# Impacts of carbon farming practices on biodiversity at the farm scale

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**Key words:** Agroecosystems, Alpha and beta diversity, Biodiversity metrics, Regenerative farming, Soil fauna, Sustainable agriculture

## Abstract

1. Biodiversity loss from intensive agriculture poses a major threat to the long-term sustainability and resilience of food production systems. Sustainable land management practices, such as carbon farming, offer promising alternatives, but their biodiversity impacts and the most effective methods for detecting these impacts remain poorly understood.
2. We surveyed 19 farms in boreal Finland to assess the effects of four carbon farming practices—cover crops, all-in mixes, adaptive grazing and ley mixtures—on plants, arthropods, nematodes and birds. We evaluated biodiversity responses using four alpha diversity metrics (abundance, species richness, Shannon diversity and Pielou's evenness) and two beta diversity metrics (Bray–Curtis and Chi-square dissimilarity).
3. Biodiversity responses were strongly context-dependent, varying by farming practice, taxonomic group and diversity metric. Abundance emerged as the most sensitive alpha metric across taxa, often detecting changes not reflected in community composition metrics. These findings suggest that abundance may serve as a useful early indicator of ecological change in managed landscapes.
4. Arthropods were particularly responsive to adaptive grazing, while ley mix and adaptive grazing supported higher nematode abundances. All diversity metrics except species richness detected changes in at least one species group, practice or field status, but abundance consistently captured the broadest responses.
5. Synthesis and applications. Carbon farming practices can support biodiversity when tailored to species group and context. Monitoring approaches that incorporate multiple metrics—and prioritise abundance as a sensitive and early indicator—can improve the detection of ecological responses to sustainable farming interventions.

## Introduction

The greatest global challenge of our time is to mitigate simultaneous climate change and biodiversity loss, both of which pose significant threats to the health of humans and ecosystems (Butchart et al., 2010, Seibold et al., 2019, Pecl et al., 2017). Agricultural practices are among the most significant drivers of these two interlinked processes (Raven & Wagner, 2021, Lal, 2004, Mena et al., 2020, Rigal et al., 2023, Yang et al., 2024). To alleviate the release of greenhouse gasses, particularly carbon dioxide $CO_2$, and to increase carbon sequestration and storage to soil organic matter and plants, carbon farming practices have recently been implemented in many areas of the world (Bradford et al., 2019, Chenu et al., 2019, Paustian et al., 2019). Carbon farming (and regenerative farming) consists of methods that increase vegetation cover (e.g., cropping season length, vegetation density, adaptive grazing of paddocks) and diversity (e.g., cover crops, inter-cropping, crop mixtures), amend soil to minimize disturbance (e.g., no-till planting), and increase soil organic matter (e.g., organic fertilizers, perennial cropping) (Mattila et al., 2022, Paustian et al., 2019, Teague & Kreuter, 2020). Given that these practices enhance vegetation diversity and green cover as well as minimize soil disturbance, they also have the potential to enhance biodiversity. However, knowledge on the realized impacts of carbon farming practices on biodiversity is scarce and mixed (Cozim-Melges et al., 2024). Furthermore, applying the limited existing knowledge to different soil and climate conditions can be challenging, as data collection has primarily focused on a few crop species in tropical and subtropical regions (Cozim-Melges et al., 2024).

Major declines in terrestrial fauna—including birds (Rigal et al., 2023), arthropods (Seibold et al., 2019, Lacroix et al., 2017), amphibians and reptiles (Cordier et al., 2021) — have been linked to agricultural land use, and there is an urgent need for agricultural management practices that enhance biodiversity (Pe'er et al., 2019). With agriculture covering nearly 40% of global arable land (Ramankutty et al., 2018), identifying practices that support diverse taxonomic groups could unlock significant opportunities for biodiversity enhancement. However, different taxa may respond differently to given cultivation practice (Gabriel et al., 2010) and hence, assessment of biodiversity impacts of cultivation practices requires simultaneous measurement of responses across multiple taxa (Cozim-Melges et al., 2024).

Robust indicators are required for quantifying biodiversity responses to agricultural practices. Biodiversity in this context refers to the composition and structure of species communities, which can be characterized using a range of indices derived from species abundances and distributions. Alpha diversity describes variation within a community and includes basic metrics such as species richness—the number of species present—and abundance, the number of individuals in the community. Shannon's diversity index (Shannon & Weaver, 1949) is widely used to incorporate both richness and evenness, while Pielou's evenness index (Pielou, 1966) specifically quantifies how evenly individuals are distributed across species. Beta diversity, which compares species composition between communities, is commonly measured using Bray-Curtis dissimilarity (Ricotta & Podani, 2017), based on relative abundances, or Chi-square distance (Legendre & De Cáceres, 2013), which compares species occurrence matrices. Biodiversity can also be evaluated through indicator species whose presence or abundance is considered to reflects responses to critical features of the habitat (Birkhofer et al., 2018, Boetzl et al., 2023). While these metrics provide valuable insights into different dimensions of biodiversity, they vary in their capacity to detect shifts in species interactions under anthropogenic change (Morris et al., 2014). To understand the ecological importance of carbon farming for biodiversity, information is needed on multiple taxonomic groups and biodiversity metrics, at both local and regional scales.

The aim of our study was to test whether carbon farming practices differ in their biodiversity impacts and whether biodiversity indices vary their ability to detect responses across taxonomic groups. More specifically, we surveyed 19 farms located in Central and Southern Finland to compare the impacts of four carbon farming practices on plants, arthropods, nematodes, and birds. These taxonomic groups were selected for their ecological relevance. Plants form the foundation of biodiversity, comprising the majority of biomass (Díaz & Malhi, 2022) and serve as the base of food webs in agroecosystems (Bohan et al., 2011, Price, 2002). Arthropods are highly diverse (Díaz & Malhi, 2022) and are well-known for their rapid responses to environmental changes (Ebeling et al., 2018). Birds are widely used in agroecological research as indicators of land-use change impacts (Butler et al., 2010). Nematodes contribute to nutrient recycling and to the regulation of microorganism populations in the soil having a significant role in the food web (Turbé et al., 2010, Yeates et al., 1993). Their community structure and functional responses have suggested to be used as bioindicators of soil health (Teshita et al., 2024). We then used four alpha diversity indices: abundance, species richness, Shannon's index, Pielou's evenness, and two beta diversity indices: Bray-Curtis dissimilarity, and Chi-square dissimilarity, to test which indices best describe change in diversity for each taxonomic group. For birds, we also tested whether we could assign indicator species for carbon farming fields using indicator species analysis. We found that the effects of carbon farming practices on both alpha and beta diversity were context-dependent, with the direction and magnitude of the effects varying among practices and across taxonomic groups. Jointly our results suggest that abundance index is the most sensitive index to be used in the biodiversity surveys at the farm scale and carbon farming practices differ in their biodiversity enhancement potential with adaptive grazing showing most impacts on arthropod taxa.

## Material and Methods

The Carbon Action experiment, initiated by the Baltic Sea Action Group, was set up by a network of 105 farms across Finland in the beginning of growing season in 2018 as a five year experiment to test how different carbon farming methods impact on carbon sequestration in fields (Mattila et al., 2022). We selected 19 farms from the set of 105 farms to be surveyed. The selected farms located in Southern and Central Finland represented four different carbon farming practices: under sown cover crops with a crop (caraway, pea, rye or oats; hereafter cover crop; 6 farms), ley mixture (5 farms), adaptive grazing aiming at maximizing carbon sequestration (hereafter adaptive grazing; 4 farms), and all-in (4 farms; Supplementary Fig S1). The farms selected for our study had applied one of the four carbon farming practices in a 1.5-hectare field plot since 2019 and adjacent to the plot with carbon farming methods was a control plot where a similar crop or pasture was grown following conventional farming practices. The all-in treatment was a combination of cover crops and locally tailored soil amendment practices (no tillage, subsoiling). Its control treatment involved the same crop, but without undersown cover crops or soil amendments. Adaptive grazing was a livestock management practice that used high stock density, frequent herd rotation, and long, adaptive plant recovery periods (Schmid et al., 2024). The control treatment for adaptive grazing was continuous grazing. In both adaptive grazing treatment and its control, the crop was a mixture of gramineous and leguminous species. In the control treatment for cover crops, the same crop (caraway, pea, rye or oats) was grown without an undersown cover crop. The control treatment for the ley mix included a maximum of three sown plant species.

