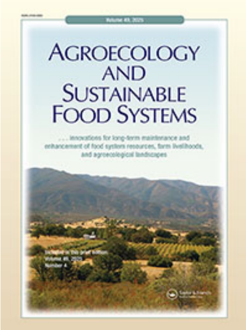


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# Factors affecting the net ecosystem productivity of agroecosystems on mineral soils: a meta-analysis

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**ABSTRACT**

To optimize agricultural land management for soil carbon sequestration, it is necessary to identify whether agroecosystems are accumulating or gaining carbon. This can be done by determining an agroecosystem's net ecosystem productivity (NEP). This study collated data from 40 papers, containing 242 annual measurements of NEP, to assess the impact of climate, soil type and management on the annual NEP of croplands and managed grasslands. Croplands lost significantly more carbon (110 g Cm$^{-2}$) than managed grasslands (29.9 g Cm$^{-2}$) and there was little statistical influence of climate, soil, or management practice on annual NEP. For agroecosystems to sequester carbon, there should be a shift in focus toward implementing management practices that increase carbon retention within agroecosystems.

**KEYWORDS**
Cropland; grassland; management; net ecosystem exchange; soil

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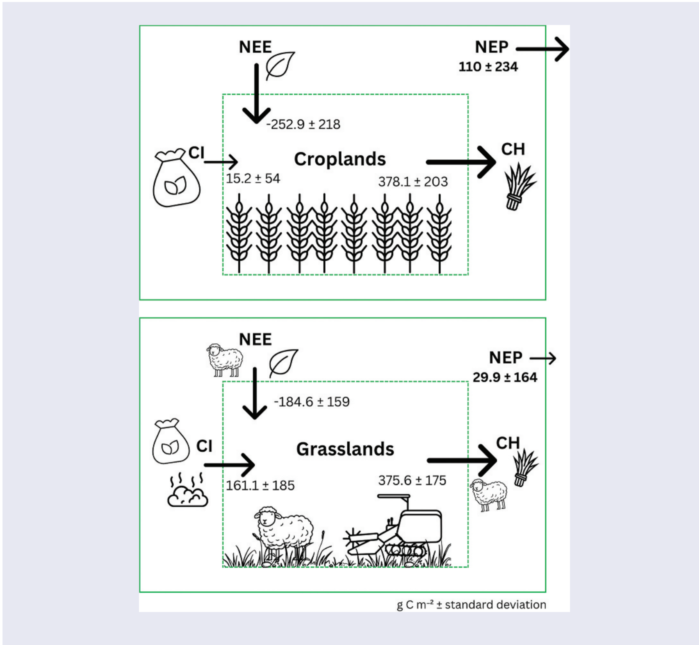


## Introduction

Soil is a major component of the global carbon (C) cycle; the top three meters store around 2500 Gt of soil organic C (SOC), which exceeds that stored by the atmosphere and vegetation combined (Jobbágy and Jackson 2001; Scharlemann et al. 2014). Soil organic C is important for soil structure, nutrient provision and ecosystem functioning (Billings et al. 2021), and can help mitigate against drought by increasing soil water-holding capacity (Iizumi and Wagai 2019). Global land use change, particularly the conversion of non-agricultural land to agricultural land, has led to an estimated loss of 50 Gt C, equivalent to 186 Gt carbon dioxide ($CO_2$), between 1860 and 2020 (Smith et al. 2016). The decline in SOC is due to an increase in the decomposition rate of soil organic matter (SOM) and a decrease in the amount of C being returned to the soil. In agriculture, this can be attributed to tillage, which disturbs and increases the oxygenation of the soil profile, and biomass removal via harvesting or grazing, which reduces the amount of litter returned to the soil (Stavi and Lal 2013). Plants assimilate C during photosynthesis, however increased rates of biomass removal associated with increased yields from agricultural intensification mean that less organic matter, and therefore organic C is being returned to the soil, thus reducing net C storage within agroecosystems (Haberl et al. 2007; Ray and Foley 2013). Furthermore, higher stocking densities, nutrient fertilization and mowing frequency associated with the intensification of livestock farming has increased grass utilization (Manning et al. 2015; Soussana and Lemaire 2014). Severe depletion of the

SOC pool is of global concern, as it degrades soil quality, leading to a decline in soil fertility and crop yield, and an increased reliance on fertilizer application. Such declines in soil health also compromises soil hydraulic functioning – i.e., infiltration, water storage, and runoff – which increase the risk of soil erosion and flooding (Ogle et al. 2019). This can subsequently increase greenhouse gas (GHG) emissions from the agricultural sector, further contributing to anthropogenic climate change.

To meet climate targets, including the UK's aim to achieve net zero GHG emissions by 2050 or earlier (Climate Change Committee 2020), a global reduction in GHGs together with an increase in SOC storage, called "negative emissions" or "$CO_2$ removal," is required. Whilst the agricultural sector currently contributes to climate change, it also has considerable potential to mitigate against it. Policies such as the 4 per 1000 Initiative place a strong focus on the use of agricultural soils for GHG removal via SOC sequestration (Minasny et al. 2017). There are several "climate-smart" farming practices which have been shown to enhance SOC sequestration under certain conditions (Chapman et al. 2018). Such practices include minimal or zero tillage (Nunes et al. 2020), the use of cover crops during intercropping periods (Lugato, Leip, and Jones 2018), greater crop residue retention (Qiu et al. 2020), increasing plant species diversity to include those with deeper roots and greater root mass (Smith 2004), and rotational grazing or mixed agriculture (Albanito et al. 2022). Soil organic C sequestration has additional benefits of improving soil health and food security (Lal 2016), however the rate of sequestration depends on soil texture, soil drainage characteristics, climate, and the length of time that the management practices have been implemented for. To understand where and how agricultural emissions can be reduced and soil C sinks increased, the C sequestration potential of climate-smart management practices across contrasting soils and climate conditions must be evaluated.

To establish whether an ecosystem is acting as a source or sink of $CO_2$, net ecosystem exchange (NEE) is determined as the difference between the $CO_2$ flux assimilated by photosynthesis (gross primary productivity – GPP) and respired from plant and soil processes (total ecosystem respiration – TER) (Eugster and Merbold 2015). The magnitude of GPP and TER are controlled by a combination of crop type, climate, soil type, and management (Davidson and Janssens 2006). Climate conditions and soil texture regulate SOM mineralization; warmer and wetter climates and fine-textured soils create favorable conditions for soil microbial activity and subsequently increase TER (Dilustro et al. 2005; Jäger et al. 2011; Shakoor et al. 2021). Temperature influences crop growth rate and GPP (Baly 1935). Intensively managed grasslands typically have higher SOC stocks than croplands as they have longer periods of vegetation cover and less frequent or intense soil disturbance (Ciais et al. 2010; Guo and Gifford 2002). Vegetation type influences GPP due to variations in photosynthetic rate, phenology, and length of the growing season (Prade, Kätterer, and Björnsson 2017; Wohlfahrt et al. 2008), and TER can be enhanced by greater soil disturbance via intensive tillage (Abdalla et al. 2013; Mohammed et al. 2021). Furthermore, grazed grasslands are likely to have a faster turnover of C than cut grasslands, as non-digestible C is returned to the soil via excreta (Chang et al. 2015).

