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PAPER

Measuring the transition to regenerative agriculture in the UK with a co-designed experiment: design, methods and expected outcomes

Katherine Berthon[1,11,*], Coline C Jaworski[2,11,*], Jonathan D Beacham[8], Peter Jackson[5], Jonathan Leake[6], Niamh M McHugh[7], Lucy Capstick[7], Tim Daniell[6], Anna Krzywoszynska[3,4], Duncan Cameron[5], John Holland[7], Sue Hartley[6], Nicolas Desneux[2], Kelly Jowett[9], Yu Zhao[6], Penelope J Watt[6] and Lynn V Dicks[1,10,*]

1 Department of Zoology, University of Cambridge, Cambridge, United Kingdom
2 Université Côte d'Azur, INRAE, UMR ISA, Nice, France
3 Department of Geography, University of Sheffield, Sheffield, United Kingdom
4 History, Culture and Communications, University of Oulu, Oulu, Finland
5 Institute for Sustainable Food, University of Sheffield, Sheffield, United Kingdom
6 School of Biosciences, University of Sheffield, Sheffield, United Kingdom
7 Game and Wildlife Conservation Trust, Fordingbridge, United Kingdom
8 University of Bristol Business School, Bristol, United Kingdom
9 Rothamsted Research, Harpenden, United Kingdom
10 School of Biological Sciences, University of East Anglia, Norwich, United Kingdom
11 These authors equally contributed to this manuscript.
* Authors to whom any correspondence should be addressed.

E-mail: kb809@cam.ac.uk, coline.jaworski@inrae.fr and lv22@cam.ac.uk

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Abstract

Regenerative agriculture is promoted as a farming system that can improve agricultural sustainability, address soil degradation, and provide ecosystem service benefits. However, there remains limited evidence for the quantifiable benefits of a widespread transition to regenerative agriculture on soil, biodiversity, and crop quality, particularly at the landscape scale, and poor integration of findings across disciplines. Social and cultural aspects of the transition, such as the positioning of regenerative agriculture as a grassroots movement, farmers' perspectives on defining regenerative practices, and social or political barriers to implementation, are harder to quantify and often overlooked in evidence-based approaches. Here, we present the detailed methodology for our interdisciplinary, co-designed landscape-scale experiment measuring changes in soil health, biodiversity, yield, and grain quality, as well as social and political dimensions of the implementation of regenerative practices. Our unique approach, through the co-production process, the landscape-scale, and the focus on a systemic transition instead of individual practices, will bring strong evidence of the benefits of regenerative agriculture for sustained agricultural productivity, the mitigation of climate change and biodiversity depletion in agroecosystems. Our research aims to guide future studies transforming theoretical ecology into testable hypotheses in real-world systems and provide actionable evidence to inform agricultural policies in the UK and beyond.

1. Introduction

Agricultural soil degradation is a global, unprecedented crisis, caused by intensive agriculture practices, including deep soil cultivation and synthetic inputs for annual, resource-demanding but high-yielding crops. Intensive farming practices have led to reduced agricultural productivity through loss of organic matter and soil structure, and to biodiversity loss (Graves *et al* 2015, Evans *et al* 2020, Raven and Wagner 2021). In response to these issues and to address climate change, a range of alternative farming systems have been devised and evaluated, with organic farming, conservation agriculture, and regenerative agriculture concepts all emerging in the 1980–90 s (Holland *et al* 1994). Organic farming prohibits the use of synthetic inputs (pesticides and fertilisers) but has been criticised in its industrial form as failing to promote environmental processes and requiring more land to maintain agricultural production than conventional agriculture (Tscharntke *et al* 2021). Conservation agriculture has been promoted for soil sustainability, seeking to minimise soil disturbance through reducing tillage, providing permanent soil cover and diversified rotations (Hobbs *et al* 2008, Giller *et al* 2015). However, it has been criticised for its narrow focus on cropping operations and its reliance on costly technological solutions, preventing widespread adoption (Bless *et al* 2023).

Regenerative agriculture originated from the Rodale Institute in the USA in the 1980s (Rodale 1983). More recently, it has been promoted in response to frustrations with previous paradigms, and a desire to restore natural processes that support soil health and reduce dependence on costly inputs rather than a narrow focus on maintaining productivity (LaCanne and Lundgren 2018, Sherwood and Uphoff 2000, Tittonell *et al* 2022, Bless *et al* 2023). In the UK context, regenerative agriculture seeks to achieve its goals by mobilising nature-based solutions instead of artificial inputs, commonly through adherence to five key principles: (i) reduce soil disturbance, (ii) increase crop diversity, (iii) keep the soil covered, (iv) keep living roots all year round, and (v) introduce livestock (Ritz 2021, Farm of the Future: Journey to Net Zero 2022, Khangura *et al* 2023). These practices enable the regeneration of root-derived carbon in much larger amounts than provided by annual crops, and enhancement of soil biodiversity, in turn supporting mycorrhizal fungi and symbiotic nitrogen fixation to reduce requirements for phosphorus and nitrogen fertilizers. Increases in earthworm and mycorrhizal activity regenerate soil macroaggregates (soil structure), and therefore soil carbon sequestration capacity (Guest *et al* 2022). These soil chemical and biological changes are expected to have broader environmental benefits, including climate change mitigation, improvements to water quality and biodiversity (Fenster *et al* 2021, O'Donoghue *et al* 2022); as well as economic and human health impacts through changing yields, input use, and nutritional quality of crops (Montgomery *et al* 2022, Khangura *et al* 2023).

The regenerative agriculture concept is gaining political importance, as shown by the peer-reviewed grant awards to major research programmes funded by UK Research and Innovation (Jackson *et al* 2021, Doherty *et al* 2022), but also in the EU with a recent assessment of regenerative agriculture by the European Academies Science Advisory Council (EASAC policy report 44, 2022). It is also considered an important way to reach 'Net Zero Emissions' goals, through enhanced carbon sequestration in agricultural soils, while also improving agricultural sustainability (Defra 2023, Khangura *et al* 2023). Aspects of regenerative agriculture are being incentivised under UK agricultural policy and in particular the Sustainable Farming Incentive (SFI) of the current UK Environmental Land Management schemes (ELMs) for England (Defra 2023). Notably, SFI schemes subsidise practices that are considered to increase carbon sequestration such as introducing herbal leys in arable rotations and the use of winter cover crops (Lal *et al* 2007, Virto *et al* 2015).

However, the grassroots origin of regenerative agriculture has led to various and often incongruous definitions of regenerative practices and principles (Newton *et al* 2020, Bless *et al* 2023, Jaworski *et al* 2023a). On the one hand, regenerative agriculture can be defined as an open-ended and potentially radical farmer-led paradigm shift (Beacham *et al* 2023), and on the other, a process and outcomes-based approach to farming (Newton *et al* 2020) that can be quantified and normalised in progressive policy agendas (Gordon *et al* 2023). Consideration of regenerative agriculture as a movement values flexibility in definition of regenerative practices, allowing for adaptation to the local context (O'Donoghue *et al* 2022), and adoption of different suites of practices by farmers in different environments (Jaworski *et al* 2023a) that do not necessarily share core values (Tittonell *et al* 2022). Flexibility in definition, combined with limited scientific evidence for the link between process and outcomes (e.g. reduced tillage leading to increased soil carbon sequestration, Tittonell *et al* 2022) has prevented a unified definition of which practices contribute to restoring soil health and should be promoted as part of the movement (Tittonell *et al* 2022, Khangura *et al* 2023).

