163-175. https://doi.org/10.1111/j.1744-7348.2005.040078.x

Foley, P. et al (2011). Whole-farm systems modelling of greenhouse gas emissions from pastoral suckler beef cow production systems.Agriculture, Ecosystems & Environment, 142: 222-230, https://doi.org/10.1016/j.agee.2011.05.010

Follett, R.F. (1995) Fate and Transport of Nutrients: Nitrogen. USDA Working paper no. 7. [online] https://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/landuse/crops/?cid=nrcs143_014202

Foresight. The Future of Food and Farming (2011) Final Project Report. The Government Office for Science, London. https://www.gov.uk/government/publications/future-of-food-and-farming

Forestry Commission (2017) “The UK Forestry Standard: The governments’ approach to sustainable forestry”. https://www.forestry.gov.uk/pdf/FCFC001.pdf/$FILE/FCFC001.pdf

Fornara, D. and Tilman, D. (2008). Plant functional composition influences rates of soil carbon and nitrogen accumulation. Journal of Ecology, 9, 314-322. https://doi.org/10.1111/j.1365-2745.2007.01345.x

Freibauer, A. *et al*. (2004) Carbon sequestration in the agricultural soils of Europe. Geoderma 122: 1-23 https://doi.org/10.1016/j.geoderma.2004.01.021

Frelih-Larsen, A. *et al*. (2008) Deliverable D11. Climate change mitigation through agricultural techniques. Policy recommendations, PICCMAT. https://www.ecologic.eu/8602

Garnett, T. (2009) Livestock-related greenhouse gas emissions: impacts and options for policy makers. Environmental Science and Policy 12: 491-503. https://doi.org/10.1016/j.envsci.2009.01.006

Gattinger, A. *et al.* (2012) Top soil carbon stocks under organic farming. Proceedings of the National Academy of Sciences 109: 18226-18231. http://doi.org/10.1073/pnas.1209429109

Gillings, S. *et al.* (2010) Assessing the implications of the loss of set- aside for farmland birds. Ibis 152: 713–723. https://doi.org/10.1111/j.1474-919X.2010.01058.x

Gouslon, D. *et al.* (2015) Bee declines driven by combined stress from parasites, pesticides, and lack of flowers. Science 347, Issue 6229, 1255957. https://doi.org/10.1126/science.1255957

Gove, B., Flower, K.A. and Bradbury, R.B. (2010) A review of environmental consequences of biomass production for UK energy consumption, RSPB Research Report No. 38

Hargreaves, K.J., Milne, R. & Cannell, M.G.R. (2003). Carbon balance of afforested peatland in Scotland. Forestry 76: 299–317. https://doi.org/10.1093/forestry/76.3.299

Henderson I.G., Ravenscroft N., Smith G. & Holloway S. (2009) Effects of crop diversification and low pesticide inputs on bird populations on arable land. Agriculture, Ecosystems & Environment, 129: 149-156. https://doi.org/10.1016/j.agee.2008.08.014

HM Government (2018a) "A Green Future: Our 25 Year Plan to Improve the Environment”. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/693158/25-year-environment-plan.pdf

HM Government (2018b) “Landmark Agriculture Bill to deliver a Green Brexit”. https://www.gov.uk/government/news/landmark-agriculture-bill-to-deliver-a-green-brexit

Hole, D.G. *et al.* (2005) Does organic farming benefit biodiversity? Biological Conservation 122: 113–130. http://doi.org.uk/10.1016/j.biocon.2004.07.018

Holm-Nielsen, J., Al Seadi, T. and Oleskowicz-Popiel, P. (2009). The future of anaerobic digestion and biogas utilization. Bioresource Technology, 100(22), pp.5478-5484. https://eva.fing.edu.uy/pluginfile.php/130290/mod_resource/content/1/The_future_of_anaerobic_digestion_and_biogas_utilization_-_Holm.pdf

