Realising grassland potential: an ecosystem services toolkit for farmers

Literature review

Introduction

Grassland covers 40% of England and 70% of England's farmland. High-quality, well-maintained grassland, now only 5% of the total, represents a hugely diverse ecosystem that can be teeming with abundant wildlife whilst also delivering benefits for climate and watercourses. And yet, despite these benefits, there are a very limited number of routes through which farmers can potentially attract private finance into grassland restoration and maintenance, reflected in the low levels of current engagement with private finance amongst dairy and livestock farmers.

This project aims to develop a grassland-specific ecosystem services toolkit focused on smaller farms. The tool kit will equip farmers with the knowledge to engage effectively with emerging private finance opportunities for transitioning to more regenerative grassland management such as Biodiversity Net Gain (BNG), carbon, and in some locations water management benefits.

This literature review is the first stage in the project. It aims to collate the available evidence for delivery of ecosystem services from UK grasslands, associated soils, grazing/cutting regimes and to identify knowledge gaps. This information will then be used to devise a grassland typology and inform the development of the toolkit.

Methods

A literature search was undertaken to collate evidence for the delivery of ecosystem services (ES) from UK grasslands. ES categories follow Bengtsson et al. (2019) and focus on those considered to have potential links to emerging financial markets e.g. carbon credits or biodiversity net gain.

For the purpose of this project, grassland is defined as plant communities included in Volume 3 of the National Vegetation Classification (NVC) (Rodwell, 1998), plus those included in the 'purple moor grass and rush pastures' priority habitat type definition (JNCC, 2011) i.e. NVC communities M23, M24, and M25 (Rodwell, 1991) when occurring on organo-mineral soils or shallow (<50 cm) peats.

Literature was sourced from Web of Science Core Collection using the keyword search in Table 1 and filtered by region to include peer reviewed papers published from UK or northern European countries only. Papers were screened by reviewing titles and then abstracts for relevance and retained for detailed review where they included content either specifically relating to the delivery of ES by UK or relevant European grasslands, or wider factors where of relevance to the UK context e.g. soil processes.

Additional material was sourced via citation searching of retained papers with Litmaps[1], recommendations from expert stakeholders, and keyword searches of the DEFRA[2] and Natural England[3] evidence archives.

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[1] Litmaps | Your Literature Review Assistant
[2] https://sciencesearch.defra.gov.uk/
[3] https://publications.naturalengland.org.uk/

Table 1: Search string applied to Web of Science Core Collection

Operator	Field	String
	Topic	"Natural capital" OR "Ecosystem services" OR "Regulating services" OR "supporting services" OR "provisioning services" OR "Natures contribution to people" OR "NCP"
AND	Abstract	("Natural capital" OR "Ecosystem services" OR "Regulating services" OR "supporting services" OR "provisioning services" OR emissions OR carbon* OR flood* OR erosion* OR soil* OR "climate regulation" OR water* OR runoff OR "natural hazard protection" OR "nutrient cycling" OR "habitat provision" OR biodiversity OR "food production" OR "crop pollination") **AND** (Grassland* OR meadow OR pasture OR hay)
NOT	Title	(forest* OR Japan* OR Chin* OR Mongolia* OR Iceland* OR Mediterranean OR Australia* OR Kenya* OR Africa* OR Tibet* OR Brazil* OR "New Zealand" OR Namibia* OR "United States" OR US OR USA OR America* OR California* OR Tropical OR seagrass* OR Zostera* OR Patagonia* OR eelgrass* OR Posidonia* OR Pakistan* OR India* OR "blue carbon" OR steppe OR Russia* Or Columbia* OR Turk* OR Ecuador* OR arid OR Peru* OR Coastal* OR beaver OR waterbird OR waterfowl)

Evidence for individual ecosystem services provided by UK grasslands.

Climate regulation

Given the extensive literature on carbon and other greenhouse gases, this section draws primarily on several recent literature reviews synthesising the available data for UK grasslands (Anderson, 2024; Jordon et al., 2024; Manning et al., 2023; Doar and Sutherland, 2022; Gregg et al., 2021), with additional recent studies included where relevant.

The UKNEA (Bullock et al., 2011) highlighted challenges in interpreting data on soil carbon stocks for semi-natural grasslands due to limited sampling depths in large soil datasets, such as the 2007 Countryside Survey (Carey, 2008) and the grouping of habitats or land use types in various studies (Carey, 2008; Bradley et al., 2005; Janssens et al., 2005). Despite much research in the interim, these issues have persisted, as noted in several recent syntheses of carbon data for UK habitats (Anderson, 2024; Jordon et al., 2024; Gregg et al., 2021).

Quantifying and comparing carbon storage and sequestration across UK grassland types remains challenging due to methodological inconsistencies and data gaps. As noted in the UKNEA and elsewhere, many analyses are limited to the top 15 or 30 cm of soil, leaving the significant amounts of carbon stored in deeper layers unaccounted for and underestimating total grassland carbon stocks (Anderson, 2024; Gregg et al., 2021; Bullock et al., 2011). Even where large datasets and sophisticated statistical modelling are used to address confounding factors, there is still high variability in carbon stocks within and between land uses, including grasslands (Levy et al., 2024).

Gregg *et al* (2021) also highlight the lack of standardized protocols for measuring greenhouse gas fluxes across habitats. Differences in sampling techniques and the inclusion or exclusion of methane and nitrous oxide fluxes in calculations lead to inconsistencies in reported values, making drawing meaningful comparisons difficult. In addition, while most studies focus on short-term changes in stocks, long-term dynamics and legacy effects of past management practices remain poorly understood, particularly at deeper levels in the soil profile (Ward et al., 2016a).

These factors make it difficult to accurately predict existing carbon stocks, project future carbon storage potential, and assess the long-term effectiveness of different management strategies.

Soil carbon stocks

Data presented by Anderson (2024), Doar and Sutherland (2022), and Gregg et al (2021), show broad ranges of soil carbon stocks for different grassland types and management regimes. These data are drawn from multiple studies with varying methodologies, including differences in the use of bulk density parameters which can be problematic (Levy et al., 2024; Smith et al., 2020; Poeplau et al., 2017; Lee et al., 2009). The variability in the reported values highlights the difficulty in making meaningful comparisons, even where some quantitative data are available. The authors also note a particular lack of UK data for both calcareous and marshy grasslands.

Although quantitative comparisons between specific grassland communities are not possible, the literature qualitatively suggests that whilst complex and depending on the interaction of various factors including land use history, management, and physical properties, certain soil types are broadly considered to have higher total carbon stocks than others including:

- ***Organo-mineral soils*** with high levels of soil organic matter (SOM) in the surface horizons typically contain more carbon than mineral soils (Berdeni and Williams, 2021; Gregg et al., 2021).

- ***Floodplain soils*** store large amounts of carbon from multiple sources: buried carbon, continuous addition of allochthonous carbon from sediment deposits, and carbon sequestered by vegetation (Sear et al., 2023).

- ***Clay-rich soils*** are associated with higher levels of carbon storage than sandy soils due to the role of clay particles in the formation of mineral-associated organic matter (MAOM) (Anderson, 2024; Jordon et al., 2024)

- ***Deep soils*** physically have more material for carbon storage than shallow soils, although in soils that are not alluvial or aeolian, carbon density decreases exponentially with depth due to root biomass distribution and litter decomposition dynamics (Levy et al., 2024).

- ***Undisturbed soils*** tend to have higher levels of carbon compared to examples of similar soil types that have been subject to carbon-releasing activities such as ploughing or drainage (Anderson, 2024). Soils with good structure i.e. high frequency and distribution of 30–150 µm pores are associated with higher long-term carbon stores (Kravchenko et al., 2019).

Soil carbon storage and sequestration

The carbon-sequestration rates across grassland types and management regimes collated by Anderson (2024) and Gregg et al. (2021) are highly variable. Doar and Sutherland (2022) present best estimates for various land-use changes, which also vary hugely for some interventions. For example, the potential effect of creating lowland meadow from intensive grassland on organo-mineral soils presented ranges from a gain of 5.23 to a loss of 9.4 t $CO2_e$ ha^{-1} yr^{-1}. Warner et al (2020) also present a mixed picture for grassland options within Countryside Stewardship. The authors note the evidence base is not sufficiently robust to make quantitative comparisons of carbon fluxes between different habitats and management regimes and highlight particularly sparse data for calcareous and acid grasslands. However, as with soil-carbon stocks, some broad principles can be drawn from the literature that may help inform land-use change decisions.