Therefore, understanding the efficacy of a transition to regenerative agriculture requires measuring whole-system changes, where farmers fully redesign their farming system to incorporate new crops and farming practices, as well as reshaping perceptions, value-systems, and worldviews (Gordon *et al* 2022, Miller-Klugesherz and Sanderson 2023, Seymour and Connelly 2023). This entails the adoption of a combination of practices that are expected to act synergistically (Li *et al* 2020, Jaworski *et al* 2023a), as well as understanding the economic and pragmatic incentives to their adoption (Beacham *et al* 2023). However, most practices have only been tested individually, at best at the field scale (reviewed in Khangura *et al* 2023), and less quantifiable aspects of the transition such as motivations, interrelationships and regenerative mindsets are rarely considered in process-based definitions (Gordon *et al* 2023, Jaworski *et al* 2023a). In addition, many of the agroecological processes that regenerative agriculture could restore, including

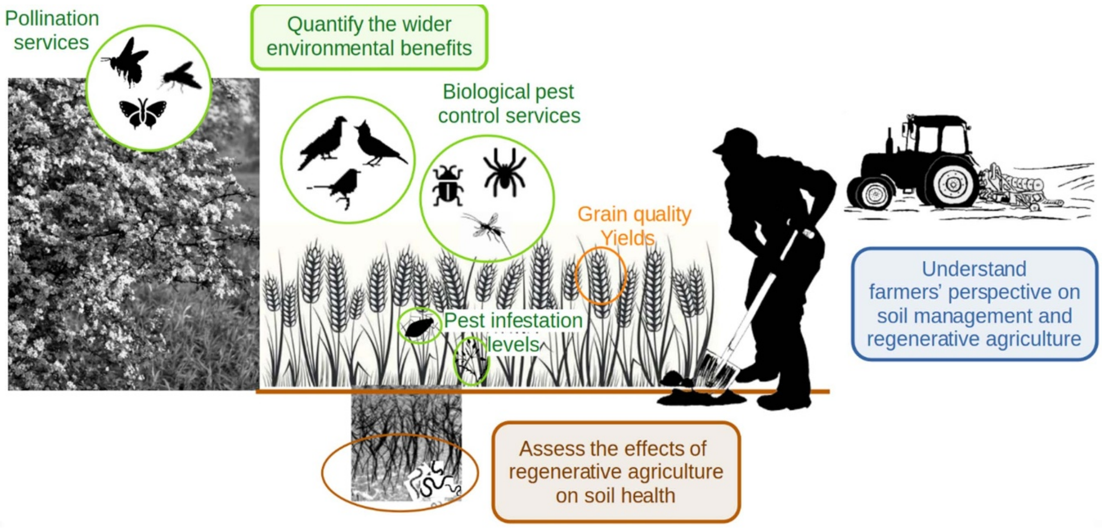

Figure 1. Outline of the study objectives and transdisciplinary approach.

provisioning and regulating ecosystem services such as biological pest control and pollination play out at larger spatial scales (Rusch *et al* 2016, Harrison *et al* 2019, Jaworski *et al* 2022), and require translating field-scale outcomes to landscape-scale processes.

We argue that measuring the systemic transition to regenerative agriculture must be achieved through locally-based knowledge co-production between farmers and researchers. As a farmer-led movement, regenerative agriculture centres farmers both as knowledge producers, leading the inquiry into the efficacy and development of regenerative farming practice (Krzywoszynska 2019) as well as decision-makers, adapting their farm systems to this new direction (Kleijn *et al* 2019, Norström *et al* 2020). Quantifying the impacts of regenerative agriculture practices is seen as important by regenerative agriculture practitioners, as it will enable them to benefit from shifting policy and market frameworks (Krzywoszynska 2019).

Here, we present our transdisciplinary, co-design approach to quantify the broad environmental benefits of a transition to regenerative agriculture from conventional arable and mixed farming practices. While we engage with a process-based definition of regenerative farming, we worked together with two groups of UK farmers to design a tailored set of practices that did not lose sight of regenerative agriculture as a movement. We approach this study in a holistic manner integrating ecological benefits (impacts on soil health, biodiversity, and ecosystem services), economic factors (yield, input costs and crop quality), and the social, political, and cultural dimensions of the transition (figure 1). We work beyond field scale with a replication unit of 60 ha, which captures shifts in agricultural rotations and farming functioning across fields, and, where the theoretically optimal design is adapted to fit real-world farming (Lacoste *et al* 2022). Specifically, we investigate the following questions:

1. What is the impact of a systemic transition to regenerative agriculture on key soil physical, chemical, and biological properties?
2. What are the wider environmental benefits of such a transition for biodiversity and associated ecosystem services, using farmland birds, pollinators, insect pests and natural enemies as indicators?
3. What is the effect of such a transition on cereal crop yields and grain quality?
4. What are farmers' perspectives on regenerative agriculture in the current policy and market environments?

2. Material and methods

We established a four-year co-designed quasi-experiment in 25 sites managed by 17 farmers across two landscapes in the UK. The co-production process is enabled by fortnightly meetings of a working group, composed of farmer cluster leaders and coordinators, and scientists from social sciences, soil sciences, and ecology. This experiment has been approved by an ethical agreement (ref. PRE.2021.055 delivered by the Cambridge Psychology Research Ethics Committee on 10 August 2021). The experiment itself started after harvest in August 2021, and will continue for four years (until harvest 2025). Years 1–4 are defined from post-harvest to the next harvest. Data has been collected in Year 1, the baseline year (August–September 2021 to July–August 2022); new datasets will be collected in Years 3 and 4 for all metrics except otherwise stated below and following the same methodology.

2.1. Farmer cluster selection

To achieve strong and consistent input into the research from farmer participants and design a landscape-scale experiment, the research relied on farmer clusters, i.e. voluntary farmer associations dedicated to knowledge exchange between farmers in a specific local area (GWCT 2019). We used the following criteria to guide the selection of farmer clusters; ideally farmer clusters should:

- encompass a diversity of farming systems, from more conventional to more regenerative;
- have a historic interest and enthusiasm for sustainable agriculture in general, and for regenerative agriculture in particular;
- have established working relationships with researchers with sites accessible for frequent visits.

Two farmer clusters were formally enrolled in the research: one in the East and one in the Southwest of England. These farmer clusters represent a diversity of farming systems, including arable only farms and mixed arable and livestock farms. These farms may also experience different climates (the Southwest has higher rainfall and colder winters) and are situated on different soil types (either thin, chalk-dominated soils, or heavy clay soils). An academic team met with each farmer cluster in July 2021 to share the objectives and requirements of the experiment and collect informed consent, assuring full anonymity of all data collected, but also participants' right to retract at any time. For confidentiality reasons, neither the farmer clusters nor the individual farms or farmers are identified here.