At the field scale, eddy covariance (EC) flux towers are widely used to determine NEE (Moncrieff et al. 1997). In agroecosystems, however, NEE does not account for lateral C fluxes, which are important for understanding whether a system is accumulating or losing C, and thus its potential to mitigate climate change. Net ecosystem productivity (NEP) provides an estimate of the C sink or source strength of an ecosystem and considers lateral fluxes of C – C imported via organic amendments and livestock excreta, and C exported in harvested or grazed aboveground biomass – as well as NEE (Evans et al. 2021). Alternatively, the net ecosystem C balance (NECB) can be calculated, which accounts for all possible lateral C fluxes (Ciais et al. 2010; Smith et al. 2010). In addition to the lateral fluxes in NEP, NECB considers exported C as dissolved organic and inorganic C in leachate and C in volatile organic emissions, and imported C as dissolved organic and inorganic C in precipitation and C in seeds. Net ecosystem productivity therefore provides a more accessible estimate of whether an agroecosystem is accumulating or losing C, as the lateral C fluxes it considers are considerably larger and easier to measure than those included in NECB (Ceschia et al. 2010; Chapin et al. 2006). Amongst the literature, NEP is reported less frequently than NEE, however it urgently needs to be measured to gain a comprehensive overview of the C sink or source strength of agroecosystems.

How NEP varies as a result of climate, soil type, land use, or the agricultural management practices used is poorly understood, yet without this knowledge it is difficult to identify the practices that promote C sequestration, and this information is urgently needed for effective policy decision-making. To truly understand how agriculture can contribute to increased C sequestration, we first need an appreciation of the net C sink or source strength of agroecosystems from a combination of climates, soil types, and management practices. This study collated published data to (i) assess the impact of climate, soil, and agricultural management (including land use, crop cover, tillage intensity, fertilization, and grassland management) on the annual NEP of global croplands and managed grasslands, and (ii) identify directions for future research.

## Methods

### *Data collection*

Publications were collated from Web of Science (Clarivate 2022) using three separate search terms (Table 1) to conduct a rapid meta-analysis. All

**Table 1.** Overview of search terms used to collate publications. TS is used to specify the topics being searched for.

| Search term | Number of results |
|---|---|
| TS = (Eddy covariance) AND TS = (net ecosystem exchange) AND TS = (agricultur* OR crop* OR grass* OR pasture) | 573 |
| TS = (Eddy covariance) AND TS = (net ecosystem carbon balance) AND TS = (agricultur* OR crop* OR grass* OR pasture) | 52 |
| TS = (Eddy covariance) AND TS = (net ecosystem productivity) AND TS = (agricultur* OR crop* OR grass* OR pasture) | 94 |

publications considered were peer-reviewed journal articles published before September 1st, 2023. The search terms were designed to focus the output of the literature search to identify the most relevant publications for this meta-analysis. The authors acknowledge, however, that due to the specific search terms used (Table 1), some publications containing relevant information may not have been identified by the literature search and subsequently not included in this review. The initial search produced 719 publications. Given the overwhelming evidence that the C source or sink strength of peat is primarily controlled by drainage and the water table level (i.e., lowering the water table of peat soils can effectively reduce $CO_2$ emissions (Evans et al. 2021)), publications that measured C fluxes of agroecosystems on peat were discarded, and thus only those on mineral soil were considered. Additionally, publications measuring C fluxes of agricultural land used to grow perennial grasses for bioenergy production were excluded, as the focus of this analysis is on food and fodder production systems. In instances where some measurements included in a publication fulfilled the criteria and some did not (i.e., multiple sites were measured with some on mineral soil and some on peat, or multiple crops were measured with some grown for bioenergy and some for food), only the measurements from site years that fulfilled the criteria were included.

Each publication was then screened against the following criteria: (1) the publication contained primary data and was not a review or meta-analysis; (2) the publication reported data measured in the field (i.e., results were not taken from an online database); (3) the publication reported NEE, or GPP and TER which could be used to calculate NEE (Equation 1); (4) the publication reported the components necessary to calculate NEP at the field scale (Equation 2) on an annual basis (i.e., measurements were taken over a 365-day period) so that comparisons could be made across sites. If a publication measured data over multiple years, each measurement year was recorded separately; (5) the publication reported a value >0 for C export. This is necessary as, by definition, there will always be C export from a cropland or managed grassland as harvested produce or grazed biomass. Studies that reported crop yield and not C export were excluded, as crop yield alone does not provide an indication of C export, as it may not consider all components of the aboveground biomass removed from the field. A cropland site and cut grassland site could be included if it reported C export (>0) and no C import (i.e., no organic amendments are added), however grazed grassland sites had to report both C export and C import (>0) to be included as there would be an import of C via livestock excreta; (6) the publication used EC to measure annual NEE or GPP and TER (i.e., not chambers or the flux gradient method); (7) the study site is either a cropland growing a food or fodder crop, or a managed grassland (i.e., cut for fodder, grazed, or both cut and grazed); (8) the publication includes information on soil texture or reports the sand, silt and clay content so that soil type could be calculated (Table 2); (9) the publication specifies the crop or vegetation type grown during the measurement period; (10) the publication presents annual NEE (or GPP and TER), C export, and C import (if applicable) in a numeric format; (11) the publication is written in or has been translated into English.

Occasionally, identical measurements were reported across multiple publications and so only one measurement per study site per year was recorded to avoid duplication.

## *Data extraction*

The screening activity identified 40 publications from which relevant data were extracted to compile a database of 242 annual NEP measurements and associated meta-data (Tables A1 and A2). Data were digitized manually from tables or from within the text.

Where Köppen climate classification was not reported, this information was extracted from mindat.org (Hudson Institute of Meteorology 2022) based on the latitude and longitude of the study location. Where soil texture was not reported it was estimated using the sand, silt, and clay percentages provided within the publication using a soil texture calculator (United States Department of Agriculture 2017). Each observation was then given a corresponding soil classification based on its textural class according to Hill et al. (2018) (Table 2). Irrigation management was not included as a management practice within the meta-analysis, as irrigation management was only acknowledged by 10 of the 40 papers and the irrigation amount reported by 8 of these 10. Therefore, the requirement for irrigation management data would have significantly limited the size of the dataset. Soil organic C content was not included as a potential driver of the annual NEP as it was reported by only 12 of the 40 papers and thus would have significantly limited

**Table 2.** Soil classification as described by Hill et al. (2018).

| Soil texture or type | Soil classification |
|---|---|
| Loam, loamy sand, sandy, sandy loam, silt, silt loam | Light |
| Clay loam, sandy clay loam, silty clay, silty clay loam | Medium |
| Clay, sandy clay | Heavy |

the size of the dataset had it been a requirement. In addition, the quantities of phosphorus (P) and potassium (K) applied as fertilizer were not considered as this data was reported by a limited number of papers. It is acknowledged that P and K fertilization have an impact on C sequestration (e.g., Chaplot and Smith 2023); however, P and K management were not reported by a sufficient number of papers to be included in the analysis. Furthermore, very few papers reported grazing intensity or the number of cuts for the managed grasslands ($N = 9$ and $N = 7$ respectively). These variables were therefore not included as potential drivers of annual NEP, as the small sample sizes for each group would be insufficient for robust analysis.