Hooper, D.U. et al (2005) Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. Ecological Monographs 75: 3-35. https://doi.org/10.1890/04-0922

House of Commons (2018) Agriculture Bill (as amended in Public Bill Committee). https://publications.parliament.uk/pa/bills/cbill/2017-2019/0292/18292.pdf

IGER & ADAS (2007) A Review of Research to Identify Best Practice for Reducing Greenhouse Gases from Agriculture and Land Management, AC0206 report prepared for DEFRA. https://docplayer.net/25272563-A-review-of-research-to-identify-best-practice-for-reducing-greenhouse-gases-from-agriculture-and-land-management.html

IPCC (2018) Summary for Policymakers. In: Global warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty. World Meteorological Organization, Geneva, Switzerland, 32 pp. https://www.ipcc.ch/sr15/chapter/summary-for-policy-makers/

Jarvie, H.P. *et al.* (2008) Stream water chemistry and quality along an upland-lowland rural land-use continuum, south west England. Journal of Hydrology, 350, 215-231. https://doi.org/10.1016/j.jhydrol.2007.10.040

Jarvie, H.P. *et al.* (2010) Streamwater phosphorus and nitrogen across a gradient in rural-agricultural land use intensity. Agriculture Ecosystems & Environment 135: 238-252. https://doi.org/10.1016/j.agee.2009.10.002

Jefferson, R.G. *et al.* (2011) Semi-natural grasslands. In: The UK National Ecosystem Assessment Technical Report. UK National Ecosystem Assessment, UNEP-WCMC, Cambridge. http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx

Joint Research Centre of the European Commission. (2014). Carbon accounting of forest bioenergy. [online] Luxembourg: Publications Office of the European Union. Available at: http://publications.jrc.ec.europa.eu/repository/bitstream/JRC70663/eur25354en_online.pdf

Jones, L. *et al.* (2011) Coastal Margins. *In:* The UK National Ecosystem Assessment Technical Report. UK National Ecosystem Assessment, UNEP-WCMC, Cambridge. Available at http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx

Kaparaju, P. and Rintala, J. (2011). Mitigation of greenhouse gas emissions by adopting anaerobic digestion technology on dairy, sow and pig farms in Finland. Renewable Energy, 36(1), pp.31-41. https://doi.org/10.1016/j.renene.2010.05.016

Kemp, D.R. and Michalk, D.L. (2007) Towards sustainable grassland and livestock management. J. Agric Sci 145: 543-564. https://doi.org/10.1017/S0021859607007253

Kirchman, H. *et al*. (2002) Possibilities for Reducing Nitrate Leaching from Agricultural Land. Ambio 31(5): 404-408. https://doi.org/10.1579/0044-7447-31.5.404

Kirk, G. *et al*. (2012), "Quantification of the potential changes in soil carbon in England from soil protection measures within the Soil Protection Review 2010", Final report to Defra Sub-Project SP1106: Soil carbon studies to explore greenhouse gas emissions and mitigation, Defra, London. http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=17323

Klumpp, K. and Soussana, J.F. (2009) Using functional traits to predict grassland ecosystem change: a mathematical test of the response-and-effect trait approach. Global Change Biology 15: 2921-2934. https://doi.org/10.1111/j.1365-2486.2009.01905.x

Koh, L. and Ghazoul, J. (2008). Biofuels, biodiversity, and people: Understanding the conflicts and finding opportunities. Biological Conservation, 141(10), pp.2450-2460. http://www.unece.lsu.edu/biofuels/documents/2008Oct/bf08_16.pdf

Kretschmer *et al.* (2011). Securing biomass for energy – developing an environmentally responsible industry for the UK now and into the future. A Report by IEEP Commissioned by the Royal Society For The Protection Of Birds (RSPB), Friends of the Earth England, Wales and Northern Ireland, Greenpeace UK and the Woodland Trust. https://www.researchgate.net/publication/267383309_Securing_biomass_for_energy-developing_an_environmentally_responsible_industry_for_the_UK_now_and_into_the_future .