2.2. Experimental design and site selection

The first co-production step was to establish a list of regenerative agriculture practices, the effects of which would be studied. Rather than prescribing an established set of principles and practices, we asked farmers to assemble a list of practices that they considered to represent regenerative farming within their farming contexts. This list was refined by the working group, based on consensus across clusters, and potential for improvements to soil health (Lal *et al* 2007, Virto *et al* 2015). This resulted in a list of nine practices:

1. Use no-till/minimum non-inversion tillage, with direct drilling as a standard and not just on certain crops;
2. Use reduced compaction techniques e.g. low tyre PSI, control traffic farming;
3. Introduce crop diversity in the rotation (especially legumes);
4. Use multi-species cover and catch cropping in the rotation;
5. Increase spring cropping in the rotation;
6. Retain crop residues in field;
7. Introduce livestock to arable rotations;
8. Use organic manures/ green compost/ digestate/ compost;
9. Introduce herbal leys in the rotation.

To test the effects of changing practices on soil health and on biodiversity, and to understand how farmers make that systemic transition, a first experimental design was proposed by researchers. It followed a BACI design (before-after-control-impact; Osenberg *et al* 2006, Christie *et al* 2019) and consisted of three treatments each with four replicates per cluster:

- Control: conventional farming: characterised by short rotations with low crop diversity and intensive tillage;
- Regenerative: farmed according to the regenerative principles, using practices defined above for at least four years before the start of the study;
- Change (or 'impact' group): conventional farming shifting to regenerative farming after one year into the study (the 'before' baseline year) by adopting some combination of the nine practices above.

The regenerative group was added to the BACI design due to the short-term nature of the study, making it difficult to measure significant changes in soil structure and chemistry, and potentially biodiversity, in less than four years (Puerta *et al* 2018, Guest *et al* 2022). During the co-design process, the before-after structure was altered to accommodate a diversity of transition trajectories: farmers in the Change group could have already implemented some of the practices at the baseline year, and each Change farmer defines their own regenerative agenda (e.g. what practices they would adopt immediately after the baseline year, and what

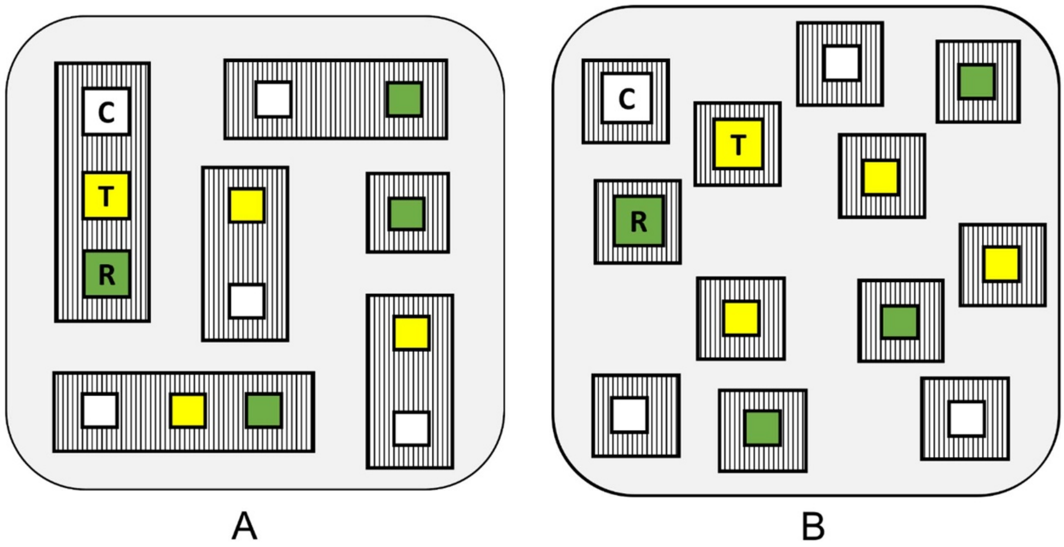

Figure 2. Experimental design and layout of the experimental sites in the southwest cluster (A) and the eastern cluster (B). Each striped square represents a farm, and each coloured square represents one site of ⩾60 ha of farmland. In the Southwest, farmers have contributed one to three sites each, in a paired design, whereas in the Eastern cluster, each farmer will be managing one site in the experiment. 'C' (white) Control group (farmed with conventional practices); 'R' (green) Regenerative group (farmed with regenerative practices); 'T' (yellow) treatment (Change) group, with practices changing from conventional to regenerative over the course of the experiment.

practices they would implement later). These alterations make the timeline less stringent than in a theoretical BACI experiment, but still allow a farm effect to be modelled, in which measures repeated through time can be compared with earlier measures of the same farm only instead of being compared with a past mean effect across all farms.

In each cluster, farmers enrolled a part of the land they farm (hereafter 'site') in one of the three treatments (Control, Regenerative and Change). The site was to be ⩾60 ha in size, and, ideally, composed of multiple fields instead of a single large field, each representing different stages in the rotation and a diversity of crops (including one winter wheat field for the first year, if possible). This led to a total of thirteen sites managed by six farmers, each providing access to between one and three sites, in the South–West cluster, and twelve sites managed by eleven farmers in the East cluster, each providing access to one site, except for one farmer who provided two sites on separate farms (figure 2).

2.3. Farming practices data

To get a better understanding of the legacy effects from past practices and monitor practices used in the experimental sites over the four years of the study, we collect information on cropping rotations, cultivation practices, and pesticide and fertiliser inputs per farm (where available) from 2016 to 2022 ('historical' practices) and 2023–2025 (during the study) on a per-field, per-year basis in each field of the 25 sites included in the landscape-scale experiment. This information is collected through a combination of farmer interviews, pdf-form surveys and from record-keeping software used by farmers (e.g. Gatekeeper), and includes crops grown, inputs used and implementation of the regenerative practices in section 2.2. We also map Countryside Stewardship options and other ELMs, and their rotations across the study period, as well as record some general information about the remainder of the farm outside the study area (site) but in less detail—i.e. presence of regenerative practices, farm size and type (arable or mixed). These data will be used to explain the variation in our observed biodiversity and soil health outcomes and capture the gradient of implementation of regenerative practices (section 3).

2.4. Farmer, societal and policy contexts to adoption of regenerative agriculture

To understand and contextualise change over time in mindsets as some farmers have adopted more regenerative practices, we conduct two rounds of one-on-one semi-structured interviews with farmers involved in the management of one or more farms in the study; once at the beginning of the project (Spring 2022) and once after the transition has occurred (Spring 2025). The first year of interviews established a baseline against which to compare changes over the next three years of the experiment. Interview topics included the nature of the farm business, their understanding of the term 'regenerative agriculture', sources of information and advice, the changing policy environment, and the future of UK farming (Beacham *et al* 2023). We also attended a series of farmer cluster meetings to understand how farmers talked amongst themselves about the adoption of, and challenges associated with, regenerative practices. This included meetings with invited presentations from agronomists and other 'experts', arranged and organised by the farmers themselves.