### Biodiversity surveys

We conducted three biodiversity surveys in the selected farms: bird surveys in May and July, vegetation and invertebrate (arthropods and nematodes) surveys in early July 2023. The survey for vegetation and invertebrates was conducted within a 12 days' time on 26 June – 7 July 2023 and at each farm we sampled both the carbon farming treatment and the control plot. To assess the vegetation diversity within the field plots three one square meter grids were placed randomly to the fields and the vascular plants including the main crop within the grid were identified and their coverage was estimated. To measure arthropod diversity, we collected a suction sample using a leaf blower from each field plot from a 2-meter transect of vegetation selected by random. The suction sample was the collected to a plastic bag and cooled immediately to 5 °C, and in stored in -20 °C until the arthropods were identified. In the laboratory, the arthropods were identified to 10 taxonomic groups that represent three functional groups (predators or parasites including *Acarina, Aranea, Hymenoptera,* neutrals including *Collembola, Coleoptera, Diptera,* and herbivores including *Aphidoidea, Heteroptera, Homoptera, Thysanoptera;* the neutral group includes taxa with diffuse trophic level or presumably neutral relationship to plants) and individuals in each group were counted. Variation in topsoil nematode abundance and community composition was assessed by pooling four sub samples taken with four-meter distances using a soil drill from 0 to 4 cm depth. The soil samples were placed into a plastic bag and immediately cooled to 5 °C. In the laboratory, nematodes were extracted using wet funnels (Sohlenius, 1979) over 23h at room temperature + 1h with heating and light provided by 60W lamps 12 cm above the funnels. Total abundance was counted and a subsample of up to 50 nematodes were identified from formaldehyde-preserved samples to five trophic groups: herbivores, bacterivores, fungivores, omnivores, and predators based on their mouth parts (Yeates et al., 1993). To express nematode abundance as individuals per g of dry soil, soil water content of each sample was determined by drying ca. 10 g (FW) of homogenized soil for 24 h in +105°C and re-weighing the dry weight of samples.

The bird survey was performed within a 10 days' time in 2-12 May 2023. The bird survey was performed as point survey method in the carbon farming treatment and control fields between 4 and 9 am. Briefly, the surveyor walked a transect line through the field and in 350-meter interval the surveyor identified and counted for 10 minutes time all birds present within 50-meter radius. To understand whether the carbon farming practices impact on field birds that nest in the fields (hereafter called field birds), we assigned the observed birds to field species and non-field species based on farmland bird classification in Vepsäläinen (2007).

**Statistical analyses**

Here, we use alpha diversity when referring to local diversity indices abundance, species richness and diversity and beta diversity when referring to Bray-Curtis dissimilarity and Chi-square dissimilarity but please see Tuomisto (2010) for further discussion on terminology. We used R software package *vegan* (Oksanen et al., 2018) to calculate six biodiversity indices: species richness, number of individuals, Shannon's index, Pielou's evenness, Bray-Curtis dissimilarity, and Chi-square dissimilarity. Shannon's index was calculated as follows:

*H = -SUM[(pi) * ln(pi)]*

where *SUM* is summation, and *pi* is prevalence as summed number of species/ within in the population. Pielou's evenness index (Pielou, 1966) was calculated as follows:

$$J' = \frac{H'}{\ln(S)}$$

where H' is the Shannon's diversity, S is the total number of species. Bray-Curtis dissimilarity was calculated as follows:

$$\text{Bray-Curtis}_{ij} = 1 - \frac{2C_{ij}}{A_i + A_j}$$

where Cij is sum of shared minimum abundances for each species and $A_i$ and $A_j$ is abundance in sample *i* and *j*. Chi square dissimilarity was calculated as follows:

$$D_{\chi^2}(i,j) = \sum_{k=1}^{S} \frac{(x_{ik} - x_{jk})^2}{x_{ik} + x_{jk}}$$

Where $x_{ik}$ and x*jk* are counts or abundances of species *k* in samples *i* and *j*. S is the total number of species. Each term in the sum compares one species across the two samples.

To understand whether carbon farming induces changes in biodiversity, separate analyses were performed for five species groups: plants, arthropods, arthropod functional groups, nematodes, and birds. To analyze changes in alpha diversity indices, a generalized linear mixed model was used, with each index value as a response variable. For abundance and species richness a Poisson distribution of error was assumed except for plants where abundance was measured as coverage and a Beta distribution of error was used. For Shannon’s diversity and Pielou’s evenness, a Gaussian distribution of error was assumed. The four carbon farming practices and field status (carbon farming practice or control) were used as categorical explanatory variables. Field status was nested under carbon farming practice and the best model was selected based on Akaike information criterion (Symonds & Moussalli, 2011). Farm ID was used as random variable in all analyses. Post hoc analysis was performed to test the differences between diversity indices between carbon farmed and control fields within carbon farming practices.

To test whether species communities differ among the carbon farming practice t and field status (carbon farming practice or control), we ran a set of ANOSIM analyses using Bray-Curtis and Chi-square dissimilarities. The analyses were conducted in R using the vegan package (Oksanen et al., 2018). Analyses were performed separately for each of the four species groups: birds, plants, arthropods, and nematodes. Due to the small sample size, ANOSIM was not performed on the field bird data.

To understand whether the abundance of the nematode functional groups differed among carbon practices as well as within practices between the carbon field vs. the control field, a set of analyses were run in Generalized linear mixed model framework. The abundance of individuals within a functional group was used as a linear response variable and a Poisson distribution of error was assumed. The four carbon practices and field status (carbon practice or control) were used as categorical explanatory variables. Field status was nested under carbon practice type and the best model was selected based on Akaike information criterion (Symonds & Moussalli, 2011). To test the impact of carbon farming practices on the abundance of the arthropod species and functional groups, altogether 12 models were run with a similar model structure. *Aphidoids* were excluded from the species group level analyses due their rare occurrence. Post hoc analysis was performed to test the differences between abundances between carbon field and control fields within carbon farming practices.

To test whether indicator species analysis can reveal bird species that are significantly associated to the fields where carbon farming is practiced, we run analyses using *indicspecies* package (De Caceres & Legendre, 2009) in R.

## Results

**Alpha and beta diversity indices explaining change in biodiversity**

We found altogether 198 plant species from the surveyed vegetation plots. The abundance of plants, measured by vegetation cover, differed between carbon farming and control fields (Table 2). For plant abundance, the ley mixture and the adaptive grazing differed significantly from the control field, exhibiting lower abundance (Fig 1). Surprisingly, plant species richness did not differ between the carbon farming practice used or between field status i.e., control vs carbon practice (Table 2). However, Shannon's diversity and Pielou's evenness differed between control vs carbon farming status of the field (Table 2; Fig 1). There were higher diversity and evenness in the ley mixture treatment compared to its control (Fig 1). Bray-Curtis dissimilarities differed among the carbon farming practices and there was a significant difference between ley mixture and control (Table 2). Plant community composition differed significantly based on Bray-Curtis dissimilarities, indicating shifts in species presence and abundance across treatments (Table 2).

The abundance of arthropods depended on carbon farming practice used and field status (Table 2; Figs 1-2). Abundance was markedly higher in adaptive grazing than its control but in other carbon practices arthropod abundance was significantly lower than in their controls (Figs 1-2). Other alpha diversity metrics showed no significant effects on arthropod communities, either on the higher taxonomic or the more general functional group level (Table 2; Fig 1). We identified significant dissimilarities (Bray-Curtis and Chi square) in the taxonomic arthropod groups among carbon practices, but not between control vs carbon practice within a single farm (Table 2). Among the functional groups, only the Chi square index revealed significant dissimilarities among carbon practices (Table 2). When we assessed the abundances of the nine individual taxonomic groups and the three functional groups of arthropods instead of their diversity indices, we found that field status had both positive and negative effects on the abundances of all analyzed groups (Supplementary Table S1; Fig 2).

We found that nematode abundance was highest in the ley mixture and lowest in the all-in treatment (Table 2; Fig 2). Compared to control fields, nematode abundance was lower in carbon practice fields under cover crops and all-in treatments, but higher under adaptive grazing (Table 1; Figs 1, 2). The other alpha diversity metrics and beta diversity metrics of nematode data did not reveal any significant differences among the treatments (Table 2, Fig 1). Among individual nematode functional groups, bacterivores were less abundant in the all-in and cover crop treatments compared to their respective controls (Supplementary Table S2; Fig. 1). Herbivorous nematodes were more abundant in the two perennial systems—adaptive grazing and ley mixture—than in all-in and cover crop treatments, but their abundance did not differ between carbon practices and their controls (Supplementary Table S2; Fig. 2). For the other functional groups, we found no significant differences among the carbon practices or between treatments and controls (Supplementary Table S2; Fig. 3).