Kröger S, Parker R, Cripps G & Williamson P (Eds.) 2018. Shelf Seas: The Engine of Productivity, Policy Report on NERC-Defra Shelf Sea Biogeochemistry programme. Cefas, Lowestoft. https://doi.org/10.14465/2018.ssb18.pbd

Lamichhane, J.R., Barzman, M., Booij, K. (2015) Agron. Sustain. Dev. 35: 443. https://doi.org/10.1007/s13593-014-0275-9

Lanigan, G. *et al.* (2018). "An Analysis of Abatement Potential of Greenhouse Gas Emissions in Irish Agriculture 2021-2030**".** Teagasc Report, Oak Park, Carlow. https://www.teagasc.ie/media/website/publications/2018/An-Analysis-of-Abatement-Potential-of-Greenhouse-Gas-Emissions-in-Irish-Agriculture-2021-2030.pdf

Lawton, J.H., *et al.* (2010) Making Space for Nature: a review of England's wildlife sites and ecological network. Report to Defra. Available online at https://www.researchgate.net/publication/268279426_Making_Space_for_Nature_A_Review_of_England%27s_Wildlife_Sites_and_Ecological_Network

Lefebvre, M., Langrell, S.R.H., Gomez-Y-Paloma, S. (2015) Incentives and policies for integrated pest management in Europe: a review. Agronomy for Sustainable Development, Springer Verlag/EDP Sciences/INRA, 35:27-45. https://doi.org/10.1007/s13593-014-0237-2

Legume Futures (2014). Legume-supported cropping systems for Europe. General project report. Available at www.legumefutures.de

Leifeld, J. *et al.* (2013) PNAS 110 (11) E984. https://doi.org/10.1073/pnas.1220724110

Lindsay, R. (2010) Peatbogs and carbon: a critical synthesis, RSPB Scotland. http://ww2.rspb.org.uk/Images/Peatbogs_and_carbon_tcm9-255200.pdf

Lorenz, K., Lal, R., 2014. Soil organic carbon sequestration in agroforestry systems. A review. Agron. Sustain. Dev. 34, 443–454. http://doi:10.1007/s13593-014-0212-y

Lukehurst, C., Frost, P and Al Seadi, T. (2010) Utilisation of digestate from biogas plants as biofertiliser. IEA Bioenergy. https://energiatalgud.ee/img_auth.php/4/46/IEA_Bioenergy._Utilisation_of_digestate_from_biogas_plants_as_biofertiliser._2010.pdf

Lüscher, A., Mueller-Harvey, I., Soussana, J., Rees, R. and Peyraud, J. (2014). Potential of legume-based grassland-livestock systems in Europe: a review. Grass and Forage Science, 69(2), pp.206-228. https://doi.org/10.1111/gfs.12124

Newton, I. (2004). The recent declines of farmland bird populations in Britain: an appraisal of causal factors and conservation actions. IBIS Vol 146 (4) 579-600. https://doi.org/10.1111/j.1474-919X.2004.00375.x

NRDC (2015). [online] Available at: https://www.nrdc.org/sites/default/files/bioenergy-modelling-IB.pdf

MacDonald, M.A. (2006) The indirect effects of increased nutrient inputs on birds in the UK: a review, RSPB Research Report 21. http://cescos.fau.edu/gawliklab/papers/MacDonaldMA2006.pdf

MacDonald, M.A. *et al.* (2017) Benefits of coastal managed realignment for society: Evidence from ecosystem service assessments in two UK regions. Estuarine, Coastal and Shelf Science. http://doi.org/10.1016/j.ecss.2017.09.007

MacLeod, M. et al. (2010) Review and update of UK marginal abatement cost curves for agriculture. Final Report to the Committee on Climate Change. https://www.theccc.org.uk/wp-content/uploads/2010/12/pr_supporting_research_SAC_agriculture.pdf