Where annual NEE was not explicitly reported in the publication, but annual GPP and TER were, it was calculated as follows:

$$NEE = GPP - TER \tag{1}$$

where GPP is gross primary productivity and TER is total ecosystem respiration. The micrometeorological sign convention is used for annual NEE; a positive NEE indicates that $CO_2$ is lost from the agroecosystem to the atmosphere, and a negative NEE indicates a net uptake of $CO_2$ from the atmosphere by the agroecosystem (Baldocchi 2003).

Annual NEP was calculated as follows (adapted from Evans et al. 2021):

$$NEP = NEE + C_H - C_I \tag{2}$$

where NEE is annual net ecosystem exchange, $C_H$ is the C in harvested and/or grazed biomass and $C_I$ is the C in imported organic amendments such as farmyard manure and/or excreta returned to grassland by grazing livestock. As in Evans et al. (2021), we use the micrometeorological sign convention for annual NEP, where a positive NEP indicates the agroecosystem is losing C and a negative NEP indicates the agroecosystem is accumulating C.

For each annual NEP measurement, information on the climate, soil type, and agricultural management practices used during the measurement period were recorded into categories and groups to understand their effects on annual NEP (Table 3). For analysis purposes, the amount of nitrogen (N) fertilizer added was converted from a continuous to a categorical variable, with categories increasing in 100 kg N ha$^{-1}$ yr$^{-1}$ increments. Where applicable, data were converted into standardized units to enable comparison between studies (i.e., components of the annual NEP converted to g Cm$^{-2}$ and N fertilizer rate to kg N ha$^{-1}$ yr$^{-1}$). For data classified as cropland, the crop type (i.e., annual or perennial) was assigned based on the crop grown during the measurement period – if the crop lived for only one growing season it was classified as annual, however if the crop was able to regrow it was classified as perennial (Figure A1).

## Data analysis

Data were analyzed using The R Language and Environment for Statistical Computing V4.1.3 (The R Foundation for Statistical Computing 2021). To determine the effect of environmental and management factors (Table 3) on annual NEP, we conducted tests for statistically significant differences between the annual NEP of climate and soil type, and management groups. First, normality tests were conducted using the Shapiro–Wilk method. Tests for statistically significant differences between groups within categories were conducted using independent t-tests, Wilcoxon tests, Kruskal Wallis tests, Dunn's tests, one-way ANOVA or Tukey tests as appropriate, depending on the normality of the data and the number of groups being compared.

Mixed effects models were used to assess the variable importance of climate, soil and management practices on the annual NEP of croplands and managed grasslands. As the model requires complete cases of data, data where one or more of the variables of interest were not reported by the publication were removed. The size of the croplands dataset for analysis was $N = 75$ and for

**Table 3.** Categories and groups used to classify data.

| Data | Category | Groups |
|---|---|---|
| Croplands and managed grasslands | Agricultural land use | Cropland |
| | | Managed grassland |
| | Köppen climate classification | Aw: Wet tropical savannah |
| | | BSk: Cold semi-arid (steppe) |
| | | BWk: Cold desert |
| | | Cfa: Humid subtropical |
| | | Cfb: Temperate oceanic |
| | | Csb: Warm-summer Mediterranean |
| | | Cwa: Monsoon-influenced humid subtropical |
| | | Dfa: Hot-summer humid continental |
| | | Dfb: Warm-summer humid continental |
| | | Dfc: Subarctic |
| | | Dwa: Monsoon-influenced hot-summer humid continental |
| | Amount of N fertiliser added (kg N $ha^{-1}$ $yr^{-1}$) | 0 |
| | | 1–100 |
| | | 101–200 |
| | | 201–300 |
| | | 301–400 |
| | | >401 |
| | Soil type | Light |
| | | Medium |
| | | Heavy |
| Croplands only | Inclusion of cover crops | Yes |
| | | No |
| | Crop type | Annual |
| | | Perennial |
| | Residues retained | Yes |
| | | No |
| | Tillage | Conventional tillage |
| | | Reduced tillage |
| | | No-till |
| Managed grasslands only | Management | Cut |
| | | Grazed |
| | | Cut + grazed |

managed grasslands was $N = 98$. As the datasets contained some data that was collected from the same site over multiple years, the site and measurement year were included as random effects in the model. Environment and management variables were included as fixed effects in the model; for croplands the fixed effects were as follows: Köppen climate classification, soil type, amount of N fertilizer added, inclusion of cover crops, residue retention and tillage method, and for managed grasslands the fixed effects were as follows: Köppen climate classification, soil type, management method and amount of N fertilizer added. Crop type (i.e., annual or perennial) was not included as a fixed effect in the croplands model as data were from sites growing annual crops only once incomplete cases had been removed.

A correlation matrix was used to assess the relationship between the fixed-effects variables used in the mixed-effects model, for croplands and grasslands data separately. First, the data was converted to dummy variables using one-hot encoding, and this encoded data was used to produce a correlation matrix. Spearman correlation was used due to the variables being a combination of ordinal and nominal data and significance to a level of 0.05 was determined. Commentary on the relationships between the fixed-effects variables is limited due to the small sample sizes of the croplands and grasslands datasets, meaning that robust conclusions based on this data are not possible.

## Results

### *Overview of the dataset*

A total of 242 individual annual NEP measurements and corresponding metadata were obtained from the 40 publications (Tables A.1 and A.2): N = 141 for croplands and N = 101 for managed grasslands. The measurements were from a total of 11 countries with the majority from the USA and Germany (Tables A1, A2, and A3); compared to temperate regions, tropical regions were underrepresented. Of the 40 publications: 5 measured the annual NEP of one field for 1 year; 12 measured the annual NEP of one field over multiple years; 6 measured the annual NEP of multiple fields over 1 year; and 17 measured the annual NEP of multiple fields over multiple years. Very few of the studies within our dataset were designed to specifically test the influence of environmental conditions or management practices on annual NEP. Annual NEP values ranged from 764.8 g $Cm^{-2}$ (highest C loss) for an annual cropland growing a cover crop, silage maize and winter wheat in Germany with a temperate oceanic climate (Cfb) and light soil (silt loam) receiving no organic amendments (Poyda et al. 2019) to −499 g $Cm^{-2}$ (highest C gain) for a cut grassland in Japan with a warm-summer humid continental climate (Dfb) and light soil (silt loam) receiving 770 g $Cm^{-2}$ of organic amendments (Hirata et al. 2013). The mean (± standard deviation) annual NEP across the dataset was $76.6 \pm 211$ g $Cm^{-2}$. Graphical summaries of the annual NEP of croplands and managed grasslands grouped by Köppen climate classification, soil type, and agricultural management are presented in Figures 1 and 2.

### *Annual NEP of arable croplands and managed grasslands*

A t-test showed a significant difference between the mean annual NEP (± standard deviation) of croplands ($110 \pm 234$ g $Cm^{-2}$) and managed grasslands ($29.9 \pm 164$ g $Cm^{-2}$) ($p = 0.02$). The annual NEP of croplands had a greater range than of managed grasslands (Figure 3). For both land uses, there were more sites with positive annual NEPs than negative (i.e., most sites were losing C); there were a greater proportion of croplands with a positive annual NEP (69%) than managed grasslands (65%).

