OXFORD MARTIN RESTATEMENT 8

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Grassland management, grazing livestock and soil carbon storage

Oxford Martin Restatement 8:

A restatement of the natural science evidence base concerning grassland management, grazing livestock and soil carbon storage.

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This paper was published in January 2024 in the Proceedings of the Royal Society B. It deals with grassland management, grazing livestock and soil carbon storage potential.

Approximately a third of all annual greenhouse gas emissions globally are directly or indirectly associated with the food system, and over a half of these are linked to livestock production. In temperate oceanic regions, such as the UK, most meat and dairy is produced in extensive systems based on pasture. There is much interest in the extent to which such grassland may be able to sequester and store more carbon to partially or completely mitigate other greenhouse gas emissions in the system. However, answering this question is difficult due to context-specificity and a complex and sometimes inconsistent evidence base. This Restatement describes a project that set out to summarize the natural science evidence base relevant to grassland management, grazing livestock and soil carbon storage potential in as policy-neutral terms as possible. A series of evidence statements are listed and categorized according to the nature of the underlying information, and an annotated bibliography is provided in the electronic supplementary material.

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Author for correspondence:
H. Charles J. Godfray
e-mail: charles.godfray@oxfordmartin.ox.ac.uk

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A restatement of the natural science evidence base concerning grassland management, grazing livestock and soil carbon storage

Matthew W. Jordon[1], Jean-Charles Buffet[1], Jennifer A. J. Dungait[2,3], Marcelo V. Galdos[4], Tara Garnett[5], Michael R. F. Lee[6], John Lynch[7], Elin Röös[8], Timothy D. Searchinger[9], Pete Smith[10] and H. Charles J. Godfray[1,11]

[1]Oxford Martin School, University of Oxford, 34 Broad Street, Oxford OX1 3BD, UK
[2]School of Geography, College of Life and Environmental Sciences, University of Exeter, Stoker Road, Exeter EX4 4PY, UK
[3]Carbon Management Centre, Scotland's Rural College (SRUC), Peter Wilson Building, The King's Buildings, West Mains Road, Edinburgh EH9 3JG, UK
[4]Rothamsted Research, West Common, Harpenden, Hertfordshire AL5 2JQ, UK
[5]TABLE, Environment Change Institute, University of Oxford, 3 South Parks Road, Oxford OX1 3QY, UK
[6]Harper Adams University, Newport TF10 8NB, UK
[7]Nature-Based Solutions Initiative, Department of Biology, University of Oxford, 11a Mansfield Road, OX1 3SZ, UK
[8]Swedish University of Agriculture Sciences, Ulls hus, Almas allé 8, Uppsala, SE-750 07, Sweden
[9]School of Public and International Affairs, Princeton University, 318 Robertson Hall, NJ 08544-1013, USA
[10]Institute of Biological and Environmental Sciences, University of Aberdeen, 23 St Machar Drive, Aberdeen AB24 3UU, UK
[11]Department of Biology, University of Oxford, 11a Mansfield Road, Oxford OX1 3SZ, UK

J-CB, 0000-0003-2062-9546; PS, 0000-0002-3784-1124

Approximately a third of all annual greenhouse gas emissions globally are directly or indirectly associated with the food system, and over a half of these are linked to livestock production. In temperate oceanic regions, such as the UK, most meat and dairy is produced in extensive systems based on pasture. There is much interest in the extent to which such grassland may be able to sequester and store more carbon to partially or completely mitigate other greenhouse gas emissions in the system. However, answering this question is difficult due to context-specificity and a complex and sometimes inconsistent evidence base. This paper describes a project that set out to summarize the natural science evidence base relevant to grassland management, grazing livestock and soil carbon storage potential in as policy-neutral terms as possible. It is based on expert appraisal of a systematically assembled evidence base, followed by a wide stakeholders engagement. A series of evidence statements (in the appendix of this paper) are listed and categorized according to the nature of the underlying information, and an annotated bibliography is provided in the electronic supplementary material.

1. Introduction

Approximately 50% of the UK's land area is managed as pasture or grassland used for ruminant livestock production [1]. The equivalent global figure is about 25% [2]. The extent of such grassland, and how it is managed, is very significant for national and global greenhouse gas fluxes, and hence climate change. Pasture and grasslands contain substantial stocks of carbon, though can also act as carbon sources, particularly if overgrazed [2]. There are also direct emissions of different types of greenhouse gases from the livestock that use these areas. Understanding the dynamics of carbon cycling and greenhouse gas emissions in pastures is critical in limiting climate change [3].

The extent to which climate change may be mitigated by producing and consuming fewer livestock products (meat and dairy) is highly contested. Some disagreements reflect the presence of vested interests or participants with strong ideological standpoints, but there is also considerable uncertainty around the natural science evidence base of relevance to policymakers. An important question is the extent to which carbon can be sequestered in the soil of grasslands grazed by livestock. This is difficult to answer for several reasons. First, carbon sequestration and stocks are influenced by the chemical and physical make-up of the soil matrix, by the local environment and by how the land is managed [4]. The rate of carbon sequestration may also vary greatly over time. For example, rates can initially be very high on degraded soils that are depleted in soil carbon, but then decline as carbon builds up over time and ultimately reach an equilibrium. Second, in calculating the net benefits of pasture as carbon stores, account needs to be taken of the emissions associated with the livestock that feed on it, as well as further emissions in the meat and dairy food processing chains. This calculation is complicated by the fact that unlike many other sources of greenhouse gases, livestock production is associated with not only carbon dioxide emissions but also considerable amounts of methane and nitrous oxide [5,6]. In formulating policies, it is important to take into account the different dynamics of these gases in the atmosphere. Third, the counterfactual of how much carbon could be stored by the land if it was managed in a different way, for example if it was forested, is relevant to policy; as is the question of whether any advantage of reducing meat or dairy production in one area might be reduced or reversed by increased production elsewhere which may be more or less carbon emission intensive. The wide range of estimates in the peer-reviewed scientific literature of the amount of carbon stored in pasture soil reflect these complexities. Outside the scientific mainstream there are some remarkable claims about the extent to which anthropogenic carbon emissions may be mitigated by sequestration in pastures and rangeland that need to be subjected to critical scrutiny.

Here, we attempt to summarize the science evidence base concerning carbon storage in pasture land used for livestock production, in a policy-neutral manner that is accessible to policymakers who have some background in this area but are not subject specialists. The format of the review is a 'Restatement' where the summary is given as numbered paragraphs in an appendix. Because of the size and complexity of the topic, we focus our attention on carbon storage in pasture and grassland used for livestock production in the UK, though we hope the review will also be useful in other countries.

2. Material and methods

The relevant literature on grassland management, grazing livestock and soil carbon storage was reviewed with particular focus on studies in the UK and a first draft evidence summary produced by a subset of the authors. The statements and their assessments were subsequently debated via correspondence until a consensus was achieved. The near-final draft was then sent to a wide set of stakeholders (see Acknowledgements) for comments. We use the following restricted terms to describe the evidence, indicated by abbreviated codes, which are similar to those used in previous Restatements [7]. Codes at the end of paragraphs after full-stops indicate that they apply to the whole previous section; codes preceding full-stops or within a sentence apply to that sentence or clause alone.

[B] Uncontentious background material.

[S] Strongly supported by a substantial evidence base where further information is unlikely to change the current consensus.

[L] Less strongly supported by the existing evidence base and where further information might alter the current consensus.

[E] Expert opinion based on information from related sources or general principles from different fields of science.

3. Results

The summary of the natural science evidence base relevant to grassland management, grazing livestock and soil carbon storage policymaking in the UK is given in the appendix, with an extensive annotated bibliography provided as electronic supplementary material.

4. Discussion

We comment here on several general themes that emerged from our attempt to summarize a disparate and sometimes contradictory literature.

First, part of the disagreement about the extent to which carbon can be sequestered in grassland is due to different methodologies in measuring carbon in the soil. Efforts to provide standards and platforms to allow more meaningful comparisons are valuable and important. There are also differences in methodologies to assess the totality of emissions from a grassland production system, and to make valid comparisons with other more intensive ways to produce meat and dairy. Very different results are obtained if emissions are compared per hectare of land versus per kg of product. It is also not straightforward to integrate factors such as point-emissions from the manure produced in intensive systems. Finally, there are no agreed ways to account for the indirect effects of different production systems such as displaced or replaced production, even though ignoring these can greatly distort the climate impact of different policies. Progress in developing better, standardized carbon accounting tools, as well as accounting for indirect effects, would greatly assist policymakers.

Second, developing policy around carbon sequestration in grasslands inevitably has an important temporal component. Carbon build up in soils can be very rapid when heavily degraded arable soils are laid to grass but the rate of accumulation eventually approaches zero. Some of the most dramatic estimates of the carbon sequestration potential of grasslands in the advocacy space take such initial accumulation figures and assume they can be applied to all pastures and in perpetuity. Measures of total emissions from grassland have to grapple with the different dynamics of carbon dioxide and methane in the atmosphere. Carbon dioxide accumulates in the atmosphere causing progressive warming while methane, produced by ruminant enteric fermentation or methanogens in waterlogged organic soils, relatively quickly reaches an equilibrium concentration so that constant rates of emissions cause a fairly stable warming. Reducing emissions of any greenhouse gas can contribute to climate change mitigation, but the evidence base and the broader policy context must together determine which particular policies to consider and/or prioritize. For example, a broader policy focus might be the urgent need to prevent future warming in the next few decades to avoid potential Earth-system tipping points and give time for technological advances to help decarbonize other sectors. In that case, policymakers might place a high premium on measures to get carbon dioxide out of the atmosphere now (carbon capture in heavily degraded soils) and on reducing methane emissions (which acts to cool the environment). Landowners and farmers who make these land use changes or reduce livestock emissions are producing 'public goods' (benefits to society), an argument for public funding. A further complication is the possibility that limiting meat or dairy production in one place is compensated for by increased production elsewhere which may be less (or more) carbon efficient.

Third, though the evidence base is inevitably not as comprehensive as desirable, it does suggest some obvious no-regret policies. A variety of management practices on pasture, discussed in this Restatement, have been shown to have both environmental and economic benefits to the landowner. Many farmers and landowners are implementing them but more would be encouraged to if provided with better advice and guidance. A different type of policy where the evidence base is clear, concerns the importance of maintaining peatlands as carbon stores and ensuring that if they are used for grazing, livestock densities are kept below a level at which peat damage causes greenhouse gas emissions.

All high-income countries, to differing degrees, subsidize their agricultural and land sectors, in large part because their labour costs are higher than in lower-income countries. Subsidies may be explicit as in the area-related cash-transfers in the European Union's Common Agricultural Policy, or more indirect as in the United States' Farm Bills' provisions which, for example, provides low-cost insurance against yield losses. Some will argue that it is economically inefficient to support a particular sector in this way, but the political reality is that these transfers will continue. Accepting this, there is the opportunity for the state to get more out of its investment. The challenge is to design the means to incentivize the provision of public goods in a way that is transparent, easily implementable, avoids major transaction costs and takes into account indirect effects. To do this successfully, policymakers need to be able to access summaries of the evidence base that are policy neutral in the sense of not being designed to support a particular advocacy position. A Restatement seeks to summarize the complex literature in an area in a way that is useful to policymakers who have knowledge of the issue but are not deep subject specialists.

Data accessibility. The data are provided in the electronic supplementary material [8].

Declaration of AI use. We have not used AI-assisted technologies in creating this article.

Authors' contributions. M.W.J.: data curation, investigation, methodology, writing—original draft, writing—review and editing; J.-C.B.: data curation, writing—review and editing; J.A.J.D.: investigation, writing—review and editing; M.V.G.: investigation, writing—review and editing; T.G.: investigation, methodology, writing—review and editing; M.R.F.L.: investigation, writing—review and editing; J.L.: investigation, writing—review and editing; E.R.: investigation, writing—review and editing; T.D.S.: investigation, writing—review and editing; P.S.: investigation, writing—review and editing; H.C.J.G.: conceptualization, investigation, methodology, writing—original draft, writing—review and editing.

All authors gave final approval for publication and agreed to be held accountable for the work performed therein.

Conflict of interest declaration. We declare we have no competing interests.

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Appendix A

(a) Aims and scope

(1) Substantial carbon is stored in pasture systems managed for livestock production. An important policy question is whether changes in land use and management practices can affect carbon sequestration in pastures and potentially contribute to climate change mitigation. Analyses of these issues have to consider the complete greenhouse gas budget of livestock production as well as the counterfactuals of using the land for other purposes.

(2) This 'Restatement' aims to summarize the science evidence base relevant to the development of policy around carbon sequestration in pastures used for livestock production. There is a focus on evidence of greatest relevance to the UK, where the climate, topography and soil types are particularly suited to pastoral agriculture.

(3) This Restatement is structured as follows. The greenhouse gas emissions from livestock production are described and summarized (section (b)) followed by a description of soil carbon dynamics (c). Estimates of carbon stocks in pastures and what they might be under alternative land uses is then summarized (d) with the next two sections examining the evidence for the influence of grazing management (e) and other management practices (f) on soil carbon. The last two sections detail indirect effects of pasture management decisions (g) and other considerations (h) of which policymakers should be aware.

(b) Livestock emissions

(4) The global food system is responsible for approximately 34% of all greenhouse emissions annually. Direct emissions from food production, land use change associated with agriculture and supply chain activities, each account for approximately a third of this total. Globally, meat and dairy production are responsible for around 57% of all food system emissions. [S]

(a) In addition to carbon dioxide (CO_2) emissions, livestock production leads to the emission of methane (CH_4), a potent greenhouse gas but with a short average atmospheric residence time of about 9 years, and nitrous oxide (N_2O), which is more potent than CH_4 and has an average atmosphere residence time of approximately 115 years. CO_2 persists for a very long time in the atmosphere, so from an emissions policy perspective, this gas can be treated as cumulative (see ¶46). [B]

(b) Metrics have been developed to allow the effects on climate of emissions of different greenhouse gases to be compared and combined, typically reporting the warming effects of other greenhouse gases as 'CO_2 equivalents' (CO_2e). The most commonly used metric is the 100-year Global Warming Potential (GWP_{100}) defined as the amount of CO_2 that would result in the same marginal climate impact over a 100 year period post-emission. A criticism of GWP_{100} is that it does not well capture the different dynamics of short-lived and long-lived greenhouse gases. To address this, an alternative emission metric, GWP*, has been developed to reflect the fact that constant emissions of a short-lived gas lead to an equilibrium concentration of that gas in the atmosphere which makes a stable contribution to global warming, whereas constant emissions of a long-lived gas results in the gas accumulating in the atmosphere and having an increasing effect on temperatures. [B]

(c) Livestock emissions are highly variable across geographical regions and production systems, meaning care must be taken when applying global-average figures to specific contexts [B]. For example, beef produced in western Europe generates only one-third of the average global emissions levels (CO_2e per kg of carcass weight) [S].