We observed 31 bird species across both carbon and control fields. Four of these were field species: *Alauda arvensis, Anthus pratensis, Numenius arquata,* and *Vanellus vanellus.* However, analyses of both the complete bird dataset and the subset of field species revealed no significant differences in any of the alpha or beta diversity indices (Table 2; Figs. 1, S2).

**Bird indicator species analysis**

Indicator species analysis showed that one species, Eurasian skylark, *Alauda arvensis*, was significantly associated to all-in and cover crops treatments (*stat* = 0.548; *P* = 0.043; Supplementary Table S3).

## Discussion

Biodiversity has declined significantly because of intensive agricultural land use (Rigal et al., 2023, Raven & Wagner, 2021, Tsiafouli et al., 2015), posing a major challenge to long-term ecosystem health and resilience in these human-managed environments. As we transition toward more sustainable food production systems, such as carbon farming, it is critical to understand how these practices affect biodiversity—and which metrics best capture these changes. Here, we tested how different biodiversity indices capture changes in four taxonomic groups—plants, arthropods, nematodes, and birds—in response to various carbon farming practices. Overall, we found that the biodiversity impacts of carbon practices were context-dependent, varying by both the index used and the taxonomic group studied. The four carbon farming practices in our study—all-in, cover crops, adaptive grazing, and ley mixture—differed in their effects on the species groups. Below, we discuss how biodiversity responses to carbon farming were shaped by the type of practice, the taxonomic group, the analytical method, and the diversity metric results for each of these factors –and consider their relevance for promoting biodiversity in sustainable farming systems.

We found that carbon farming practices differed in the species groups they affected. Our results align with a recent meta-analysis by Cozim-Melges et al. (Cozim-Melges et al., 2024), concluding that no single agricultural practice enhances biodiversity across all taxonomic groups. In our study, the two annual treatments- all-in and cover crops - produced directionally similar responses in alpha diversity for nematodes and arthropods—

two taxonomic groups known to respond rapidly to cropping practices. For example, a Swedish study on inter-cropping treatments in cereals reported changes in both nematode and predatory arthropod taxa (Boetzl et al., 2023). Among the perennial treatments (adaptive grazing and ley mixture), we observed a decrease in plant abundance. However, the effects on other species groups were more variable. Notably, adaptive grazing increased arthropod abundance, while the ley mixture treatment showed only a minor increase in arthropods. These findings are consistent with a recent North American study that reported increases in cross-taxa in response to adaptive grazing (Rigal et al., 2023) even if mixed responses are reported across various environments (Morris, 2021).

All of the biodiversity indices used in our study detected differences in at least one species group, carbon farming practice, or field status. Abundance was the most sensitive metric, showing responses to carbon farming practices across all species groups except birds. Shannon diversity and Pielou's evenness revealed changes in plant communities between ley mixture treatment and its control. Both beta diversity indices—Bray-Curtis and Chi-square dissimilarity—detected changes in arthropod communities across carbon practice types. Responses in Chi-square dissimilarity were particularly dependent on arthropod functional groups, even though only three such groups were included. In contrast, Bray-Curtis dissimilarity detected changes in plant communities across both carbon practice type and field status. Our findings are in line with a study by Morris et al. (2014), which found that abundance was the only metric consistently sensitive to changes across multiple species groups in response to land-use intensity. However, their path analysis also showed that Shannon's and Simpson's diversity indices provided a better model fit for those same groups.

Arthropods abundances showed the strongest responses to carbon farming practices among the studied groups, and the direction of the response varied depending on the management practice. Often the direction of responses was consistent across both functional group abundances (predators, herbivores, neutrals), and the abundances of individual taxonomic groups. However, for example *Acarina* and *Aranea*, both predators, responded in opposing ways to the ley mixture treatment, and *Collembola* responded differently to cover crops than the other neutral taxa. This suggests that taxonomic group-level analyses may provide more accurate insights than broader functional groupings when evaluating biodiversity responses to management practices. Like arthropods, nematode abundance varied significantly by carbon farming practice. Bacterivores were sensitive to the carbon practice vs. control contrast (field status) in the two annual practices, while herbivores were more abundant in farms with perennial management practices. The higher proportion of bacterivorous nematodes in farms following annually renewed management practices (more frequent disturbance or vegetation-free periods) might reflect the early colonization status of these soil organisms (Archidona-Yuste et al., 2025) while the reduced total nematode abundance in cover crop and all-in treatments compared to control suggests that carbon-farming practices based on undersown cover crops can influence soil communities. In contrast, birds did not show significant differences among treatments. This pattern was also observed in field bird species, although the absence of detectable effects may be due to the small sample size. These results contrast with previous studies that have found field birds to be more responsive to agricultural practices  (Doxa et al., 2010, Rigal et al., 2023). For plants, we observed changes in several diversity indices other than species richness. Plants were the only species group directly manipulated by growers, so variation among treatments was expected. The apparent low variability in species richness may be explained by the dynamics of different treatment types: in the perennial practices (adaptive grazing and ley mixture), five years of consistent management may have homogenized vegetation across fields, while in the annual practices (cover crops and all-in), increased weed emergence may have increased species diversity (Henckel et al., 2015).

**Conclusions** Jointly our findings reveal that biodiversity responses to carbon farming practices are highly context-dependent, shaped by the type of management, the taxonomic group examined, and the diversity metric used. Notably, alpha diversity metrics, particularly abundance, were often more sensitive in detecting changes than beta diversity measures. This suggests that shifts in the number of individuals within species groups may occur even when overall community composition remains relatively stable (Winfree et al., 2015). Ecologically, changes in abundance can have significant implications: increased abundance may enhance ecosystem functions such as nutrient cycling, pest control, and pollination (Liu et al., 2021, Winfree et al., 2015). The effects may be particularly strong when changes affect key functional groups (Crawford et al., 2021). The high responsiveness of abundance highlights the importance of not overlooking simple metrics when monitoring biodiversity outcomes (Gotelli & Colwell, 2001).

The variation in detection power among metrics also emphasizes the need for careful selection of biodiversity indicators when evaluating the ecological impacts of sustainable farming practices (Sutherland et al., 2013). While beta diversity provides important insights into turnover and community restructuring (Anderson et al., 2011), alpha diversity metrics may be more immediately informative for detecting early responses to management changes in agroecosystems. This study offers new evidence on the effectiveness of biodiversity indices for assessing biodiversity at the farm scale (Herzog & Franklin, 2016) Developing reliable monitoring tools at this scale is especially important for policymakers seeking to implement biodiversity-friendly agricultural policies (Targetti et al., 2016).

Importantly, our results suggest that even modest adjustments in farming practices can promote biodiversity, but the outcomes are strongly taxon-specific. Promoting practices that maintain or enhance organism abundance, particularly in underrepresented groups such as soil fauna, could strengthen the resilience and multifunctionality of agricultural ecosystems (Tscharntke et al., 2005, Bardgett & van der Putten, 2014). Our findings are particularly valuable for boreal biomes, where data on the biodiversity impacts of agricultural practices have been limited (Cozim-Melges et al., 2024). Thus, carbon farming and related diversification strategies have clear potential to contribute not only to climate mitigation goals but also to enhance farmland biodiversity within production landscapes. Strategic monitoring and adaptive management, informed by appropriate diversity metrics, will be key to achieving these dual objectives (Bommarco et al., 2013, Kremen & Merenlender, 2018)

# Tables

**Table 1. Number of sampled units, individuals, species, and functional groups in this study.**

| | Sampling units | Observed individuals | Observed species number | Functional groups |
|---|---|---|---|---|
| **Plants** | 114 | 1405* | 198 | - |
| **Arthropods** | 38 | 6669 | - | 3 |
| **Nematodes** | 38 | 19418 (1934**) | - | 5 |
| **Birds** | 76 | 109 | 31 | - |
| **Field birds** | 76 | 15 | 4 | - |

*Plant species observations within a grid.