A t-test showed the mean annual *in-situ* NEE (± standard deviation) of croplands ($-252.9 \pm 218$ g $Cm^{-2}$) was significantly more negative than that of managed grasslands ($-184.6 \pm 159$ g $Cm^{-2}$) ($p = 0.005$); more atmospheric $CO_2$ was being taken up by croplands than managed grasslands during periods of active growth. A Wilcoxon test showed that the mean C imported (± standard deviation) was significantly lower, by around 10 times, for croplands ($15.2 \pm 54$ g $Cm^{-2}$) than for managed grasslands ($161.1 \pm 185$ g $Cm^{-2}$) ($p = <0.001$). The mean C exported (± standard deviation) from croplands ($378.1 \pm 203$ g $Cm^{-2}$) was similar to that from managed grasslands ($375.6 \pm 175$ g $Cm^{-2}$) ($p = 0.73$). The mean amount of C exported from croplands was considerably greater than the mean annual $CO_2$ being assimilated as NEE and the mean C imported via organic amendments, so mean annual NEP was positive and there was overall C loss. The mean amount of C exported from managed grasslands, however, was similar to the mean $CO_2$ that was assimilated as NEE and the mean C imported via organic amendments and excreta from grazing livestock, so NEP was close to neutral.

### *Environmental drivers of annual NEP*

#### *Climate*

The majority of the annual NEP measurements in our dataset (95%) were from temperate and continental climate zones. Standard deviation of mean annual NEP was high for most climatic zones, ranging from 28 to 407 g $Cm^{-2}$ (Table 4), as the sample size of each Köppen climate zone was highly variable. For croplands, the mean annual NEP (± standard deviation) of sites with a warm-summer Mediterranean (Csb) climate ($-119.7 \pm 177$ g $Cm^{-2}$) was significantly lower than that of sites with a warm-summer humid continental (Dfb) climate ($326 \pm 312$ g $Cm^{-2}$) ($p = 0.01$) and a Monsoon-influenced hot-summer humid continental (Dwa) climate ($537.3 \pm 48$ g $Cm^{-2}$) ($p = 0.0007$); and the mean annual NEP (± standard deviation) of sites with a hot-summer

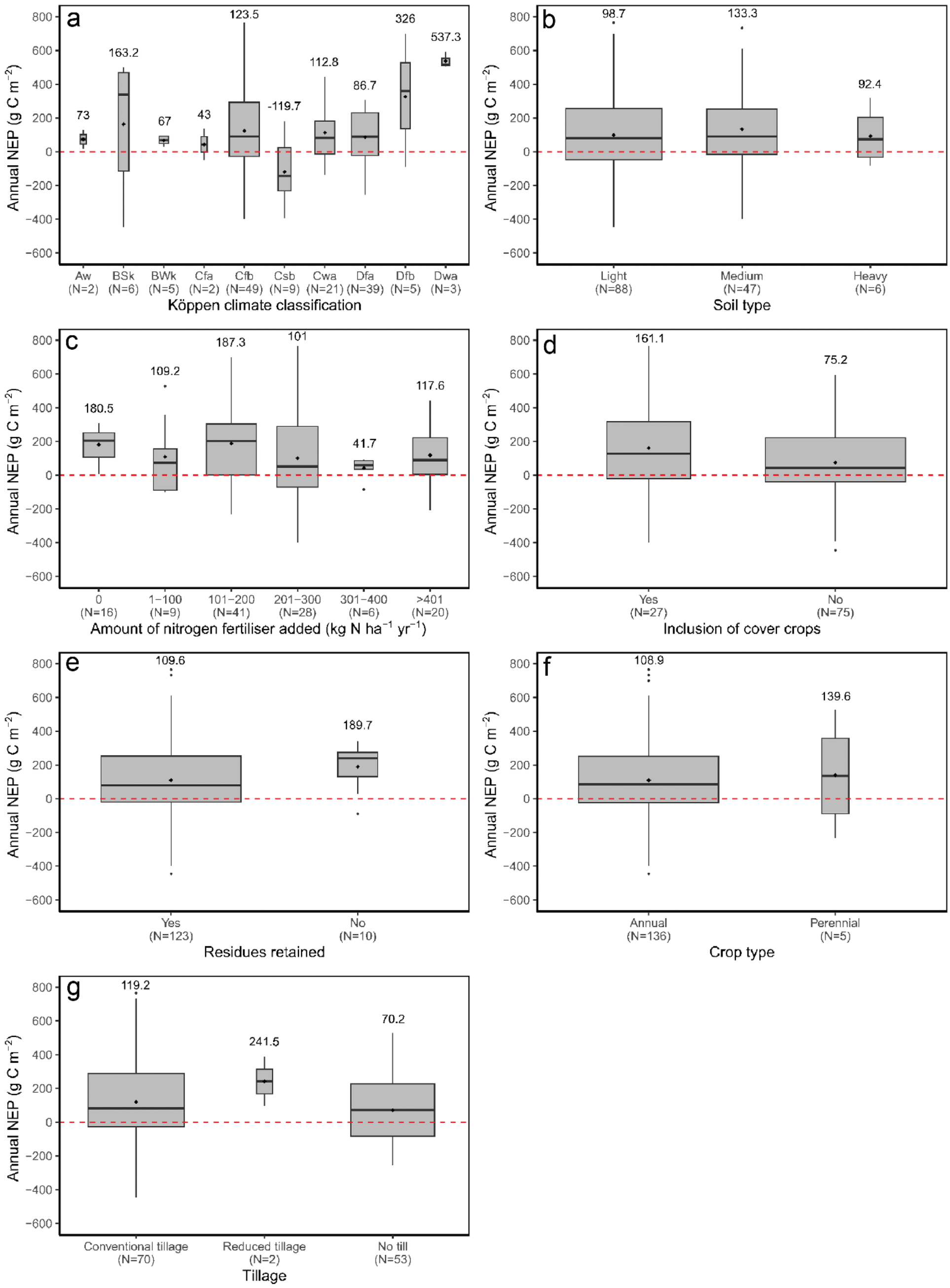


**Figure 1.** Boxplots summarising the annual NEP database for croplands, displaying the range of annual NEP measurements grouped by: Köppen climate classification (a), soil type (b), amount of N fertiliser added (c), use of cover crops or not (d), crop residue retention or removal (e), crop type (i.e., annual or perennial) (f) and type of tillage (g). *N*= indicates the number of observations within each group, and C, D, E and G only display data from observations that reported information on that category. The width of each boxplot is proportional to the number of samples in each group; the diamond within each box and the value associated with each box represent the mean of the group. Positive values indicate C loss from and negative values indicate C accumulation within the agroecosystem. See Tables 4, 5, 6 and A1 for further information.