(5) Livestock associated emissions in the UK mainly come from enteric fermentation, fertilizer manufacturing and application, manure storage and application, feed production, and the energy used in agriculture and the food supply chain. Globally, land conversion from natural or semi-natural vegetation to agriculture for livestock grazing or feed production is also a significant source of emissions. [B]

(a) Ruminant animals (cattle, sheep, goats) use enteric fermentation to digest grass and other coarse plant material [B]. This produces CH_4, which together with livestock manure, accounts for one-third of global anthropogenic CH_4 emissions [S].

(b) CH_4 and N_2O are emitted both when livestock excreta are deposited directly onto pastures, and when manure from indoor systems is stored and then spread onto fields as fertilizer. Emissions per unit of excreta produced are lower in pasture than in indoor systems, due to the fermentation dynamics of stored manure. Manufactured fertilizers applied to pasture and feed crops also lead to significant N_2O emissions. Some nitrogen (N) from livestock excreta and fertilizers is also lost via leaching into water and volatization into the atmosphere as ammonia (NH_3), followed by deposition onto land elsewhere, which both result in indirect N_2O emissions. The production of manufactured fertilizers is energy intensive and a source of CO_2. [B]

(c) CO_2 emissions arise from land use change, energy generation for on-farm machinery, supply chain transport, feed and food processing, and other on-farm and food-chain activities. [B]

(I) Worldwide, deforestation and conversion of other habitats to create pasture and cropland to grow animal feed are responsible for approximately 10% of global livestock-associated emissions. [S]

(6) The greenhouse gas emissions per kg of meat or dairy vary with the production system and feed type. Globally, 65% of all livestock emissions at present come from beef and dairy cattle. [S]

(a) There are approximately 9.6 million (M) cattle in the UK, with 1.5 and 1.9 M breeding females in the beef and dairy herds, respectively. The UK has 33 M sheep, of which 16 M are breeding females. Agriculture accounted for 11% of UK emissions in 2020 (reflecting food produced rather than consumed), of which 45% was due to enteric fermentation by sheep and cattle, and 12% due to livestock manure. [B]

(b) For meat produced in the UK, typical greenhouse gas emissions are substantially lower than global averages. UK-specific emissions have been estimated at: beef, 11–25; lamb, 17; pork, 6.4 and chicken, 4.6. All figures given as kg CO_2e per kg of carcass weight including bones from 'cradle' to farmgate. A litre of milk is responsible for 1.1 kg CO_2e. [S]

(I) Emissions are sometimes expressed per kg of edible protein or per total (or digestible) amino acids or micronutrient content [B]. Such metrics can be useful in comparing among substitutable food types, but selecting measures that favour particular food types has also been used for advocacy purposes [E].

(7) A number of different measures and interventions have been proposed or are being researched to reduce the direct greenhouse gas emissions from livestock. Improved animal husbandry and genetics can increase production efficiency leading to lower emissions per kg of meat or dairy produced, while livestock nutrition and feed supplements may reduce methane production. [S]

(a) CH_4 is produced from fermenting cellulose-rich material, so increasing the digestibility and starch or fructan content of forage by manipulating plant species or variety composition, or plant age at time of grazing, can reduce emissions. Cereal-based and 'concentrate' feeds also lead to lower methane emissions. [S] Although, full life-cycle assessments are required to assess the overall benefits of different strategies [E].

(b) Manipulation of rumen chemistry and microbiota can also reduce emissions.

(I) Feed additives, such as nitrates and ionophores, inhibit methanogenic bacteria or provide alternative metabolic pathways to those ending with CH_4. Feed additives can be difficult to deliver when livestock are not fed in confined systems. [L]

(II) Feeding cattle diets rich in lipids, condensed tannins or seaweeds, particularly red algae, also reduces CH_4 emissions. Condensed tannins are naturally available in some forage species such as sainfoin (*Onobrychis* sp.) and birdsfoot trefoil (*Lotus* sp.), as well as in willow trees (*Salix* sp.) which can be browsed by livestock in agroforestry systems. There is uncertainty about the overall emissions reduction potential in farm contexts, as well as possible animal welfare and health, and environmental impacts. For example, the bromoform and iodine contents of seaweed currently limit how much can be safely fed to animals, thus restricting their emissions reduction potential. [L]

(c) Genetic breeding programmes can produce animals that grow faster, have greater muscle volume or milk yield, improved fertility and lower maintenance requirements. [B]

(I) For example, the use of female-sexed semen in UK dairy cows has reduced the number of pure dairy calves required to breed replacement cows, enabling higher meat yield beef sires to be used on much of the herd, resulting in more meat produced for the same number of animals. [B]

(II) Genetic breeding for increased efficiency may have negative welfare impacts or result in animals less suited to outdoor production systems; for example, by reducing robustness and resilience to variable environmental conditions. [B]

(III) CH_4 production in ruminant animals is a heritable trait (heritability of 0.14–0.26), mediated through host genomic influences on the rumen microbiome, suggesting the potential for breeding programmes to substantially reduce CH_4 production over a small number of generations. [L]

(c) Soil carbon dynamics

(8) Carbon exists in the soil in two broad forms—organic and inorganic. Soil organic carbon (SOC) is derived from the remains of living organisms, while soil inorganic carbon (SIC) comes from carbonate rocks and minerals, such as chalk and limestone. Both components influence the global carbon cycle, but the dynamics of SOC are more rapid and more relevant to the degree to which soils can be managed to store and sequester carbon. [B]

(9) The existing level of SOC in the soil is referred to as the *stock*, and is typically measured in tonnes of carbon per hectare (t C ha^{-1}). Any increase in soil carbon stocks is referred to as *sequestration*. Soil carbon sequestration achieves a net removal of CO_2 from the atmosphere and is typically measured in tonnes of CO_2-equivalent per hectare per year (t CO_2e ha^{-1} yr^{-1}). Maintaining existing soil carbon stocks is important to avoid releasing additional CO_2 into the atmosphere, but only soil carbon sequestration removes CO_2 from the atmosphere and hence contributes to climate change mitigation. [B]

(10) CO_2 is absorbed from the atmosphere by photosynthesis in green plants. Stocks of SOC in the soil are increased by the addition of dead plant material from above-ground plant litter, growth of below-ground roots, root exudates from living plants and through the addition of any animal excreta containing partly digested plant material. Microorganisms in the soil, which make up a small (2–4%) but important living pool of SOC, use other SOC components for energy and growth, leading to CO_2 release through respiration, while also making nutrients available to plants through mineralization. [S]

(11) The flow of carbon through different fractions of SOC is measured and modelled using different approaches. One widely adopted approach measures two broad pools of soil organic matter (SOM; approximately 50% carbon by weight and so proportional to SOC) that decompose at different rates. [B]

(a) *Particulate organic matter* (POM) is relatively undecomposed material present in large particles (53–200 µm). POM can be decomposed quickly by soil microbes with some used for energy and released as CO_2, and some used for growth (increasing microbial biomass). [S]

(b) *Mineral associated organic matter* (MAOM) is strongly bound to small soil mineral particles (less than 53 µm), particularly clay minerals, and predominantly originates from plant root exudates and microbial residues. Because microbes are tiny and live on soil particles and in the small pores between them, when they die their residues can get stuck to the soil minerals. MAOM is important for soil structure because it helps to hold soil particles together. This fraction is therefore relatively persistent and less susceptible to microbial degradation. [S]

(12) The amount of SOC generally decreases with depth and is determined by how far plant carbon (as roots, root exudates or their breakdown products—POM, MAOM and dissolved organic carbon (DOC)) can move down the soil profile. Although the top soil layers contain more SOC than the subsoil, conditions in the subsoil slow down the breakdown of SOC. [B]

(a) SOC in surface soil layers can be lost through wind or water erosion as fragments of partially broken-down plant material (POM) or bound to soil particles (MAOM), or by leaching of DOC out of the soil into waterways. This can be prevented through maintenance of plant ground cover and rooting structures which stabilize soils. [B]

(13) The difference between the rates of SOC addition and removal determine whether soils sequester or lose carbon, and an equilibrium SOC level occurs when the two rates are the same. Cropland under continuous arable cropping stores relatively less SOC at equilibrium compared with permanent pasture which stores much more (figure S1). Following a change in land use or management, soil carbon stocks can take decades to centuries to reach a new equilibrium (influenced by climate—fastest in the tropics, slower in boreal regions), but the most rapid SOC changes occur in the first 20–50 years (see figure 1 and figure S1). [S]

(a) Depletion of SOC is used as an indicator of soil degradation. There is substantial potential for SOC to accumulate if management practices change, until a new higher equilibrium is reached. [S]

(b) A related concept to soil carbon equilibrium is that of 'sink saturation', which occurs in mineral soils (on which the majority of livestock grazing occurs) once all available MAOM binding sites are occupied. At sink saturation, little further sequestration is possible, even if soil carbon inputs were to increase, limiting the impact of management interventions. [S]

(I) Approximately 80% of European grassland topsoil is currently below this sink saturation point, suggesting a considerable scope to increase carbon stocks further through sequestration. [L]

(c) In wet anoxic peaty soils and peat bogs, microbial activity is reduced and soils can continue to accumulate large amounts of carbon as partially decomposed plant material, albeit very slowly. [B]

(14) Soil carbon sequestration is reversible if organic inputs to the soil (i.e. plant residues and livestock excreta) are not maintained or if the rate of microbial degradation is increased, even after the SOC has reached sink saturation. [S]

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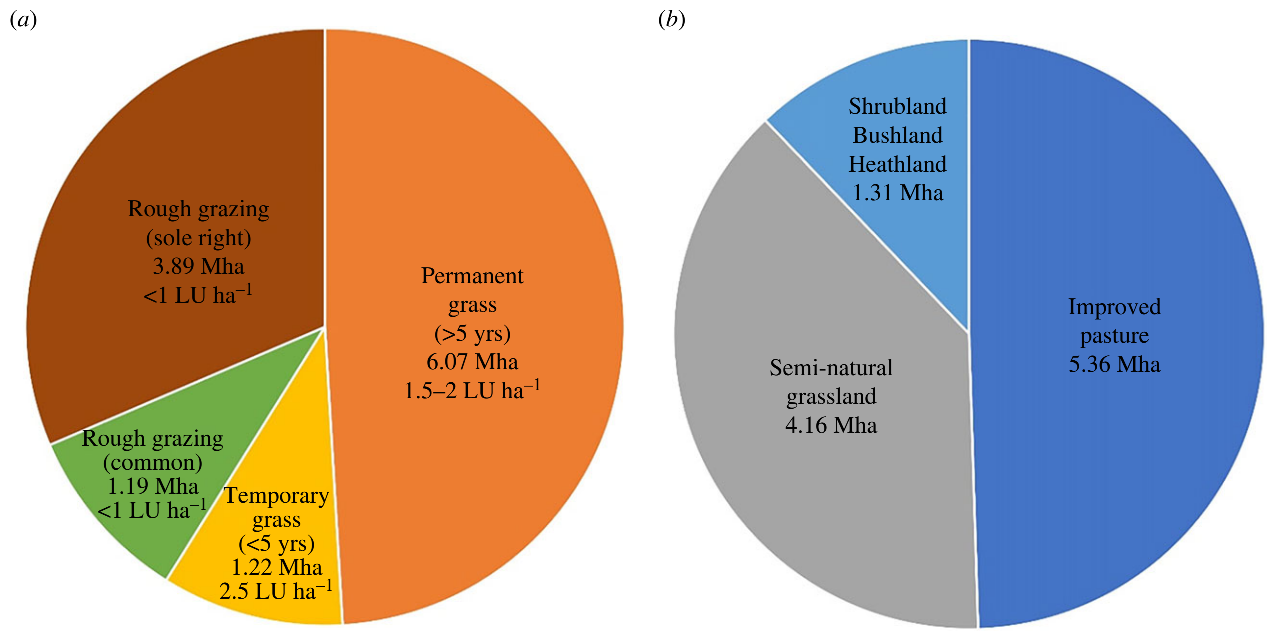

Figure 1. (*a*) UK agricultural area that is grassland or used for grazing, total 12.4 Mha (million hectares) (1). Typical stocking rates in livestock units per hectare (LU.ha−1) are also given, based on industry recommendations, where cattle over 2 years are 1 LU, and a lowland ewe with lamb is 0.12 LU. (*b*) UK land cover of habitat types that can be used for grazing (2).

(a) Soil tillage, particularly of previously uncultivated soils such as long-term pasture, results in the rapid loss of soil carbon by exposing more organic matter to microbial degradation because it stimulates microbial activity by increasing soil oxygen levels and temperatures (see ¶15(d)). [S]

(I) Where repeated tillage is replaced with no-till techniques in croplands, a redistribution of carbon within the soil rather than a substantial overall increase often occurs. Transitioning from repeated tillage to no-tillage reduces SOC losses, but where soil carbon inputs are limited (particularly in croplands where there are periods of bare soil between annual crops and limited photosynthetic activity relative to perennial systems), no-till alone is unlikely to meaningfully increase soil carbon stocks. [S]

(15) SOC formation and persistence are affected by soil characteristics which in turn are influenced by the parent material from which the soil has formed, as well as local physical conditions [B]. POM formation is mainly determined by soil temperature and moisture, although these factors separately influence plant productivity and therefore carbon inputs. N availability positively affects the formation of MAOM, while its amount is chiefly determined by soil clay content [S].

(a) Clay soils accumulate larger quantities of MAOM than do sandy and silty soils because more mineral binding sites are available. [B]

(b) Binding of SOC to minerals is inhibited in very acidic soils ($pH < 4$) [B], an effect that can be mitigated by adding lime [S].