Sequestration potential

Long-established, extensively managed grasslands on undisturbed mineral soils are important carbon stores but have a limited capacity for additional sequestration. Once soils reach their carbon-saturation point, further increases become limited. (Jordon et al., 2024; Gregg et al., 2021). However, storage is not permanent as carbon can be released through various management changes such as soil disturbance (Gregg et al., 2021).

Floodplain soils are an exception, continuously sequestering carbon via the regular deposition of sediments, which add allochthonous carbon and bury existing soil organic matter in the surface horizons, protecting it from decomposition. Broad floodplains with low erosion, high sediment loads, and frequent flooding store more sediment quickly, while narrow floodplains with high erosion and low sediment loads have lower storage (Sear et al., 2023).

Changes in management or restoration activities can promote carbon sequestration in grasslands, but sequestration capacity is highly variable dependent on soil type, land-use history, and the existing soil-carbon stocks. Broadly, soils with depleted organic matter content, e.g. those used for intensive agriculture, are likely to be furthest from their saturation point and have greatest capacity for additional carbon sequestration (Anderson, 2024; Doar and Sutherland, 2022; Gregg et al., 2021).

Sequestration rates

The rate of carbon sequestration following beneficial land use-change is not linear. Initially high rates are likely to reduce over time as the soil reaches a new equilibrium/saturation point (Jordon et al., 2024; Doar and Sutherland, 2022) and rates can change or even reverse if management or environmental conditions are altered. Shorter term studies tend to report higher sequestration rates (Conant et al., 2017), potentially leading to overestimates.

Management interventions

Species diversity & land use change

Establishing permanent grassland on arable land sequesters carbon. A field trial by Warner et al (2020) reported sequestration of 0.45 t C ha^{-1} yr^{-1} over a ten-year period following conversion of arable land to low-input grassland under Countryside Stewardship. Anderson (2024) cites a global synthesis which showed arable reversion to be amongst the interventions with the highest carbon gains (Conant et al., 2017).

Greater plant species diversity in grasslands increases carbon sequestration by creating and maintaining good soil structure, specifically a broad pore-size distribution throughout the profile, which has been shown to be key in processing and stabilising soil carbon inputs (Kravchenko et al., 2019).

The inclusion of some deep-rooting legume species is beneficial for carbon sequestration (Anderson, 2024; Bai and Cotrufo, 2022; Gregg et al., 2021; Yang et al., 2019; Cong et al., 2014). However, where fertilisation occurs alongside the addition of legumes, excess N availability can lead to increased N_2O emissions, counteracting the benefit (Jordon et al., 2024; Garnett et al., 2017; Ward et al., 2016). There is limited evidence specifically from UK semi-natural grasslands, with most data from experimental plots in neutral or intensive grasslands e.g. (Savage et al., 2021; De Deyn et al., 2011).

Grazing

Grazing management influences soil carbon storage but is site specific and effects vary depending on intensity and practices (Gregg et al., 2021; Garnett et al., 2017). It is generally accepted that sustained high stocking densities lead to carbon loss (Anderson, 2024; Berdeni and Williams, 2021; Gregg et al., 2021; Garnett et al., 2017). Studies in progress e.g. Rivero et al. (2024) show some carbon enrichment of the surface layer in response to cell grazing in the short term, but longer-term systematic studies showing the effect of mob and rotational grazing systems on the carbon content of the whole profile and its relationship with soil density are, as yet, elusive (Jordon et al., 2024; Wagner et al., 2023; Garnett et al., 2017).

Light and moderate grazing is considered better for carbon storage than intensive systems, but to what extent these systems actively sequester carbon is less clear and difficult to disaggregate from other factors (Gregg et al., 2021). Eze et al. (2018c) attribute higher levels of soil carbon under less intensive grazing regimes to increased plant litter return. However, Bai and Cotrufo (2022), whilst acknowledging uncertainty around the contribution of above and below ground plant matter to soil carbon fractions in different contexts, state that just 7% of above ground residues are converted to MAOM, suggesting that this may not transfer to long-term carbon storage.

There are limited UK specific studies on the impact of grazing on different grassland communities. Ward et al (2016 a&b) found highest levels of soil carbon under intermediate rather than extensive levels of grazing in both experimental plots at Colt Park (MG6 grassland) and analysis of a large dataset of soil samples from a range of English grasslands. They attribute this to a balance between primary productivity and decomposition rates but do not separate the influence of grazing from that of fertilisation.

Cutting

The influence of haymaking on carbon cycling has received little attention (Gregg et al., 2021) However, Myrgiotis *et al* (2022) suggested that cut grasslands may be a weaker C-sink than grazed fields in a modelling exercise covering all managed grassland in Great Britain, but based on very little primary data.

Fertilisation

Evidence for the effect of fertilisation on soil carbon is mixed. Gregg et al. (2021) point to studies in neutral grasslands where increases in carbon storage are attributed to higher primary productivity driven by the application of inorganic fertiliser (Eze et al., 2018c, 2018b; Fornara et al., 2013). Fornara et al. (2013) found N only VS NPK treatments increased root biomass and carbon storage. In contrast, Garnett et al. (2017) cite the findings of Henderson et al. (2015) that ammonium nitrate application resulted in carbon losses due to higher plant shoot VS root growth, supported by the meta-analysis of Yue et al. (2021), and/or increased rates of carbon decomposition.

Long-term application of farmyard manure (FYM) to MG6 grassland at Colt Park was found to be beneficial for soil carbon stocks but detrimental to bacterial:fungal ratios in the soil (Ward et al., 2016). This suggests short-term carbon gains may be offset given the important role of fungi in processes influencing the more stable soil carbon fractions (Li et al., 2024; Hawkins et al., 2023; Domeignoz-Horta et al., 2021).

Increases in soil-carbon sequestration associated with fertilisation may be offset by other emissions e.g. NO_2 arising from their storage and use and must be considered in the context of other potential impacts on biodiversity, water quality etc. (Jordon et al., 2024; Gregg et al., 2021). For example, both NPK and FYM treatments on MG6 grassland at Colt Park resulted in increased nutrient leaching (Ward et al., 2016).

Other greenhouse gases:

The need to account for whole greenhouse gas budgets in grassland systems, including methane (CH_4), and nitrous oxide (N_2O) emissions from the system itself, and potential impacts of displacing food production is widely acknowledged (Jordon et al., 2024; Gregg et al., 2021; Soussana et al., 2007). However, comparisons are complicated by different metrics. Blignaut et al. (2022) calculated the lifetime methane emissions of a cow and her calves to be a net source of $CO2_e$ using the standard GWP_{100} metric but a net sink with GWP*, which was designed to address concerns that the former overestimates the long-term benefits and underestimates the short-term benefits of reducing methane emissions (Allen et al., 2018, 2022).

Livestock related emissions

Sousanna et al (2007) calculated a greenhouse gas balance across a range of grassland types in Europe, concluding that the balance was effectively zero when emissions both on and off site from cattle were deducted from sequestration rates. Chang et al (2021) also show that, on a global scale, the emissions from cattle in sparsely grazed and natural grasslands cancels out the climate cooling benefits of the soil organic carbon sink.

Manning et al. (2024) developed a model for calculating greenhouse gas emissions associated with different livestock in conservation grazing systems. Based on IPCC guidelines using GWP_{100}, it suggests that the species and size (using weight as a proxy) of livestock have the greatest effect on emissions, medium ponies being the lowest overall emitters per livestock unit (0.66 t $CO2_e$ LU^{-1} yr^{-1}) and small sheep breeds the highest (3.6 t $CO2_e$ LU^{-1} yr^{-1}). Larger breeds of each species are suggested to be lower emitters per livestock unit e.g. 0.38 t $CO2_e$ LU^{-1} yr^{-1} for large breeds of cattle, versus 0.43 t $CO2_e$ LU^{-1} yr^{-1} for small breeds. The model is heavily caveated by the authors, who note that it cannot account for the impact of different forage composition on livestock emissions or consider baseline fluxes from the habitats considered (which includes only calcareous grassland) due to a lack of robust data on these factors.

Legumes containing condensed tannins such as bird's-foot trefoil can both reduce enteric methane emissions in livestock and reduce NO_2 emissions by up to 25% by increasing the proportion of N excreted in faeces rather than urine (Mueller-Harvey et al., 2019; Lüscher et al., 2014).

Habitat and soil nutrient related emissions

There is limited information available on the magnitude of methane or nitrous oxide emissions from specific UK grassland communities and management regimes.