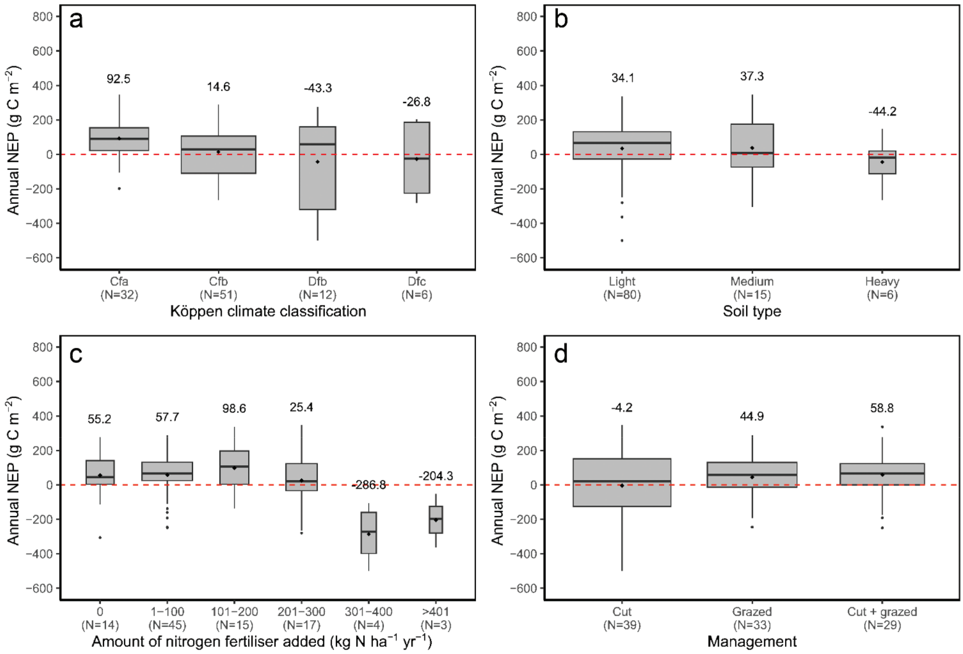


**Figure 2.** Boxplots displaying the range of annual NEP measurements grouped by: Köppen climate classification (a), soil type (b), amount of N fertiliser added (c) and grassland management (d). *N*= indicates the number of observations within each group, and C only displays data from observations that reported information on that category. See Tables 4, 5, 7 and A.2 for further information.

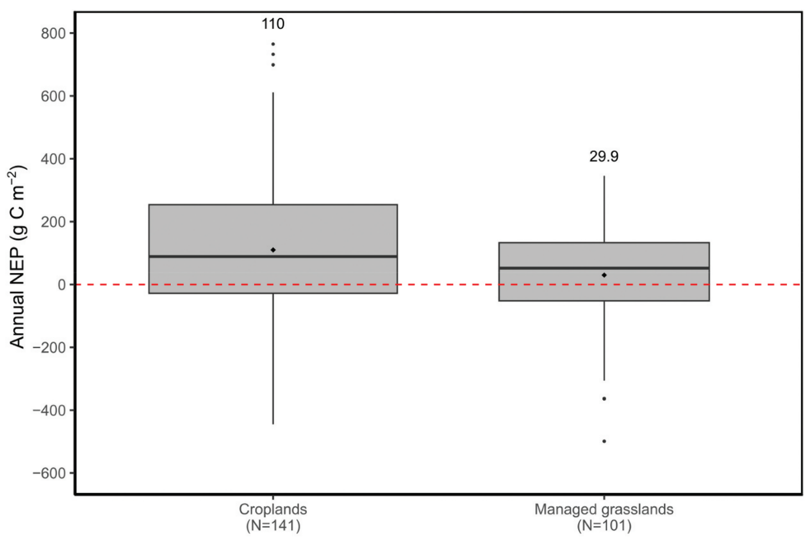


**Figure 3.** Range of annual NEP measurements for croplands and managed grasslands. The width of each boxplot is proportional to the number of samples in each group; the diamond within each box and the value associated with each box represent the mean of the group.

**Table 4.** Mean annual NEP ± standard deviation (SD), the proportion of sites with positive and negative annual NEP measurements, and an indication of significant differences between the mean annual NEPs of Köppen climate classification groups for the croplands and managed grasslands data. *N*= indicates the number of observations within each group.

| | Köppen climate classification | N= | Mean annual NEP ± SD (g $Cm^{-2}$) | % positive observations | % negative observations | Significant difference |
|---|---|---|---|---|---|---|
| Croplands | Aw: Wet tropical savanna | 2 | 73 ± 78 | 100 | 0 | Between groups ($p$=0.002): Csb and Dfb ($p$=0.01), Csb and Dwa ($p$=0.0007), Dfa and Dwa ($p$=0.03) |
| | BSk: Cold semi-arid | 6 | 163.2 ± 407 | 67 | 33 | |
| | BWk: Cold desert | 5 | 67 ± 28 | 100 | 0 | |
| | Cfa: Humid subtropical | 2 | 43 ± 132 | 50 | 50 | |
| | Cfb: Temperate oceanic | 49 | 123.5 ± 265 | 65 | 35 | |
| | Csb: Warm-summer Mediterranean | 9 | −119.7 ± 177 | 33 | 67 | |
| | Cwa: Monsoon-influenced humid subtropical | 21 | 112.8 ± 161 | 71 | 29 | |
| | Dfa: Hot-summer humid continental | 39 | 86.7 ± 158 | 72 | 28 | |
| | Dfb: Warm-summer humid continental | 5 | 326 ± 312 | 80 | 20 | |
| | Dwa: Monsoon-influenced hot-summer humid continental | 3 | 537.3 ± 48 | 100 | 0 | |
| Managed grasslands | Cfa: Humid subtropical | 32 | 92.5 ± 124 | 81 | 19 | None ($p$=0.15) |
| | Cfb: Temperate oceanic | 51 | 14.6 ± 138 | 59 | 41 | |
| | Dfb: Warm-summer humid continental | 12 | −43.3 ± 265 | 58 | 42 | |
| | Dfc: Subarctic | 6 | −26.8 ± 230 | 50 | 50 | |

humid continental (Dfa) climate (86.7 ± 158 g $Cm^{-2}$) was significantly lower than that of sites with a Monsoon-influenced hot-summer humid continental (Dwa) climate (537.3 ± 48 g $Cm^{-2}$) ($p = 0.03$). Köppen climate classification was identified by the mixed-effects model as the only variable significantly influencing the annual NEP of croplands (Figure A.2). The Csb climate zone was the only group with a negative mean annual NEP, as 67% of these sites were accumulating C; all other climate zones had a greater proportion of sites with positive mean annual NEPs than negative, indicating that most of these sites lost C. The managed grasslands sites covered fewer Köppen climate zones than the croplands (Table 4). There were no statistically significant differences between the mean annual NEPs of any of the Köppen climate zones ($p = 0.15$), and the mixed effects model showed that Köppen climate classification had no significant effect on the NEP of managed grasslands (Figure A.2). Mean annual NEP was positive for sites in temperate climates (Cfa and Cfb) and negative for sites in subtropical climates (Dfb and Dfc); there were a greater proportion of sites that lost C in temperate climates than subtropical climates.

### *Soil*

Most of the data (69%) were from sites with light soil (i.e., well-drained, high sand content); sites with heavy soil (i.e., poorly drained, high clay content) were underrepresented in our dataset (Table 5). For most soil types, standard deviation of mean annual NEP was high, ranging from 143 to 238 g $Cm^{-2}$. Soil type had no significant influence on the annual NEP of croplands or managed grasslands (Figure A.2); no significant differences were observed between the mean annual NEPs of croplands ($p = 0.71$) or managed grasslands ($p = 0.32$) when grouped by soil type. For croplands, mean annual NEP was positive for all soil types and there were a greater proportion of sites with positive annual NEPs than negative. Mean annual NEP was negative for managed grassland sites with heavy soil and positive for managed grassland sites with light and medium soils; most managed grasslands with heavy soil accumulated a small

**Table 5.** Mean annual NEP ± standard deviation (SD), the proportion of sites with positive and negative annual NEP measurements, and an indication of significant differences between the mean annual NEP of soil-type groups for the croplands and managed grasslands data. See Table 2 for soil-type classification. *N*= indicates the number of observations within each group.

| | Soil type | N= | Mean annual NEP ± SD (g $Cm^{-2}$) | % positive observations | % negative observations | Significant difference |
|---|---|---|---|---|---|---|
| Croplands | Light | 88 | 98.7 ± 237 | 68 | 32 | None |
| | Medium | 47 | 133.3 ± 238 | 70 | 30 | ($p$ = 0.71) |
| | Heavy | 6 | 92.4 ± 159 | 67 | 33 | |
| Managed grasslands | Light | 80 | 34.1 ± 162 | 85 | 15 | None |
| | Medium | 15 | 37.3 ± 182 | 53 | 47 | ($p$ = 0.32) |
| | Heavy | 6 | −44.2 ± 143 | 33 | 67 | |

amount of C, whereas those with light or medium soil lost a small amount of C.