Marañón, E., *et al*. (2011). Reducing the environmental impact of methane emissions from dairy farms by anaerobic digestion of cattle waste. Waste Management, 31(8), pp.1745-1751. https://doi.org/10.1016/j.wasman.2011.03.015

Marriott, C.A., *et al.* (2010) Impacts of extensive grazing and abandonment on grassland soils and productivity. Agriculture, Ecosystems and Environment 139; 476-482. https://doi.org/10.1016/j.agee.2010.09.005

McAdam J.H., McEvoy P.M. (2009) The Potential for Silvopastoralism to Enhance Biodiversity on Grassland Farms in Ireland. In: Rigueiro-Rodróguez A., McAdam J., Mosquera-Losada M.R. (eds) Agroforestry in Europe. Advances in Agroforestry, vol 6. Springer, Dordrecht. https://doi.org/10.1007/978-1-4020-8272-6_17

McCracken, D. *et al.* (2011) Enclosed Farmland. In: The UK National Ecosystem Assessment Technical Report. UK National Ecosystem Assessment, UNEP-WCMC, Cambridge. http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx

McSherry, M. E. and Ritchie, M. E. (2013). Effects of grazing on grassland soil carbon: a global review. Glob Change Biol, 19: 1347-1357. http://doi.org/10.1111/gcb.12144

Medina-Roldán *et al.* (2012) Grazing exclusion affects soil and plant communities, but has no impact on soil carbon storage in an upland grassland. Agriculture, Ecosystems & Environment 149: 118-123. https://doi.org/10.1016/j.agee.2011.12.012

Miles, R. and Richardson, N. (2018) Sustainable Shores (Technical Report). Royal Society for the Protection of Birds. https://www.rspb.org.uk/globalassets/downloads/projects/sustainable-shores-project---technical-report.pdf

Monteny, G. et al. (2006) Greenhouse gas abatement strategies for animal husbandry. Agriculture, Ecosystems and Environment 112: 163-170. https://doi.org/10.1016/j.agee.2005.08.015

Moraine, M. *et al.* (2014) Farming system design for innovative crop-livestock integration in Europe. Animal, 8: 1204-1217. http://doi.org/10.1017/S1751731114001189

Moran, D. et al. (2008) UK Marginal Abatement Cost Curves for the Agriculture and Land Use, Land-Use Change and Forestry Sectors out of 2022, with Qualitative Analysis of Options to 2050. Final Report to the Committee on Climate Change, SAC Commercial Ltd. https://www.theccc.org.uk/wp-content/uploads/2016/02/UK-Marginal-Abatement-Cost-Curves-for-the-Agriculture-and-Land-Use-Land-Use-Change-and-Forestry-Sectors-to-2022.pdf

Morison, J. *et al.* (2012). Understanding the carbon and greenhouse gas balance of forests in Britain. Forestry Commission Research Report. Forestry Commission, Edinburgh. https://www.forestry.gov.uk/website/publications.nsf/DocsByUnique/F892C36600EB60F580257EBB0046F9DB

Morris J. *et al.* (2010). Restoration of Lowland Peatland in England and Impacts on Food Production and Security. Report to Natural England. Cranfield University, Bedford. https://www.fensforthefuture.org.uk/admin/resources/downloads/lowland-peatland-restoration-study.pdf

Mosquera-Losada M.R., Freese D., Rigueiro-Rodríguez A. (2011) Carbon Sequestration in European Agroforestry Systems. In: Kumar B., Nair P. (eds) Carbon Sequestration Potential of Agroforestry Systems. Advances in Agroforestry, vol 8. Springer, Dordrecht. https://doi.org/10.1007/978-94-007-1630-8_3

National Atmospheric Emissions Inventory. [online] https://uk-air.defra.gov.uk/assets/documents/reports/empire/naei/annreport/annrep98/chap2_1.html. Accessed 11 Jan 2018