**Nematodes characterized to functional groups.

**Table 2**. **The Effect of Carbon Farming Practices on Alpha and Beta diversity.** The effect of carbon practice vs control and carbon practice type on alpha (abundance, species richness Shannon's diversity, Pielou's evenness) and beta (Bray's dissimilarities, Chi square dissimilarities) diversity of plants, arthropods, arthropod functional groups, nematode functional groups, all bird species (31 species) and field birds (4 species) analyzed using generalized linear mixed models. Significant differences (*P* < 0.05) are shown in bold.

| | Effect | Abundance | | | Species richness | | | Shannon's diversity | | | Pielou's evenness | | | Bray's dissimilarities | | Chi square dissimilarities | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | d.d., d.d.f. | *F* | *P* | d.d., d.d.f. | *F* | *P* | d.d., d.d.f. | *F* | *P* | d.d., d.d.f. | *F* | *P* | *R* | *P* | *R* | *P* |
| **Plants** | Carbon practice type | **3, 15** | **6.15** | **0.0078** | 3, 15 | 0.75 | 0.5376 | 3, 15 | 1.75 | 0.1993 | 3, 15 | 2.24 | 0.1258 | **0.226** | **0.002** | 0.048 | 0.154 |
| | Field status (carbon vs. control) | **4, 15** | **5.96** | **0.006** | 4, 15 | 0.51 | 0.7263 | **4, 15** | **4.34** | **0.0158** | **4, 15** | **4.17** | **0.0182** | **0.1426** | **0.019** | 0.0343 | 0.234 |
| **Arthropods** | Carbon practice type | 3, 15 | 1.2 | 0.3423 | 3, 15 | 0.93 | 0.4511 | 3, 15 | 2.44 | 0.1047 | 3, 15 | 1.91 | 0.1706 | **0.081** | **0.049** | **0.132** | **0.008** |
| | Field status (carbon vs. control) | **4, 15** | **147** | **<.0001** | 4, 15 | 0.13 | 0.9684 | 4, 15 | 0.37 | 0.8281 | 4, 15 | 0.11 | 0.9775 | 0.0038 | 0.449 | **0.522** | **0.001** |
| **Arthropod functional groups** | Carbon practice type | - | - | - | 3, 15 | 0.02 | 0.9955 | 3, 15 | 0.64 | 0.6004 | 3, 15 | 0.44 | 0.7255 | 0.048 | 0.17 | **0.086** | **0.048** |
| | Field status (carbon vs. control) | - | - | - | 4, 15 | 0.03 | 0.9975 | 4, 15 | 1.96 | 0.1519 | 4, 15 | 1.07 | 0.4065 | -0.013 | 0.564 | 0.0439 | 0.234 |
| **Nematodes** | Carbon practice type | **3, 15** | **7.95** | **0.0021** | 3, 15 | 0.13 | 0.9421 | 3, 15 | 1.29 | 0.3126 | 3, 15 | 0.98 | 0.4295 | 0.049 | 0.166 | 0.069 | 0.085 |

| | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Field status (carbon vs. control) | **3, 15** | **3.97** | **0.0216** | 4, 15 | 0.06 | 0.9934 | 4, 15 | 2.02 | 0.1435 | 4, 15 | 0.53 | 0.7184 | -0.03 | 0.623 | -0.002 | 0.472 |
| **Birds** | Carbon practice type | 3, 15 | 1.62 | 0.2271 | 3, 15 | 1.28 | 0.3156 | 3, 9 | 1.02 | 0.4173 | 3, 5 | 1.33 | 0.364 | 0.083 | 0.173 | 0.067 | 0.127 |
| | Field status (carbon vs. control) | 4, 15 | 1.04 | 0.417 | 4, 15 | 0.9 | 0.4862 | | 1.11 | 0.396 | 4, 5 | 1.31 | 0.3798 | 0.059 | 0.12 | 0.008 | 0.421 |
| **Field birds** | Carbon practice type | 3, 30 | 0.18 | 0.9476 | 3, 30 | 0.13 | 0.9705 | 3, 30 | 0.82 | 0.4915 | 3, 19 | 0.36 | 0.7811 | - | - | - | - |
| | Field status (carbon vs. control) | 4, 30 | 0.23 | 0.8773 | 4, 30 | 0.13 | 0.943 | 4, 30 | 0.84 | 0.5113 | 4, 19 | 1.4 | 0.2719 | - | - | - | - |

**Figure legends**

**Fig. 1**. **Effects of Carbon Farming Practices on Alpha Diversity.** The effect of carbon practice vs control A) on alpha diversity metrics in plants, arthropods, nematodes and birds, and B) on abundance of functional and taxonomic groups of arthropods and functional groups of nematodes. Significant differences are shown with a bold black line.

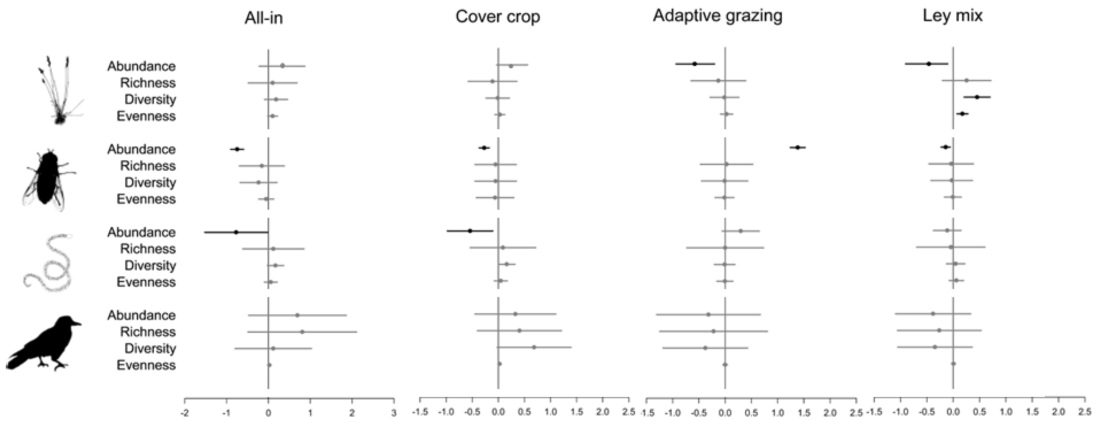
**Fig. 2. The Composition of Arthropod and Nematode Communities in Different Carbon practices.** Composition of A) arthropod and B) nematode communities sampled from fields with and without carbon practices surveyed in 19 farms in Central and Southern Finland in 2023.

**Fig. 3. Bird species occurrence in response to carbon farming practices across Finnish farms.** Occurrence of 31 bird species observed in fields with and without carbon practices surveyed in 19 farms in Central and Southern Finland 2023. The carbon practice vs control practice used is indicated with colors as in legend. An asterisk indicates species significantly associated with all-in and cover crop treatment in indicator species analysis. The four field bird species are indicated with bold text.