(c) If oxygen movement in the soil is reduced, due to compaction, limiting space between soil particles or because the soil is very wet, certain microbial activity and associated SOC loss is reduced. However, if excess N and carbon are available, N_2O emissions are enhanced by anaerobically active microbes. [B]

(I) Draining wet high-organic soils and peatlands can lead to major increases in greenhouse gas emissions through higher microbial decomposition. In addition, dry peat is susceptible to shrinkage and loss by wind erosion. [B]

(d) Higher temperatures stimulate microbial activity and hence carbon loss from soils. This may be partly mitigated by higher plant productivity and hence greater potential carbon inputs into the soil, where plant growth is not water limited. [B]

(I) In large-scale geographical comparisons, desert areas contain the smallest amounts of SOC, and boreal forests contain more SOC than tropical or temperate forests, per unit area. [S]

(II) Although tropical forests contain the most carbon in plant biomass, total plant plus soil carbon stocks per hectare are largest in boreal regions. [S]

(III) There is a concern that a warming world will increase carbon losses from soils, particularly from the POM fraction. The accumulation of dead microbes from a larger soil microbial population may partially offset this in soils where sufficient moisture, carbon and nutrients are available. Increased plant productivity due to the fertilization effect of increased atmospheric CO_2 concentrations may also occur. [L]

(e) Because MAOM levels are largely determined by microbial activity, the availability of nutrients for microbes limits the formation of SOC. Despite the importance of N availability in SOC formation, applying N through manufactured fertilizer is unlikely to be a good strategy for climate mitigation because the microbial reduction of available N to N_2O can outweigh gains in soil carbon from increasing N availability. [S]

(I) Poor soil organic matter levels in croplands limits organic N available to support crop productivity, so additions of manufactured N fertilizer may be

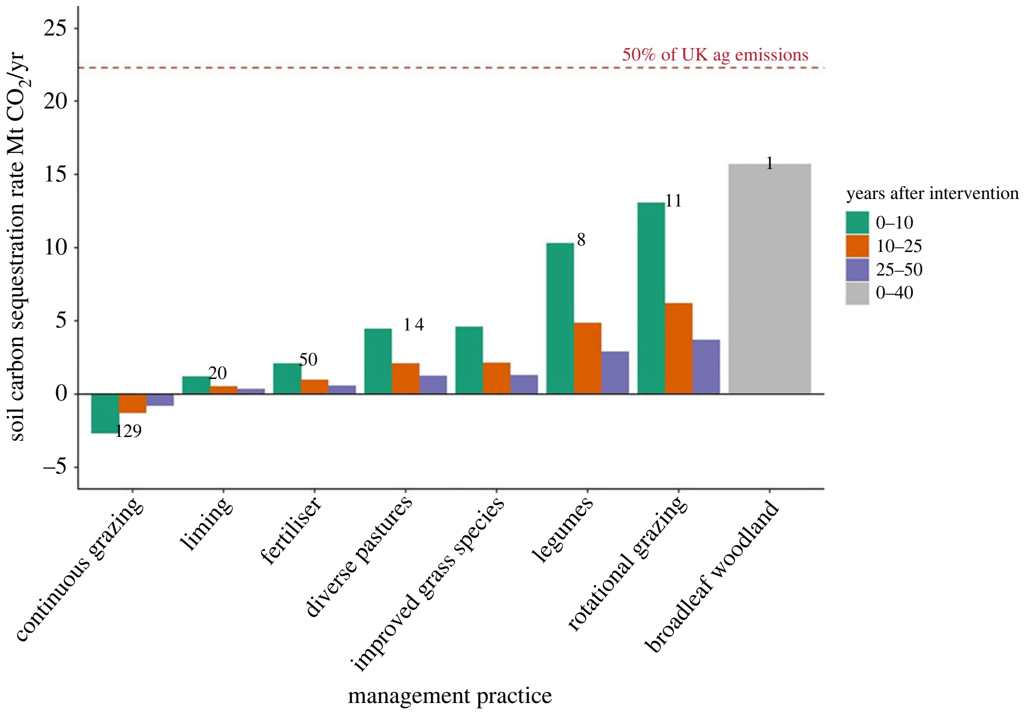

Figure 2. Carbon sequestration or emission rates from adopting grassland management practices, assumed to be adopted over 20% of UK improved pasture area from figure 1*b* (i.e. 1.07 M ha). Positive values are carbon sequestration, negative values represent emissions. Average sequestration rates presented from years 0–10, 10–25 and 25–50 following intervention adoption, demonstrating the declining rates of sequestration/emissions over time as soil carbon stocks approach a new equilibrium. The numbers above each group of bars indicate the number of primary studies underpinning that data point. The red dashed line at 22.3 Mt CO_2e.yr represents half (50%) of UK agriculture emissions in 2020 (total 44.6 Mt CO_2e (3)), included as a point of reference.

warranted regardless of the lack of a direct beneficial effect on net emissions in order to increase yields, thus maintaining food production while reducing the overall land required for agriculture. [E]

(f) The composition and diversity of soil microorganisms also influence the rates of SOC turnover and stabilization. Organic material from dead microbes is the source of around 50% of total SOC in grassland topsoil. Fungi play a greater role in SOC accumulation than bacteria, due to their close association with plant roots, greater carbon use efficiency and production of more stable types of organic matter. Fungal abundance is reduced in acidic soils (see ¶15(b)). [S]

(16) Soils can also be responsible for CH_4 and N_2O emissions. [B]

(I) Soil CH_4 emissions largely occur under anaerobic conditions. Some soil bacteria are methanotrophic and so can metabolize atmospheric CH_4 (to CO_2) but the net flux from normal aerobic soil is near zero. [S]

(II) N_2O emissions increase with the availability of excess mineral N in the soil and under anaerobic conditions. Excess soil mineral N from manufactured fertilizers, excreta and N-fixing plants such as legumes can increase N_2O emissions from agricultural soils. [B]

(d) Soil carbon in pastures and under alternative land uses

(17) Global soils contain approximately 1400 Gt of carbon to 1 m depth, as compared with approximately 480 giga tonnes (Gt) in forest biomass and litter and approximately 890 Gt tonnes as CO_2 in the atmosphere. Approximately 26% of the global land area is grassland (including rangelands, shrublands, permanent pasture and temporary pasture), which contains approximately 20% of soil carbon stocks. [S]

(18) 9.8 Gt of carbon are stored in soils in England, Wales and Scotland, 52% of which is in peatland and total greenhouse gas emissions in 2020 for England, Wales and Scotland was 0.103 Gt C-CO_2e. Permanent and temporary grasslands cover half of the UK (approximately 12.4 Mha), and accounts for 72% of agricultural land. UK grasslands can be categorized by their intensities of agricultural use, soil and vegetation types and carbon stocks (figure 2). [S]

(a) Permanent pasture, which may be semi-natural and used as 'rough grazing' or agriculturally 'improved' through the addition of forage species, lime or fertilizer, contains 1.2 Gt of carbon in the top 1 m of soil.

(b) Temporary grassland (or leys which form part of an arable rotation) are typically included in SOC estimates for cropland (total approximately 0.7 Gt for GB).

(c) Peatland covers 3 Mha, or 12% of the UK land area. Of this, 1.5 Mha is used for extensive grazing of livestock, predominantly on blanket bogs in the uplands, and a further 0.2 Mha for cropland in the lowlands. [S]

(I) The total net emissions of UK peatland are approximately 0.006 Gt C-CO_2e per year due to existing degradation. Of these emissions, 32% are from arable cropland on drained peat, and a further 27% from peatland converted to agriculturally improved pasture, predominantly in the lowlands, corresponding to 7 and 8% of UK peatland area, respectively. [S]

(II) Peatland in near-natural condition is approximately carbon neutral. Restoration of peatland currently used for livestock grazing, such as by rewetting drained areas or reduced stocking rates to allow peat-forming vegetation to recover, has substantial long-term emissions mitigation potential, although there may be short-term net emissions increases when peatland is first re-wetted. [S]

(19) Tillage of permanent pasture for reseeding or conversion to cropland leads to substantial soil carbon loss. Tillage breaks apart soil aggregates, exposing organic matter to microbial degradation, which is stimulated by associated increases in soil temperature and available oxygen. Temporary grassland and croplands also have reduced soil carbon inputs leading to smaller equilibrium SOC stocks, in part because the plants they support have less well-developed root systems and, in croplands, because plants are typically present for only part of the year and there are periods of bare soil. Converting cropland or temporary pasture to permanent pasture increases soil carbon as these processes are reversed (figure S1). [S]

(a) Reversion of cropland to grassland (grazed or otherwise) has been a major source of soil carbon sequestration in Europe between 1950 and 2010. [S]

(20) Introducing temporary grass-based leys into arable rotations leads to larger soil carbon stocks compared with continuous arable cropping but still much less than permanent pasture. [S]

(a) Leys in arable systems also have other benefits including soil stabilization and reduced erosion because of plant roots (especially if they include deep-rooted species), and increased fertility where N-fixing plants are present. [S]

(b) *Pasture cropping* involves sowing arable crops directly into pasture without tillage. This avoids significant losses of carbon, although leads to lower yields compared with no-till cropland. It is most feasible when crops that grow in the cool season are sown in pastures containing grasses that only grow in the warm season, so that competition with the crop is reduced. A practice more applicable to the UK is to *undersow* arable crops with pasture species, to avoid a period of bare ground after the crop is harvested and before a temporary ley is established, and which can maintain or even increase arable yields if an appropriate species combination is chosen. [L]

(21) Conversion of grassland to woodland typically increases total carbon stocks because, although soil stocks are typically similar, much more carbon is stored in woodland as above-ground biomass. The net change in carbon stocks is influenced by underlying soil type and depth and the form of afforestation. [B]

(a) On mineral soils with low organic content, typical of temporary pasture and cropland, woodland creation tends to increase soil carbon stocks as woodland soils contain more POM [S]. If tree planting involves major soil disturbance, it can take 5–10 years for net carbon sequestration to occur (including tree biomass) and over 20 years for the soil carbon stocks to be restored [E].

(b) On soils with high organic content such as shallow peats and organo-mineral soils, more soil carbon is lost during tree planting, so net sequestration and build-up of carbon stocks is delayed by several decades. [S]

(c) Planting trees on UK deep peat soils, now prohibited, requires drainage and results in lowering of the water table with subsequent substantial carbon losses. [S]

(d) Broadleaf trees are more likely than conifers to maintain or enhance both soil carbon stocks and biodiversity if planted on temperate pastures. [S]

(e) Planting methods that reduce soil disturbance or that allow natural regeneration to occur, including conservation grazing or rewilding approaches, reduce carbon loss from the soil. Natural approaches result in slower woodland establishment and carbon accumulation, although the intermediate scrub habitats have biodiversity benefits. [E]

(22) Silvopasture, a form of agroforestry, integrates trees into livestock pastures, either on field boundaries or as rows or clumps within the field [B]. It entails limited soil disturbance compared with woodland establishment [L] which protects below-ground carbon stocks, while later leaf deposition and above-ground biomass accumulation adds to carbon stocks [S]. Livestock grazing densities may be lower in mature silvopasture although individual animal productivity can be higher due to reduced heat- and cold stress, but the carbon consequences of displaced production needs also to be considered [L].

(a) Similarly, hedgerows can be incorporated or reintroduced into farming landscapes with minimal loss of agricultural area. Total carbon storage in hedgerows is influenced by their height and width, and whether broadleaf trees are allowed to mature at intervals, which has little additional impact on agricultural land area but positive carbon benefits. Soil carbon stocks under and in the immediate proximity to hedgerows are also increased by dead wood and litter fall. [S]

(b) It is possible to achieve greater carbon stocks *per tree* by planting them in silvopasture rather than in a woodland, due to preserved pasture soil carbon stocks plus additional tree biomass. However, the carbon stock *per hectare* is greatest in woodland. [L]

(23) The same factors that influence carbon dynamics when converting grasslands to woodland also operate when grasslands are converted to permanent bioenergy crops, such as the grass *Miscanthus* and short-rotation willow or poplar. Like commercial forestry, the above-ground biomass is harvested rather than accumulating *in situ*. The consequences for carbon budgets then depend on whether the embedded carbon re-enters the atmosphere, either rapidly via burning or more slowly via decomposition, or is captured and stored, in addition to the energy source it replaces. It is important to consider indirect effects, such as the opportunity costs of not using the land for another purpose or the alternative fuel sources displaced by biomass, when calculating overall carbon budgets. [B]

(e) Grazing management and soil carbon

(24) The presence of livestock, and how livestock are managed, have multiple effects on soil carbon dynamics and storage, as well as on plant net primary productivity (NPP), biomass removal and livestock emissions. Plant productivity in turn underpins the CO_2 flux of soils.

Livestock excreta affects soil nutrient availability, while animal trampling changes the physical properties of soil such as its density. Grazing management also influences forage quality (proportion of vegetative versus senesced plant material), which in turn impacts the level of ruminant enteric methane emissions (see ¶5(a)). [B]

(a) The scientific evidence base for the effects of different pasture management regimes on carbon stocks and flows is relatively limited. Medium- and long-term studies at field scales would be of great value to policymakers. [E]

(25) Continuous grazing frequently leads to net removal of vegetation biomass. Much of the embodied carbon is released as CO_2 through animal respiration or CH_4 through enteric fermentation and frequently results in reduced organic matter entering the soil. Meta-analyses show that continuous grazing reduces soil carbon by 3–31% compared with grazing exclusion in moist cool climates such as the UK, with greater reductions at higher grazing intensities [S]

(a) In some circumstances, continuous grazing can increase plant productivity and therefore soil carbon inputs.

(I) Moderate grazing of low-productivity semi-natural grasslands that have evolved under intermittent grazing pressure increases soil carbon stocks [S]. This is unlikely to directly apply to UK grasslands, due to their different evolutionary history and higher productivity [E].

(II) In warmer and drier climates than the UK, continuous grazing, particularly at low intensities, can increase soil carbon. [L]

(III) Overgrazing occurs when the rate of biomass removal by livestock exceeds the capacity of the forage to recover, and typically compromises plant productivity, livestock performance and soil carbon stocks [B]. Continuous grazing, also known as set stocking, is possible without overgrazing, although this requires careful pasture management with regular assessments of forage growth and adjustments of stocking rates [E].

(26) Rotational, as opposed to continuous, grazing is the practice of moving livestock around subunits of pasture to create alternating periods of grazing and no grazing [B]. When well-managed, these respites from grazing allows vegetation to regrow, which can lead to an overall rise in NPP with increased root development and exudates thus maintaining or increasing SOC. [L]

(a) One meta-analysis suggested that rotational grazing could increase SOC by 21% in the first 3 years of implementation compared with grazing exclusion (although only the top 5 cm of soil was considered in this analysis) [L]. This rate of sequestration will decrease after 5–20 years as SOC stocks approach a new equilibrium, and the overall SOC increase will depend on soil type, particularly clay content, and initial condition, as degraded soils have higher capacity for sequestration [B].

(b) A subset of rotational practices grazes high densities of livestock for short periods of time with long rest and recovery intervals; this can be called mob, tall grass, holistic planned or adaptive multi-paddock grazing. Grazing in this way can result in some plant foliage being trampled down rather than eaten, creating a layer of decomposing vegetation at the soil surface which can increase soil organic matter. [L]

(c) The increased NPP in rotationally grazed systems can allow higher stocking rates which may lead to increased direct emissions, but possible reduced indirect emissions from displaced production. Alternatively, it can allow for a reduction in inputs (fertilizer, feed), which decreases direct and indirect emissions associated with inputs. [B]

(I) Different management objectives can result in differing outcomes from adopting rotational grazing approaches. For example, a focus on livestock performance prioritizes maintaining forage in a highly palatable vegetative stage, and typically utilizes shorter rotations to prevent forage senescing. Other systems prioritize improvements in soil health and balancing the energy to fibre ratio in livestock diet for optimum ruminant functioning. In these systems, longer recovery periods without grazing are used to maximize forage accumulation, root growth and forage is frequently allowed to reach maturity before grazing. The precise soil-plant-livestock emissions balance across such variations in management regimes are highly context specific and have not been well quantified. [E]

(d) The impact of rotational grazing depends on environmental conditions (temperature, rainfall, soil type) being suitable for vegetation regrowth between grazing episodes and this may limit its benefits. [B]

(I) By contrast, adaptive and holistic grazing approaches aim to respond to changing environmental conditions. For example, areas of pasture can be removed from or added to the grazing rotation to match rapid versus slow rates of forage growth, respectively, at different points in the grazing season. However, 'stockpiling' forage biomass in this way can result in a trade-off with forage digestibility and therefore livestock performance and hence emissions per unit of meat or dairy output. [E]

(e) Although well-managed rotational grazing approaches can increase soil carbon and agricultural productivity, the extent and magnitude of possible benefits is contested. [E]

(I) Bold claims have been made by some advocates that approaches such as Holistic Planned Grazing could reverse desertification and fully mitigate anthropogenic climate change. However, the proposed mechanisms underpinning these assertions lack an evidence base [S], and the scale of the claims are implausible [E].