Natural England (2007) Environmental impacts of land management. NERR030. http://publications.naturalengland.org.uk/publication/30026

Natural England (2010) England's peatlands. Carbon storage and greenhouse gases. http://publications.naturalengland.org.uk/publication/30021

Naylor, R., *et al.* (2005). Losing the links between livestock and land. Science 310: 1621-1622. http://doi.org/10.1126/science.1117856

Newell Price, J.P. *et al.* (2011) "An Inventory of Mitigation Methods and Guide to their Effects on Diffuse Water Pollution, Greenhouse Gas Emissions and Ammonia Emissions from Agriculture". Prepared as part of Defra Project WQ0106. https://www.climatenorthernireland.com/cmsfiles/Images/An-Inventory-of-Mitigation-Methods-and-Guide-to-their-Effects_2011.pdf

Newton, I. (2004). The recent declines of farmland bird populations in Britain: an appraisal of causal factors and conservation actions. IBIS Vol 146 (4) 579-600. https://doi.org/10.1111/j.1474-919X.2004.00375.x

O'Brien, D., Shalloo, L., Buckley, F., Horan, B., Grainger, C. and Wallace, M., 2011. The effect of methodology on estimates of greenhouse gas emissions from grass-based dairy systems. Agriculture, Ecosystems & Environment 141, 39-48. https://doi.org/10.1016/j.agee.2011.02.008

Organic Research Centre (2015) 'Diverse legume leys: pollinator benefits'. Summary available online at https://agricology.co.uk/resources/diverse-legume-leys-pollinator-benefits; full guide not accessible as at Jan 2018.

Ostle, N.J. *et al.* (2009) UK land use and soil carbon sequestration. Land Use Policy, 26, S274-S283. https://doi.org/10.1016/j.landusepol.2009.08.006

Paveley, N. *et al.* (2008) Can disease management reduce greenhouse gas emissions? Arable Cropping in a Changing Climate. HGCA R&D Conference 23 - 24 January. https://www.researchgate.net/publication/237731081_Can_disease_management_reduce_greenhouse_gas_emissions

Pelletier, N. and Tyedmers, P. (2010) Forecasting potential global environmental costs of livestock production 2000–2050. http://doi.org/10.1073/pnas.1004659107

Perkins A.J. *et al.* (2011) Adaptive management and targeting of agri-environment schemes does benefit biodiversity: a case study of the corn bunting *Emberiza calandra*. Journal of Applied Ecology. http://doi.org/10.1111/j.1365-2664.2011.01958.x

Phoenix, G.K. *et al.* (2008) Sustaining ecosystem services in ancient limestone grassland: importance of major component plants and community composition. Journal of Ecology 96: 894-902. https://doi.org/10.1111/j.1365-2745.2008.01403.x

Plieninger, T. (2011) Capitalizing on the Carbon Sequestration Potential of Agroforestry in Germany's Agricultural Landscapes: Realigning the Climate Change Mitigation and Landscape Conservation Agendas, Landscape Research, 36: 435-454, http://doi.org/10.1080/01426397.2011.582943

Powlson, D. S., Whitmore, A. P. and Goulding, K. W. (2011), Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and the false. European Journal of Soil Science, 62: 42-55. http://doi.org/10.1111/j.1365-2389.2010.01342.x

Quinton, C.D. *et al.* (2018) Prediction of effects of beef selection indexes on greenhouse gas emissions. Animal, 12: 889-897. http://doi.org/10.1017/S1751731117002373

Read, D.J., Freer-Smith, P.H., Morison, J.I.L., Hanley, N., West, C.C., Snowdon, P. (eds). 2009. Combating climate change – a role for UK forests. An assessment of the potential of the UK's trees and woodlands to mitigate and adapt to climate change. The Stationery Office, Edinburgh. https://www.forestry.gov.uk/pdf/SynthesisUKAssessmentfinal.pdf/$FILE/SynthesisUKAssessmentfinal.pdf