## The influence of agricultural management practices on annual NEP

### Croplands

Mean annual NEP (± standard deviation) was not significantly different between croplands as a result of the amount of N fertilizer added, the inclusion of cover crops, residue retention, crop type (i.e., annual or perennial) or tillage method ($p = >0.05$) (Table 5). None of these variables had a significant influence on annual NEP (Figure A.2). All management practices had a greater proportion of sites with a positive mean annual NEP than negative; standard deviation of mean annual NEP was high, ranging from 67 to 312 g $Cm^{-2}$ (Table 5).

### Managed grasslands

Significant differences in mean annual NEP were observed between managed grasslands as a result of the amount of N fertilizer added ($p = <0.05$) but not as a result of the grassland management practice used ($p = 0.5$) (Table 6). Mean annual NEP was significantly higher from sites fertilized with 1–100 kg N $ha^{-1}$

**Table 6.** Mean annual NEP ± standard deviation (SD), the proportion of sites with positive and negative annual NEP measurements, and an indication of significant differences between the mean annual NEP of management practices for the croplands data. *N*= indicates the number of observations within each group. Papers classified as "unknown" are those where the amount of N fertilizer, inclusion of cover crops, residue retention or tillage method is not reported and thus statistics cannot be calculated.

| | | N= | Mean annual NEP ± SD (g $Cm^{-2}$) | % positive observations | % negative observations | Significant difference |
|---|---|---|---|---|---|---|
| Amount of N fertiliser added (kg $ha^{-1}$ $yr^{-1}$) | 0 | 16 | 180.5 ± 95 | 100 | 0 | None ($p$ = 0.45) |
| | 1–100 | 9 | 109.2 ± 217 | 67 | 33 | |
| | 101–200 | 41 | 187.3 ± 238 | 76 | 24 | |
| | 201–300 | 28 | 101 ± 292 | 54 | 46 | |
| | 301–400 | 6 | 41.7 ± 67 | 83 | 17 | |
| | >401 | 20 | 117.6 ± 178 | 75 | 25 | |
| | Unknown | 21 | | | | |
| Inclusion of cover crops | Yes | 27 | 161.1 ± 295 | 67 | 33 | None ($p$ = 0.17) |
| | No | 75 | 75.2 ± 213 | 68 | 32 | |
| | Unknown | 39 | | | | |
| Residues retained | Yes | 123 | 109.6 ± 233 | 69 | 31 | None ($p$ = 0.27) |
| | No | 10 | 189.7 ± 135 | 90 | 10 | |
| | Unknown | 8 | | | | |
| Crop type | Annual | 136 | 108.9 ± 232 | 69 | 31 | None ($p$ = 0.77) |
| | Perennial | 5 | 139.6 ± 312 | 60 | 40 | |
| Tillage | Conventional tillage | 70 | 119.2 ± 267 | 66 | 34 | None ($p$ = 0.37) |
| | Reduced tillage | 2 | 241.5 ± 204 | 100 | 0 | |
| | No till | 53 | 70.2 ± 188 | 66 | 34 | |
| | Unknown | 16 | | | | |

**Table 7.** Mean annual NEP ± standard deviation (SD), the proportion of sites with positive and negative annual NEP measurements, and an indication of significant differences between the mean annual NEP of management practices for the managed grasslands data. *N* = indicates the number of observations within each group. Papers classified as "unknown" are those where the amount of N fertilizer is not reported and thus statistics cannot be calculated.

| | | N= | Mean annual NEP ± SD (g $Cm^{-2}$) | % positive observations | % negative observations | Significant difference |
|---|---|---|---|---|---|---|
| Amount of N fertiliser added (kg N $ha^{-1}$ $yr^{-1}$) | 0 | 14 | 55.2 ± 149 | 71 | 29 | Between groups ($p$ = <0.006): 1–100 and 301–400 ($p$ = 0.04), 101–200 and 301–400 ($p$ = 0.02) |
| | 1–100 | 45 | 57.7 ± 119 | 80 | 20 | |
| | 101–200 | 15 | 98.6 ± 148 | 73 | 27 | |
| | 201–300 | 17 | 25.4 ± 181 | 53 | 47 | |
| | 301–400 | 4 | −286.8 ± 179 | 0 | 100 | |
| | >401 | 3 | −204.3 ± 156 | 0 | 100 | |
| | Unknown | 3 | | | | |
| Management | Cut | 39 | −4.2 ± 204 | 54 | 46 | None ($p$ = 0.5) |
| | Grazed | 33 | 44.9 ± 123 | 73 | 27 | |
| | Cut + grazed | 29 | 58.8 ± 139 | 72 | 28 | |

$yr^{-1}$ (57.7 ± 119 g $Cm^{-2}$) and 101–200 kg N $ha^{-1}$ $yr^{-1}$ (98.6 ± 148 g $Cm^{-2}$) than with 301–400 kg N $ha^{-1}$ $yr^{-1}$ (−286.8 ± 179 g $Cm^{-2}$) (p = 0.04 and p = 0.02 respectively). The amount of N fertilizer applied had the greatest (and only significant) influence on the annual NEP of managed grasslands (Figure A.2). Mean annual NEP was positive for most of the management practices – excluding those fertilized with 301–400 and >401 kg N $ha^{-1}$ $yr^{-1}$, and those that were cut. Standard deviation of mean annual NEP was high across all groups, ranging from 119 to 204 g $Cm^{-2}$.

## Discussion

This study compiled data from 40 publications that measured land-atmosphere and lateral C fluxes to evaluate how environmental conditions and management practices control the annual NEP of agroecosystems; the dataset comprised a total of 242 individual annual NEP measurements and associated meta-data. The mean annual NEP (± standard deviation) of our dataset was slightly positive (76.6 ± 211 g $Cm^{-2}$), although the standard deviation of the mean was high, which reflects the large range of values. Sixty-seven percent of the sites in our dataset had a positive annual NEP (69% of cropland sites and 65% of managed grassland sites), confirming that on average these agroecosystems lost C, as found by Smith et al. (2007). The mean annual NEP (± standard deviation) of croplands (110 ± 234 g $Cm^{-2}$) was significantly higher than that of managed grasslands (29.9 ± 164 g $Cm^{-2}$); croplands lost over 3.5 times more C than managed grasslands. Our results are similar to those reported by Ceschia et al. (2010), who found that European crop sites lost, on average, 138 ± 239 g $Cm^{-2}$ $year^{-1}$ and that 70% of sites within their dataset lost C. Based on this C loss, they predict that 2% of SOC content is being lost from European croplands annually (Ceschia et al. 2010). Our results show that the implementation of best management practices made no statistical difference to the NEP of croplands and that the NEP of the managed grasslands was only significantly influenced by N fertilizer rate.