Warner et al. (2020) provide some figures on methane and nitrous oxide fluxes different land uses, but for broad categories e.g. a methane emission of 0.37 t $CO2_e$ ha^{-1} yr^{-1} for 'intensive grassland on drained peat' and an uptake of 0.03 t $CO2_e$ ha^{-1} yr^{-1} for 'Grassland (managed) on peat'. They note that complex and varied factors influence NO_2 emissions from agricultural soils.

Floodplain meadows are dynamic systems that can be either a source or sink of methane depending on water table height, duration of soil anoxia, and the presence of aerenchymatous plant species (Peacock et al., 2024). Wet or waterlogged soils also favour denitrification, leading to NO_2 emissions which may peak during flood events (Gregg et al., 2021). Warner et al. (2020) estimate that moving livestock to fields less prone to flooding or waterlogging during winter reduces the risk of raised NO_2 emissions by lowering the amount of nitrogen deposited onto anaerobic soil via animal excreta by 0.001 – 0.007 t $CO2_e$ LU^{-1} per month.

Data Gaps:

There are multiple data gaps with respect to carbon storage and sequestration in UK grasslands including.

- Limited easily comparable data on carbon stocks and sequestration rates of different UK grassland communities, particularly marshy and calcareous grasslands. Difficulties arise where samples are taken from various depths, with some only considering the superficial layer, and soil densities are not uniformly assessed.

- Limited comparable data on the specific impacts of different management on carbon dynamics on and between different UK grassland types e.g. including grazing intensity, mowing regimes, fertilization, and restoration techniques.

- Limited comparable data on the full greenhouse gas balances of different UK grasslands and management regimes e.g. methane and nitrous oxide emissions which could potentially offset sequestration of atmospheric CO_2.

- Limited long-term data on the impacts of management changes on different UK grassland types, particularly with respect to turnover of the labile fraction of soil organic matter and its potential to negate the benefits of more stable carbon fractions.

Water services

Water quantity and flow regulation

Grasslands contribute to the regulation of water resources via the infiltration and retention of water in their soils. These processes can support groundwater recharge by replenishing aquifers and maintaining baseflows (Weatherhead and Howden, 2009), prevent or mitigate the impacts of flooding by reducing surface runoff and, in the case of floodplain grasslands, temporary storage of water during flood events (Vári et al., 2022; Bengtsson et al., 2019; Posthumus et al., 2010). The relative importance of these services is likely to be geographically driven with water stress more likely in the drier south and east and flooding in the wetter north and west (Weatherhead and Howden, 2009).

The water regulation potential of semi-natural grasslands is poorly understood, particularly at larger scales. In many modelling studies grasslands are treated as a single category, assuming properties attributed to intensive grasslands, such as low infiltration rates and limited water-holding capacity, can be applied to all grassland types (Ellis, Anderson, et al., 2021).

Whilst the influence of land use and management on flood risk at small scales is acknowledged, the effects are highly context dependent. Factors such as soil type, management practices, and the scale of flood events interact in complex ways. This complexity, coupled with limited long-term monitoring, makes it challenging to scale findings or to isolate the impacts of land use from climate variability (Dadson et al., 2017; Pattison and Lane, 2012).

Limited quantitative data specifically relating to water quantity and flow regulation from different UK semi-natural grassland types are available, with most studies examining the impact of different land uses and/or management types more broadly. However, these do offer some evidence that semi-natural grasslands can deliver these services more effectively than arable land or intensive grassland. For example, a recent meta-analysis of European studies concluded that permanent grasslands, defined as those not included in crop rotations for at least five years, had lower bulk density, higher hydraulic conductivity, and lower runoff than arable soils (Milazzo et al., 2023).

The infiltration and water storage capacity of grasslands is controlled by soil characteristics including type, texture, depth, and structure (Marley et al., 2024; Alaoui et al., 2018; Gregory et al., 2015a; Gilman, 2002; Gowing et al., 2002; Boorman et al., 1995), but can be mediated by various factors including vegetation type and management, which has implications for potential grassland restoration (Ellis, 2021).

Management

The high stocking rates and tracking of farm machinery associated with intensive grassland and arable management can lead to the degradation of soil structure via compaction, particularly when undertaken on wet soils (Hargreaves et al., 2019; Gregory et al., 2015, Anon, 2007; O'Connell, 2004).

Compaction is linked to increased bulk density, disconnection between top and subsoils, decreased porosity, and reduction in earthworm populations, resulting in reduced infiltration and hydraulic conductivity, and promoting surface water pooling and runoff (Alaoui et al., 2011; Batey, 2009, Anon, 2007). Gregory et al. (2015) report increases in bulk density of up to 0.18 Mg/m^3 and decreases in soil porosity of 10–25% from studies of compacted UK soils. The same review also notes the importance of soil organic matter (SOM) for porosity, water retention, and infiltration, citing reductions of 10-40% in aggregate stability for 1% less SOM, and up to 10% decrease in water retention for differences in SOM content of 3-7% (Gregory et al., 2015).

The DEFRA project BD2304 (Anon, 2007) and O'Connell (2004) draw a clear link between increased agricultural intensification and higher compaction levels and subsequent reductions in infiltration and increases in runoff rates. The reports cite figures from various studies, including Heathwaite et al. (1990,) who measured an 80% reduction in infiltration capacity and double the amount of surface runoff from heavily grazed compared to lightly grazed permanent grassland in Devon.

Modelling of catchment monitoring data at Pontbren in upland Wales over four years demonstrated areas with higher proportions of agriculturally 'improved' VS more natural grassland had a flashier response, although the authors acknowledged the need for further research to unpick the relative influence of soil type, drainage, management, and vegetation (McIntyre and Marshall, 2010). Plot scale experiments at the same project recorded 48% less runoff and lower bulk density at 2-5 cm depth in ungrazed VS sheep grazed grassland plots after five years (Marshall et al., 2014).

More recently, rainfall simulation experiments in Devon demonstrated that soils in extensively managed unimproved 'culm grasslands' (M23, M24, & M25) took longer to reach saturation point and exhibited slower soil-moisture release compared to species-poor swards under intensive management (MG6 & MG7). The authors attribute this to lower O horizon bulk density, higher organic matter, and significantly lower levels of compaction (Ellis, 2021; Ellis, Brazier, et al., 2021).

Differences in land use and management can also contribute to the resilience of soils and soil processes to extremes of flooding and drought. Grasslands have been shown to exhibit greater resilience to flooding than arable land, experiencing less severe impacts on soil properties and earthworm populations, and recovering more quickly. (Kiss et al., 2021; Harvey et al., 2019).
Lavallee et al. (2024) demonstrated that intensive grassland management favours the development of stress-tolerant bacterial soil microbial taxa, but has the opposite effect on fungal taxa, reducing their resilience to drought and potentially contributing to less efficient carbon and nitrogen cycling.

Vegetation

Species richness has been shown to positively influence soil stability and structure. Greater plant diversity has been shown to stimulate the development of soil pores (Kravchenko et al., 2019) and diversity of plant root traits including depth, length density, diameter, and exudation are key for good soil structure and stability (Bardgett et al., 2014) and therefore for infiltration and water holding capacity.

In field and mesocosm experiments, Gould et al. (2016) demonstrated a positive link between species diversity and soil properties through complementary sets of root traits, highlighting the importance of grasses for aggregate stability and legumes for hydraulic conductivity and strength. Pérès et al (2013) demonstrated similar results independent of soil texture in mesocosm experiments. Plot experiments comparing different forage species found infiltration rates in plots sown with red and white clover to be eight times higher than those with a perennial ryegrass monoculture (Marley et al., 2024).

Various studies on afforestation show increased vegetation surface roughness can slow the flow of water across the land surface, leading to increased water infiltration and reduced surface runoff (Dadson et al., 2017). For grasslands this may be of relevance on shallow or highly compacted soils where saturation is reached quickly, however, there are limited studies investigating this for different grassland types. In addition, Dadson et al. (2017) note that in floodplains, increasing surface roughness usually has minimal impact on flood levels in all but the largest floods, where flows reach a similar speed to the river channel.

Two small studies investigated surface roughness with specific reference to the flood mitigation potential of semi-natural grasslands. Research in Devon demonstrated that vegetation density and surface roughness affected runoff volume at the field scale with 1.4 times longer surface flow pathways calculated in M24 than in MG6, resulting a significant difference in drainage density at areas over 1,225 m^2 (Ellis, 2021; Ellis, Brazier, et al., 2021).

Plot-scale flume experiments in Swindale, attribute differences in mean overland flow velocity between four grassland/management types to surface roughness and seasonal management. Rank (ungrazed) grassland was found to have the slowest flow (0.026 m/s), then rushes (0.029 m/s) and low-density grazing (0.032 m/s), with hay meadows having the highest velocity (0.041 m/s), the authors suggest this could result in a >1hr delay in overland flow for rank grassland VS hay meadow in an 18 mm storm. Flow velocity from the hay meadow also significantly increased from 0.006 m/s in July to 0.054 m/s in September following the hay cut (2020).