(II) It has been suggested that adaptive, holistic or mob grazing approaches could help form 'new' soil and there is some evidence for increases in topsoil depth. True soil creation occurs very slowly and requires bedrock weathering and mineralization, and the observed increases in topsoil depth are likely to result from: (a) increase in the fine root mat typical of grasses at the soil surface and deeper roots increasing organic inputs to the subsoil, (b) partially decomposed plant litter accumulating on the soil surface and (c) improved soil particle aggregation and soil porosity reducing soil density for an equivalent mass. It is implausible that perpetually high rates of soil carbon sequestration will occur due to the saturation point of MAOM eventually being reached. [E]

(27) Allowing livestock to graze temporary pasture in ley-arable rotations leads to a greater increase (2–20%) in SOC than if grass is cut and removed [L]. Livestock excreta also increase nutrient availability for subsequent crops and may benefit soil biodiversity, although the direct and indirect emissions associated with livestock must be considered [S]. Livestock can also be used to graze over-winter cover crops in arable rotations, although there is currently insufficient evidence to determine the impact of this on soil carbon stocks [E].

(28) In assessing the net benefits of soil carbon sequestration due to grazing, it is important to include the direct and indirect greenhouse gas emissions from the livestock involved. [B]

(a) Where soils are initially degraded, improved livestock management, such as reducing grazing intensity when overgrazing has occurred (¶26b), adopting rotational rather than continuous grazing (¶26) or integrating livestock into arable systems, can lead to an increase in soil carbon stocks particularly over the short term. Understanding the full carbon-budget implications requires considering system productivity and any indirect effects of higher or lower outputs. [E]

(b) If introducing livestock into ley-arable rotations results in an overall increase in their numbers, then the overall carbon-budget effect is likely to be negative. But if livestock numbers do not change, positive outcomes may occur, for example because: (i) the temporary pasture that is used for grazing or forage production replaces other feed sources and their associated emissions, (ii) land is released for targeted sequestration projects, (iii) inorganic fertilizer and herbicide use is reduced on cropland due to livestock manure and grazing, respectively. Such seasonal reallocation of existing livestock from pasture areas to temporarily graze cropland is gaining traction with UK practitioners, but is limited by the spatial separation of the main cropland and livestock areas in the UK. [E]

(c) In calculating the net climate benefits of altering grazing patterns to promote carbon sequestration, using the GWP_{100} CO_2e metric may weight too highly the negative effects of CH_4 emission (if livestock numbers do not change there will be no net increased warming through this route). Using the alternative GWP* metric allows the direct effects of changes in CO_2 emissions and sequestration, and of changes in CH_4 emissions, on climate warming over different time scales to be better integrated (see also ¶(47)). [S]

(29) Relatively low intensity ‘conservation grazing’ by domestic livestock at appropriate times can be an important management tool in maintaining a variety of semi-natural habitats of high biodiversity value (figure 1*b*), in some cases substituting for wild herbivores that are locally absent or extinct [B]. Because of their low stocking rate, the effects on soil carbon dynamics and net emissions are small [E].

(a) Some semi-natural habitats are very sensitive to the presence of domestic livestock. For example, upland peatbog ecosystems can be damaged by densities of sheep above 1 per 2.5 hectares, which is the typical stocking density in these environments. Peat-forming vegetation is especially susceptible to trampling although limited grazing can still be desirable. [L]

(f) Pasture management and soil carbon

(30) There are multiple ways to impact pasture soil carbon dynamics, some via grazing livestock, and some by utilizing other interventions. This section explores the latter and their interaction with grazing interventions.

(31) Cutting or mowing grasslands with removal of biomass may reduce organic inputs to soil, but can also stimulate plant regrowth and therefore NPP, thus increasing root exudates and turnover which can lead to increased soil carbon. A recent meta-analysis indicated that grassland mowing with no biomass removal has no overall effect on SOC, although there is some evidence that increasing cutting frequency can result in SOC increase. [L]

(32) Low productivity ‘weedy’ grass species tend to accumulate over time in old intensively managed pastures, particularly if optimal soil pH for grass growth is not maintained. Pasture productivity and quality of forage for livestock production can be improved by reseeding with higher-quality species, including legumes (see (¶33)) and deep-rooted herbs (see (¶34)) [S]. Pasture rejuvenation can increase SOC by 1–2% per annum [L], particularly where seeding techniques are used that avoid tillage, such as direct drilling or ‘overseeding’, thus preventing soil carbon losses (see ¶(19)) [B].

(33) Introducing N-fixing legumes (clovers, trefoils, vetches) can increase plant productivity and SOC, as well as improve forage quality thus benefiting livestock productivity [S]. Inclusion of legumes increases N_2O emissions due to higher soil N availability, which typically negates about 30% of the benefits of increased carbon sequestration in western European contexts, although legumes also influence soil structure and microbial activity which may lead to proportionally lower N_2O emissions for a given N input [L]. However, where legume N-fixation replaces applications of manufactured N fertilizer, this both avoids the emissions from fertilizer manufacturing and reduces soil N_2O emissions for similar forage dry matter production [B].

(34) Plants with deep roots, both herbs and grasses, are proposed to improve SOC stocks by delivering root exudates and dead organic matter to deeper soil horizons, as well as stimulating microbial activity at lower depths, although there is currently limited evidence from temperate regions to support this [E]. Deep roots can have other benefits such as improving drought resilience and facilitating water infiltration by increasing soil porosity which reduces runoff [L].

(35) Pastures with a high diversity of plant species, cultivars and functional groups can increase biomass production by around 30% and soil carbon stocks by approximately 18% [S]. High plant diversity can also increase mineral supply to livestock improving animal health and productivity. This happens because a diverse set of

species can use available resources more efficiently and are affected in different ways by environmental stress thus conferring resilience. [E]

(36) The application of manufactured fertilizer frequently improves productivity and can increase SOC by around 10%, although pasture diversity is reduced due to fertilizer favouring the most competitive species. However, any increase in carbon sequestration from N fertilizer is more than outweighed by the increased N_2O emissions after application, plus the emissions involved in its manufacture (via the Haber–Bosch process, responsible for 1.4% of global emissions). [S]

(37) The application of farmyard manure and slurry is a source of N and other key macro- and micronutrients, and contains organic matter that can be incorporated into the soil, thus increasing soil carbon stocks. This increase must be set against CH_4 and N_2O emissions during storage and application, especially in wet conditions. [S]

(a) Long-term experiments in the UK indicate that there is little SOC benefit from applying manure to permanent pastures where these soils are already carbon saturated [L]. By contrast, applications on cropland increase soil carbon stocks by 20–30% [S].

(I) In the UK, manure is increasingly being returned to cropland from livestock farms, often in exchange for straw for animal bedding, as arable farmers seek to restore cropland soil organic matter levels and reduce manufactured fertilizer usage. However, associated machinery compaction needs to be managed to avoid negatively impacting crop productivity or soil emissions, and the geographical separation of the main cropland and livestock regions currently limit uptake of this practice. [E]

(b) Using manure applications to increase SOC stocks can risk simply redistributing organic matter, rather than leading to an overall increase. This is because if the components of manure, particularly crop residue used for bedding, were left on the field where they were grown, this would have increased soil organic matter in those locations anyway. [B]

(c) Options to reduce emissions from manure include treatments such as anaerobic digestion or aerobic composting, storage under cover and reducing storage time, and application methods such as sub-surface injection.

(I) Anaerobic digestion of manure and slurry often reduces CH_4 emissions from storage, although emissions can be higher if digested manure is stored in warm conditions over summer, but may increase N_2O emissions after application by making the N present more readily available to microbes. The overall reduction in emissions from manure storage and application due to anaerobic digestion ranges from approximately 20–60%. Fossil fuel usage can be displaced with the CH_4 captured via the digestion process. Aerobic composting and aeration of slurry similarly reduces CH_4 emissions which more than outweighs any increase in N_2O emissions in CO_2e. Treatment of manure with acids reduces ammonia, but may increase N_2O emissions after application due to higher N retention and could increase field liming requirements to maintain optimal soil pH for plant growth. [L]

(II) Injection of slurry just below the soil surface can reduce ammonia by 85% compared with conventional splash plate slurry application. The improved soil N retention reduces manufactured fertilizer requirements but N_2O emissions are correspondingly increased. [L]

(III) Modifying livestock diets, such as by reducing or changing the form of protein included, improving dietary protein:energy balance, and altering the way in which N is excreted can also reduce N in manure by around 20%, which reduces N_2O emissions due to the lower availability of N. [L].

(38) Ground limestone can be applied to acidic soils to increase pasture productivity and improve soil structure (liming) [B]. Soil carbon sequestration is inhibited in highly acidic soils ($pH < 4$) and liming can increase SOC by around 6% [L].

(a) Liming reduces N_2O and CH_4 emissions [L]. However, the net impact on greenhouse gas emissions varies between studies, due to a simultaneous increase in soil CO_2 emissions from increased microbial activity as soil acidity is neutralized. Emissions associated with the production, transport and application of lime also need to be accounted for.

(g) Indirect effects

(39) The creation or destruction of pastures, and how they are managed, affects greenhouse gas fluxes and the amount of carbon stored above and below ground. In estimating the overall benefits or downsides of these changes, it is essential to consider any indirect changes to emissions and storage elsewhere. [B]

(40) Conversion of cropland to pasture will result in reduced crop production and increased or redistributed ruminant livestock production. However, where arable soils are significantly degraded, temporary grass leys can recover soil organic matter and fertility thus enabling a return to more productive arable cropping on rotation (see ¶20 and ¶27). Changing management practices to increase soil carbon stores may affect food production, positively or negatively. Reduced production in one place will tend to stimulate more production elsewhere, in the absence of policies to influence consumption practices. [E]

(a) It is seldom possible to identify precisely where and how compensatory responses take place and instead they must be estimated by food system models that take into account how prices and demand are affected by changes in supply, and which incorporate realistic assumptions about within-nation and international trade. [E]

(b) Deforestation, especially in the tropics, is a major source of greenhouse gas emissions, biodiversity loss and soil degradation, so where compensatory production to reduced UK outputs involves clearing tropical forests, the net effects on climate and other environmental outcomes are likely to be very unfavourable. [S]

(41) There is substantial variation in the emissions produced from different beef production systems. Grass-fed systems typically incur higher CH_4 emissions and require

more total land than grain-fed systems, although less cropland. N_2O and CO_2 emissions are normally higher in grain-fed systems. There are not major differences in emissions per kg of animal liveweight or meat between the two production systems, though a great range of values within each. [S]

(a) Limiting livestock production to grass-based ruminants, known as 'livestock on leftovers', would reduce the amount of cropland needed to produce feed and avoid competition between human food versus animal feed production. It cannot be applied to egg, pork and chicken production because pig and poultry production systems rely on cereal-based feeds, though they can utilize food waste. Though grass-fed ruminants utilize pastures unsuitable for arable crops the counterfactual of using the land for other purposes such as woodland creation for carbon sequestration or for biodiversity should be considered. [S]

(h) Policy implications

(42) Many grasslands around the world are severely degraded and better management, particularly reductions in overgrazing, could lead to the sequestration of 1.65 Gt CO_2e yr^{-1} globally (approximately 3% of annual global emissions) [S]. There is some opportunity for increasing carbon sequestration in grassland soils degraded from overgrazing and tillage in the UK, although well-managed permanent pastures that are already high in soil organic matter will have limited capacity for further sequestration though if maintained in good condition they can act as substantial carbon stores [E].

(a) The overall technical potential for soil carbon sequestration from appropriate pasture management in the UK has been estimated at 2.95 Mt CO_2e per year (approximately 0.7% of UK current emissions), informed over 20 years. [L]

(43) The permanence of carbon storage needs to be considered when developing land use policy to increase carbon sequestration. Much carbon in soils, especially the POM fraction, can be quickly lost if appropriate management is not continued or if the land is ploughed or severely disturbed [S].

(a) It will be important to explore the resilience of soil carbon stores to climate change. For example, changes in temperature and moisture will influence soil carbon dynamics (see ¶(25)), while extreme weather events can affect soil microbial and plant communities and lead to episodes of soil loss and erosion. Climate change may also have an indirect effect through its influence on forage production and livestock husbandry. [E]

(44) Ruminant livestock production systems in the UK are predominantly pasture based with little recent history of land use change and only moderate use of imported feed crops, meaning direct emissions from land use change are much less than the global average. UK and Eire CH_4 emissions, per kg unit output, are also well below global average, due to existing high productivity and efficiency, and a suitable climate. [E]

(a) A reduction in UK livestock production could have net negative climate implications if the displaced production was made up from less carbon efficient production systems elsewhere (and for the same reason increased production might be positive). Whether this occurs depends on the details of trade networks and different product substitutability. [E]

(b) For similar reasons a switch from more animal- to more plant-based diets in the UK, which would reduce emissions, need not imply a corresponding fall in livestock production if UK livestock products are competitive on global markets, especially were carbon trading to be widely introduced in the food sector. [E]

(45) Greenhouse gas emissions from the livestock sector can be reduced by supply-side measures (¶7), but for nations and the world to halt global warming demand-side interventions such as constraining global *per capita* meat consumption and reducing food loss and waste are unavoidable. [B]

(a) People can be encouraged to reduce their consumption of high-emissions food types by education, persuasion, fiscal measures and regulation. The effect of the change depends on the quantity and emission intensity of the foods that they switch to. On average, appropriate plant-based substitutes such as pulses have substantially lower emissions than animal-based foods and so policies to reduce demand for meat and dairy would result in lower emissions. [S]

(b) Some groups in low-income countries rely on animal-based products for their nutrition and should not be subject to meat consumption reduction policies in the absence of alternatives. The availability of minerals and vitamins differs between animal- and plant-sourced food which needs to be considered in formulating population nutrition policy. [E]

(46) As stated above (¶ 4(b)28(c)) emissions policies around livestock production need to take into account the different residence times of CH_4, N_2O and CO_2 in the atmosphere. [E]

(a) To stop further temperature increases, the world must achieve net zero CO_2 emissions, but only prevent greater than present emissions of agricultural CH_4. If livestock production were to remain constant with the same emissions intensity then its CH_4 emissions would not contribute substantially to further global temperature increases beyond the amount of warming already caused by livestock CH_4 emissions, though any CO_2 and N_2O emissions would. [E]

(b) A sustained reduction in livestock production today would reduce the equilibrium concentration of CH_4 in the atmosphere which would reduce the level of temperature increase from livestock. By contrast, the concomitant reduction in CO_2 emissions would only limit further warming. [E]

(c) Different metrics (GWP_{100}, GWP*) report the warming effects of CO_2, CH_4 and other gases in different ways, and the most suitable metric is influenced by the precise policy goal it is intended to support. The widely used GWP_{100} can overestimate the long-term advantages of reducing CH_4 emissions and underestimate the short-term benefits, an issue that GWP* or climate modelling approaches can address. [E]

(47) There are always multiple objectives when managing landscapes and grasslands, and these need to be considered in decisions around increasing carbon sequestration. For example, well-managed grasslands can improve water quality and reduce flood hazard. Both globally and in the UK, grasslands used as pasture produce food and support livestock farmers and allied rural businesses, and pastoralists with strong heritage and cultural value. Some grasslands also have important biodiversity value, and species-rich grasslands are threatened in the UK, although these frequently require specific management interventions that reduce their value for livestock production. [S]

(a) Climate change, changing weather patterns and extreme weather events are likely to impact the ecosystem service of temperate grasslands. For example, changes in temperature and moisture will influence soil carbon dynamics (¶15(b-c)) and the resilience of forage productivity, while extreme weather events can drive shifts in soil microbial and plant communities.