Mean annual NEE was negative for both agroecosystems, though the *in-situ* uptake of $CO_2$ was greater for croplands than managed grasslands. This was compensated for by the significantly greater mean annual C import to managed grasslands, which was around ten times greater than that to croplands. The mean annual C export was similar from and accounted for the largest proportion of mean annual NEP in both agroecosystems. For the croplands, the mean annual C export was considerably greater than the C added to the system (via plant photosynthesis and organic amendments), meaning that, on average, croplands lost C. For managed grasslands, the mean annual C export was only slightly higher than the C imported to the system (via plant photosynthesis, organic amendments and excreta), however, meaning that overall managed grasslands were near C-neutral and lost only a small amount of C.

Multiple studies have proposed that soil C loss is greater from croplands compared to managed grasslands, which tend to accumulate C or be C-neutral (Altimir et al. 2016; Prescher, Grünwald, and Bernhofer 2010). Croplands typically experience greater soil disturbance via tillage and the inclusion of bare soil or fallow periods within annual crop rotations, both of which have been shown to increase $CO_2$ emissions (Ciais et al. 2010; Jansson et al. 2021; Oertel et al. 2016) and NEP. We found that, on average, croplands did lose more C than managed grasslands, although was not solely attributed to the influence of management practices on NEE, as suggested above, and instead was largely influenced by the amount of C imported to the agroecosystem. Furthermore, there is a large potential for uncertainty when calculating C exports and C imports, which is larger than the uncertainty associated with NEE measurement by EC (Ceschia et al. 2010); this was likely to be a factor contributing to the large variation in our results.

## Environmental drivers of NEP

### Climate

Köppen climate classification was the only variable, of those we considered, shown to have a significant influence on the mean annual NEP of croplands. Croplands with a warm-summer Mediterranean (Csb) climate accumulated three times as much C, on average, than those with a warm-summer humid continental (Dfb) climate, and five times as much as those with a monsoon-influenced hot-summer humid continental (Dwa) climate, both of which lost C. Contradictorily, managed grasslands with temperate climates (Cfa and

Cfb), on average, lost C, while managed grasslands with subtropical climates (Dfb and Dfc) accumulated C, although the differences in mean NEP were not significant. Subtropical climates are usually warmer than temperate climates, and agroecosystems in warmer regions have been observed to have higher rates of microbial activity, SOM decomposition, and TER, and subsequently a higher NEE and NEP (Bandaru 2022; López-Garrido et al. 2014; Maia et al. 2019). Other studies have observed higher C loss from croplands and managed grasslands in warmer climates compared to those in colder climates (Waldo et al. 2016).

### *Soil*

Soil type had no statistical influence on the mean annual NEP of the croplands or managed grasslands within our dataset. It is notable, however, that the proportion of managed grassland sites that accumulated C increased with increasing soil clay content; on average managed grasslands with light and medium soils lost C, whereas those with heavy soils accumulated C. Clay particles protect SOC from decomposition, and it has been observed that soils with a higher clay content have lower $CO_2$ emission compared to lighter soils (Béziat, Ceschia, and Dedieu 2009; Li et al. 2010; Maia et al. 2019; Mangalassery et al. 2015; Prout et al. 2022), which can increase NEP (i.e. reduce overall C loss). The majority of the sites in our dataset were on light soil, and so the lack of significant difference in mean annual NEP between the soil types can probably be explained by the small number of sites with heavy and medium soils.

## *The influence of management practices on annual NEP*

The cropland sites in our dataset spanned a variety of crop types (see Crop Species in Table A.1) and management practices, although due to the spatial disparity in our dataset were dominated by crops grown in Europe and North America. The managed grassland sites were dominated by multi-species mix, which predominantly consisted of ryegrass, and were either managed for cutting, grazing, or both cutting and grazing.

None of the management practices we considered – crop type (i.e., annual or perennial), residue management (i.e., retention or removal), the inclusion of cover crops, the amount of N fertilizer added, or the tillage method – had a statistical influence on the annual NEP of croplands. For the managed grasslands, the amount of N fertilizer added had a statistically significant influence on the mean annual NEP, however the grassland management method (i.e., cut, grazed or cut + grazed) did not.

### Croplands

The mean annual NEP of the croplands was not significantly influenced by the type of tillage, crop type (i.e., annual or perennial), retention of crop residues, the inclusion of cover crops, or the amount of N fertilizer added, suggesting that the adoption of other best management practices, such as increasing the quantity of C imports, may have greater success in reducing C losses. Relative to conventional tillage, no till aims to reduce SOM decomposition and soil $CO_2$ losses by disturbing the soil structure less (Olson, Lang, and Ebelhar 2005; Smith 2004; Stavi and Lal 2013). Numerically, our results evidence this, as sites managed with conventional tillage lost more C than those managed with no till, although the difference was not significant. Tillage practices and crop residue management are often interlinked, with no till and crop residue retention often promoted in conservation agriculture to improve soil health (Farhate et al. 2019). Crop residues that are left on the field can be incorporated into the soil with tillage or left on the soil surface if no till is adopted (Fernández, Sorensen, and Villamil 2015) and can improve soil quality by reducing erosion and providing an input of organic C (Nunes et al. 2020; Oertel et al. 2016). There is a large consensus across the literature, however, that retaining crop residues, regardless of the tillage method used, can increase $CO_2$ emissions (Brye, Longer, and Gbur 2006; Sainju, Jabro, and Caesar-TonThat 2010): combining crop residue retention with conventional tillage can oxidize older SOC and release it as $CO_2$ (Ruan and Robertson 2013; Ussiri and Lal 2009; Wegner et al. 2018), whereas retaining residues and using no till leaves biomass to decompose on the soil surface, where is becomes more available to microorganisms for use as a substrate for priming, and is then released as $CO_2$ (Mangalassery et al. 2015; Wegner et al. 2018). Our results corroborate this; the croplands sites in our dataset tended to lose C, and the amount of C lost was not significantly different between sites with residues retained and residues removed. The crop type (i.e., annual or perennial) also had no statistical influence on the variability of annual NEP. Sites growing annual crops often have higher C loss than those growing perennial crops (Amiro et al. 2017; Sarauer and Coleman 2018), as perennial crops have longer growing seasons and extensive root systems which add slowly-decaying C into the soil and increase SOC (Ostle et al. 2009; Pausch and Kuzyakov 2017; Smith 2004). Furthermore, annual cropping systems are associated with more frequent tillage, as the soil is often plowed after harvest, which reduces C sequestration potential (Flynn et al. 2012; Ledo et al. 2020). Our results do not corroborate this, however, although this may be due to the large difference in sample sizes between the annual and perennial sites in our dataset. To improve the understanding of the influence of crop type on annual NEP, further investigation should therefore consider crop type more specifically (i.e., by species (see Crop Species in Table A.1.) or rotation). The literature evaluating the impact of cover crops on C fluxes is conflicting. Cover crops can decrease annual NEP by providing an addition of C to offset some of the C lost at harvest, and can reduce soil erosion and thus $CO_2$ emission (Abdalla et al. 2013; Cates and Jackson 2019). Alternatively, some studies observe higher $CO_2$ emissions from soils with cover crops compared to bare soils (Sanz-Cobena et al. 2014). Cover crop biomass is often left on the soil surface after termination, which is likely to have a similar effect on annual NEP as crop residue retention, increasing C losses as a result of priming (Wegner et al. 2018). The average annual NEP of sites with cover crops shows that these sites lost over twice as much C as those without cover crops, which supports the findings of Abdalla et al. (2013) and Cates and Jackson (2019), although the difference was not significant.