Repo, A. *et al*. (2011). Forest bioenergy climate impact can be improved by allocating forest residue removal. GCB Bioenergy, 4(2), pp.202-212. https://doi.org/10.1111/j.1757-1707.2011.01124.x

Rhodes, C.J. (2013) Peak phosphorus - Peak food? The need to close the phosphorus cycle. Science Progress 96 (Pt 2): 109-52. http://doi.org/10.3184/003685013X13677472447741

Robinson, R. A. and Sutherland W.J. (2002) Post-war changes in arable farming and biodiversity in Great Britain. Journal of Applied Ecology 39: 157-176. https://doi.org/10.1046/j.1365-2664.2002.00695.x

Rochon, J.J. *et al.* (2003) Grazing legumes in Europe: a review of their status, management, benefits, research needs and future prospects. Grass and Forage Science 59: 197–214. https://doi.org/10.1111/j.1365-2494.2004.00423.x

Rook, A.J. and Tallowin, J.R.B. (2003) Grazing and pasture management for biodiversity benefit. Animal Research 52: 181-189. https://doi.org/10.1051/animres:2003014

Rooney, D. (2018) "Early Bird Survey of GB cropping intentions ~ provisional figures**".** [online] https://cereals.ahdb.org.uk/markets/market-news/2018/november/30/early-bird-survey-of-gb-cropping-intentions-~-provisional-figures.aspx

Rotz, C.A. (2004) Management to reduce nitrogen losses in animal production. Journal of Animal Science 82: E119-E137. https://www.researchgate.net/profile/Ca_Rotz/publication/8243583_Management_to_Reduce_Nitrogen_Losses_in_Animal_Production/links/549af0830cf2b803713716b2.pdf

Scherer-Lorenzen, *et al.* (2003) The role of plant diversity and composition for nitrate leaching in grasslands. Ecology, 84, 1539-1552. https://doi.org/10.1890/0012-9658(2003)084[1539:TROPDA]2.0.CO;2

Schlesinger, W. H. (2000) Carbon sequestration in soils: some cautions amidst optimism. Agriculture, Ecosystems and Environment 82: 121-127. https://doi.org/10.1016/S0167-8809(00)00221-8

Schulte, R. and T. Donnellan (2012). "A Marginal Abatement Cost Curve for Irish Agriculture", Teagasc Report, Dublin, Ireland. https://www.teagasc.ie/media/website/publications/2012/1186_Marginal_Abatement_Cost_Curve_for_Irish_Agriculture.pdf

Silcock, P., Brunyee, J. and Pring, J. (2012). "Changing livestock numbers in the UK Less Favoured Areas – an analysis of likely biodiversity implications". Report to the RSPB. https://www.rspb.org.uk/globalassets/downloads/documents/positions/agriculture/changing-livestock-numbers-in-the-uk-less-favoured-areas---an-analysis-of-likely-biodiversity-implications.pdf

Sitaula, B.K, Hansen, S., Sitaula, J.I. and Bakken, L.R. (2000) Methane oxidation potentials and fluxes in agricultural soils: Effects of fertilisation and soil compaction. Biogeochemistry, 48, 323-339

Smith, P. *et al*. (2000) Meeting Europe's climate change commitments: quantitative estimates of the potential for carbon mitigation by agriculture. Global Change Biology 6: 525-539. https://doi.org/10.1046/j.1365-2486.2000.00331.x

Smith, P. *et al*. (2008) Greenhouse gas mitigation in agriculture. Phil Trans R Soc B 363: 789-813. https://doi.org/10.1098/rstb.2007.2184

Smith, J., Pearce, B.D. and Wolfe, M.S. (2013) Reconciling productivity with protection of the environment: Is temperate agroforestry the answer? Renewable Agriculture and Food Systems 28: 80–92. https://doi.org/10.1017/S1742170511000585