Water quality

Semi-natural grasslands can contribute to improved water quality through nutrient retention and cycling, and erosion/sediment control.

Nutrient retention and cycling

Nutrient transport, particularly of phosphorus (P) and nitrogen (N) into freshwater via leaching or runoff from agricultural systems is a major driver of eutrophication (Dungait et al., 2012).

The Environment Agency (2019) state that excess P levels are the leading cause of waterbodies in England failing to achieve the Water Framework Directive's 'Good Ecological Status.' In cases where agriculture and land management are primary pressures, P accounts for 55% of failures.

Rothwell et al. (2022) identified an 89.5 Gg surplus of P in UK agricultural soils, with 8.38 Gg lost to aquatic environments in 2018, largely from livestock manure. In the Stour catchment, Withers et al. (2024) attributed 43% of total P flux to agriculture, with areas dominated by livestock farming showing the highest P surplus (9.3 kg ha^{-1} yr^{-1}) and flux (0.92 kg ha^{-1} yr^{-1}) due to high manure levels and feed imports. They suggest reducing this surplus could lower annual river P levels by a third. In addition, legacy P losses from soils with a history of over-application remain a significant long-term eutrophication source Dungait et al. (2012)

Agriculture is also the dominant source of freshwater nitrogen (N) pollution in the UK, contributing an estimated 81% nationally, but is highly site specific with impacts ranging from 1-100% across different waterbodies (Zhang et al., 2014). Dissolved organic N increases with agricultural intensity, correlating strongly with cattle stocking density and arable land percentage (Jarvie et al., 2008, 2010). Dungait et al. (2012) further demonstrates the link between intensification and leaching of N, citing lower losses from low-input, extensively managed grasslands (<20 kg ha^{-1} yr^{-1}) compared to intensively managed, drained grasslands (up to 150 kg ha^{-1} yr^{-1}).

A recent analysis suggests that targeting nutrient reductions in headwater streams would offer significant water quality benefits. Jarvie et al. (J2018) found P limitation was more common in rivers and N limitation in headwaters, though upland, low-alkalinity streams often showed co-limitation, and suggest reducing P in lowland headwaters and N in upland rivers could deliver the greatest improvements.

Extensively managed grasslands avoid contributing to water quality issues by functioning as low-nutrient systems with lower stocking densities, minimal or no inputs, and reduced soil compaction, which is associated with less runoff as discussed in the previous section. However, they can also actively enhance water quality by retaining and cycling nutrients.

de Vries et al. (2012) provide clear evidence that extensively managed, species-rich neutral grasslands (MG3, MG4, and MG5) have better retention and lower leaching of N compared to intensively managed, species-poor grasslands (MG6 and MG7) across a range of soil types. This is attributed to higher fungal:bacterial ratios in soils, which immobilize and enhance retention of N. In experimental calcareous grasslands, Phoenix et al. (Phoenix et al., 2008) found grass only communities had similar, low levels of N leaching compared to mixed communities, suggesting grasses may be the main driver behind the low leaching rates.

In experimental settings on calcareous bedrock, species-richness in grasslands have been demonstrated to improve the efficiency of P uptake via root trait diversity and associations with arbuscular mycorrhizal fungi (AMF). In field studies, this effect was shown to be mediated or overridden by management intensity as fertilization and/or intensive grazing caused a shift to direct P uptake by dominant, fast-growing species. This provides evidence that extensively managed, species-rich grasslands promote more efficient P cycling, and can reduce P losses (Oelmann et al., 2011, 2021).

In floodplain grasslands, haymaking can significantly improve water quality by exporting P and N in the biomass of hay crop, reducing excess nutrients in the soil that are vulnerable to leaching. Measurements from a range of traditionally managed floodplain meadows harvested in June indicate that median values of 6.91 kg ha^{-1} of P and 61 kg ha^{-1} of N can be removed in hay (Gowing et al., n.d.). A single hay cut is typically sufficient to balance the annual nutrient inputs from flood sediments, although this depends on the volume and nutrient composition of the sediments deposited, which varies for a given site depending on weather conditions and physical and land management features within the catchment. Replacing autumn grazing with a second hay cut can reduce nutrient levels further, removing 8.0 kg ha^{-1} of P and 86.9 kg ha^{-1} of N (Bowskill et al., 2023).

Sediment

Erosion of soil and subsequent transport of sediment into watercourses contributes to water quality issues by carrying nutrients, but is also a factor in and of itself, blocking out light, reducing oxygen levels, degrading aquatic habitats, and affecting a range of wildlife including spawning fish (Holden et al., 2017; Bilotta and Brazier, 2008). Agriculture has been estimated to be responsible for 72% of sediment pollution in UK rivers (Zhang et al., 2014).

No studies were found comparing sediment runoff rates for or between specific UK semi-natural grassland types or between semi-natural grasslands and other land uses. Most research has focused on comparisons between arable and intensive grasslands or examined the impact of management intensity more generally. For example, Glendell and Brazier (2014) compared two adjacent, contrasting catchments in southwest England. The predominantly agricultural catchment, containing a mixture of arable land, short-rotation grassland, and permanent pasture was found to export higher concentrations of suspended sediment than the semi-natural catchment dominated by semi-natural habitats including woodland and moorland at 76.32 t km^{-2} and 44.99 t km^{-2} respectively.

Whilst arable land generally exhibits higher sediment runoff than grassland (Milazzo et al., 2023; Evans et al., 2017), intensively managed grasslands can also be significant sources, particularly those with high stocking rates, soil compaction, outwintering animals, and subsurface drainage (Evans et al., 2017; Bilotta et al., 2007, 2008). For example, Peukert et al. (2014) recorded erosion rates of 182.2–527.4 kg ha^{-1} yr^{-1} suspended sediment from intensively managed grasslands on the North Wyke Platform (cattle or sheep grazing with 65-78 kg ha^{-1} N inputs). Additional research at North Wyke found no significant impact of cattle and sheep grazing on sediment loss when best practice grazing management was employed. This included stocking rates of 0.15–1.00 UK livestock units (LU) ha^{-1} for sheep, and 0–0.77 LU ha^{-1} for cattle, with reduced stocking rates when soils are waterlogged, indoor housing of livestock in winter, and fencing of ditches and streams (Pulley et al., 2021) .

Floodplain grasslands can also contribute to water quality through the capture and storage of sediments, as well as uptake and export of the nutrients contained within them as discussed above. Sediment deposition is widely variable dependent on individual catchments and the frequency of flood events but can be substantial, a study of ten floodplain meadow sites across five catchments (Thames, Severn, Trent, Derwent, and Great Ouse) following the 2007 summer floods showed the meadows retained up to 40 t ha^{-1} (Rothero et al., 2016).

Data gaps

Data gaps relating to water quality, quantity and flow regulation include:

- Limited quantitative data on infiltration rates of different UK semi-natural grassland types across different soil types and management regimes.

- Limited research on the effect of vegetation surface roughness in grasslands on flood mitigation, particularly for different grassland types.

- Absence of comparative studies on sediment runoff rates between UK semi-natural grasslands and other land uses

Pollination and pest control

Pollination

Crop pollination is in high demand in Europe (Bengtsson et al., 2019), with the value of pollination services to UK agriculture recently estimated at between £189 million and £379 million per year (Scarpa et al., 2022). Whilst no sources directly quantifying the contribution of grasslands or specific grassland types to this figure were found, it is widely recognised that grasslands affect crop pollination and agricultural yields via spillover (Bullock et al., 2021; Klaus et al., 2021; Orford et al., 2016; Woodcock et al., 2016). Wild pollinator populations are critical to UK agriculture given that managed honeybees are thought to supply only approximately a third of the pollination required to support national food production (Vanbergen et al., 2014). In one field experiment c. 0.4 t ha^{-1} of yield gains in an oilseed rape crop were attributed to insect pollinators (Woodcock et al., 2016).

Direct evidence of differences in pollination service delivery between specific UK grassland communities is limited. However, several factors have been shown to influence the delivery of pollination services from UK grasslands more generally.

Grassland diversity and composition

Species-rich and semi-natural grasslands typically support a greater abundance and diversity of pollinators compared to species-poor examples by providing a wider range of resources for pollinators, catering to a greater variety of species (Larkin and Stanley, 2023; Holland et al., 2017; Orford et al., 2016; Woodcock et al., 2014). Examples of restored calcareous grassland can achieve similar species occurrence and abundance of moths to semi-natural examples, and more than eight times that of arable fields, within short timescales (3 years) (Alison et al., 2017).