## Figures

### Figure 1.

**A**




**B**

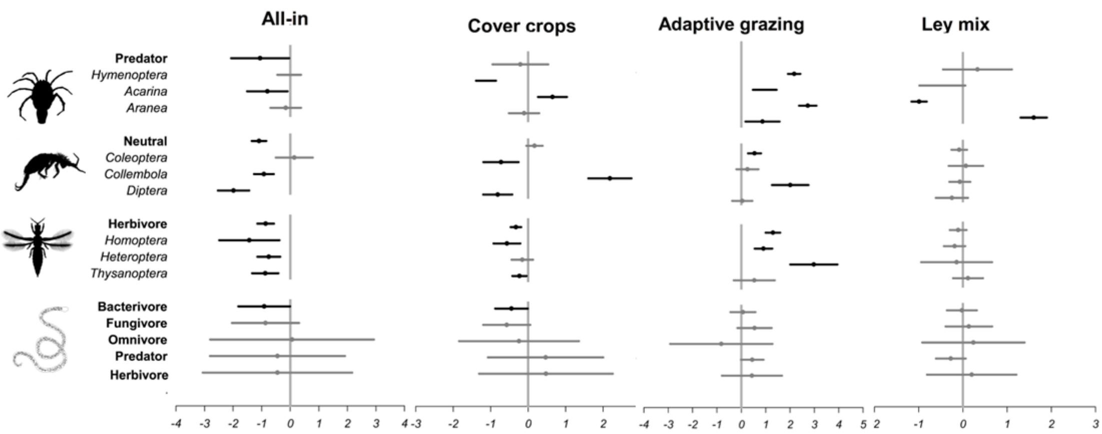
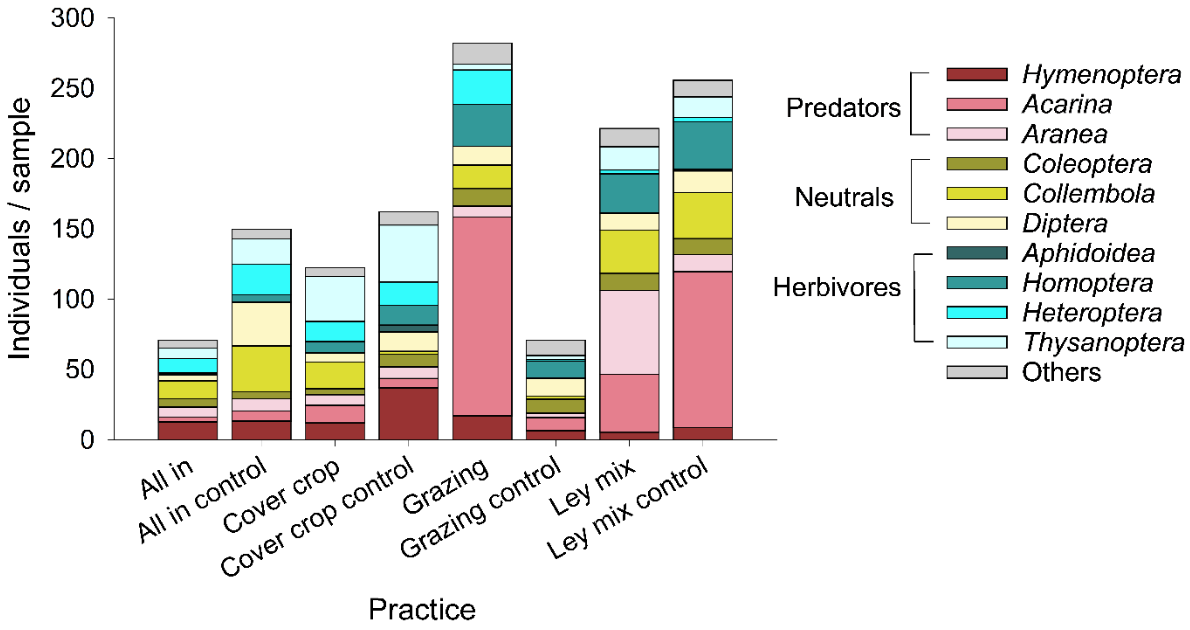



**Fig. 1**. **Effects of Carbon Farming Practices on Alpha Diversity.** Effect sizes from Tukey's test comparing carbon farming practices to controls are shown for: A) alpha diversity metrics of plants, arthropods, nematodes, and birds, and B) the abundance of functional and taxonomic groups of arthropods and functional groups of nematodes. Effect sizes are represented by points, with error bars indicating 95% confidence intervals. Statistically significant differences are highlighted with bold black lines.

Figure 2.

A




B

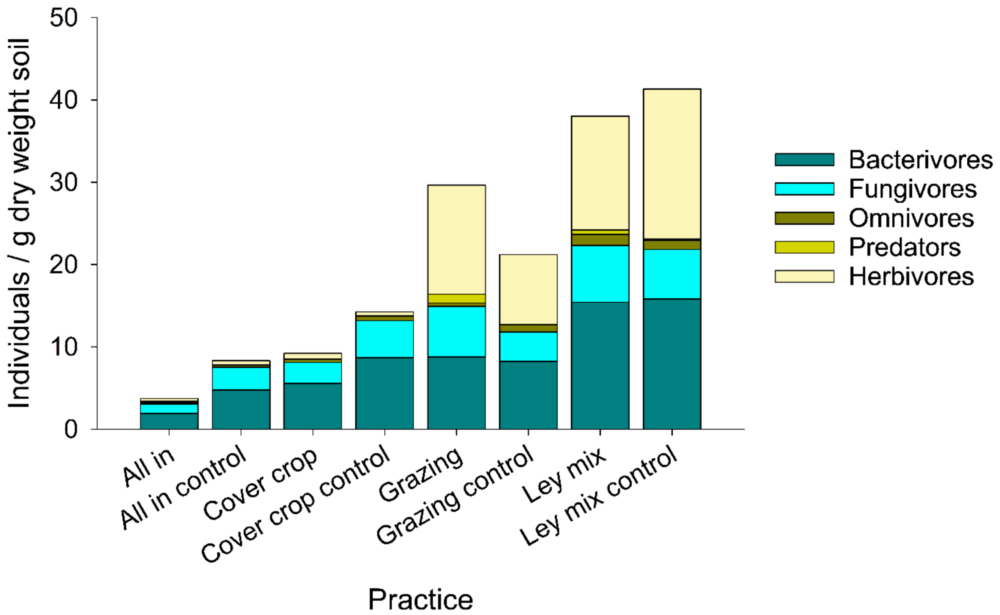



**Fig. 2. The Composition of Arthropod and Nematode Communities in Different Carbon practices.** Composition of A) arthropod and B) nematode communities sampled from fields

with and without carbon practices surveyed in 19 farms in Central and Southern Finland in 2023.

**Figure 3.**

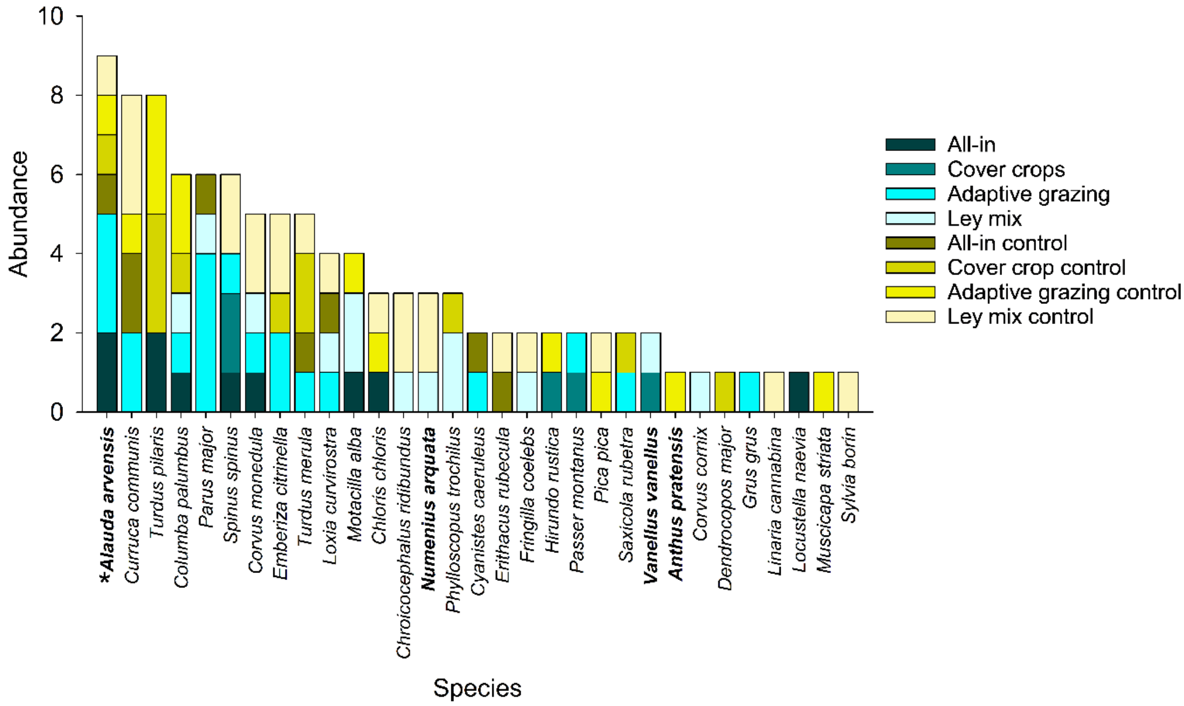



**Fig. 3. Bird species occurrence in response to carbon farming practices across Finnish farms.** Occurrence of 31 bird species observed in fields with and without carbon practices surveyed in 19 farms in Central and Southern Finland 2023. The carbon practice vs control practice used is indicated with colors as in legend. An asterisk indicates species significantly associated with all-in and cover crop treatment in indicator species analysis. The four field bird species are indicated with italics.