In addition, we will undertake a series of 12–15 in-depth interviews with actors in the policy domain (representatives from state agencies, NGOs and charities) to understand their perspective on the changing policy environment in which farmers are operating. These will include the shift from the Common Agricultural Policy, based on payments per hectare of land under cultivation, to ELMs, based on the payment of 'public money for public goods' (Defra 2023). The interviews will allow us to identify the different 'epistemic communities' (Gough and Shackley 2001) at play in this terrain: these are communities that differ not only in terms of how problems around the status quo are constructed, but also in their objectives, core beliefs, and the extent to which regenerative agriculture represents a favourable response.

2.5. Quantifying the impacts of regenerative agriculture on soil health

In line with a recent recommendation to the UK Government Soil Health Enquiry, we have measured topsoil water-stable macroaggregates (>2 mm) and the proportion of soil organic C and N in these aggregates as an integrated measure of soil health in arable landscapes (Guest *et al* 2022). Water-stable macroaggregates, which are generated by the interacting effects of roots, earthworms, and mycorrhizal fungi, are the critical structures within which organic carbon and nitrogen are sequestered, and macropore space maintained, ensuring good soil infiltration, water storage and drainage (Guest *et al* 2022). They are therefore highly responsive, early indicators of improvements in soil health, including carbon sequestration, fertility-building, improved biological and hydrological functioning (Puerta *et al* 2018, Guest *et al* 2022). To corroborate biological enhancement, we measure earthworm biodiversity, and use bulk density profiles to assess soil compaction, along with pH, all of which are common indicators of the influence of farming practices on soil health (Hallam *et al* 2020, Liptzin *et al* 2022, Khangura *et al* 2023). Comparison of these measures will be used to establish guideline values for different soil types and a methodology for landscape-scale soil monitoring of topsoil health.

Soil is sampled in February-March of Years 1 and 4, when the soil is moist, and under optimal weather conditions (not rainy, temperatures between 8 °Cand 15 °C). Soil sampling follows a standard 'W' shape composed of eight locations per field (figure 3(A)). For one field per site and at each of the eight locations, standard 100 cm^3 bulk density cores are collected to four different depths: 0–7, 7–14, 14–21, and 21–28 cm, which cover the topsoil, and plough layer. Earthworm densities are sampled at the same time as the soil coring, at three locations per field (figure 3(A)) using the method by Römbke *et al* (2006). A block of topsoil of 18 × 18 × 20 cm is extracted and placed on a blue plastic sheet, and earthworms are sorted manually, starting with those visible on the walls of the pit. Earthworms are counted and identified as juveniles versus adults (presence of a clitellum). In addition, a Visual Evaluation of Soil Structure, hereafter VESS, score is performed on a 4 × 18 × 20 cm slice extracted from the same pit and following the method and scoring by Vidacycle (https://soils.vidacycle.com/soil-tests/vess-visual-evaluation-of-soil-structure/).

2.6. Quantifying the wider environmental benefits of regenerative agriculture

We investigate the potential benefits of regenerative agriculture on three aspects of biodiversity and their related ecosystem services: winter farmland birds, particularly farmland-specialist and insectivorous birds; insect natural enemies (spiders, carabid beetles, ladybirds, parasitoids, and hoverflies) and their related pest control functions; and pollinators and associated pollination services. We have chosen these groups as they are responsive to changes in land management (e.g. Zielonka *et al* 2024), have purported responses to the regenerative practices given in 2.2 (see section 3.2, table 3) and provide important ecosystem services to agriculture (Issacs *et al* 2009). For example, regenerative agriculture has the potential to reverse declining population trends in farmland birds (Robinson and Sutherland 2002) by recreating favourable habitats through use of diverse rotations, winter stubbles and cover crops, and less productive stages in the rotations with reduced inputs (e.g. herbal leys; that support greater invertebrate populations; Donald *et al* 2002). Below we provide in brief the sampling methods chosen as appropriate for each group, but for more information on detailed sampling protocols see the Supplementary Methods.

2.6.1. Farmland bird sampling

We focus on winter farmland birds because winter is typically the time when birds become more reliant on seed and plant food, and earthworms in fields (Holland *et al* 2006). Birds are monitored from November to February of Years 1, 3 and 4, using three timed 600 m transects placed in fixed locations across the sampling site (see supplementary material for more information on placement rules). Each transect is composed of one 200 m section of field-margin and two 200 m in-field sections, each at least 50 m away from field corners (figure 3(A)). This was established as the best compromise to (i) record the widest bird diversity but

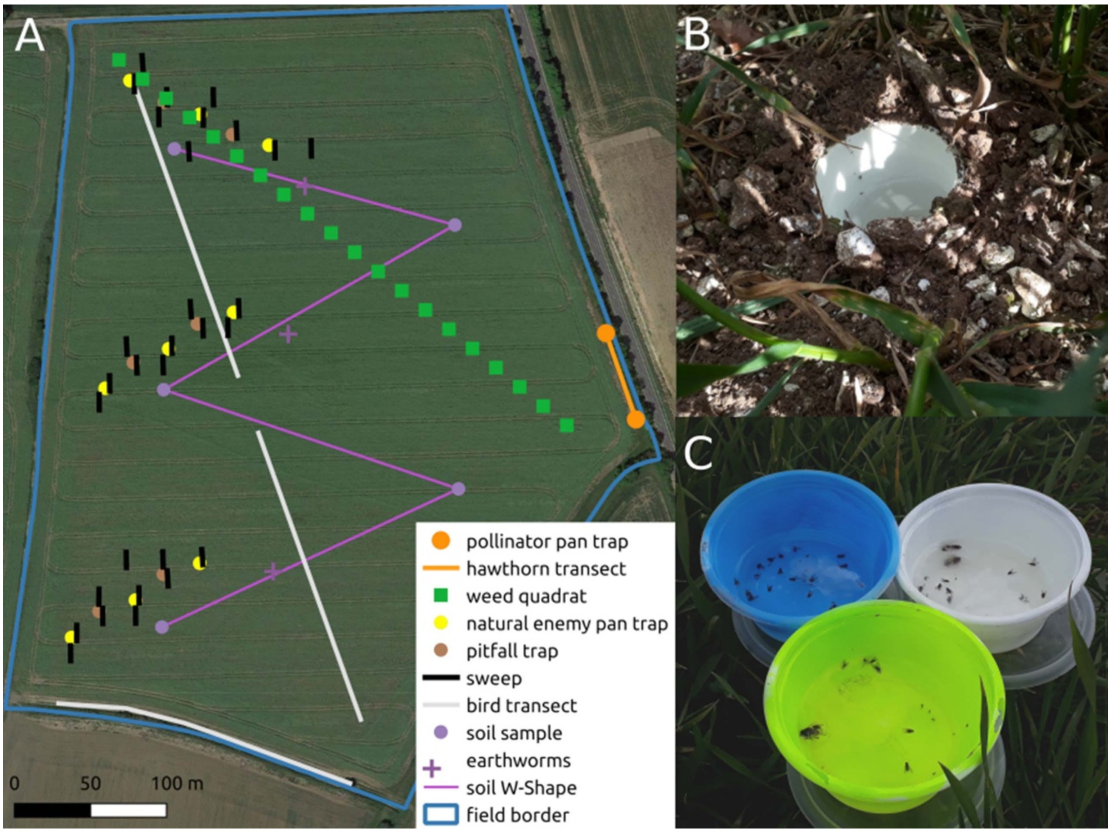

Figure 3. (A) An example of the location of soil and biodiversity samplings in a field. Locations of soil sampling and bird transects are mapped precisely using ViewRanger, while the other sample locations are approximative. (B) A pitfall trap and (C) a pan trap used for natural enemy sampling.