Floral abundance and temporal availability

Plant communities themselves do not necessarily influence the diversity or abundance of pollinators. Instead, flower abundance and agricultural intensification were shown to be important factors in hoverfly species-richness and abundance, whilst soil moisture influenced hoverfly community composition in a of four grassland types (MG5, MG6, M24, and MG10) in West Wales (Lucas et al., 2017). Increased availability of flowers is associated with a greater abundance of pollinators, and increased pollination services (Bullock et al., 2021a; Woodcock et al., 2019; B. A. Woodcock et al., 2014; Woodcock, 2010; Potts et al., 2003, 2009).

The availability of flowers throughout the active season provides a consistent food supply for pollinators, supporting persistence of populations and subsequently pollination services (B. A. Woodcock et al., 2014; Bäckman and Tiainen, 2002). Late-season nectar availability is particularly important for bumblebee colonies, as it coincides with the development of reproductive larvae (Pywell et al., 2011). Diverse grasslands support a range of species with a variety of flowering times and provide a good continuity of food resources for pollinators throughout the season depending on how they are managed. (Doležal et al., 2019; B. A. Woodcock et al., 2014).

Plant diversity and species composition.

Functional traits of pollinators, e.g. tongue length in some bee species, affect which plant species they are attracted to and can effectively pollinate (*effect traits*)(Woodcock et al., 2019; Goulson et al., 2005; Carvell, 2001). Plant diversity in grasslands is positively correlated with functional diversity of pollinators (Orford et al., 2016). This, in turn, is positively correlated with diversity of effect traits which has been shown to increase pollination levels and yields in oilseed rape, even when accounting for overall pollinator abundance (Woodcock et al., 2019). The partitioning of floral resources based on traits like tongue length also allows different functional groups of pollinators to coexist, contributing to more stable and effective pollination services (Woodcock et al., 2019; Garibaldi et al., 2011; Carvell, 2001).

Plant species composition in grasslands also directly affects the abundance and diversity of pollinators. Calcareous grassland moth species were shown to be more abundant on restored sites where plant indicator species of the target habitat had established (Alison et al., 2017) and several studies have found that the presence of key species can be more important than overall botanical diversity for attracting wild bees and hoverflies (Nichols et al., 2019; Warzecha et al., 2018).

Additionally, field experiments in the UK showed that adding seed mixtures containing legumes and non-leguminous forbs to agriculturally improved grasslands significantly enhanced pollinator species-richness, abundance, and functional diversity (Orford et al., 2016; Woodcock et al., 2014; Pywell et al., 2011). These studies propose that targeted use of key plant species in simple seed mixtures offers a cost-effective approach to improving pollination services in grassland creation or enhancement schemes. However, the beneficial effects on pollination of adding red clover or mixed seed to semi-improved MG6 grassland was found to be negated by the application of either NPK or FYM fertiliser (Ward et al., 2016).

Management

The factors discussed above can be significantly influenced by the management practices, or lack thereof, applied to individual grasslands. Cutting, grazing, fertilization, and abandonment all affect the floral resources available to pollinators, which in turn affect their abundance, diversity, and ability to deliver pollination services. There appears to be a no single management approach that is universally optimal for pollinators, choices between cutting and grazing should therefore be based on local conditions and pollinator community composition (Rosa García and Fraser, 2019).

Cutting

Cutting and removal of biomass, such as in hay or silage production, can either benefit or disadvantage pollinators depending on the timing and frequency of cutting. The literature does not identify a single optimal cutting schedule and given the variability in both pollinator communities and in grasslands across edaphic and geographic gradients, a one-size-fits-all approach is unlikely to be effective (Woodcock, 2010; Potts et al., 2009).

However, several general principles have been linked to increased species-richness and abundance of pollinators. These include timing cuts to promote sward species and structural diversity, ensure floral abundance, and maintain continuity of flowering throughout the pollinator-active period, all of which can be influenced by plant community composition (Savage et al., 2021b; B. A. Woodcock et al., 2014; Woodcock, 2010; Potts et al., 2009). Additionally, retaining refuge or nesting areas is important for certain species, such as the brown-banded carder bee and the ringlet butterfly (Potts et al., 2009; Carvell, 2001). However, total abandonment of management soon leads to the loss of insect-pollinated plant species, reducing the value of grassland for pollinators (Sõber et al., 2024).

A study on Romanian hay meadows showed that by reflowering in late summer/early autumn, early cut meadows (late June) were an important late-season food source for pollinators, suggesting that restricting cutting of high diversity grasslands to late summer can be detrimental (Johansen et al., 2019).

Based on these principles, extensive management, with some spatial and/or temporal variation in cutting tailored to the plant and invertebrate communities present, appears most likely to optimise pollination services across a range of grassland types.

Grazing

The impact of grazing regimes on pollination is complex and highly variable, as different grazing intensities and stock types can lead to contrasting outcomes for pollinator communities and the plants they rely on.

Intensive grazing with high stocking densities can reduce floral resources and simplify sward structures in grasslands, reducing their suitability for many pollinator species but particularly those with specialised dietary or nesting requirements (Rakosy et al., 2022; B. A. Woodcock et al., 2014; Redpath et al., 2010). In contrast, extensive grazing systems with lower stocking rates and periods of stock exclusion, can increase plant diversity, provide higher floral abundance, and foster a heterogeneous sward structure (B. A. Woodcock et al., 2014; Woodcock, 2010; Potts et al., 2009; Tallowin et al., 2005). However, even within extensive grazing systems, the specific timing and intensity of grazing can significantly influence pollinator communities. For example, livestock exclusion during peak summer flowering times, has been shown to be particularly beneficial for pollinators in grasslands (Woodcock et al., 2014).

Fertilisation

Fertilization is widely recognized to reduce plant species richness in grasslands over time, thereby diminishing overall floral resources (Bullock et al., 2011). Both mineral and organic fertilizers, including farmyard manure (FYM), have been shown to negatively impact pollinators (Ward et al., 2016). However, a UK study comparing different fertilization treatments found that FYM had a less detrimental effect on pollinator diversity than mineral fertilization (NPK). The study concluded that FYM provides the most effective balance between maintaining biodiversity, including pollinator abundance, and supporting forage productivity and livestock nutrition (Villa-Galaviz et al., 2023).

Landscape context.

The ability of UK grasslands to deliver pollination services is influenced by landscape composition and spatial context. Loss and fragmentation of species-rich grasslands reduces the availability of forage plants supporting wild pollinator populations, and subsequently their abundance, functional diversity, and capacity to pollinate crops and influence agricultural yields (Woodcock et al., 2014; Garratt et al., 2014; Carvell et al., 2006; Goulson et al., 2005).

Spillover of pollinators into arable or horticultural land is affected by the proportion, size of and connectivity to semi-natural habitat in the surrounding landscape but the importance of these factors varies for different taxa at different scales (Larkin and Stanley, 2021, 2023; Botham et al., 2015). Alison et al. (2017) found no significant effect between connectivity of restored and existing semi-natural calcareous grassland and overall moth abundance at up to 7 km. In contrast, Klaus et al. (2021) demonstrated solitary bee visits to potted *Phacelia* plants declined by 61% between 0-400 m from semi-natural calcareous grasslands but observed a smaller effect on bumblebees and hoverflies. Similarly, experiments in oilseed rape crops in England showed visitation rates to the crop by bumblebees, solitary bees, and flies significantly declined within 200 m of field margins, but honeybee visitation rates were unaffected by distance (Woodcock et al., 2016).

Garratt et al. (2014) demonstrate that field beans are primarily pollinated by bumblebees, while oilseed rape is pollinated by a range of taxa.

Pest control

No UK-specific data are available on the pest control benefits of semi-natural grasslands or between different grassland types. Most research has focused on spillover of natural enemies from field margins, the impact of agri-environment schemes, and the role of invertebrate functional diversity.

For example, (Woodcock et al., 2016) demonstrate that spillover of natural predators from field margins has a significant effect on extinction rates of aphid colonies in adjacent wheat crops, with species-rich rather than grass-only margins extending the reach of this effect up to 50 m into the crop. The authors suggest that for a 12 ha field, inclusion of species-rich margins could enhance pest control services for 50% of the total area. The study also indicated additive effects of different predator guilds, with the combined effect of canopy-active and ground-active predators resulting in faster aphid population decline than where only one guild was present. Bullock et al. (2021b) report similar results from a study on the effectiveness of sown wildflower margins within agri-environment schemes.