## References

Anderson, M. J., Crist, T. O., Chase, J. M., Vellend, M., Inouye, B. D., Freestone, A. L., Sanders, N. J., Cornell, H. V., Comita, L. S., Davies, K. F., Harrison, S. P., Kraft, N. J. B., Stegen, J. C. & Swenson, N. G. 2011. Navigating the multiple meanings of β diversity: a roadmap for the practicing ecologist. *Ecology Letters* **14**: 19-28.

Archidona-Yuste, A., Ciobanu, M., Kardol, P. & Eisenhauer, N. 2025. Divergent alpha and beta diversity trends of soil nematode fauna along gradients of environmental change in the Carpathian Ecoregion. *Communications Biology* **8**: 587.

Bardgett, R. D. & van der Putten, W. H. 2014. Belowground biodiversity and ecosystem functioning. *Nature* **515**: 505-511.

Birkhofer, K., Rusch, A., Andersson, G. K. S., Bommarco, R., Dänhardt, J., Ekbom, B., Jönsson, A., Lindborg, R., Olsson, O., Rader, R., Stjernman, M., Williams, A., Hedlund, K. & Smith, H. G. 2018. A framework to identify indicator species for ecosystem services in agricultural landscapes. *Ecological Indicators* **91**: 278-286.

Boetzl, F. A., Douhan Sundahl, A., Friberg, H., Viketoft, M., Bergkvist, G. & Lundin, O. 2023. Undersowing oats with clovers supports pollinators and suppresses arable weeds without reducing yields. *Journal of Applied Ecology* **60**: 614-623.

Bommarco, R., Kleijn, D. & Potts, S. G. 2013. Ecological intensification: harnessing ecosystem services for food security. *Trends Ecol Evol* **28**: 230-8.

Bradford, M. A., Carey, C. J., Atwood, L., Bossio, D., Fenichel, E. P., Gennet, S., Fargione, J., Fisher, J. R. B., Fuller, E., Kane, D. A., Lehmann, J., Oldfield, E. E., Ordway, E. M., Rudek, J., Sanderman, J. & Wood, S. A. 2019. Soil carbon science for policy and practice. *Nature Sustainability* **2**: 1070-1072.

Butchart, S. H. M., Walpole, M., Collen, B., van Strien, A., Scharlemann, J. P. W., Almond, R. E. A., Baillie, J. E. M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K. E., Carr, G. M., Chanson, J., Chenery, A. M., Csirke, J., Davidson, N. C., Dentener, F., Foster, M., Galli, A., Galloway, J. N., Genovesi, P., Gregory, R. D., Hockings, M., Kapos, V., Lamarque, J.-F., Leverington, F., Loh, J., McGeoch, M. A., McRae, L., Minasyan, A., Morcillo, M. H., Oldfield, T. E. E., Pauly, D., Quader, S., Revenga, C., Sauer, J. R., Skolnik, B., Spear, D., Stanwell-Smith, D., Stuart, S. N., Symes, A., Tierney, M., Tyrrell, T. D., Vié, J.-C. & Watson, R. 2010. Global Biodiversity: Indicators of Recent Declines. *Science* **328**: 1164-1168.

Chenu, C., Angers, D. A., Barré, P., Derrien, D., Arrouays, D. & Balesdent, J. 2019. Increasing organic stocks in agricultural soils: Knowledge gaps and potential innovations. *Soil and Tillage Research* **188**: 41-52.

Cordier, J. M., Aguilar, R., Lescano, J. N., Leynaud, G. C., Bonino, A., Miloch, D., Loyola, R. & Nori, J. 2021. A global assessment of amphibian and reptile responses to land-use changes. *Biological Conservation* **253**: 108863.

Cozim-Melges, F., Ripoll-Bosch, R., Veen, G. F., Oggiano, P., Bianchi, F. J. J. A., van der Putten, W. H. & van Zanten, H. H. E. 2024. Farming practices to enhance biodiversity across biomes: a systematic review. *npj Biodiversity* **3**: 1.

Crawford, M. S., Barry, K. E., Clark, A. T., Farrior, C. E., Hines, J., Ladouceur, E., Lichstein, J. W., Maréchaux, I., May, F., Mori, A. S., Reineking, B., Turnbull, L. A., Wirth, C. & Rüger, N. 2021. The function-dominance correlation drives the direction and strength of biodiversity–ecosystem functioning relationships. *Ecology Letters* **24**: 1762-1775.

De Caceres, M. & Legendre, P. 2009. Associations between species and groups of sites: indices and statistical inference. *Ecology* **90**: 3566-74.

Doxa, A., Bas, Y., Paracchini, M. L., Pointereau, P., Terres, J.-M. & Jiguet, F. 2010. Low-intensity agriculture increases farmland bird abundances in France. *Journal of Applied Ecology* **47**: 1348-1356.

Gabriel, D., Sait, S. M., Hodgson, J. A., Schmutz, U., Kunin, W. E. & Benton, T. G. 2010. Scale matters: the impact of organic farming on biodiversity at different spatial scales. *Ecology Letters* **13**: 858-869.

Gotelli, N. J. & Colwell, R. K. 2001. Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. *Ecology Letters* **4**: 379-391.

Henckel, L., Börger, L., Meiss, H., Gaba, S. & Bretagnolle, V. 2015. Organic fields sustain weed metacommunity dynamics in farmland landscapes. *Proceedings of the Royal Society B: Biological Sciences* **282**: 20150002.

Herzog, F. & Franklin, J. 2016. State-of-the-art practices in farmland biodiversity monitoring for North America and Europe. *Ambio* **45**: 857-871.

Kremen, C. & Merenlender, A. M. 2018. Landscapes that work for biodiversity and people. *Science* **362**.

Lacroix, C., Seabloom, E. W. & Borer, E. T. 2017. Environmental Nutrient Supply Directly Alters Plant Traits but Indirectly Determines Virus Growth Rate. *Frontiers in Microbiology* **8**: 16.

Lal, R. 2004. Soil carbon sequestration to mitigate climate change. *Geoderma* **123**: 1-22.

Legendre, P. & De Cáceres, M. 2013. Beta diversity as the variance of community data: dissimilarity coefficients and partitioning. *Ecology letters* **16**: 951-963.

Liu, D., Chang, P.-H. S., Power, S. A., Bell, J. N. B. & Manning, P. 2021. Changes in plant species abundance alter the multifunctionality and functional space of heathland ecosystems. *New Phytologist* **232**: 1238-1249.

Mattila, T. J., Hagelberg, E., Söderlund, S. & Joona, J. 2022. How farmers approach soil carbon sequestration? Lessons learned from 105 carbon-farming plans. *Soil and Tillage Research* **215**: 105204.

Mena, E., Stewart, S., Montesano, M. & Ponce de León, I. 2020. Soybean Stem Canker Caused by Diaporthe caulivora; Pathogen Diversity, Colonization Process, and Plant Defense Activation. *Frontiers in Plant Science* **10**.

Morris, C. D. 2021. How Biodiversity-Friendly Is Regenerative Grazing? *Frontiers in Ecology and Evolution* **Volume 9 - 2021**.

Morris, E. K., Caruso, T., Buscot, F., Fischer, M., Hancock, C., Maier, T. S., Meiners, T., Müller, C., Obermaier, E., Prati, D., Socher, S. A., Sonnemann, I., Wäschke, N., Wubet, T., Wurst, S. & Rillig, M. C. 2014. Choosing and using diversity indices: insights for ecological applications from the German Biodiversity Exploratories. *Ecology and Evolution* **4**: 3514-3524.

Oksanen, J., Blanchet, G., Friendly, M., Roeland, K., Legendre, P., McGlinn, D. J., Minchin, P. R., O'Hara, R. B., Simpson, G. L., Solymos, P., Stevens, M. H. H., Szoecs, E. & Wagner, H. (2018) vegan: Community Ecology Package. pp.

Paustian, K., Larson, E., Kent, J., Marx, E. & Swan, A. 2019. Soil C Sequestration as a Biological Negative Emission Strategy. *Frontiers in Climate* **1**.

Pe'er, G., Zinngrebe, Y., Moreira, F., Sirami, C., Schindler, S., Müller, R., Bontzorlos, V., Clough, D., Bezák, P., Bonn, A., Hansjürgens, B., Lomba, A., Möckel, S., Passoni, G., Schleyer, C., Schmidt, J. & Lakner, S. 2019. A greener path for the EU Common Agricultural Policy. *Science* **365**: 449-451.