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8. Jordon MW *et al.* 2023 A restatement of the natural science evidence base concerning grassland management, grazing livestock, and soil carbon storage. Figshare. (doi:10.6084/m9.figshare.c.6996767)

An annotated bibliography of the natural science evidence base concerning grassland management, grazing livestock, and soil carbon storage

Matthew W. Jordon,[1] Jean-Charles Buffet,[1] Jennifer A.J. Dungait,[2,3] Marcelo V. Galdos,[4] Tara Garnett,[5] Michael R.F. Lee,[6] John Lynch,[7] Elin Röös,[8] Timothy D. Searchinger,[9] Pete Smith,[10] H. Charles J. Godfray[*,1,11]

[1]Oxford Martin School, University of Oxford, 34 Broad Street, Oxford, OX1 3BD, UK.

[2]School of Geography, College of Life and Environmental Sciences, University of Exeter, Stoker Road, Exeter, EX4 4PY, UK.

[3]Carbon Management Centre, Scotland's Rural College (SRUC), Peter Wilson Building, The King's Buildings, West Mains Road, Edinburgh, EH9 3JG, UK.

[4]Rothamsted Research, West Common, Harpenden, Hertfordshire, AL5 2JQ, UK.

[5]TABLE, Environment Change Institute, University of Oxford, 3 South Parks Road, OX1 3QY, UK

[6]Harper Adams University, Newport, TF10 8NB, UK.

[7]Nature-Based Solutions Initiative, Department of Biology, University of Oxford, 11a Mansfield Road, OX1 3SZ, UK.

[8]Swedish University of Agriculture Sciences, Ulls hus, Almas allé 8, Uppsala, SE-750 07, Sweden.

[9]School of Public and International Affairs, Princeton University, 318 Robertson Hall, NJ 08544-1013, USA.

[10]Institute of Biological and Environmental Sciences, University of Aberdeen, 23 St Machar Drive, Aberdeen, AB24 3UU, Scotland, UK.

[11]Department of Biology, University of Oxford, 11a Mansfield Road, OX1 3SZ, UK.

Author for correspondence: Charles Godfray (Charles.godfray@oxfordmartin.ox.ac.uk).

This annotated bibliography summarises the evidence supporting each section of the main restatement, cross-referenced by section number. Where statements are background information [B] or supported by citations in immediately preceding sections, these statement numbers are intentionally left blank (or noted as Author's summary) in the annotated bibliography below.

(a) Aims and scope

(1) Author's summary.
(2) Author's summary.
(3) Codes used in previous Restatements provided by Dadson et al. (2017b), Goddard et al. (2022), Godfray et al. (2014), Godfray et al. (2015), Godfray et al. (2013), Godfray et al. (2019), McLean et al. (2017).

(b) Livestock emissions

(4) For overall greenhouse gas emissions and climate change implications of the global food system, see Clark et al. (2020). Crippa et al. (2021) estimate the emissions breakdown of the global food system. Xu et al. (2021) find that of emissions from global food production, livestock account for 57%.

(a) Author's summary.

Description and discussion of different GHG metrics and methane are available at Allen (2015), Allen et al. (2018), Lynch (2019a), also Box 7.3 in IPCC AR6 WG1 (Forster et al., 2021), and Cross-chapter Box 2 in IPCC AR6 WG3 (Dhakal et al., 2021). To summarise in brief: there are two factors that determine the impact of different GHGs on atmospheric warming: the 'strength' of radiative forcing, which in turn causes warming, per molecule of each gas, and the amount of time each gas stably persists in the atmosphere. In terms of the three GHGs most relevant to ruminant livestock production: carbon dioxide (CO_2) has a relatively low radiative forcing per molecule but persists for a long time (1000+ years), methane (CH_4) has a high radiative forcing per molecule but breaks down rapidly (half-life of 12 years), and nitrous oxide (N_2O) both has a high radiative forcing per molecule and persists for a long time. To enable comparison between emissions of different GHGs, various metrics have been developed to express emissions of all gases as carbon dioxide equivalents (CO_2e).

The most commonly used metric is the Global Warming Potential over 100 years (GWP_{100}) of a one-off emission of a tonne of gas. This expresses the total amount of radiative forcing over 100 years relative to the number of tonnes of CO_2 that would be required to give the same total warming. Therefore, the GWP_{100} of CO_2 is set as 1. Using this method, one tonne of methane emissions is equivalent to 27 tonnes of CO_2, and one tonne of nitrous oxide emissions is equivalent to 273 tonnes of CO_2 (Forster et al., 2021).

The warming impact of methane can better-expressed by comparing small ongoing emissions of methane with a one-off pulse of carbon dioxide, using a recently developed metric called GWP*. This accounts for the high radiative forcing but short atmospheric lifespan of methane (Lynch et al., 2021).

Domestically, UK emissions are reported as aggregated GWP100 CO_2e, and the Climate Change Committee (CCC) sets net-zero pathways within this accounting framework (Climate Change Committee, 2019). Physically, however, there is no way to universally combine all GHGs: at the very least short- and long-lived gases must be separated in order to determine temperature change contributions (Allen et al., 2022).

For this Restatement, we generally refer to '[CO_2-equivalent] emissions', and note their relevance for overall UK policy purposes, but in some cases instead note 'warming', and details of dynamic temperature change over time. The reader should note these two concepts are not necessarily aligned. We make no judgments over where or whether different approaches should be preferred, and in most cases simply relay information as reported in the original citations.

(b) GHG emissions intensities for beef production in different regions of the world sourced from Figure 8 of Gerber et al. (2013b).

(5) For an overview of emissions sources from global livestock production, see Gerber et al. (2013b). For a breakdown of global methane emissions, see Saunois et al. (2020).

(a) Ruminants have three fore-stomachs (the rumen, reticulum and omasum) and one true stomach (the abomasum). In the rumen, food is initially digested by microbes to produce volatile fatty acids, in anaerobic (oxygen-free) conditions in a process known as enteric fermentation. This produces hydrogen gas as a by-product, which another set of microbes (methanogenic archaea) convert into methane gas (CH_4) which is then eructed (belched) by the ruminant. The majority of this enteric methane production occurs in the rumen and reticulum, although 2-10% can also occur in the hindgut (Hristov et al., 2013a). This disposal of hydrogen gas is important for healthy rumen functioning, as a build-up of hydrogen can cause animal bloat and reduce the rumen pH below a critical threshold which inhibits microbial digestion. Therefore, this creates an important biochemical limit on the extent to which methanogenic microbes can be directly inhibited without impairing animal welfare or performance.

(b) Author's summary.

(c) Author's summary.

(I) (IPCC, 2019).

(6) (Gerber et al., 2013b).

(a) Ruminant livestock numbers as of June 2021 sourced from Defra (2021a). UK greenhouse gas emissions for 2020 sourced from BEIS (2022).

(b) UK-specific emissions values for individual food products were sourced from CIEL (2020). In some circumstances other functional units may be chosen to express emissions, such as per kg of edible protein, *e.g.* as available in Figure 3 of Gerber et al. (2013b). However, in these instances it is also important to account for amino acid composition and digestibility, in addition to bioavailable micronutrient content, see Lee et al. (2021b), McAuliffe et al. (2023), McAuliffe et al. (2018a), McAuliffe et al. (2020b).

(I) Expert's opinion.

(7) For an overview of supply-side GHG mitigation options for UK livestock production, see CIEL (2020), CIEL (2022).

(a) (Hristov et al., 2013a, Herrero et al., 2016). Enhancing digestibility will improve microbial protein production through increasing the balance of readily available energy in the rumen and rapidly degradable plant protein, which will improve animal performance and nutrient use efficiency. Starch and to a degree fructan move the rumen microbial population to more amylolytic activity, favouring propionate formation in the rumen which is a hydrogen sink thus reducing methane production via substrate supply reduction. (Cummins et al., 2021).

(b) For reviews of dietary additives to reduce enteric methane emissions, see Hristov et al. (2013b), Hristov et al. (2013a), Gerber et al. (2013b), Gerber et al. (2013a), Patra et al. (2017), Beauchemin et al. (2022), also Marques and Cooke (2021) for ionophores. Methane-suppressing feed additives can be administered at pasture through routine bolusing or drenching, or through inclusion in mineral licks or buckets, although there are challenges around ensuring persistence of effect and adequate uptake, respectively.

(I) See Rabiee et al. (2012) for dietary lipids, and Jayanegara et al. (2012), Cardoso-Gutierrez et al. (2021), Goel and Makkar (2012), Kingston-Smith et al. (2010), Luscher et al. (2014), Mueller-Harvey et al. (2019) for condensed tannins. Regarding seaweeds, see Maia et al. (2016), Abbott et al. (2020), Min et al. (2021) for reviews of the potential and challenges of seaweed for enteric methane mitigation. Muizelaar et al. (2021) find that the active component from seaweed, bromoform, is transferred to dairy cattle urine and milk when included in the diet. Furthermore, there is a risk that large-scale feeding of seaweed to livestock could increase the release of inorganic bromoform to the atmosphere, which could contribute to atmospheric ozone depletion, although the overall rate of ozone depletion from this effect is thought to be minimal (Glasson et al., 2022).

(II) Author's summary.

(c) Author's summary.

(I) Author's summary.

(II) Author's summary.

(III) Brito et al. (2018) conduct a meta-analysis on methane emissions heritability in sheep and cattle, finding that this trait is inder moderate genetic control (h^2 estimates from 0.14-0.26 across 18 articles). Lopez-Paredes et al. (2020) identify heritabilities of enteric methane concentration and production of 0.11 and 0.12, respectively, in Spanish Holstein dairy cattle. Examples of studies that investigate heritability of sheep enteric methane emissions using respiration chambers and portable accumulation chambers in New Zealand and Australia include Pinares-Patino et al. (2013), Goopy et al. (2015), Robinson et al. (2020), Rowe et al. (2019), Johnson et al. (2022) which estimate heritabilities of 0.13 to 0.32. Martinez-Alvaro et al. (2022) identify an element of cattle genomic control of rumen microbiota and enteric methane production in Scottish beef steers, and therefore potential for microbiome-based breeding in emissions reduction. They estimate reduction in emissions of -1.43 +/- 0.14 to -3.32 +/- 0.77 g CH_4/kg dry matter intake (DMI) per generation, depending on selection intensity (1.16 to 2.67, respectively). Selection based on cattle genomic breeding values for rumen microbiome composition also has the advantage of avoiding the need for costly direct measurement of methane emissions on farm.

(c) Soil carbon dynamics

(8) Author's summary.

(9) Author's summary.

(10) The time taken for SOC to be decomposed by soil microbes ranges from seconds to centuries, and its release from soils as CO_2 depends on a combination of its form, where it is in the soil, how the soil is managed, and climate. For an introduction to soil organic matter, see White (2006). For the rate that organic inputs are mineralised to CO_2, see Angers et al. (2022). Current understanding of SOC pools is not based on fast vs slow pools, but rather accessibility to decomposition by microbes (Campbell and Paustian, 2015, Zhang et al., 2021, Dungait et al., 2012b). For the rate of root carbon vs aboveground biomass carbon stabilisation, see Jackson et al. (2017), Bai and Cortrufo (2022).

(11) For an overview of soil carbon analysis methods, see White (2006), also Grewal et al. (1991), Pribyl (2010). For minimum differences detectable through measurement, see Schrumpf et al. (2011), Smith (2004). For a comparison between soil carbon models, see Smith et al. (1997). For a description of soil organic carbon fractions and particulate vs mineral-associated organic matter, see Lehmann and Kleber (2015), Lavallee et al. (2020), Cotrufo et al. (2019), also White (2006), Badgery et al. (2014), Just et al. (2021), Kelleher and Simpson (2006), Simpson et al. (2007), Poeplau et al. (2018). Residues of dead microbes, or 'necromass', are a key pool of resistant SOC in grassland soils (Bai and Cortrufo, 2022).

(a) (Witzgall et al., 2021).

(b) (Witzgall et al., 2021).

(12) Soil fauna, particularly 'macrofauna' like earthworms, play a major role in the vertical movement and mixing of organic carbon; they help aerating the soil, break down plant litter and speed up decomposition which can increase CO_2 emissions, but mixing of organic carbon and soil in soil fauna guts can help form very persistent pools of SOC excreted in the casts of earthworms and faeces of other soil animals. For a review of the role of soil fauna in SOC distribution and stabilisation, see Filser et al. (2016).

(a) Over 2 Mha of soil in England and Wales is estimated to be at risk of erosion (Environment Agency, 2019). For the costs of soil degradation, see Graves et al. (2015).

(13) Following a beneficial change in land management, sequestration rates are high and steadily decline until a new equilibrium is reached (Smith, 2014, Poulton et al., 2018). SOC levels can take around 100-750 years to reach a new equilibrium when increasing, and 50-150 years for a decrease (Falloon et al., 2006). Therefore, where soil carbon sequestration is still being detected decades after a change in land use or management, this is a legacy effect of the historic change rather than evidence for perpetual sequestration (Smith, 2014, Foster et al., 2003). For example, Fornara et al. (2016) found that a 43-year-old grassland in Northern Ireland was still sequestering carbon after being reseeded in 1969. Similarly, Bellamy et al. (2005) identified ongoing declines in SOC stock in UK arable soils between 1978 and 2003 following conversion from grassland following the Second World War. Some misleadingly high values for potential increases in soil carbon stocks are reached by incorrectly assuming that initial high rates of soil carbon sequestration would be sustained in a linear fashion. This cannot be achieved in practice. When considering soil carbon sequestration or losses, it is necessary to convert from soil carbon stocks to carbon dioxide equivalents (CO_2e) by multiplying by a factor of 3.66.