Several reviews acknowledge the potential of grasslands to support natural predator populations and provide pest control services in agricultural landscapes, but highlight the limited specific studies in Europe and worldwide, with most European studies focused on grass margins in agricultural systems (Nagy et al., 2020; Holland et al., 2016, 2017b). However, some research covers factors that may broadly affect the extent to which UK grasslands can deliver pest control services. Given many hoverfly species have predatory larvae (Woodcock et al., 2022), the factors discussed for pollinators above may also be of relevance in some contexts.

Landscape factors.

A synthesis of European and North American studies demonstrated that landscape simplification, negatively impacts natural pest control. Average pest control levels were 46% lower in homogeneous landscapes compared to those with a higher proportion of semi-natural habitats, including grasslands (Rusch et al., 2016). In the UK, deficits in functional diversity of ground beetles were found to be correlated with areas with high proportions of arable land (Woodcock et al., 2014b).

A French study found that a 50% increase in the proportion of grassland (meadows and set-aside areas) within 500 m of cereal fields resulted in a 38% increase in seed predation and a 20% increase in aphid predation, but distance displayed no significant effect.

Martin et al. (2019) show that whilst high edge density is beneficial overall and favours many natural predator groups, particularly those overwintering outside crops, responses can vary depending on functional traits and overall landscape composition. For example, ground dispersing species that can overwinter within crop fields were associated with a higher proportion of arable land and lower edge density. Boetzl et al. (2024) and Venturo et al. (2024) indicate similar differences with carabid beetles, demonstrating that distance functions are heavily influenced by functional traits, crop type, and adjacent habitat.

Habitat factors

Age

Boetzl et al. (2021) found that older, well-established habitats under agri-environmental schemes in Europe supported greater species richness of most taxa, including those associated with pest control, suggesting that older grasslands, are more likely to support higher levels of biodiversity and associated pest control services. Similarly, restored grasslands in Austria attracted pollinators within three years of establishment, but predatory species were slower to respond (Hussain et al., 2021) and carabid beetles in restored *Nardus* grasslands in Belgium did not fully resemble the community or traits expected (DeCock et al., 2023).

Structural Complexity

A review of European research, cites studies suggesting tall, herbaceous vegetation and tussock forming grasses are attractive to some predatory invertebrates groups and can increase winter survival rates, but notes different species are likely to have different preferences (Holland et al., 2016).

An Irish study found increased species-richness and abundance of predatory arthropods in fenced grass margins compared to grazed fields (Anderson et al., 2013).

Woodcock and Pywell, (2010) report a negative correlation between sward height and predatory species-richness in lowland calcareous grassland but acknowledge the impact of winter only grazing and reliance on pitfall trapping on this finding and suggest that variation in sward structure should benefit overall invertebrate diversity.

Data gaps

The literature suggests that the ability of UK grasslands to deliver pollination and pest control services will vary with what invertebrate taxa they can support, local landscape factors, and the types of crops under cultivation. However, there is limited information focussed on the delivery of either service by UK semi-natural grassland types specifically, with most research focussed on agri-environment scheme interventions such as wildflower margins or addition of legume-rich seed mixes in intensive grassland or arable systems.

Biomass cropping

The Biomass Strategy (Department for Energy Security & Net Zero, 2023) outlines the UK Government's plan to use biomass in support of the nation's net-zero commitment. However, it does not specifically address biomass cropping from permanent grasslands.

Numerous studies focus on perennial bioenergy crops, such as Miscanthus and short rotation coppice (SRC), in response to the goal of expanding planting to 700,000 hectares by 2050, as set out in the Sixth Carbon Budget (Committee on Climate Change, 2020). However, multiple sources acknowledge potential trade-offs between expanding dedicated bioenergy crops in the UK, food production and the extent of high biodiversity and/or carbon-rich habitats such as semi-natural grasslands (Hodgson et al., 2024; Committee on Climate Change, 2020; French, 2019; Milner et al., 2016).

Several European studies exploring the potential for biomass cropping from permanent grasslands, outside the requirements of livestock husbandry, conclude they could serve as a significant resource for bioenergy and biofuel production, while also providing biodiversity and other ecosystem service benefits (Ketzer et al., 2017; Smyth et al., 2009, 2011; Charlton et al., 2009).

However, the use of biomass from semi-natural grasslands presents challenges for both anaerobic digestion and combustion techniques. The material can include plants with high lignin content and, particularly if cut late in the growing season, high fibre content, making it difficult to break down during anaerobic digestion and resulting in low energy yields (Boob et al., 2019; Sawatdeenarunat et al., 2015; Bühle et al., 2012). Additionally, semi-natural grasslands often have high mineral contents, particularly potassium, chlorine, and sulphur which can cause issues during combustion, including slagging, fouling, and corrosion in boilers (Joseph et al., 2018; Bühle et al., 2012; Frank Hensgen et al., 2012). High nitrogen content in the biomass can also lead to increased NOx emissions during combustion (Joseph et al., 2018; F. Hensgen et al., 2012).

The integrated generation of biogas and solid fuel from biomass (IFBB) technique (Wachendorf et al., 2009) was developed to overcome these issues. It has been demonstrated to be effective on a commercial scale in Germany (Joseph et al., 2018). The technology has recently been successfully implemented at farm-scale demonstration sites in the UK through the COMBINE[4] and REDIRECT[5] projects but is not yet available on a commercial scale.

With respect to the bioenergy potential of specific UK grassland types and the influence of different management regimes, empirical data are limited. The National Non-Food Crops Centre (NNFCC)[6] lists 630 operational anaerobic digestion plants in the UK, with 115 listing 'grass' or 'grass silage' as a feedstock. However, no definition or descriptions are provided for these terms with respect to the grasslands of origin in potential biogas yield figures[7].

Various European studies have investigated the energy yield of different semi-natural vegetation types (Van Meerbeek et al., 2014, 2015). However, the majority use broad vegetation classifications that are difficult to align with UK grassland types.

Methane yields per hectare for Annex I grassland types, sampled in Germany but broadly corresponding to four UK grassland communities, are reported by (Hensgen et al., 2014; Fig 1). This study indicated

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[4] http://www.combine-nwe.eu/index.php%3Fid=9.html

[5] http://cwmharry.org.uk/redirect/

[6] https://www.biogas-info.co.uk/resources/biogas-map/

[7] https://www.biogas-info.co.uk/about/feedstocks/

that using the IFBB technique with biomass collected from two cuts in mid-July and mid-October, Lowland hay meadow (MG4) and Mountain hay meadow (MG3) were the highest yielding, followed by Molinia meadow (M24 & M26) and then Species-rich Nardus grasslands (CG10 & CG11).

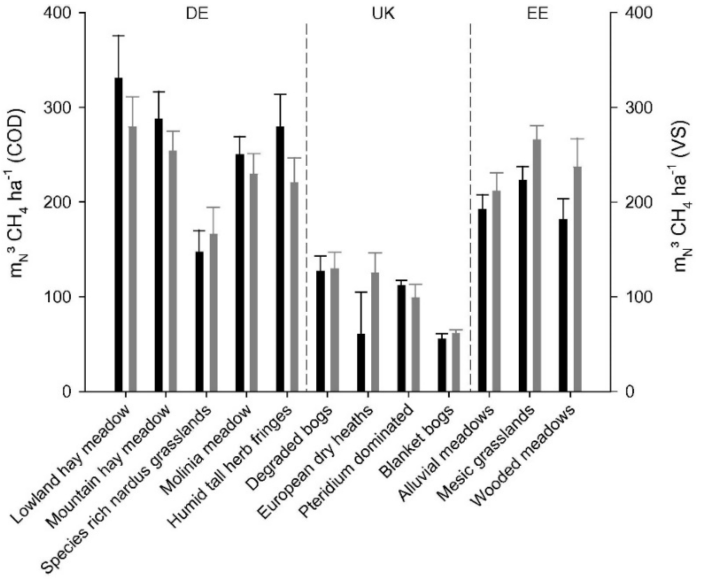

Fig 1: Area related methane yield of press fluids from IFBB (arithmetic means with standard error of means, two-year data) for twelve European semi-natural grassland habitat types (Hensgen et al., 2014)

In the UK context, a comparison of the bioenergy potential of biomass from grasslands managed for nature conservation in Oxfordshire (including SSSIs, restored species-rich grasslands, and improved grasslands under various agri-environment scheme options) showed the grasslands sampled had up to 50% less lignin content than other bioenergy crops, with an estimated yield of 527.2 ± 39 m^3 of biogas per tonne of dry matter (DM), up to 160% more than cereals or crop waste, and only slightly less than Miscanthus, which yielded 653.9 ± 84.2 m^3 / t DM (French, 2019). However, biogas yields were calculated using the Buswell Formula (Symons and Buswell, 1933) which is known to overestimate Biochemical Methane Potential (BMP) (Lesteur et al., 2010), and therefore these values may be somewhat inflated.