Pecl, G. T., Araújo, M. B., Bell, J. D., Blanchard, J., Bonebrake, T. C., Chen, I.-C., Clark, T. D., Colwell, R. K., Danielsen, F., Evengård, B., Falconi, L., Ferrier, S., Frusher, S., Garcia, R. A., Griffis, R. B., Hobday, A. J., Janion-Scheepers, C., Jarzyna, M. A., Jennings, S., Lenoir, J., Linnetved, H. I., Martin, V. Y., McCormack, P. C., McDonald, J., Mitchell, N. J., Mustonen, T., Pandolfi, J. M., Pettorelli, N., Popova, E., Robinson, S. A., Scheffers, B. R., Shaw, J. D., Sorte, C. J. B., Strugnell, J. M., Sunday, J. M., Tuanmu, M.-N., Vergés, A., Villanueva, C., Wernberg, T., Wapstra, E. & Williams, S. E. 2017. Biodiversity redistribution under climate change: Impacts on ecosystems and human well-being. *Science* **355**: eaai9214.

Pielou, E. C. 1966. Measurement of Diversity in Different Types of Biological Collections. *Journal of Theoretical Biology* **13**: 131-&.

Ramankutty, N., Mehrabi, Z., Waha, K., Jarvis, L., Kremen, C., Herrero, M. & Rieseberg, L. H. 2018. Trends in Global Agricultural Land Use: Implications for Environmental Health and Food Security. *Annual Review of Plant Biology* **69**: 789-815.

Raven, P. H. & Wagner, D. L. 2021. Agricultural intensification and climate change are rapidly decreasing insect biodiversity. *Proceedings of the National Academy of Sciences* **118**: e2002548117.

Ricotta, C. & Podani, J. 2017. On some properties of the Bray-Curtis dissimilarity and their ecological meaning. *Ecological Complexity* **31**: 201-205.

Rigal, S., Dakos, V., Alonso, H., Auniņš, A., Benkő, Z., Brotons, L., Chodkiewicz, T., Chylarecki, P., de Carli, E., del Moral, J. C., Domşa, C., Escandell, V., Fontaine, B., Foppen, R., Gregory, R., Harris, S., Herrando, S., Husby, M., Ieronymidou, C., Jiguet, F., Kennedy, J., Klvaňová, A., Kmecl, P., Kuczyński, L., Kurlavičius, P., Kålås, J. A., Lehikoinen, A., Lindström, Å., Lorrillière, R., Moshøj, C., Nellis, R., Noble, D., Eskildsen, D. P., Paquet, J.-Y., Pélissié, M., Pladevall, C., Portolou, D., Reif, J., Schmid, H., Seaman, B., Szabo, Z. D., Szép, T., Florenzano, G. T., Teufelbauer, N., Trautmann, S., van Turnhout, C., Vermouzek, Z., Vikstrøm, T., Voříšek, P., Weiserbs, A. & Devictor, V. 2023. Farmland practices are driving bird population decline across Europe. *Proceedings of the National Academy of Sciences* **120**: e2216573120.

Schmid, R. B., Welch, K. D., Teague, R. & Lundgren, J. G. 2024. Adaptive Multipaddock (AMP) Pasture Management Increases Arthropod Community Guild Diversity Without Increasing Pests. *Rangeland Ecology & Management* **94**: 141-148.

Seibold, S., Gossner, M. M., Simons, N. K., Blüthgen, N., Müller, J., Ambarlı, D., Ammer, C., Bauhus, J., Fischer, M., Habel, J. C., Linsenmair, K. E., Nauss, T., Penone, C., Prati, D., Schall, P., Schulze, E.-D., Vogt, J., Wöllauer, S. & Weisser, W. W. 2019. Arthropod decline in grasslands and forests is associated with landscape-level drivers. *Nature* **574**: 671-674.

Shannon, C. E. & Weaver, W. 1949. *Mathematical Theory of Communication*. University of Illinois Press, Urbana.

Sohlenius, B. 1979. A carbon budget for nematodes, rotifers and tardigrades in a Swedish coniferous forest soil. *Ecography* **2**: 30-40.

Sutherland, W. J., Freckleton, R. P., Godfray, H. C. J., Beissinger, S. R., Benton, T., Cameron, D. D., Carmel, Y., Coomes, D. A., Coulson, T., Emmerson, M. C., Hails, R. S., Hays, G. C., Hodgson, D. J., Hutchings, M. J., Johnson, D., Jones, J. P. G., Keeling, M. J., Kokko, H., Kunin, W. E., Lambin, X., Lewis, O. T., Malhi, Y., Mieszkowska, N., Milner-Gulland, E. J., Norris, K., Phillimore, A. B., Purves, D. W., Reid, J. M., Reuman, D. C., Thompson, K., Travis, J. M. J., Turnbull, L. A., Wardle, D. A. & Wiegand, T. 2013. Identification of 100 fundamental ecological questions. *Journal of Ecology* **101**: 58-67.

Symonds, M. R. E. & Moussalli, A. 2011. A brief guide to model selection, multimodel inference and model averaging in behavioural ecology using Akaike's information criterion. *Behavioral Ecology and Sociobiology* **65**: 13-21.

Targetti, S., Herzog, F., Geijzendorffer, I. R., Pointereau, P. & Viaggi, D. 2016. Relating costs to the user value of farmland biodiversity measurements. *Journal of Environmental Management* **165**: 286-297.

Teague, R. & Kreuter, U. 2020. Managing Grazing to Restore Soil Health, Ecosystem Function, and Ecosystem Services. *Frontiers in Sustainable Food Systems* **4**.

Tscharntke, T., Klein, A. M., Kruess, A., Steffan-Dewenter, I. & Thies, C. 2005. Landscape perspectives on agricultural intensification and biodiversity – ecosystem service management. *Ecology Letters* **8**: 857-874.

Tsiafouli, M. A., Thébault, E., Sgardelis, S. P., de Ruiter, P. C., van der Putten, W. H., Birkhofer, K., Hemerik, L., de Vries, F. T., Bardgett, R. D., Brady, M. V., Bjornlund, L., Jørgensen, H. B., Christensen, S., Hertefeldt, T. D., Hotes, S., Gera Hol, W. H., Frouz, J., Liiri, M., Mortimer, S. R., Setälä, H., Tzanopoulos, J., Uteseny, K., Pižl, V., Stary, J., Wolters, V. & Hedlund, K. 2015. Intensive agriculture reduces soil biodiversity across Europe. *Global Change Biology* **21**: 973-985.

Tuomisto, H. 2010. A diversity of beta diversities: straightening up a concept gone awry. Part 1. Defining beta diversity as a function of alpha and gamma diversity. *Ecography* **33**: 2-22.

Vepsäläinen, V. (2007) Farmland Birds and Habitat Heterogeneity in Intensively Cultivated Boreal Agricultural Landscapes. In: *Department of Biological and Environmental Sciences*, Vol. PhD. pp. 52. University of Helsinki, Helsinki.

Winfree, R., W. Fox, J., Williams, N. M., Reilly, J. R. & Cariveau, D. P. 2015. Abundance of common species, not species richness, drives delivery of a real-world ecosystem service. *Ecology Letters* **18**: 626-635.

Yang, Y., Tilman, D., Jin, Z., Smith, P., Barrett, C. B., Zhu, Y.-G., Burney, J., D'Odorico, P., Fantke, P., Fargione, J., Finlay, J. C., Rulli, M. C., Sloat, L., Jan van Groenigen, K., West, P. C., Ziska, L., Michalak, A. M., Team, t. C.-A., Lobell, D. B., Clark, M., Colquhoun, J., Garg, T., Garrett, K. A., Geels, C., Hernandez, R. R., Herrero, M., Hutchison, W. D., Jain, M., Jungers, J. M., Liu, B., Mueller, N. D., Ortiz-Bobea, A., Schewe, J., Song, J., Verheyen, J., Vitousek, P., Wada, Y., Xia, L., Zhang, X. & Zhuang, M. 2024. Climate change exacerbates the environmental impacts of agriculture. *Science* **385**: eadn3747.

Yeates, G. W., Bongers, T., De Goede, R. G., Freckman, D. W. & Georgieva, S. S. 1993. Feeding habits in soil nematode families and genera-an outline for soil ecologists. *J Nematol* **25**: 315-31.