especially that of species using in-field habitat; (ii) exhaustively sample sites in a relatively limited time window (Wilson and Gillings 2002, Atkinson *et al* 2006); and (iii) maximise the detectability of the treatment effects (farming practices affect birds using in-field habitat; Donald *et al* 2002). The three transects per site can be used as statistical units, that is pseudo-replicates of bird foraging activity measures.

Sampling occurs twice per year, first in November–December and then in January–February, with all sites of each farmer cluster sampled over a maximum time period of one month to minimise temporal variability, and with surveys starting one hour after dawn and finishing one hour before dusk to avoid counting birds moving to/from roost sites. Weather conditions, start and end of time walk for each transect as well as sunrise and sunset times are also recorded. In addition, vegetation height and percentage of bare soil are recorded in a 0.5×0.5 m quadrat at the end of each transect section (12 quadrats per site) to account for reduced visibility of birds on the ground.

2.6.2. Pest and natural enemy sampling

As proxies to estimate the effects of farming practices on biological pest control services, we measure abundances of aphid pests and their natural enemies in cereal crops (Chaplin-Kramer *et al* 2013, Jaworski *et al* 2015, Holland *et al* 2021). We focus on cereals as the economically most important cash crop in both farmer clusters; cereal crops are important both in arable and mixed UK farming systems, comprising 76% of the land area cropped in the UK (Defra 2022); and break crops (i.e. crops used to break cereal diseases, and which come after multiple years of cereal cropping) often differ between regenerative (e.g. peas, beans, leys) and conventional rotations (e.g. oil-seed rape). We also target several natural enemy functional groups with contrasting ecologies to disentangle the various mechanisms of regenerative practices effects e.g. changing aphid resources, alternative resources, and reduced insecticides. This includes ladybirds (generalist predators but high preference for aphids; Sloggett 2008, Pan *et al* 2020), ground-dwelling predators (carabid beetles and spiders—often generalists but less efficient at controlling aphids; Lövei and Sunderland 1996, Pywell *et al* 2015), hoverflies (specialist aphid predators; Chabert and Sarthou 2017) and parasitoid wasps (specialist aphid parasitoids; Monticelli *et al* 2019).

When present in a site and in each sampling year, one field of winter wheat, winter barley and/or spring barley are selected for pest and natural enemy sampling. Unlike the bird transects, these are not fixed from year-to-year but follow cereal fields as they move in the rotation. We use a variety of sampling methods to best capture abundances and species richness of pests and natural enemies (McCravy 2018, figure 3(A)), i.e. pitfall trapping to measure the activity-density of and catch ground-dwelling predators (Hohbein and Conway 2018, figure 3(B)), pan trapping to measure the abundances of aphid parasitoid wasps and hoverflies (O'Connor *et al* 2019, figure 3(C)) and sweep-netting to assess aphid and parasitized aphid abundances, as well as ladybird abundances.

We use four sampling rounds per year to capture seasonal fluctuations of natural enemies throughout the cropping year (from crop emergence to harvest) and allow for estimating the major demographic trends: (i) late October to early November, when winter cereals have grown enough biomass to host quantifiable insect populations (i.e. 5–10 cm high); (ii) first three weeks of April; (iii) second half of May; and (iv) late June-early July (before harvest). For logistical reasons (unusually late harvest and sowing in 2022), the Autumn round was not carried out in Year 1. The Autumn round coincides with aphid immigration from grass margins and other green bridges into crop fields (first leaves emerged, growth stage GS13-GS19, plant height ∼5–8 cm; Bayer Aphid Expert Guide 2013, AHDB Wheat Growth Guide 2021); at this time of the year the predominant natural enemies are spiders but there also remain parasitoid wasps, some beetle species and hoverflies looking for food resources and overwintering sites (Legrand *et al* 2004, Raymond *et al* 2014). In early April (growth stage GS31, winter wheat stem elongation), the crop starts to grow again after winter dormancy phase (AHDB Wheat Growth Guide 2021). This is also when natural enemies emerge from overwintering (Legrand *et al* 2004, Raymond *et al* 2014). In late May, pest and natural enemy populations are at their highest growth rate, while in late June, populations have reached their highest levels or started to decline again, due to the decline in suitable plant resources soon before harvest (Holland *et al* 2005, Ciss *et al* 2014, Raymond *et al* 2014).

2.6.3. Pollinator sampling

The main pollinator-dependent crops in our farming systems are oilseed-rape, beans, and other break crops (Defra 2022), which differ between control and regenerative sites and therefore cannot be used to compare the abundance and diversity of visiting pollinators nor pollination levels across the differently farmed landscapes. For this reason, we focused on a common non-crop pollinator-dependent plant, hawthorn *Crataegus monogyna* Jacq. (García and Chacoff 2007, Jacobs *et al* 2009) which allows pollinator monitoring in any year, irrespective of the crop rotation. This species is abundant in site hedgerows of both landscapes and has shown pollination deficiency, providing an opportunity to measure improvement due to higher levels of floral resources for pollinators (Jacobs *et al* 2009).

To quantify the abundance and diversity of pollinators in each site, transect walks and pan traps are used along one fixed hedgerow transect per site twice a year during Years 1, 3 and 4: at the time of hawthorn flowering peak (May), and of immature fruit development (July) (figure 3(A)). In most cases, the hedgerow with the highest abundance of hawthorn among the three hedgerows used in bird transects was used; if none of these three had hawthorn, an alternative hedgerow with the highest hawthorn abundance was selected. Each hedgerow transect is 60 m long, and hedgerow characteristics (adjacent habitat type, orientation, management) have been recorded. On each sampling period the transect is walked twice; first to record general abundance of pollinators, and then to record pollinators specifically visiting hawthorn flowers. Two pan traps are placed at each end of the hawthorn transect immediately after hawthorn pollination sampling (figures 3(A) and (C)), and placed in the field for 48 h. Pan trap devices are the same as those used for natural enemy sampling (section 2.6.2) but are used in a different temporal window.