The same study (French, 2019) reported the highest estimated biogas yields from restored grasslands when comparing unimproved, restored, and improved grasslands. Whilst some studies have noted a positive correlation between species-richness and estimated biogas yield (French, 2019; Tilman et al., 2006), overall lignocellulosic composition has been shown to be key. This suggests that factors such as species composition and the balance of plant functional groups in grasslands have a greater influence on their potential bioenergy yield (Meserszmit et al., 2024; French, 2019). Management also effects bioenergy yields, as both dry matter yield and chemical composition change throughout the growing season, with some studies reporting an association between later harvesting dates and higher lignin content and lower methane yields (Van Meerbeek et al., 2015). Further research on specific UK grassland types and management regimes is required to understand the interplay of these factors on their bioenergy potential.

Grassland biodiversity and ecosystem services

The UKNEA (Bullock et al., 2011) highlights the biodiversity value of UK semi-natural grasslands in providing habitat for a wide range of rare and priority species, linking this to the cultural services of conservation and heritage. However, grassland biodiversity, i.e. plant species-richness, is primarily discussed as a positive driver of most individual ecosystem services and potentially for maximising multifunctionality of services.

Bengtsson et al (2019) report limited evidence, primarily from experiments, supporting a strong direct link between high small-scale biodiversity of semi-natural grasslands and individual ecosystem services, noting that vegetation structure and functional traits of plant species are likely to be more important for individual services. A systematic review of studies in field settings also concluded that abiotic factors and functional groups were stronger drivers for grassland ecosystem functions than biodiversity *per se* (van der Plas, 2019).

However, as different processes rely on distinct functional groups, more species may be required to support multiple ecosystem services (Bengtsson et al., 2019; Hector and Bagchi, 2007). This has been demonstrated experimentally for plant species richness (Meyer et al., 2018; Isbell et al., 2011). In a study of 150 grasslands, Soliveres et al. (2016) found that high species richness across multiple trophic groups had stronger positive effects on ecosystem services than single-group richness, particularly enhancing multifunctionality, and driving ecosystem functioning as strongly as abiotic conditions and land-use intensity.

In addition, plant species diversity is directly linked to the development of good soil structure, which is key for multiple ecosystem services including carbon sequestration, via increasing aggregate stability (Gould et al., 2016) and stimulating the development of pores (Kravchenko et al., 2019).

A literature review addressing the value of grassland for nature conservation was undertaken by the British Ecological Society on behalf of the Plantlife charity with a view to defining a category of ancient grasslands of particularly high biodiversity interest. A report summarising this review and suggesting how this group of grasslands may be defined and designated was written in 2024. It is not yet in the public domain and therefore the relevant part of that report, summarising the evidence in the literature, is reproduced here, with the kind permission of Plantlife, as Appendix 1.

Other information

In terms of bias, the authors of this review work on research projects that promote floodplain meadows. Other contributors may promote other grassland types or services or have financial interests in delivering them.

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Appendix 1

An extract from a report prepared by the British Ecologcial Society for Plantlife, which reviews the literature pertaining to the characteristics of 'ancient' grasslands relevant to potential definitions and methods of identification.

Vascular plants

Chronosequencing studies have identified species restricted to old calcareous grasslands (>100 years) (e.g., Pornon and Andalo, 2023; Wagner et al., 2019; Redhead et al., 2014; Gibson and Brown, 1991) and 'indicators of long continuity' have been suggested for the MG5 NVC community (Natural England, 2013) and more generally for lowland calcareous grasslands (Table 13 in Porley, 1988) (see Annex 1). Natural England are due to publish work on chronosequencing of grasslands in 2024 (Katey Stephen, pers. comm). However, overall cover or summed frequency of indicator species has been shown to be a more reliable indicator of grassland age than species number (Wagner et al., 2019).

Several studies indicate a positive correlation between grassland age and occurrence of stress-tolerant species, as opposed to ruderal or competitive species (e.g., Nerlekar and Veldman, 2020; Wagner et al., 2019; Redhead et al., 2014b; Gibson and Brown, 1991b). This concept has been used to assess restoration success in floodplain meadows (Rothero et al., 2020). In addition, unpublished surveys have shown that extreme stress-tolerators such as Pulsatilla vulgaris have yet to recolonise chalk grassland ploughed briefly in the 1950s even though they occur only a few metres away on the adjacent Knocking Hoe National Nature Reserve in Bedfordshire (Kevin Walker, pers. obs.).

High α diversity, community evenness, and low β diversity are associated with older semi- natural grasslands (Pornon and Andalo, 2023) and several studies suggest that whilst species richness can recover relatively quickly, restored grasslands may take decades (>60 years) to recover to near natural species composition and functional traits (e.g., Buisson et al., 2022; Nerlekar and Veldman, 2020; Redhead et al., 2014; Woodcock et al., 2011; Fagan et al., 2008; Hirst et al., 2005). However, in cases of short duration disturbance, such as past peaks in agricultural production where grasslands were ploughed and later reverted to pasture (e.g. 1790-1815, 1940-1950), these differences can be difficult to detect, and other methods may be needed to date grasslands accurately (Wagner et al., 2019). Seed banks can reflect land use history over long periods, with the seeds of some ruderal species (associated with cultivation) persisting for over 150 years (Karlík & Poschlod, 2014).

Grassland fungi

CHEGD fungi (waxcaps and allies)18 are typically associated with undisturbed, low nutrient status grasslands (Griffith et al., 2004; Feehan and McHugh, 1992). IUCN assess that several of the species found in the UK face high extinction threat (Annex1). However, given the biomass of these fungi is predominantly underground, the high weather-dependence of fungal fruiting (Griffith et al., 2013), inhibition of fruiting by long swards (Griffith et al., 2013) and lack of suitably experienced surveyors make effective assessment of diversity via fruit body surveys challenging. There are long-lived perennials, often not fruiting every year so potentially requiring several years of surveying (Newton et al., 2003).

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[1] CHEGD grasslands are those that are rich in grassland fungi including the following groups: C (Clavariaceae [fairy clubs]): *Camarophyllopsis*, *Clavaria*, *Clavulinopsis*, *Hodophilus Lamelloclavaria*, *Ramariopsis*; H (Hygrophoraceae): *Cuphophyllus*, *Gliophorus*, *Gloioxanthomyces*, *Hygrocybe*, *Neohygrocybe*, *Porpolomopsis*; E (Entolomataceae): *Clitopilus*, *Entoloma*; G (Geoglossoid fungi): *Geoglossum*, *Glutinoglossum*, *Microglossum*, *Sabuloglossum*, *Trichoglossum*; D (Dermoloma etc): *Dermoloma*, *Pseudotricholoma*.

Vascular plant richness is often a poor proxy for CHEGD diversity (Holden, 2013; Öster, 2008), in part because many pastures where these fungi fruit prolifically have reduced diversity due to heavy grazing that does not damage the underground fungal structures. Some larger, very long-lived CHEGD fungi are useful indicators of grassland age e.g.

Hygrocybe punicea, now dominant in the control plots of the long-running Park Grass field site at Rothamsted Research (last ploughed in ca. 1815). However, this obvious species was absent when the site was surveyed in 1874 suggesting that it takes >60 years, and possibly up to 150 years, to reach maturity (Gareth Griffith, pers. obs*)*.

Recent use of soil eDNA to analyse fungal populations via metabarcoding (Detheridge & Griffith, 2021), which assesses mycelial abundance, has revealed that CHEGD are the dominant soil fungi in many undisturbed grasslands (often >50% total fungal biomass). This approach avoids the problems caused by the vagaries of fungal fruiting and provides accurate identification via the curated UNITE ITS2 database (https://unite.ut.ee/; Koljalg et al., 2013). Such analyses can be conducted at any time of year and can also identify the plants present via DNA present in roots, seeds or litter (Clasen et al., 2022). However, from a species conservation perspective where the presence/abundance of mature individuals is a key factor, it is not possible to determine from eDNA whether any mycelia detected are fertile.