Snyder, C.S. *et al*. (2009) Review of greenhouse gas emissions from crop production systems and fertilizer management effects. Agriculture, Ecosystems and Environment 133: 247-266. https://doi.org/10.1016/j.agee.2009.04.021

Soussana, J.F. *et al*. (2004) Carbon cycling and sequestration opportunities in temperate grasslands. Soil Use and Management 20: 219-230. https://doi.org/10.1111/j.1475-2743.2004.tb00362.x

Soussana et al. (2007) Full accounting of the greenhouse gas (CO_2, N_2O, CH_4) budget of nine European grassland sites. Agriculture, Ecosystems and Environment 121: 121–134.

Steinbeiss, S., *et al.* (2008) Plant diversity positively affects short-term soil carbon storage in experimental grasslands. Global Change Biology, 14: 2937-2949. https://doi.org/10.1111/j.1365-2486.2008.01697.x

Stewart G.B. & Pullin A.S. (2008) The relative importance of grazing stock type and grazing intensity for conservation of mesotrophic 'old meadow' pasture. Journal for Nature Conservation, 16, 175-185. https://doi.org/10.1016/j.jnc.2008.09.005

Thyø, K.A. and Wenzel, H. (2007) Life Cycle Assessment of Biogas from Maize silage and from Manure, 2nd draft, Institute for Product Development. https://www.xergi.fr/images/dokumenter/life_cycle_assessment_report.pdf

Tilman, D., Hill, J. and Lehman, C. (2006) Carbon-negative biofuels from low-input high-diversity grassland biomass. Science, 314, 1598-1600. http://doi.org/10.1126/science.1133306

Torralba, M. *et al.* (2016) Do European agroforestry systems enhance biodiversity and ecosystem services? A meta-analysis. Agriculture, Ecosystems & Environment 230: 150-161. https://doi.org/10.1016/j.agee.2016.06.002

Torstensson, G., Aronsson, H. and Bergström, L. (2006) Nutrient Use Efficiencies and Leaching of Organic and Conventional Cropping Systems in Sweden. Agron. J. 98:603–615. http://doi.org/10.2134/agronj2005.0224

Tsonkova, P. *et al*. (2012) Agroforest Syst 85: 133. https://doi.org/10.1007/s10457-012-9494-8

Tuck, S. *et al.* (2014). Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta-analysis. Journal of Applied Ecology, 51(3), pp.746-755. https://doi.org/10.1111/1365-2664.12219

Tuomisto, H.L. *et al.* (2012) Does organic farming reduce environmental impacts? – A meta-analysis of European research. Journal of Environmental Management, 112: 309-320. https://doi.org/10.1016/j.jenvman.2012.08.018

T. Vanden Nest, B. *et al.* (2014) Effect of organic and mineral fertilizers on soil P and C levels, crop yield and P leaching in a long term trial on a silt loam soil. Agriculture, Ecosystems & Environment 197: 309-317. https://doi.org/10.1016/j.agee.2014.07.019

Underwood, E. and Tucker, G. (2016) Ecological Focus Area choices and their potential impacts on biodiversity. Report for BirdLife Europe and the European Environmental Bureau, Institute for European Environmental Policy, London. http://doi.org/10.13140/RG.2.2.12692.30085

Van der Sluijs, J.P. et al. (2015) Conclusions of the Worldwide Integrated Assessment on the risks of neonicotinoids and fipronil to biodiversity and ecosystem functioning. Environ Sci Pollut Res (2015) 22: 148. https://doi.org/10.1007/s11356-014-3229-5

Vanguelova, E. *et al.* (2018) Afforestation and Restocking On Peaty Soils - New Evidence Assessment. ClimateXChange, pp.1-43. https://www.climatexchange.org.uk/media/3137/afforestation-and-restocking-on-peaty-soils.pdf