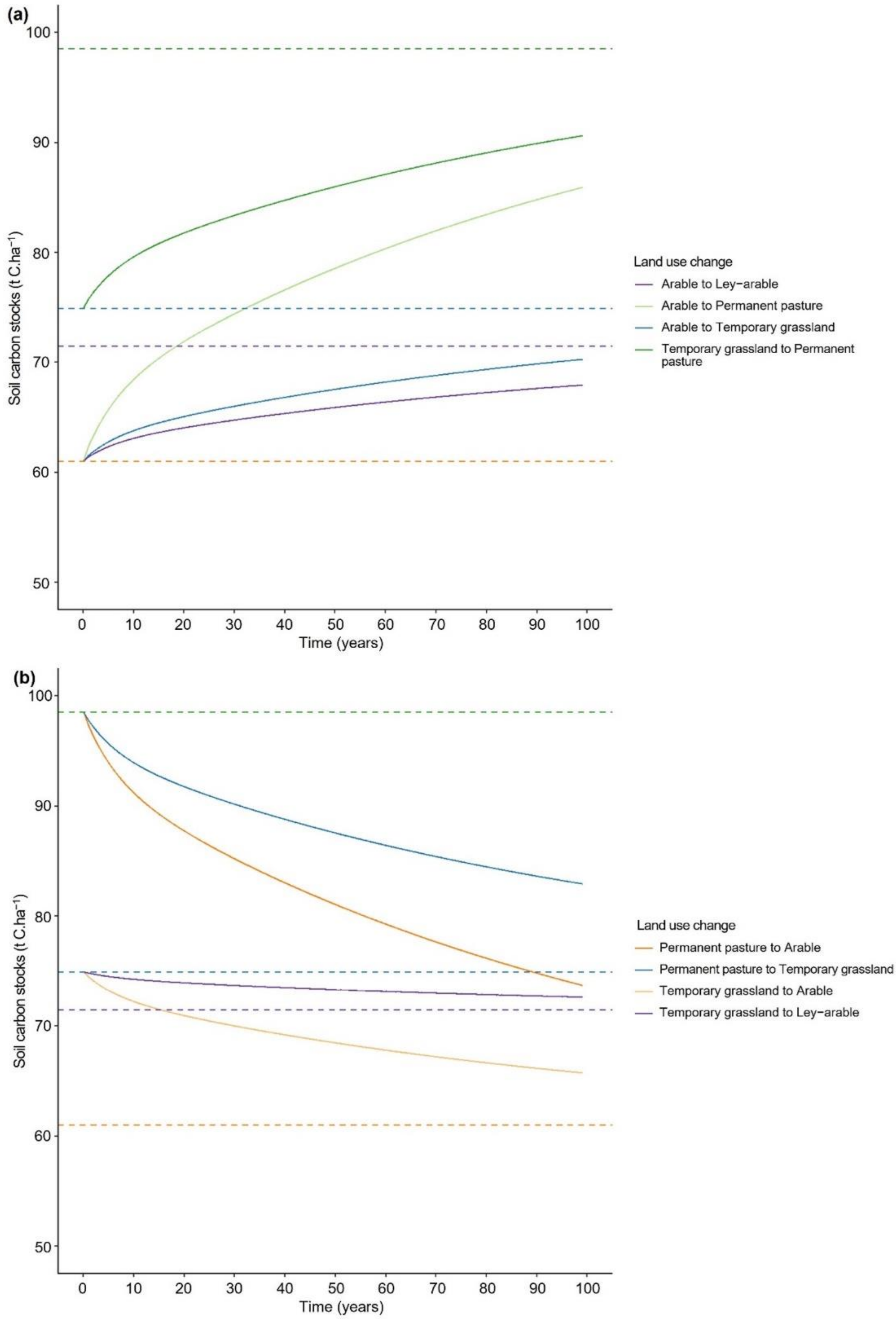

Figure S1: Trajectories of soil organic carbon (SOC) stock changes in topsoil (0–15 cm), following land use change between arable cropland (orange horizontal line), ley-arable rotations (purple horizontal line), temporary grassland (blue horizontal line), and permanent pasture (green horizontal line), in a direction that would (*a*) increase or (*b*) decrease SOC stocks. Trend lines are coloured according to the land use they were changed to (for example, the blue trend line represents a change from arable to temporary grassland). Equilibrium SOC stocks for England and Wales from Smith et al. (2010), which are in turn based on the UK soil carbon database assembled by Bradley et al. (2005); ley-arable soil carbon inputs are from Jordon et al. (2022b). Rates of soil carbon changes simulated using the RothC-26.3 model, assuming GB average clay content of 21% from WISE30sec soil maps (Batjes, 2016) and GB average weather variables (precipitation, temperature, potential evapotranspiration) for 1980-2010 from Terraclimate (Abatzoglou et al., 2018). Conversion to or from woodland is not included, as much of the carbon stored in woodland is in above- and belowground biomass in addition to soil organic matter. UK native-species broadleaf woodland is likely to have similar SOC stocks to permanent pasture. The SOC stocks for permanent pasture likely include some organic and organo-mineral soils, hence the high SOC level. High organic matter content soils likely have less capacity for further sequestration *via* improved pasture management practices. For Figure 3: Values for soil organic carbon changes are sourced from Abdalla et al. (2018), Zhou et al. (2017a), Eze et al. (2018), Byrnes et al. (2018) for continuous grazing (duration, number of studies and SOC changes averaged across these four meta-analyses), Eze et al. (2018) for liming; Conant et al. (2017) for fertiliser; Prommer et al. (2020) for diverse pastures; Conant et al. (2001) for improved grass species and legumes); Phukubye et al. (2022) for rotational grazing; Lamb et al. (2016) for broadleaf woodland which represents carbon sequestration in woody biomass as a counterfactual to soil carbon sequestration rates in pastures presented. The values reported here should be assumed to be optimistic and should be treated with caution for the following reasons. Firstly, much UK improved pasture is likely to have SOC stocks higher than 74.9 $t.ha^{-1}$ and therefore less remaining capacity to increase before the sink saturation point is reached. Secondly, percentage increases in soil carbon reported by global meta-analyses and used here are in fact unlikely to be directly applicable to the UK context, given underlying differences in initial pasture SOC stocks, climate and NPP. Thirdly, these simulations assume for simplicity no current adoption of these management practices, which is clearly incorrect. Finally, implementing multiple beneficial management practices will have a saturating rather than additive effect on soil carbon stocks, i.e. the values presented for individual management practices could not simply be added together where multiple practices are adopted simultaneously. Baseline SOC stock is assumed to be 74.9 t $C.ha^{-1}$ to 15 cm depth (temporary grassland value from Smith et al. (2010)). Sequestration rates simulated using RothC-26.3, assuming clay content of 21% from WISE30sec soil maps (Batjes, 2016) and average weather variables (precipitation, temperature, potential evapotranspiration) for Great Britain from 1980-2010 from Terraclimate (Abatzoglou et al., 2018).

(a) See Section 7 of Lal (2004).

(b) Smith (2014) demonstrate using long term experimental data that grasslands do not act as a perpetual sink for carbon sequestration.

(I) (Cotrufo et al., 2019).

(c) (Lindsay, 2010).

(14) For a discussion of the issues of reversibility of soil carbon sequestration and permanence of soil carbon gains, see Smith (2005).

(a) Meta-analyses of the effect of tillage on soil carbon include Jordon et al. (2022a), Meurer et al. (2018), Haddaway et al. (2017), Sandén et al. (2018).

(I) (Bai et al., 2019).

(15) For an overview of the effect of soil properties and climate variables on soil carbon levels, see White (2006). Also Jenkinson (1990), Yu et al. (2017), Kerr and Ochsner (2020), Patel et al. (2021). For the difference in factors influencing POM vs MAOM formation, see Bai and Cortrufo (2022), Mitchell et al. (2021).

(a) Prout et al. (2022) suggest that the ratio of soil carbon concentration to soil clay content could provide a realistic indicator for soil carbon gains achievable on UK agricultural land.

(b) Author's summary.

(c) See Dobbie and Smith (2003).

(I) Author's summary.

(d) Author's summary.

(I) For estimates of terrestrial carbon stocks in vegetation and soils across biomes, see Table 3.2 in IPCC (2001).

(II) (MALHI et al., 1999).

(III) The net effect of climate change on SOC stocks is uncertain and likely to be highly spatially variable (Smith et al., 2005). For a summary of the mechanisms by which climate change could impact soil carbon in different environments, see Bai and Cortrufo (2022)

(e) The more stable MAOM fraction of SOM has a higher carbon to nitrogen (C:N) ratio than the plant residue inputs to the soil from which it is derived (Kirkby et al., 2013). Additional inputs of N to the soil can promote the formation of inert MAOM from more easily degradable POM. Increasing soil N inputs can increase nitrous oxide emissions (Mosier et al., 1998), negating soil carbon sequestration (Powlson et al., 2011, Henderson et al., 2015) and may require extra fertiliser production (van Groenigen et al., 2017).

(I) Expert's opinion.

(f) For an overview of the role of soil fungi and bacteria in SOC dynamics, see Bai and Cortrufo (2022).

(16) Author's summary.

(I) For reviews of the actions of methanogenic and methanotrophic bacteria, see Serrano-Silva et al. (2014), Le Mer and Roger (2001), Mancinelli (1995).

(II) Soil N_2O emissions are also influenced by factors such as pH and availability of certain micronutrients.

(d) Soil carbon in pastures and under alternative land uses

(17) 1 Gigatonne (Gt) = 1 Petagram (Pg) = 1 billion tonnes or 10^{15} g. (Batjes, 2016) estimate global soil carbon stocks to 1 m depth as 1408 Gt C (standard deviation 154 Gt C), using a globally harmonised database. Estimates are also available to 30 cm, 50 cm, 1.5 m and 2 m. Pan et al. (2011) estimate current carbon stocks in the world's forests and rates of C gains and losses. Note that total C stocks are "861 ± 66 Pg C, with 383 ± 30 Pg C (44%) in soil (to 1-m depth), 363 ± 28 Pg C (42%) in live biomass (above and below ground), 73 ± 6 Pg C (8%) in deadwood, and 43 ± 3 Pg C (5%) in litter". Only the value for living biomass plus deadwood plus litter is reported in

this Restatement to avoid double counting with the global soils estimate. The National Oceanic and Atmospheric Administration (recorded the monthly average atmospheric CO_2 concentration for July 2022 as 418.9 parts per million (ppm). 1 ppm of atmospheric CO_2 corresponds to 2.13 Gt C. This equates to approximately 892.26 Gt C. The FAO provide estimates of global grassland area and carbon stocks (Conant, 2010). For Figure 2: Semi-natural grassland includes neutral, acid, and calcareous grasslands plus bracken, and shrubland, bushland and heathland includes dwarf shrub heath and montane if not covered by rock by more than 95%. Note that the areas in (a) and (b) do not sum to the same total, are calculated using different methods, and land use vs habitat classifications are not directly comparable. Shrubland in (b) will likely fall within rough grazing in (a), and improved pasture in (b) will fall within permanent and temporary grass in (a).

(18) Milne and Brown (1997) estimate carbon stocks in vegetation and soil in Great Britain. For soil carbon stocks, see Table 12 therein. This total value with an error parameter is also reported by Dawson and Smith (2007). Bradley et al. (2005) provide soil carbon stocks by land use and soil sampling depth for each country in the United Kingdom (see Table 5). The estimate for pasture and arable (includes temporary grassland) to 0-100 cm sampling depth is provided in the 18a and 18b. Defra (2021a) estimate areas of each type of UK pasture as: permanent pasture, 6.1 Mha; temporary pasture, 1.2 Mha; sole-right rough grazing, 3.9 Mha; common rough grazing, 1.2 Mha. These do not map exactly to the carbon stocks reported in the restatement as the permanent pasture carbon stock includes improved permanent pasture plus rough grazing, while there is not a carbon stock for temporary pasture that is disaggregated from cropland. UK territorial greenhouse gas emissions for 2020 were sourced from BEIS (2022).

(a) (Defra, 2021b).

(b) (Defra, 2021b).

(c) For UK peatland area and usage, see ONS (2019). Evans et al. (2017) estimate current emissions from UK peatland.

(I) (Evans et al., 2017).

(II) (ONS, 2019).

(19) See Carolan and Fornara (2016) for carbon losses from cultivating permanent pasture. See Soussana et al. (2004), Smith et al. (2010) for soil carbon gains from converting temporary to permanent pasture. For evidence that converting arable to pasture increases SOC stocks, see Ostle et al. (2009), Badgery et al. (2014), Guo and Gifford (2002), Dawson and Smith (2007), Powlson et al. (2011). Most UK grassland is located in the wetter west of the country, on high clay-content soils which retain moisture ('heavy' soils), which is one reason why they are not under arable cropping. Heavy, wet 'plastic' soils compact easily and need appropriate management to break compaction and open up the soil - benefitting grass roots, soil structure, soil biodiversity and reducing N_2O emissions (Natural England, 2016).

(a) Fuchs et al. (2016) estimate soil carbon gains from European cropland abandonment.

(20) (West and Post, 2002, Dawson and Smith, 2007, Soussana et al., 2004, Badgery et al., 2014, Jordon et al., 2022a, Conant et al., 2017, Poulton et al., 2018).

(a) (Cooledge, 2022, Jaramillo et al., 2021).

(b) Badgery et al. (2014) find that pasture cropping can allow arable crops to be grown with soil carbon stocks not significantly reduced compared to permanent pasture.

(21) For woodland carbon stocks compared to pasture, see Guo and Gifford (2002), Ostle et al. (2009), Guo et al. (2021), Barcena et al. (2014), Laganière et al. (2010).

(a) Cotrufo et al. (2019) find that European woodland soil organic matter contains proportionally more POM (labile, potential for ongoing increases) and less MAOM (stable, plateaus) than grassland soils. Richards et al. (2017) simulate conversion of permanent pasture to short rotation forestry, and find that net GHG emissions in terms of SOC loss peak 5 years after land use change, and net emissions only start to be fall (due to net sequestration) after 15 years.

(b) Matthews et al. (2020) find that where woodland is planted on organo-mineral soils (a shallow peat layer overlaying a mineral sub-soil), it may take decades before net carbon sequestration occurs, due to large initial C losses from disturbing the organic soil layer.

(c) (Sloan et al., 2018).

(d) (Guo and Gifford, 2002, Laganière et al., 2010).

(e) Harmer et al. (2001) document natural vegetation succession from arable to woodland at two sites at Rothamsted Experimental Station and identify a slow rate of woodland colonisation. There is insufficient evidence from the UK to determine how natural regeneration approaches compare to tree planting in terms of trajectory and total amount of carbon sequestration possible (Matthews, 2020, Jordon and Wentworth, 2021). Rewilding is unlikely to achieve overall carbon stocks comparable with woodlands despite its potential biodiversity benefits (Sandom et al., 2020).