Use of soil eDNA analysis has hitherto mainly focused on acidic soils in the upland fringes, driven by the need for urgent fungal biodiversity assessment, for example for grasslands targeted for tree planting where ecological surveys have generally failed to consider fungal conservation value. However, it would be useful to deploy this approach in different British grassland types to determine whether CHEGD fungi are a consistent feature of undisturbed grasslands more broadly. The fact that outside Europe CHEGD fruit bodies are more commonly found in (non-ectomycorrhizal) woodland habitats suggests that they can inhabit a broad range of soil types (Halbwachs et al., 2018).

Invertebrates

Invertebrates such as coleoptera can indicate environmental change over time (Woodcock et al., 2005; Brown and Hyman, 1986) and could potentially be used as a measure of grassland age. For example, beetle assemblages in restored floodplain meadows were shown to differ from those of long-established grasslands (Woodcock et al., 2006, 2008) and restoration of plant communities does not necessarily correspond with recolonisation of invertebrates, which may be due to factors including dispersal mechanisms, isolation of restoration sites, or microclimate (Woodcock et al., 2010, 2012; Knop et al., 2011). Ant hills have also been successfully used to predict age in calcareous grasslands (King, 1981, King & Timothy King, 2021), although this may not be a useful approach in mown grasslands.

As with fungi, lack of taxonomic expertise has hindered full appreciation of the biodiversity of grassland invertebrates. Many of these species spend most of their lives in soil (and the least known e.g. collembolans, all their lives), eDNA approaches above can be used for invertebrate taxa but are still being tested for some groups (Kirse et al, 2021). A further parallel with fungi is that whilst long-term overgrazing damages plant biodiversity, the presence of diverse soil fungal/insect populations is indicative of a healthy, undisturbed soil ecosystem, and could potentially be used to identify sites with high restoration potential.

Soil properties

Correlation between low soil phosphorus and mineral nitrogen and older grasslands is well established in the literature (Löfgren et al., 2020; Karlík and Poschlod, 2019; Schmid et al., 2017; Jaunatre et al., 2016; Fagan et al., 2008). High fungal biomass and fungal to bacterial biomass ratios have also been shown to be associated with unimproved/low input grasslands across a representative range of grassland types and regions in England (e.g., Smith et al., 2008; Bardgett et al., 2007; Bardgett and McAlister, 1999). Ellenberg N values have been shown to correlate with both nutrient levels and grassland age and could be useful as a proxy without the requirement for soil sampling (Löfgren et al., 2020; Schmid et al., 2017; Walker et al., 2004).

Undisturbed grasslands accumulate organic matter which can be dated via radiocarbon analysis, with the thermonuclear ‘bomb radiocarbon’ peak of the 1960s permitting highdating resolution for atmospheric CO_2 fixed into soils by plants over the past ca. 70 years (Leifeld et al., 2009).

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Annex 1: Potential vascular plant and fungal indicator species

Table 1: Previously noted indicator species for ancient/old lowland calcareous grassland

Species	**Source/age class**		
	Wagner et al. (2019): (>200 years)	Redhead et al., (2014): (≈136 years)	Gibson & Brown (1991) (>100 years)
Agrostis capillaris		+	
Anthoxanthum odoratum		+	
Anthyllis vulneraria		+	
Arenaria serpyllifolia		+	+
Asperula cynanchica	+	+	
Avenula pubescens	+		
Betonica officinalis		+	
Brachypodium pinnatum		+	
Briza media		+	
Bromopsis erecta			+
Bryonia dioica		+	
Campanula glomerata		+	
Campanula rotundifolia	+	+	
Carduus nutans		+	
Carex caryophyllea	+	+	
Carex flacca	+	+	
Carex humilis		+	
Carex spicata		+	
Cirsium acaule		+	
Cirsium palustre		+	

Clinopodium acinos		+	
Clinopodium vulgare		+	
Deschampsia cespitosa		+	
Festuca ovina		+	+
Filipendula vulgaris	+	+	+
Galeopsis tetrahit		+	
Galium verum			+
Genista tinctoria		+	
Gentianella amarella		+	
Gymnadenia conopsea		+	
Helianthemum nummularium	+	+	+
Helictochloa pratensis		+	
Hippocrepis comosa		+	+
Hypericum hirsutum		+	
Koeleria macrantha	+	+	+
Linum catharticum		+	
Myosotis arvensis		+	
Origanum vulgare		+	
Picris hieracioides		+	
Pilosella officinarum		+	
Poa angustifolia		+	
Polygala vulgaris		+	
Poterium sanguisorba	+	+	
Primula veris		+	
Reseda lutea		+	
Rhamnus cathartica		+	

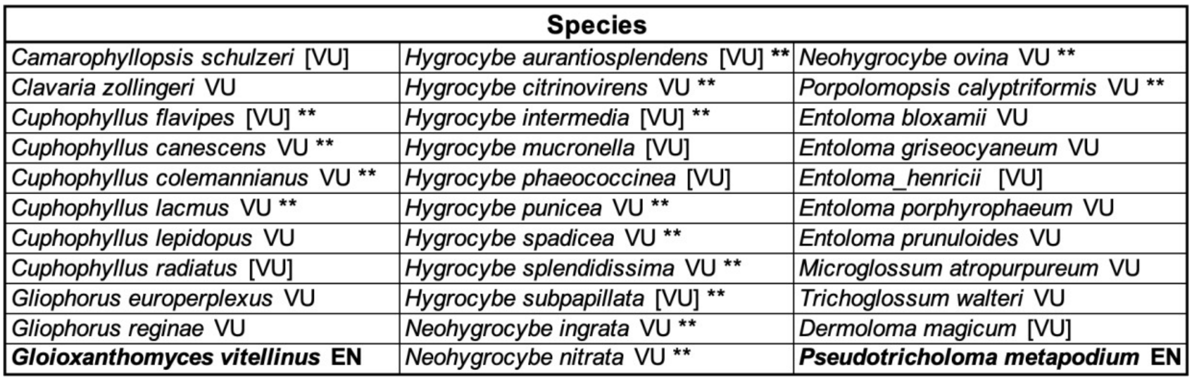
Rhinanthus minor		+	
Rubus fruticosus agg.		+	
Rumex acetosa		+	
Scabiosa columbaria		+	
Serratula tinctoria		+	
Sinapis arvensis		+	
Sonchus asper		+	
Stellaria graminea		+	
Succisa pratensis		+	
Thesium humifusum		+	
Thymus drucei		+	
Ulex europaeus		+	
Valeriana officinalis		+	
Verbascum thapsus		+	
Veronica arvensis		+	
Veronica officinalis			+
Viola hirta		+	

Table 2: Previously noted indicators of 'long continuity' for MG5 grassland (Natural England, 2013)

Species		
Anemone nemorosa	*Genista tinctoria*	*Serratula tinctoria*
Betonica officinalis	*Lathyrus linifolius*	*Silaum silaus*
Carex caryophyllea	*Pimpinella saxifraga*	*Succisa pratensis*
*Conopodium majus**	*Saxifraga granulata*	

*Can tolerate some fertiliser addition.

Table 3: Grassland fungi assessed as Globally Vulnerable (VU) or Endangered (EN) by IUCN (www.iucnredlist.org/; www.redlist.info/)



Species		
Camarophyllopsis schulzeri [VU]	*Hygrocybe aurantiosplendens* [VU] **	*Neohygrocybe ovina* VU **
Clavaria zollingeri VU	*Hygrocybe citrinovirens* VU **	*Porpolomopsis calyptriformis* VU **
Cuphophyllus flavipes [VU] **	*Hygrocybe intermedia* [VU] **	*Entoloma bloxamii* VU
Cuphophyllus canescens VU **	*Hygrocybe mucronella* [VU]	*Entoloma griseocyaneum* VU
Cuphophyllus colemannianus VU **	*Hygrocybe phaeococcinea* [VU]	*Entoloma_henricii* [VU]
Cuphophyllus lacmus VU **	*Hygrocybe punicea* VU **	*Entoloma porphyrophaeum* VU
Cuphophyllus lepidopus VU	*Hygrocybe spadicea* VU **	*Entoloma prunuloides* VU
Cuphophyllus radiatus [VU]	*Hygrocybe splendidissima* VU **	*Microglossum atropurpureum* VU
Gliophorus europerplexus VU	*Hygrocybe subpapillata* [VU] **	*Trichoglossum walteri* VU
Gliophorus reginae VU	*Neohygrocybe ingrata* VU **	*Dermoloma magicum* [VU]
***Gloioxanthomyces vitellinus* EN**	*Neohygrocybe nitrata* VU **	***Pseudotricholoma metapodium* EN**

[VU] indicates the species that are assessed as VU, but not yet formally published by IUCN. ** indicates those species considered by Bosanquet et al. (2018) to be high diversity indicators