To quantify how other flowering resources may affect hawthorn pollination levels, the abundances of all flowering species in the hedgerow as well as in the adjacent field margin are quantified in six replicates starting at position 0 of the transect and every 10 m. Hedgerow replicates are vertical quadrats of 1×2 m examined while facing the hedge, and the field margin replicates are 1×1 m horizontal quadrats laid on one side of the hedge on the hedge bank or field margin. In both quadrat types, all flowering species are identified, their number of flower units counted, and their percentage cover visually estimated (POMS 2023). In case of very abundant flower resources, the number of flower units is counted in a subsection of the plot only and then extrapolated. Counts in the vertical plot also include hawthorn. In vertical plots, the approximate height and width of the hedgerow are recorded. In the horizontal plots, the percentage cover of bare ground is recorded. In addition, the floral resources, and percent bare ground are recorded in a 2 m radius around each pan trap.

2.6.4. Fruit set and pollination efficacy

As a proxy for pollination service, we measured the fruit set of hawthorn plants within the hedgerows sampled for pollinators. We selected twenty groups of buds during the first pollinator sampling round (section 2.8), randomly positioned at intervals of 3 m and at height 0.5–2 m above the ground along the hawthorn transect (Jacobs *et al* 2009; figure 3(A)). Where this is not possible due to low hawthorn abundance, groups of buds are randomly selected within hawthorn subsections. The number of buds and/or flowers in each group of buds are counted. In July as fruits start to develop, the number of immature fruits in each marked group of buds is counted (Jacobs *et al* 2009). This provides estimates of initial pollination rates. Indeed, in fruit-producing plants, abscission of unfertilized immature fruits—which may be due to inadequate pollination—occurs soon after flowering (Jacobs *et al* 2009). Immature fruits that are most likely to mature are those that result from outcrosses (successful pollination; Stephenson 1981).

2.6.5. Weed sampling

Reducing tillage may have implications for weed management on farms. Weeds are sampled in the same field as for pests and natural enemy sampling (section 2.6.2), once a year in the first two weeks of May. We use the diagonal sampling method to fairly represent weed density and diversity (Colbach *et al* 2000, Golafshan and Yasari 2012). That is, 30 m away from field corner, weeds are sampled in a 0.5*0.5 m quadrat every 20 m and repeated 20 times (figure 3(A)). If the field size was too small to place all quadrats in a diagonal, the last quadrats are placed along the second diagonal at least 30 m away from the field corner. The percent of coverage (in cm^2, accurate to 2 cm^2) in the quadrat as well as the identification of each weed is recorded *in situ* using local botany guides: Hubbard (1992) for grasses, and Rose and O'Reilly (2006) for other flowering plants.

2.7. Yield and grain quality outcomes

To capture potential trade-offs between ecological and economic impacts of regenerative practices and to measure potential benefits of regenerative agriculture for human nutrition, in terms of agriculture product quality, we collect farmers' data on crop yields and quality from our experimental sites. At each harvest we also collect 1 kg of grain of winter wheat from each site (if available) and analyse for macro and micro-nutrients, mycotoxins and pesticide residues as indicators of grain quality. Using information on variable costs and yields, we also perform a basic economic comparison with the FarmBench online benchmarking tool.

2.8. Statistical considerations

2.8.1. Power analyses

Power analyses were conducted using R Core Team (2022) to define the sampling sizes for natural enemy and pollinator data based on expected size effects and standard deviations (number of pan traps, pitfall traps, and sweep netting bounds). We followed the approach developed by Breeze *et al* (2021). Independent power analyses were performed for the abundances of hoverflies, bees, aphids, ladybirds and carabid beetles, and tested the effects of the number of rounds, the number of replicates per field, and the range of expected size effects on the likelihood to detect a significant treatment effect (table 1). Due to the real-farm constraints which shaped the experimental design, a conservative approach was adopted, where each dataset was simulated across one landscape and eight sites with two treatments (control—4 sites: conventional farming; treatment—4 sites: expected effect sizes without considering the time since change in practices). Note that expected effect sizes listed in table 1 often relate to tillage intensity only, while regenerative systems are expected to differ along multiple dimensions relative to conventional systems, and therefore extrapolated effect sizes may be larger (section 3).

Final sampling sizes were established as a compromise between logistics (the highest sampling size feasible within time constraints), the likelihood to detect the expected effects, and environmental ethics: where adding more samples did not change the likelihood to detect an effect, the lowest sampling size was selected, to minimise the negative impact of sampling on biodiversity. For more detail on the power analysis implementation, see the Supplementary Material.

2.8.2. Arthropod identification methods

The identification methods for arthropods collected by pan- and pitfall-trapping entail trade-offs between taxonomic resolution and speed of processing samples. The objective is to design high-throughput, accurate methods to be able to process a very large number of specimens and obtain statistically and ecologically meaningful data, that is quantitative data with a large sampling size and in a limited budget and time. To this end, we focus on key taxonomic groups that are known to be associated with ecosystem service provision: Hymenoptera (containing pollinating bees, and parasitoids); Syrphidae (pollinating flies); Araneae (terrestrial spiders); and Coleoptera (predatory beetles). In table 2, we outline the expected resolution and accuracy achievable for identification in these groups, as well as the resources available.

Table 1. Summary of power analyses.

Ecological metric	Expected size effect (%)	Power (%)	References
Number of hoverflies per pan trap	+50 to +100	70–98	Chabert and Sarthou (2017), (Chabert and Sarthou 2020)
Number of bees per pan trap	+20 to +50	12–39	Williams *et al* (2010)
Number of aphids per 10 sweeps	− 45 to −75	73–95	(Kennedy *et al* (2010), Chabert and Sarthou (2017), (Chabert and Sarthou 2020)
Number of ladybugs per 10 sweeps	−25 to −35	21–50	Tillman *et al* (2004); Tamburini *et al* (2016)
Number of carabid beetles per pitfall trap	+36 to +300	21–100	Kosewska *et al* (2014); Jowett *et al* (2021)

Table 2. Identification of arthropod groups used as indicators in our study, the expected taxonomic resolution and accuracy achievable, and taxonomic keys used.

Ecosystem Service Role	Biodiversity Indicator	Taxonomic Resolution	Accuracy of Identification (%)	References
Natural Enemies	Spiders (Araneae)	Family	100	Jones-Walters (1989)
	Beetles (Carabidae)	Species	90–95	Luff (2007), Jowett (2022)
	Beetles (Coccinellidae)	Species	100	Roy and Brown (2018)
	Parasitoids	Family	80	Yeo and Corbet (2015)
Pollinators	Bees (Anthophilla)	Genus or Species	95–100	Falk (2019)
	Hoverflies (Syrphidae)	Genus or Species	90–95	Stubbs and Falk (2002)

3. Anticipated results and avenues for analysis

Our experiment will generate a large amount of quantitative data on soil structure, chemistry, and biology, on insect and bird biodiversity, and on cereal yield quantity and quality, across 25 sites in two UK landscapes. Although scientific evidence on the effects of regenerative practices on these metrics are scarce, especially relative to combinations of practices, we drew expectations from published literature to establish the experimental design (section 2.2) and to determine sampling sizes (section 2.8.1). Here, we present these expectations, and highlight avenues to analyse the experimental data that will be produced in this large, co-produced, landscape-scale experiment in commercial farms.