(22) Meta-analyses demonstrating increased both aboveground and soil carbon stocks from agroforestry include Mayer et al. (2022), Chatterjee et al. (2018), Ma et al. (2020), Kim et al. (2016). Raising livestock in agroforestry systems can also deliver some productivity and welfare benefits compared to pasture without trees (Pent, 2020, Jordon et al., 2020).

(a) For carbon storage in hedgerows, see Drexler et al. (2021), Van Vooren et al. (2017), also Axe et al. (2017) for how hedgerow height and width can be influences carbon stocks. For sustantiable carbon under hedgerows see Biffi et al. (2022)

(b) UK examples of this include Upson et al. (2016), Fornara et al. (2017), Beckert et al. (2015).

(23) Richards et al. (2017) find that bioenergy crops have variable effects on soil net GHG balance (SOC, N_2O, CH_4) compared to permanent pasture. The cultivation required to prepare ground for bioenergy crops causes SOC losses, but bioenergy crops can have higher inputs of organic matter to the soil and lower N_2O emissions from reduced requirements for N fertiliser.

(e) Grazing management and soil carbon

(24) For an overview of the impacts of grazing management on emissions from a grazing system see Mayel et al. (2021); also Hörtnagl et al. (2018) and Soussana et al. (2010a). Beyond the general themes of plant productivity driving CO_2 flux and soil carbon dynamics forage digestibility influencing ruminant enteric CH_4 emissions and soil nitrogen availability (due to manufactured fertiliser, legumes, manure and influence by soil pH, moisture and temperature) driving soil N_2O emissions, there is limited data available regarding the precise impacts of different grazing management systems on soil-plant-livestock emissions. For discussion on N2O emissions from sheep excreta on lowland grassland and grazed hill pasture see Mancia et al. (2022). The relative benefits and limitations of grasslands in terms of soil carbon stocks compared to other land uses is to a large extent independent of the ruminant livestock which are frequently grazed on them. For example, temporary grasslands are often introduced to cropland via ley-arable rotations to provide fertility and weed-control benefits, with the biomass mechanically harvested to provide feedstock for anaerobic digestor plants rather than grazed. However, livestock can influence soil carbon dynamics through a combination of:

- affecting plant productivity (net primary productivity, NPP) and therefore inputs of organic material to the soil through defoliation intensity (Milchunas and Lauenroth, 1993, Mayel et al., 2021),
- increasing availability and redistribution of nutrients such as nitrogen to plants and microbes, thus accelerating SOM turnover (Soussana and Lemaire, 2014, Mayel et al., 2021), and
- changing soil physical properties, *e.g*. increasing bulk density by reducing soil pore volume through trampling see Drewry et al. (2008), also Drewry and Paton (2010), Cournane et al. (2011), Houlbrooke and Laurenson (2013). This can result in soil compaction, particularly in wet conditions, decreasing the aeration of the soil and therefore increase N_2O emissions (Oenema et al., 1997).

(a) Expert's opinion.

(25) In general, grazing leads to carbon losses from vegetation as CO_2 from animal respiration and CH_4 from enteric fermentation (Soussana and Lemaire, 2014), thus decreasing NPP and SOC compared to grazing exclusion (Milchunas and Lauenroth, 1993, Nordborg, 2016, Manley et al., 1995). Livestock grazing only leads to increased SOC when other aspects of management increase biomass production beyond offtake by ruminants (Eze et al., 2018). For results of meta-analyses of soil carbon under continuous grazing compared to no grazing, see Table 1. High grazing intensities also lead to increased soil N_2O emissions and reduced soil CH_4 uptake through impacts on soil microbial processes (Tang et al., 2019, Tao et al., 2023). Reducing grazing intensity can boost soil carbon sequestration and reduce soil N_2O and CH_4 emissions (Chang et al., 2016).

Table 1. Results from meta-analyses of the impact of continuous grazing at different grazing intensities on soil organic carbon, relative to grazing exclusion. The impact of rotational grazing approaches are discussed in section 26. Where disaggregated results are available by climate zone, only results from the climate most comparable to the UK are extracted. Significance levels: ns not significant; * $p<0.05$

Citation	Geographical extent	Grazing intensity	Definition of grazing intensity	Number of grazed-no grazed comparisons	Impact on SOC	Sig	Article data location
(Eze et al., 2018)	Global	Overall			-15%	*	Table 2
		Light	< 5 sheep/ha	100	-6.9%	*	
		Moderate	5–10 sheep/ha	67	-13.2%	*	
		Heavy	> 10 sheep/ha	65	-27.1%	*	
(Abdalla et al., 2018)	Global, estimate for moist cool climate	Overall		9	-19.5%	*	Main text, Section 3.2
		Light	< 33% of carrying capacity[a]	9	-21%	*	Fig. 5
		Medium	33-66% of carrying capacity[a]		-31%	*	
		High	66-100% of carrying capacity[a]		-18%	ns	
(Byrnes et al., 2018)	Global	Overall		225	-8%	*	Fig. 2
		Light	Primary study reporting	56	-4%	ns	Fig. 3
		Moderate		44	-13%	*	
		Heavy		59	-15%	*	
(McSherry and Ritchie, 2013)	Global, estimates for C3 grasses	Light	Primary study reporting		5%	ns	Fig. 2
		Moderate			-8%	ns	
		Heavy			-11%	*	
(Phukubye et al., 2022)	Global	Low	0.4-2.5 animal units/ha/yr	91	-4%	ns	Fig. 2
		High	>2.5 animal units/ha/yr	25	-19%	ns	
(Zhou et al., 2017a)	Global, estimates for humid climates	Light	Author qualitative classification	9	-3%	*	Fig. 6
		Moderate		12	-13%	*	
		Heavy		12	-21%	*	

(a) Author's summary.

(I) Grazing can increase NPP and SOC on grassland sites with a long evolutionary history of grazing (Conant et al., 2001), *e.g*. the Northern Great Plains in the USA (Wang et al., 2016, Holland et al., 1992). In addition, in certain contexts, grazing can increase belowground allocation, resulting in higher root biomass and exudates, and therefore increased soil microbial biomass (Frank et al., 2002, Hamilton and Frank, 2001), which can result in higher SOC in grazed grasslands than ungrazed, *e.g*. Wilson et al. (2018).

(II) For example, Abdalla et al. (2018) found that in 'moist warm' climate zones, grazing increased SOC stocks by 7.6%, and in 'dry warm' and 'dry cool' climates by 5.8 and 16.1%, respectively, at low intensities.

(III) The FAO describe pasture degradation through overgrazing as "generally related to a mismatch between livestock density and the capacity of the pasture to be grazed and trampled" (FAO, 2006). As stocking rate increases, individual animal performance declines linearly, but total animal performance per hectare follows a bell curve by which it initially increases with stocking rate then starts to decline (Jones and Sandland, 1974). Therefore, there is an incentive for managers to avoid overgrazing in order to maximise total animal performance (Kemp and Michalk, 2007)

(26) For a definition of rotational grazing, see Briske et al. (2011b). Other terms that fall within this definition include controlled grazing, precision grazing, paddock grazing, cell grazing, strip grazing, techno grazing, mob grazing, holistic planned grazing (HPG) and adaptive multi-paddock (AMP) grazing. Biomass production has been found to increase in grazing systems with longer rest periods in recent meta-analyses such as McDonald et al. (2019), Jordon et al. (2022b). Periods of rest in rotational grazing enable plant regrowth and therefore a greater rate of photosynthesis through increasing leaf area index, which in turn promotes root development and increases inputs of root exudates into the soil (Sanderman et al., 2015, Savory and Butterfield, 2016, Voisin, 1959).

(a) Meta-analyses of soil carbon under rotational grazing include Byrnes et al. (2018), who found no change in SOC stocks under rotational grazing compared to no grazing, and Phukubye et al. (2022), who identified an average increase in SOC of 21%, although studies in this analysis are of variable quality. Mosier et al. (2021) found that increases in SOC under adaptive multipaddock grazing predominantly occurred in the MAOM fraction, although findings were very variable.

(b) Creating a layer of decomposing vegetation at the soil surface through livestock trampling may increase carbon inputs to the soil compared to standing biomass in ungrazed areas (Jones and Donnelly, 2004, Eyles et al., 2015, Rumpel et al., 2015, Piñeiro et al., 2010), It has also been found in a semi-arid rangeland context (Roberts and Johnson, 2021).

(c) Rotational grazing, and mob grazing/HPG/AMP as more extreme cases of this, increase stocking density by restricting animals to a subsection of the total pasture available (sometimes referred to as a paddock or cell). In instances where rotational grazing approaches improve biomass production, this enables the carrying capacity of the site to be increased and therefore the stocking rate possible (i.e. carry more animals for a given time, or carry the same animals for a longer time). In other words, the stocking density is the management 'input', and the stocking rate is the productivity outcome. It is possible to achieve increases in stocking density in rotational grazing systems while keeping stocking rate constant, simply by altering the number of paddocks and paddock size in the rotation to create pasture recovery time. In instances where farmers wish to use the increase in biomass production from rotational grazing to increase livestock carrying capacity (rather than, for example, reducing purchased inputs of fertiliser and animal feed or extending their grazing season), the increase in methane emissions from the additional livestock in the system would likely negate any soil carbon sequestration benefit from rotational grazing in the medium to long term.

(I) Expert's opinion.

(d) For a discussion of why the purported benefits of rotational grazing are not always realised in practice, see Briske et al. (2008), Briske et al. (2011a), Briske et al. (2011b). In spatially heterogenous rangeland environments such as the North American Great Plains, increased stocking density in rotational grazing systems has a negative linear relationship with animal daily liveweight gain for the same stocking rate (Augustine et al., 2020, Derner et al., 2021). Adaptive rotational grazing has been found to outperform non-adaptive rotational grazing, but nevertheless still achieves lower animal growth rates than continuous grazing in these environments (Derner et al., 2021).

(I) Expert's opinion.

(e) Although widespread anecdotal evidence from practitioners that rotational grazing increases productivity and soil carbon stocks has been supported by recent academic meta-analyses of studies, the benefits of rotational grazing practices in increasing biomass production and livestock productivity are contested, particularly in a rangeland context (Briske et al., 2011b, Teague et al., 2013). There are some confounding factors which can result in a disjoint between practitioner reports and empirical evidence. These include grazing intensity (*i.e.* stocking rate), which can have a greater effect than grazing system (*i.e.* rotational vs continuous) on carbon inputs to the soil, and management intensity, as managers of rotational grazing systems are often more engaged and therefore often adopt other beneficial practices which impact soil carbon (Abdalla et al., 2018, Conant et al., 2017, Jones and Donnelly, 2004, Rumpel et al., 2015, Briske et al., 2008).

(I) Allan Savory, a Zimbabwean biologist who developed Holistic Planned Grazing™ (HPG) (Savory and Butterfield, 2016), has controversially suggested that ruminant livestock grazing could help reverse land degradation and desertification in arid regions and that the soil carbon sequestration potential of HPG may be sufficient to mitigate all historic anthropogenic greenhouse gas emissions (Savory Institute, 2013). Although some studies have identified positive effects of HPG on some indicators, see Chaplot et al. (2016), Ferguson et al. (2013), Weber and Gokhale (2011), many of the mechanisms promoted by Savory remain largely unevidenced (Hawkins, 2017, Holechek et al., 2000, Chamane et al., 2017), and conclusions drawn from these around reversing both desertification and anthropogenic climate change have been widely refuted, see Nordborg (2016), Carter et al. (2014), also see the following exchange Briske et al. (2014), Briske et al. (2013), Teague (2014). However, this does not negate the emerging evidence base that well-managed rotational grazing approaches (including HPG) at appropriate stocking rates can deliver improvements in livestock productivity and modest increases in soil carbon stocks compared to continuous grazing approaches in temperate regions.

(II) Expert's opinion.

(27) For greater SOC increases due to livestock, see Hanley and Ridgman (1979), Schulz et al. (2017), Johnston et al. (2017). Hanley and Ridgman (1979) found that grazing leys increased SOC concentration by 0.04 $g.100\,g^{-1}$ (±0.006) (around 2% increase) compared to

mowing. Johnston et al. (2017) found that SOC was 0.21 g.100 g^{-1} higher after 33 years of grazed rather than ungrazed leys (around 20% increase). Schulz et al. (2017) found that a mixed livestock-arable system increased in SOC over 17 years, whereas a stockless ley-arable decreased in SOC from a similar baseline line, resulting in 14% higher SOC in the system with livestock. These studies tend to compare grazing with harvesting and removal of forage, rather than grazing vs ungrazed standing biomass. For enhanced nutrient availability, see Watson et al. (2000), Poffenbarger (2010). Note that livestock do not add new or additional nutrients to a system, but their process of digestion means that some nutrients ingested as part of their diet are excreted in more plant-available forms, than would otherwise be the case if that forage decomposed as plant litter rather than passing through a ruminant. However, nutrient cycling by livestock is also leaky due to their uncoupling of the C and N cycles, leading to a risk of environmental pollution, *e.g*. N from urine and manure present in surface runoff or volatised as N_2O (Soussana and Lemaire, 2014, Gerber et al., 2014, Bouwman et al., 2013). Furthermore, more N is now cycled through ruminant livestock due to applications of N fertiliser to pastures (Dungait et al., 2012a).

(28) Author's summary.

(a) Expert's opinion.

(b) Expert's opinion.

(c) Although adopting improved management practices, including livestock grazing, on very degraded soils can achieve high initial rates of SOC sequestration that in some cases may fully mitigate ruminant GHG emissions (in CO_2 equivalents) in the short term (Stanley et al., 2018, Rowntree et al., 2016), such high rates of sequestration are unlikely to be maintained as soil carbon stocks approach a new equilibrium. Only partial mitigation of ruminant livestock emissions is achievable long term from grassland soil carbon sequestration under GWP_{100} accounting (Soussana et al., 2010b). Hammar et al. (2022) use a climate modelling approach to explore temperature impacts over time of livestock emissions and soil carbon sequestration for hypothetical Swedish suckler cow farm under a number of different management scenarios. They demonstrate that soil carbon sequestration could offset 15-22% of the warming caused by the livestock emissions, depending on production intensity and where the system boundary was drawn for emissions accounting purposes. There is some evidence that current domestic ruminant livestock numbers may be approximately 'equivalent' to now-extinct wild ruminant megafauna, in terms of contribution to atmospheric methane levels and therefore warming (Smith et al., 2016).

(29) Semi-natural habitats in the UK which are important for biodiversity include natural and semi-natural grasslands, saltmarsh, sand dune systems, heathlands, grass moorlands, coastal and floodplain grazing marsh, wood pastures, and rush pastures. These also contain significant stores of soil carbon (Field et al., 2020).