Vickery, J.A. *et al*. (2001) The management of lowland neutral grasslands in Britain: effects of agricultural practices on birds and their food resources. Journal of Applied Ecology 38: 647-664. https://doi.org/10.1046/j.1365-2664.2001.00626.x

Waghorn G.C., Woodward S.L., Tavendale M. And Clark D.A. (2006) Inconsistencies in rumen methane production – effects of forage composition and animal genotype. International Congress Series, 1293, 115–118. https://doi.org/10.1016/j.ics.2006.03.004

Wedlock, D.N. *et al.* (2013) Progress in the development of vaccines against rumen methanogens. Animal 7 Suppl 2:244-52. http://doi.org/10.1017/S1751731113000682

Weigelt, A. *et al*. (2009) Biodiversity for multifunctional grasslands: equal productivity in high-diversity low-input and low-diversity high-input systems. Biogeosciences, 6: 1695-1706. https://doi.org/10.5194/bg-6-1695-2009

Weiske, A. (2005) Survey of technical and management-based mitigation measures in agriculture. MEACAP. http://minisites.ieep.eu/assets/244/WP3D7a_mitigation.pdf

Weiske, A. *et al*. (2006) Mitigation of greenhouse gas emissions in European conventional and organic dairy farming. Agriculture, Ecosystems and Environment 112: 221-232. https://doi.org/10.1016/j.agee.2005.08.023

Weiske, A. (2007) Potential for carbon sequestration in European agriculture. MEACAP. https://ieep.eu/publications/meacap-potential-for-carbon-sequestration-in-european-agriculture

Wichelns, D. and Oster, J. D. (2006) Sustainable irrigation is necessary and achievable, but direct costs and environmental impacts can be substantial. Agricultural Water Management 86: 114-127. https://doi.org/10.1016/j.agwat.2006.07.014

Williams, A.G., Audsley, E. and Sandars, D.L. (2006) Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main report. Defra Research Project ISO205. Bedford: Cranfield University and Defra. http://randd.defra.gov.uk/Default.aspx?Module=More&Location=None&ProjectID=11442

Wilson, P., Sparkes, D. (2007) Carbon Footprint of Weed Control, BCPC Annual Review of Weed Control

Wilson, D. *et al.* (2013) Rewetted industrial cutaway peatlands in western Ireland: a prime location for climate change mitigation? Mires and Peat 11, Article 01, 1–22. http://pixelrauschen.de/wbmp/media/map11/map_11_01.pdf

Winspear R. and Davies G. (2005) A management guide to the birds of lowland farmland. The RSPB, Sandy

Woolf, D, *et al*. (2010). Sustainable biochar to mitigate global climate change. Nature Communications, 1: 1-9. https://www.nature.com/articles/ncomms1053

Wulf, S., Maeting, M. and Clemens, J. (2002) Application technique and slurry co-fermentation effects on ammonia, nitrous oxide and methane emissions after spreading: II. GHG emissions. J. Environ. Quality 31, 1795-1801. https://www.researchgate.net/profile/Joachim_Clemens/publication/11001595_Application_technique_and_slurry_co-fermentation_effects_on_ammonia_nitrous_oxide_and_methane_emissions_after_spreading_II_Greenhouse_gas_emissions/links/00b4951fb88c60c32a000000.pdf

Zanchi, G., Pena, N., Bird, N. (2010). The upfront carbon debt of bioenergy. Joanneum Research. https://www.transportenvironment.org/publications/upfront-carbon-debt-bioenergy

Zavaleta, E.S. *et al*. (2010) Sustaining multiple ecosystem functions in grassland communities requires higher biodiversity. Proceedings of the National Academy of Sciences of the United States of America, 107, 1443-1446. https://doi.org/10.1073/pnas.0906829107

We welcome feedback and comments about the content of this report. Please contact Mair Floyd-Bosley at Mair.FloydBosley@rspb.org.uk for more details

Image credits: Front cover, Andy Hay (rspb-images.com)