3.1. Expectations

3.1.1. Soil physical, chemical and biological properties

We expect that long-term Regenerative fields will show improvements in soil physical, chemical, and biological properties compared to Conventional fields (table 3). Reduced soil disturbance using no-till techniques, combined with maintaining soil cover, is expected to increase soil organic carbon in the surface layers (Haddaway *et al* 2017; Ogle *et al* 2019) and promote biological activity of earthworms (Briones and Schmidt 2017). The increased use of N–fixing legumes in rotation will also increase chemical fertility (Sánchez-Navarro *et al* 2019). Incorporating soil residues, and integrating livestock is considered good practice under both conventional and regenerative systems (Khangura *et al* 2023), and can enhance soil carbon sequestration (Li *et al* 2019). Similarly, inclusion of cover crops, and diversifying crop rotations—often through the inclusion of spring crops—can enhance nitrogen and carbon storage in soil (Bai *et al* 2018, Abdalla *et al* 2019, Jian *et al* 2020). Conversely, soil compaction may initially increase with reduced tillage, due to a delay in biological activity recovery. However, it should decrease in the longer term via a diversity in rooting depth (Blanco-Canqui and Wortmann 2020), and the promotion of earthworm activity in soils receiving carbon inputs through living roots all year around, and experiencing reduced disturbance with less intensive tillage (Prendergast-Miller *et al* 2021).

However, the influence of no-till on soil carbon varies across climates (Ogle *et al* 2019, Abbas *et al* 2020, Sun *et al* 2020), between soil types (Page *et al* 2020, Shakoor *et al* 2021), and down the soil profile (Haddaway *et al* 2017). Our two farmer clusters differ in soil type and farming system characteristics, and this has implications for how soil characteristics may change in response to regenerative practices. For example, in

Table 3. The expected change in each of our measured response variables in regenerative fields compared to conventionally managed fields. Variables may either increase (↗), decrease (↘) or be unaffected (∿) relative to conventional sites—the direction of change is not indicative of the desirable state for that measure. Short term trends are those we expect to see in the duration of our study (< 4 years), long term trends may appear after >4 years of regenerative practices.

		Expected trend		
Category	Outcome/Response	Short term	Long term	References
Soil properties	Water stable aggregates	↗	↗	Guest *et al* (2022)
	Soil organic carbon	∿	↗	Haddaway *et al* (2017), Ogle *et al* (2019), Li *et al* (2019)
	Bulk density	↗	↘	Li *et al* (2019)
	Total N	↗	↗	García and Chacoff (2007), Garba *et al* (2022)
	pH	∿	∿	Zhao *et al* (2022)
	Earthworm density	↗	↗	Briones and Schmidt (2017), Bai *et al* (2018)
Biodiversity	Birds	∿ / ↗	↗	Donald *et al* (2002), Stoate *et al* (2003)
	Pollinators	↗ / ∿	↗	Williams *et al* (2010), Roulston and Goodell, (2011), Ullman *et al* (2016), Antoine and Forrest (2021)
	Beetles	↗ / ∿ / ↘	↗ / ∿ / ↘	Jowett *et al* (2021), Muller *et al* (2022)
	Spiders	∿	∿ / ↗	Tahir *et al* (2012)
	Parasitoids	↘	↘	Andow (1991)
	Pests	↘ / ↗	↘	Kendall *et al* (1991), Holland *et al* (2004), Bryan *et al* (2021)
Grain yield and quality	Yield	↘	∿	Pittlekow *et al* (2015), Haung *et al* (2018), Pearsons *et al* (2022)
	Quality	↗ / ∿	↗	Wozinak *et al*, (2014), Darguza and Gaile (2019), Montgomery *et al* (2022)
	Inputs	↗ / ↘	↘	Khangura *et al* (2023)

the Eastern cluster, heavy clay soils are likely to have higher carbon storage than the sandy and shallow chalk-dominated soils of the Southwest (Page *et al* 2020). Similarly, nitrogen dynamics may differ across soil types, for example, minimum tillage can reduce nitrous oxide emissions in finer textured soils but tends to increase emissions on heavier soils in wet climates (Pelster *et al* 2023). Further, all farms in the Southwest are already mixed cropping systems (with integration of livestock), while this is not the case in the East.

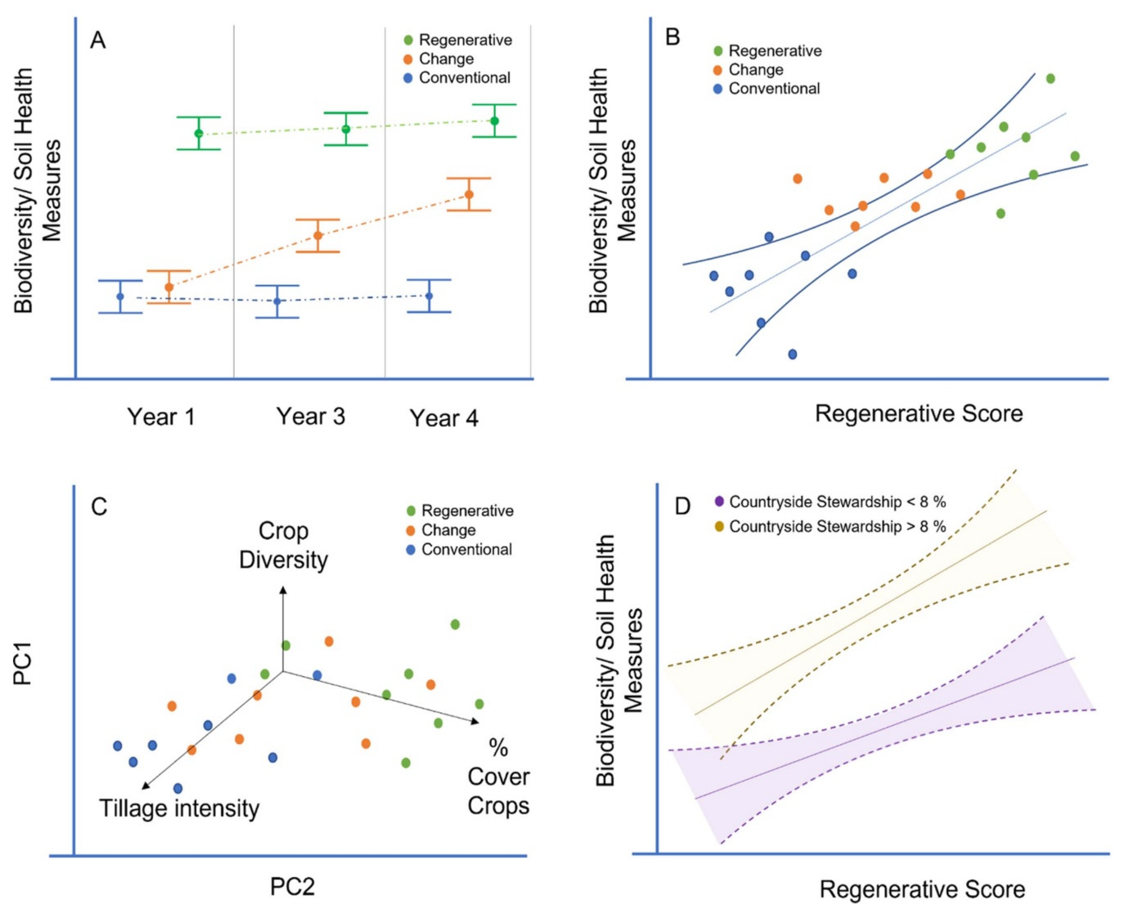

Figure 4. Hypothetical measures of the transition to regenerative agriculture through (A) categorical approaches characterising the farming system and time series data, (B) creation of a regenerative score based on the nature and intensity of practices used and their combination or (C) multivariate analysis linking farming practices data with outcomes. Using hierarchical or multivariate modelling approaches, we can also measure the interaction between implementing regenerative practices and the presence of farm- or landscape-scale features such as the presence of agri-environment schemes or semi-natural habitats (D). All the trends represented here are speculative, based on expectations presented in section 3.1.