# Supplementary Material

## Supplementary Tables

**Supplementary Table S1. The effect of carbon practice vs control and carbon practice type on occurrence of taxonomic and functional groups of arthropods.** Statistically significant (*P* < 0.05) effects are indicated in bold.

| Taxonomic groups | *Hymenoptera* | | | *Arachnida* | | | *Aranea* | | | *Coleoptera* | | | *Collembola* | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | d.d., d.d.f. | *F* | *P* | d.d., d.d.f. | *F* | | d.d., d.d.f. | *F* | *P* | d.d., d.d.f. | *F* | *P* | d.d., d.d.f. | *F* | *P* |
| **Carbon practice type** | 3, 15 | 0.53 | 0.6654 | 3, 15 | 1.19 | 0.3459 | 3, 15 | 0.49 | 0.6956 | 3, 15 | 1.59 | 0.2337 | 3, 15 | 3.25 | 0.0516 |
| **Field status (carbon practice type)** | **3, 15** | **25.86** | **<.0001** | **3, 15** | **34.07** | **<.0001** | **3, 15** | **105.95** | **<.0001** | **3, 15** | **3.41** | **0.0356** | **3, 15** | **31.98** | **<.0001** |
| | *Diptera* | | | *Homoptera* | | | *Heteroptera* | | | *Thysanoptera* | | | | | |
| | d.d., d.d.f. | *F* | *P* | d.d., d.d.f. | *F* | | d.d., d.d.f. | *F* | *P* | d.d., d.d.f. | *F* | *P* | | | |
| **Carbon practice type** | 3, 15 | 1.59 | 0.2337 | **3, 16** | **7.54** | **0.0026** | 7.54 | 0.0026 | 0.0796 | 3, 15 | 1.25 | 0.3253 | | | |
| **Field status** | **3, 15** | **3.41** | **0.0356** | **3, 16** | **11.86** | **0.0002** | **11.86** | **0.0002** | **<.0001** | **3, 15** | **6.4** | **0.0033** | | | |
| **Functional groups** | **Herbivores** | | | **Predators** | | | **Neutrals** | | | | | | | | |
| | d.d., d.d.f. | *F* | *P* | d.d., d.d.f. | *F* | *P* | d.d., d.d.f. | *F* | *P* | | | | | | |
| **Carbon practice type** | 3, 15 | 0.59 | 0.634 | 3, 15 | 2.03 | 0.1535 | 3, 15 | 1.95 | 0.1657 | | | | | | |
| **Field status (carbon practice type)** | **3, 15** | **36.71** | **<.0001** | **3, 15** | **92.9** | **<.0001** | **3, 15** | **26.24** | **<.0001** | | | | | | |

**Supplementary Table S2. The effect of carbon practice vs control and carbon practice type on occurrence of functional groups of nematodes.** Statistically significant (*P* < 0.05) effect is indicated in bold.

| Nematodes | Herbivores | | | Predators | | | | Omnivores | | | Bacteriovores | | | Fungivores | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | d.d., d.d.f. | *F* | *P* | d.d., d.d.f. | *F* | | *P* | d.d., d.d.f. | *F* | *P* | d.d., d.d.f. | *F* | *P* | d.d., d.d.f. | *F* | *P* |
| **Carbon practice type** | **3, 15** | **8.46** | **0.0016** | 3, 15 | 0.42 | 0.7412 | 0.4633 | 3, 15 | 3.1 | 0.0585 | **3, 15** | **4.24** | **0.0234** | 3, 15 | 2.8 | 0.0758 |
| **Field status (carbon practice type)** | 3, 15 | 1.91 | 0.1617 | 3, 15 | 0.68 | 0.6134 | 0.1994 | 3, 15 | 0.32 | 0.8621 | 3, 15 | 2.92 | 0.0573 | 3, 15 | 2.1 | 0.1316 |

**Supplementary Table S3.** The results of indicator species analysis on birds on four carbon practices and their controls. Statistically significant (P < 0.05) association is indicated in bold.

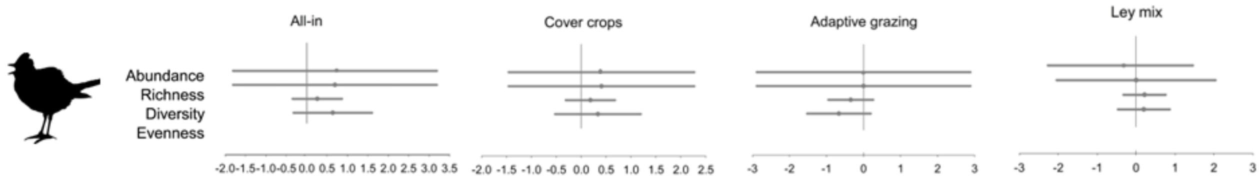
| | Carbon | | | | Control | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | All-in | Cover crop | Adaptive grazing | Ley mixture | All-in | Cover crop | Adaptive grazing | Ley mixture | index | *stat* | *P* |
| ***Alauda arvensis*** | **1** | **1** | **0** | **0** | **0** | **0** | **0** | **0** | **9** | **0.548** | **0.043** |
| *Anthus pratensis* | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.475 | 0.444 |
| *Chloris chloris* | 0 | 1 | 1 | 0 | 0 | 0 | 1 | 0 | 61 | 0.389 | 0.542 |
| *Chroicocephalus ridibundus* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 7 | 0.413 | 0.463 |
| *Columba palumbus* | 0 | 1 | 1 | 1 | 0 | 1 | 0 | 0 | 129 | 0.346 | 0.741 |
| *Corvus cornix* | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 6 | 0.386 | 1 |
| *Corvus monedula* | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 0 | 29 | 0.433 | 0.258 |
| *Curruca communis* | 0 | 1 | 1 | 0 | 1 | 0 | 1 | 0 | 133 | 0.356 | 0.562 |
| *Cyanistes caeruleus* | 0 | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 16 | 0.406 | 0.723 |
| *Dendrocopos major* | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 6 | 0.386 | 1 |
| *Emberiza citrinella* | 1 | 1 | 0 | 0 | 0 | 1 | 0 | 1 | 106 | 0.309 | 0.991 |
| *Erithacus rubecula* | 0 | 1 | 0 | 0 | 1 | 0 | 0 | 0 | 18 | 0.406 | 0.726 |
| *Fringilla coelebs* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 8 | 0.607 | 0.065 |
| *Grus grus* | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.475 | 0.429 |
| *Hirundo rustica* | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 13 | 0.406 | 0.736 |
| *Linaria cannabina* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 8 | 0.424 | 0.66 |
| *Locustella naevia* | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.386 | 1 |
| *Loxia curvirostra* | 1 | 0 | 1 | 0 | 0 | 1 | 1 | 0 | 115 | 0.360 | 0.606 |
| *Motacilla alba* | 0 | 0 | 1 | 1 | 1 | 0 | 0 | 0 | 73 | 0.375 | 0.474 |
| *Muscicapa striata* | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 3 | 0.475 | 0.437 |
| *Numenius arquata* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 8 | 0.366 | 0.859 |
| *Parus major* | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 9 | 0.431 | 0.292 |
| *Passer montanus* | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 15 | 0.423 | 0.461 |
| *Phylloscopus trochilus* | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 0 | 31 | 0.486 | 0.212 |
| *Pica pica* | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 0 | 25 | 0.447 | 0.255 |
| *Saxicola rubetra* | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 13 | 0.406 | 0.726 |
| *Spinus spinus* | 1 | 1 | 0 | 0 | 1 | 0 | 1 | 1 | 181 | 0.294 | 0.785 |
| *Sylvia borin* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 8 | 0.424 | 0.66 |
| *Turdus merula* | 1 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 55 | 0.511 | 0.101 |
| *Turdus pilaris* | 0 | 0 | 1 | 1 | 0 | 0 | 0 | 1 | 76 | 0.395 | 0.429 |
| *Vanellus vanellus* | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 35 | 0.380 | 0.949 |

**Supplementary Figures**

**Supplementary Figure S1**


**Supplementary Figure S2. Locations of the 19 Study Farms in Finland.** Locations of the study farms implementing different carbon farming treatments: all-in, cover crops, adaptive grazing, and ley mix.

**Supplementary Figure S2.**




**Supplementary Figure S1. Fig. 1**. **Effects of Carbon Farming Practices on Alpha Diversity.** Effect sizes from Tukey's test comparing carbon farming practices to controls are shown for alpha diversity metrics in field birds. Effect sizes are represented by points, with error bars indicating 95% confidence intervals. Statistically significant differences are highlighted with bold black lines.