### *Managed grasslands*

Managed grasslands that received over 301 kg N $ha^{-1}$ $yr^{-1}$ gained C on average, whereas those that received less than this lost C. Our findings contradict those of De La Motte et al. (2016) who found lower C losses from a managed grassland in years when less N fertilizer was added, but corroborate those of Hirata et al. (2013) who found C uptake increased with N fertilization rate. In addition, managed grasslands fertilized with 0 kg N $ha^{-1}$ $yr^{-1}$ had just over twice the C loss of those fertilized with 201–300 kg N $ha^{-1}$ $yr^{-1}$, showing greater C loss with lower N fertilization and corroborating the findings of Hirata et al. (2013). Whilst it is accepted that a sufficient soil N supply required for C sequestration in agroecosystems (Dmuchowski, Baczewska-Dabrowska, and Gworek 2022; Flechard et al. 2005; Moinet et al. 2017; Soussana et al. 2007), the relationship between N fertilizer rate and annual NEP has not been widely researched. High rates of N fertilization could increase vegetation growth and photosynthesis, increasing annual $CO_2$ uptake and lowering annual NEP, as found by Liu et al. (2019), but could also result in increased C exports via biomass removal.

When comparing the grassland management method used, cut + grazed grasslands had the highest C losses, followed by grazed grasslands, and cut grasslands had a small uptake of C; all were close to C neutral and not significantly different from one another, however. Rutledge et al. (2015) and Carswell et al. (2019) propose that C export is usually higher from managed grasslands that involve cutting compared to grazing, although C imports may be higher when livestock are present as excreta will be returned to the soil in addition to any organic fertilizer. Concomitantly, the presence of livestock within the EC footprint is likely to increase NEE, as the $CO_2$ respired by grazing animals will be measured by the flux tower (Senapati et al. 2014). These factors may partially explain the numerically higher mean annual NEP from cut + grazed grasslands. Carbon fluxes from managed grasslands are also highly likely to vary as a result of the management intensity (Zeeman et al. 2010) – i.e., stocking density and harvest frequency – however these management practices were only reported by a small number of the managed grasslands studies in our dataset and thus not considered as variables affecting NEP in our statistical model. To further understand the controls on the NEP of managed grasslands, our dataset would therefore benefit from sufficient information on the grazing intensity, grazing species, number of cuts, yield, and amount of C removed with each cut.

It is important to consider the challenges and potential error introduced when calculating C exported as grazed biomass and C imported as livestock excreta. Multiple methods were used to calculate these values across the publications used in this analysis. The C exported via grazing was calculated by multiplying the C content of the grass by either the difference in height of a measured area of grass before and after grazing (De La Motte et al. 2016; Laubach et al. 2019, 2023; Skinner 2008, 2013) or by a standardized pasture utilization value of 0.85 (Rutledge et al. 2017; Wall et al. 2019, 2020, 2023). All publications considered the C imported as excreta as a proportion of the C ingested via grazing, however the proportion itself is variable: Skinner (2008, 2013) assumes 37% of ingested C to be returned as dung; Rutledge et al. (2015) assume this to be 34%; and other studies use a more comprehensive calculation which includes the non-digestible fraction of the grazed biomass and the amount of time livestock spend on the paddock (De La Motte et al. 2016; Laubach et al. 2019, 2023; Rutledge et al. 2017; Wall et al. 2019, 2020, 2023).

Several significant relationships were identified between the fixed effects variables in the croplands and grasslands datasets (Table A.4 and Table A.5 respectively). Due to the low sample size of the datasets, it is not possible to make robust conclusions on the influence of environmental factors on land management practices. A greater amount of data would mean that these relationships could be more reliably and confidently identified and understood.

## Recommendations for future research and policy

The existing research is limited both spatially and temporally, as NEE, and C exports and C imports for the calculation of NEP are not reported consistently across the literature. To provide a more comprehensive and robust understanding of the controls on the annual NEP of agroecosystems, we propose the following recommendations for future research: (i) more measurements from sites in different climates, with different soil types and management practices; (ii) standardized reporting of NEE, C exports and C imports for the calculation of NEP, taking measurements on an annual timescale, and reporting sufficient meta-data to make more direct comparisons between sites. These meta-data should include but not be limited to mean air temperature and total precipitation during the study period, soil texture, SOC content and stock, grassland management, crop or vegetation type (including for managed grasslands), vegetation yield, N fertiliser rate, amount and type of C imported, amount of C exported (i.e., in grain yield and harvested residue), tillage management, grazing species, grazing duration, grazing intensity, the weight of harvested residues, whether cover crops were grown, number of harvests, and any management (i.e., tillage or fertilization) occurring during the non-growing season; (iii) use before-after control-impact type paired studies, such as in Zenone et al. (2013) and Skinner (2013), to provide more direct evidence of how altering management practices could influence NEP (i.e., conventional versus no till, cover crops versus no cover crops, residue retention versus residue removal); (iv) measure SOC at sites where EC is used to measure NEE to directly compare the impacts of management and land use practices, and the relationship between NEE and SOC. This would require longer measurement periods – i.e., 5–10 years – to identify changes to SOC; (v) measure NEP over an entire crop rotation, as also suggested by Ceschia et al. (2010), as C imports may not occur in every year; (vi) to reduce uncertainties in the global GHG balance of croplands, systematically measure other GHG fluxes (i.e., $N_2O$) at the plot scale to update emission factors for a range of field operations (Ceschia et al. 2010; Osborne et al. 2010; Smith et al. 2010); (vii) introduce one standardized method to determine the amount of grass ingested by grazing livestock and the C returned to the soil via excreta.

The agricultural sector would benefit from more targeted policy recommendations as to which agricultural practices will reduce soil C loss; our results show that using no till and growing cover crops do not always necessarily result in soil C gain, and so their effectiveness may be dependent on the environment in which they are grown. Guidance on the combinations of climate, soil type and management practices that are more likely to increase soil C sequestration would help farmers take more targeted action, although much of the ability to do this is dependent on evidence from research that uses the recommendations proposed above. Furthermore, greater communication on the importance of adding organic amendments to agricultural soils to provide an input of C would be beneficial (Bruni et al. 2022), as is currently being done in the UK Sustainable Farming Incentive and the international 4 per 1000 Initiative.

## Conclusion

Our results show that, on average, global agroecosystems are behaving as C sources, despite the implementation of best management practices which are encouraged as methods to increase soil C sequestration. On average, the croplands in our dataset lost C, whereas the managed grasslands were close to C neutral. However, over 65% of all sites in both categories had positive NEP values. There was little influence of climate, soil type, and agricultural management on the NEP of croplands and grasslands, with croplands only being influenced by climate and grasslands only being influenced by N fertilization.

Future research can build on this by more consistent data reporting across the literature and increasing the number of field studies. For agroecosystems to sequester C, there should be a shift in focus toward implementing management practices that increase C retention within agroecosystems, and encourage the addition of C where appropriate.

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