(a) Upland peatlands such as blanket bogs are often utilised for livestock grazing, but grazing has, at best, a limited role in maintaining healthy peatlands. Only very low stocking rates are sustainable on bog ecosystems, equivalent to 0.4 sheep per hectare (Lindsay et al., 2014). This figure is based on UK grazing studies such as Hulme and Birnie (1997) and Rawes and Hobbs (1979) that assess blanket bog vegetation under different stocking densities and grazing exclusion, reviewed in Lindsay (2010). Only limited stocking is possible both due to the low productivity of bog ecosystems in terms of supplying forage dry matter for livestock grazing, but also because peatland vegetation is very sensitive to livestock trampling. Key species such as Sphagnum mosses can withstand only very infrequent trampling (less than once or twice per year). Trampling damage results in the loss of peat-forming species and causes the peat to dry out. This increases susceptibility to erosion and can turn bog ecosystems from a carbon sink to a carbon source. Reducing stocking rates to <0.4 sheep/ha, seasonally removing livestock in winter, and grazing smaller breeds of livestock can minimise the risk of further damage and enable degraded peatland to start to recover and carbon storage to increase (Lindsay et al., 2014, Ward et al., 2007). Where peatlands have historically been drained, rewetting these by blocking drains is a key component of restoration but may further reduce their suitability for grazing.

(f) Pasture management and soil carbon

(30) Author's summary.

(31) Mayel et al. (2021) review the mechanisms by which mowing influences SOC. Soussana et al. (2007) calculated the net emissions balance of European grasslands, and found that mown grasslands sequestered soil carbon, but that if the harvested biomass was fed to livestock, accounting for the resulting enteric CH_4 emissions resulted in no net warming reduction overall. Phukubye et al. (2022) meta-analyse nine studies and find that mowing has no impact on soil carbon compared to ungrazed and uncut management. Kramberger et al. (2015) find that cutting frequency does not influence SOC stocks when residue was removed, whereas Poeplau et al. (2016) found that SOC was 12% higher when city lawns were mown eight times a year compared to once a year in Sweden and residues retained.

(32) Reseeding or otherwise rejuvenating pastures to re-establish productive varieties of forage plants has the dual benefit of increasing carbon inputs to the soil and therefore potentially increasing soil carbon stocks (Conant et al., 2001), alongside improving livestock productivity and therefore reducing livestock direct emissions (Hristov et al., 2013a, Herrero et al., 2016). For a study analysing plots, switch between grassland types see Norton et al. (2022). Note, many old species-rich meadows should be retained because of their high biodiversity value.

(33) N availability is frequently a limiting factor in plant growth, therefore N fixation by legumes can increase pasture biomass production by increasing fertility (Jordon et al., 2022b, Suter et al., 2015, Luscher et al., 2014). As such, introducing legumes to pastures is a well-established means to increase SOC stocks (Conant et al., 2017, Henderson et al., 2015, Fornara and Tilman, 2008). In addition, stable mineral-associated organic matter in the soil has a higher carbon to nitrogen content than plant residue inputs to the soil, so additional N inputs from legumes can promote MAOM formation and stable SOC increases (Rumpel et al., 2015). As part of a forage mixture with grasses, legumes increase forage quality including protein content, and therefore boost livestock growth rates (Jordon et al., 2022b), reducing livestock direct emissions per unit of animal product. Increasing N inputs to soils through legumes increases nitrous oxide emissions which negates around 30% of the soil carbon sequestration benefit from legumes (Henderson et al., 2015). However, legumes still have a substantial net C sequestration potential despite this and integrating legumes into swards to fix nitrogen can displace

synthetic fertiliser applications, which both avoids emissions from the energy intensive Haber-Bosch process (Luscher et al., 2014) and reduces soil nitrous oxide emissions for the same forage dry matter production (Murphy et al., 2018) and animal daily liveweight gain (McAuliffe et al., 2018b), also see McAuliffe et al. (2020a) for mechanisms by which legumes may reduce soil N_2O emissions.

(34) For the effect of rooting depth on soil carbon, see Whitehead (2020), Dodd et al. (2011), Kell (2011). Although there is limited evidence in temperate regions for this plausible mechanism, deep rooting grasses have been found to increase SOC at depth in South American savannahs (Fisher et al., 1994). Deeper rooted forage species also increase forage production due to improved water and nutrient uptake, and milk yields due to higher nutritive value, in temperate regions (Jordon et al., 2022b, Cranston et al., 2015, McCarthy et al., 2020). Some herb species have high concentrations of second metabolites in their foliage, such as condensed tannins, which may reduce methane and nitrous oxide emissions from ruminant livestock (see Livestock emissions section of Restatement).

(35) Increasing the number of forage species in pasture in appropriate abundances increases biomass production (Tilman et al., 2001, Hector et al., 1999, Nyfeler et al., 2009, Finn et al., 2013, Weisser et al., 2017) and soil carbon levels (Prommer et al., 2020, Skinner and Dell, 2016, Chen et al., 2018, Cong et al., 2014, Fornara and Tilman, 2008, Steinbeiss et al., 2008, Yang et al., 2019), compared to pastures with one or a small number of species. This is because the presence of multiple species: i) increases the resilience of productivity across varying environmental conditions (Skinner and Dell, 2016, Sanderson et al., 2005), ii) prevents large drops in productivity if a single important species is lost, due to functional redundancy (Weisser et al., 2017), and iii) increases levels of root exudation plus reduces water evaporation due to denser vegetation covering the topsoil, which both promote soil microbial activity (Lange et al., 2015). Deep rooted grass cultivars have also been demonstrated to increase rates of water infiltration into soils, but this potential may be limited due to compaction caused by livestock trampling (Stoate et al., 2021). For advocacy for more unconventional research see Pierret et al. (2016).

(36) For estimates of increased SOC from fertiliser applications, see Conant et al. (2017), Henderson et al. (2015), Eze et al. (2018), Fornara et al. (2016), Kätterer et al. (2013). Henderson et al. (2015) also estimate nitrous oxide emissions and therefore demonstrate overall net emissions from fertiliser application in Western Europe. The Haber-Bosch process required to produce nitrogen fertiliser generates 1.4% of global greenhouse gas emissions (Capdevila-Cortada, 2019).

(37) Storage and application of manure and slurry is a significant source of methane and nitrous oxide emissions (Gerber et al., 2013b, Zhou et al., 2017b), although N_2O emissions from applications may only be significant in wet conditions (Ball et al., 2014).

- (a) Powlson et al. (2012) discuss data from Rothamsted long term experiments in the UK that find that grassland treated with manure is only slightly higher, or no different, in SOC over 100 years than un-manured grassland. Powlson et al. (2012) review UK experiments, Sandén et al. (2018) meta-analysed European long term experiments, and Maillard and Angers (2014) meta-analysed global studies, all finding that manure and slurry increase cropland SOC stocks compared to mineral fertiliser with equivalent N content.
  - (I) Expert's opinion.
- (b) Organic matter in manure typically has an alternative fate (Powlson et al., 2011). For example, straw used for livestock bedding could have been retained on the arable field instead, which benefits cropland SOC (Powlson et al., 2012, Sandén et al., 2018), and cutting grassland to harvest hay or silage for indoor animal feed can reduce SOC compared to leaving biomass unharvested. More broadly, availability of manure is likely to be a key constraint to using this practice for increasing soil carbon stocks at scale (Poulton et al., 2018).
- (c) Options to reduce CH_4 and N_2O emissions from manure are summarised in Box 2 of Gerber et al. (2013b), also Gerber et al. (2013a), Montes et al. (2013), Chadwick et al. (2011), Petersen et al. (2013).
  - (I) Möller (2015) review the impact of anaerobic digestion of manure on nitrogen availability and nitrous oxide emissions. Scott and Blanchard (2021) use IPCC methodologies plus data from a Northern Ireland dairy farm to estimate the impact of anaerobic digestion on capturing CH_4 emissions during storage, but lower N_2O emissions during storage are offset by higher N_2O emissions during application. Scott and Blanchard (2021) estimate the total GHG emissions reduction potential of anaerobic digestion of being 16.6-24%, depending on the system the cows are kept in. Amon et al. (2006) found that anaerobic digestion reduced total CO_2e emissions from storage and application of cattle slurry by ~60%. However, Rodhe et al. (2015) found that anaerobically digested cattle slurry had higher CH_4 emissions during summer storage than non-digested slurry in Sweden. Føreid et al. (2021) find that anaerobic digestion increases plant available N compared to undigested cattle manure when applied to wheat in Sweden.
  - (II) Chadwick et al. (2000) show that trailing shoe and band spreading slurry application also reduce ammonia emissions by 75 and 39%, respectively, compared to splash plate. See Duncan et al. (2017) for evidence of decreased ammonia emissions but increased nitrous oxide emissions from slurry injection compared to broadcast spreading.
  - (III) Gerber et al. (2013a), Montes et al. (2013) review livestock dietary manipulation options to reduce emissions from manure. Lee et al. (2002) demonstrate the importance of ration energy:protein balance on N digestion.

(38) Regarding the impact of liming, see Abdalla et al. (2022) for pasture productivity, Mayel et al. (2021) for soil aggregate stability and Eze et al. (2018) for soil carbon.

- (a) Abdalla et al. (2022) identified insufficient studies to meta-analyse the impact of liming grassland on greenhouse gas emissions, but their qualitative summary of available data concluded that liming either decreases or has no effect on soil N_2O and CH_4 emissions. Although liming can increase CO_2 emissions, particularly when applied in excess, evidence from cropland suggests that the overall effect on net GHG emissions is approximately neutral due to reductions in more potent N_2O and CH_4 emissions (Wang et al., 2021). Conversely, a survey of UK livestock farms found that liming to achieve the recommended pH of 6 would increase CO_2 emissions four times above that saved from reduced N_2O emissions in CO_2-equivalents (Gibbons et al., 2014, Goulding, 2016). Life cycle analyses from Australian cropland estimate pre-farm emissions from production and transport of lime to account for around 7-17% of emissions from crop production. The lower figure is expressed per tonne of wheat by Brock et al. (2012), and the upper per hectare of cropland by Barton et al. (2014).

(g) Indirect effects

(39) Author's summary.

(40) A major risk from decreasing arable crop yields through adopting measures which increase cropland soil carbon stocks (*e.g.* ley-arable rotations (Hu and Chabbi, 2022)) is that this leads to compensatory cultivation of pasture elsewhere, also known as indirect land use change, thus leading to net SOC losses (Powlson et al., 2011, Carlton et al., 2011). Furthermore, globally, deforestation to create pastures and cropland to rear ruminant livestock is a major source of emissions from animal agriculture (McAlpine et al., 2009, Godde et al., 2018). Thus, without accompanying demand-side measures, interventions that decrease UK ruminant livestock production (*e.g.* by creating woodland on pastures) could simply lead to 'offshoring' this production (for example through trade deals with insufficient environmental safeguards) and may contribute to deleterious land use changes such as deforestation overseas (de Ruiter et al., 2017, de Ruiter et al., 2016), thus again resulting in an overall negative net environmental impact.

(a) Expert's opinion.

(b) (Pearson et al., 2017).

(41) For meta-analyses of lifecycle analyses of grass-fed vs grain-fed beef production, see Lynch (2019b), Clark and Tilman (2017), de Vries et al. (2015). For current global livestock cereal consumption and arable land required, see Mottet et al. (2017). Tillgren (2021) estimates that much cereal grown for livestock feed in Sweden is of human food quality. However, small amounts of human-edible feed could be strategically used to complement forage diets to meet specific livestock nutritional requirements (Wilkinson and Lee, 2018).

(a) Papers estimating the meat production possible from 'livestock on leftovers' systems, where ruminants are reared only on grassland and crop by-products, include Karlsson and Röös (2019), Röös et al. (2017), Röös et al. (2016), Van Zanten et al. (2018), Schader et al. (2015).

(h) Policy implications

(42) Around half of total global grassland is estimated to be degraded (Bardgett et al., 2021). The FAO estimate that restoring degraded grasslands could sequester 0.15 — 0.7 Gt CO_2 yr^{-1} depending on the price of carbon credits (and therefore incentive to land managers to cease overgrazing or similar) (Conant, 2010). Conant and Paustian (2002) estimate the sequestration potential from restoring all overgrazed grasslands as 1.65 Gt CO_2 yr^{-1}. Also see Ogle et al. (2004) for management coefficients impacting carbon sequestration. Prevention of desertification by reducing overgrazing in drylands could also deliver significant soil carbon sequestration (Lal, 2001, Oldeman, 1992).

(a) (Smith, 2012), based on mitigation estimates from Smith et al. (2008).

(43) For more information on the issues of reversibility of soil carbon sequestration and permanence of soil carbon gains, see Smith (2005).

(a) Expert's opinion.

(44) Expert's opinion.

(a) See Royal Society Multifunctional Landscape Report (2023).

(b) Expert's opinion.

(45) Author's summary

(a) For potential reductions in greenhouse gas emissions from dietary change, see Willett et al. (2019), Hallström et al. (2015), Aleksandrowicz et al. (2016), Poore and Nemecek (2018). Lee et al. (2021a) demonstrate the risk of potential "meat taxes" resulting in unintended negative consequences, such as through the misallocation of resources on a national scale.

(b) Expert's opinion.

(46) See (4b) above.

(a) Expert's opinion.

(b) Expert's opinion.

(c) Expert's opinion.

(47) DeLonge and Basche (2017) meta-analyse the impact of grazing management practices on soil water infiltration rates, finding that improved grazing practices may improve water infiltration rates. Such alleviation of soil compaction caused by grazing livestock trampling could reduce flood hazard (Dadson et al., 2017a). However, soil compaction and trampling by grazing livestock currently incur significant economic costs in terms of accelerated soil erosion and higher flood hazards (Graves et al., 2015, Benaud et al., 2020). For a discussion of the positive and negative impacts of grazing on water quality, see Bilotta et al. (2007). Cole et al. (2020) review the positive role of grass riparian buffer strips in reducing nutrients and sediments entering water courses. For the importance of livestock in supporting pastoralist livelihoods, see Rota and Sperandini (2009). For a discussion of the importance of grazing livestock to the UK rural economy, heritage and culture, see National Sheep Association (2016) also Natural England (2013). Approximately 90% of the UK's lowland species-rich semi-natural grassland has been lost since the 1940s, predominantly through land use change to cropland or intensification to agriculturally-improved pasture (Bullock and et al, 2011). Schils et al. (2022) systematically review the multifunctionality of European permanent grasslands, and find that reduced management intensity improves biodiversity, climate regulation and water purification outcomes. Schils et al. also highlight the importance of protecting low-input permanent pasture from conversion to other land uses such as cropland in ensuring continued delivery of multiple ecosystem services.

(a) (Dodd et al., 2023, Schirpke et al., 2017).

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Oxford Martin School
University of Oxford
34 Broad Street
Oxford OX1 3BD
United Kingdom

Tel: +44 (0)1865 287430
Email: info@oxfordmartin.ox.ac.uk
Web: http://www.oxfordmartin.ox.ac.uk/
X: @oxmartinschool
LinkedIn: https://www.linkedin.com/company/oxford-martin-school/