Therefore, we expect differences in the outcome of the regenerative practices between our two landscapes, and the Eastern cluster may show a more pronounced change based on their soil type and experimental introduction of livestock into the farming system.

Improvements in soil properties from a transition to regenerative agriculture are known to increase over time (Mondal and Chakraborty 2022). This may be the case not only for the Change fields, but also for the Regenerative fields in our experiment, which have transitioned to regenerative agriculture at different times and different speeds, using different combinations of practices (and not necessarily all nine practices yet), and may still be experiencing changes in practices and in soil physical and biological properties. Overall, we expect that, by the end of our experiment, the Change fields will achieve an intermediate outcome between Control and Regenerative fields, in relation to our indicators (figure 4). This means for some indicators, such as soil carbon, the short term changes (i.e. in the duration of our study) may not be visible (table 3).

3.1.2. Biodiversity

Overall, we anticipate that both abundance and diversity of biodiversity will respond positively to regenerative practices (table 3), through provision of an increased diversity of plant resources (Stoate *et al* 2003, Jaworski *et al* 2023b) and reducing disturbance on sensitive life stages, especially in the soil (Christmann 2022). For example, winter foraging birds benefit from seed-bearing cover crops (Stoate *et al* 2003), ground-nesting bees and some carabids may benefit from reduced tillage (Williams *et al* 2010, Roulston and Goodell 2011, Ullman *et al* 2016), and crop diversity at the landscape scale has been shown to increase pollinator diversity (Power *et al* 2016, Aguilera *et al* 2020, Raderschall *et al* 2021). Increased plant diversity in regenerative systems can also promote a diversity of natural enemies and therefore pest control services (Nicholls and Altieri 2013, Letourneau *et al* 2011; Aguilera *et al* 2020, Smith *et al* 2020, Jeavons *et al* 2023). Many other soil aspects, including chemistry, biology, and structure, relative to the farming systems studied, may affect pollinators, although this remains understudied (Antoine and Forrest 2021, Carvalheiro *et al* 2021).

However, biodiversity responses may not be immediate, nor universal. For example, increased tillage intensity negatively impacts carabid abundance and diversity (Muller *et al* 2022), but several species show no response to tillage treatments and may be more abundant due to competitive advantage over species affected by particular tillage timings in breeding interruption, or preference for surface crop residues associated with no tillage (Baguette and Hance 1997, Blubaugh and Kaplan 2015, Jowett *et al* 2019, Jowett *et al* 2021). Similarly, spiders may not have strong responses to tillage (Tahir *et al* 2012), but may be positively impacted by cover crops or diverse plant cover along field boundaries (Sunderland and Samu 2000, Jaworski *et al* 2023b). As a result of increased predator activity, aphid abundances are expected to be lower in Regenerative fields, and parasitoid abundances are expected to strongly correlate to aphid abundances (resource concentration hypothesis; Andow 1991).

Other practices, such as the use of pesticides and adoption of agri-environment schemes may also impact biodiversity in Regenerative fields (figure 4(D)). For example, pest chemical pest management may have adverse effects on the persistence of natural enemies (Ruberson *et al* 1998), pollinators (Woodcock *et al* 2016b) and birds (Rigal *et al* 2023). Similarly, landscape composition and the presence of agri-environment schemes that enhance floral resources are also likely to impact pollinators (Sutter *et al* 2017, Jones and Rader 2022), natural enemies (Pywell *et al* 2015, Bullock *et al* 2021) and birds (Staggenborg and Anthes 2022). There may be interactive effects between regenerative practices and implementation of agri-environment schemes (Jaworski *et al* 2023b). For instance, the presence of semi-natural areas can enhance the effectiveness of in-field practices, such as the effects of crop diversity for promoting beetle and pollinator diversity (Aguilera *et al* 2020). While many regenerative farms reduce their use of insecticides, this is not a core principle of regenerative agriculture, making associated changes in biodiversity difficult to predict (Fenster *et al* 2021). We will collect detailed records of chemical inputs and agri-environment schemes from our experimental farms to control for their impact on our biodiversity outcomes (figure 4).

3.1.3. Grain yield and quality

There are expected trade-offs between biodiversity in agricultural fields and maintaining high yields and grain quality (Khangura *et al* 2023). However, the influence of tillage on crop yields varies depending on the crop, the climatic conditions and soil type (Pittlekow *et al* 2015, Haung *et al* 2018), and can be mediated by nutrient additions (Pittlekow *et al* 2015, Pearsons *et al* 2022), residue retention practices (Pittlekow *et al* 2015), and crop rotations (Wozinak *et al* 2014). For example, a meta-analysis by Pittlekow *et al* (2015) showed a marked decrease in yields within cereal and rice crops in response to no-till practices, but not legumes, oilseed or cotton. However, nutrient addition and crop residue retention narrowed the difference in crop yield between conventional and no-till management. Contrastingly, a meta-analysis by Huang *et al* (2018) showed a limited influence of tillage on barley yield, except in dry and alkaline soils where there was up to a 49% yield increase with adoption of no-till management.

The measurements of grain quality used by grain merchants are determined by its end use such as milling, baking, or brewing and comprise physical and chemical properties. Physical measurements include moisture levels, test (bushel) weight and grain damage. Chemical measurements include amount of protein, toxins, and malt extract for barley. Diversifying crop rotations may increase the suppression of weeds (Sharma *et al* 2021) and crop diseases (Khangura *et al* 2023). However, some farmers worry that grain damage from insects may increase under regenerative practices due to decreased pesticide inputs (Beacham *et al* 2023). Similarly increases in humid conditions under regenerative practices (Ogle *et al* 2019) may influence presence of disease and other toxins. Destruction of cover crops and management of pests and weeds is often performed chemically in regenerative agriculture, due to a reduction in mechanical (till-based) control methods (Khangura *et al* 2023). Herbicide use is also increasing across the UK (Pesticide Collaboration 2023). Therefore, we may expect to see agrochemical residues in grains regardless of treatment, especially in the short-term.

The influence of farming practices on nutrient content is highly varied, with some studies showing increases (Darguza and Gaile, 2019; Montgomery *et al* 2022), decreases (Wozinak, 2014) or no change in nutrient content (Pearsons *et al* 2022) compared to conventional practices. The nutrient content of crops is heavily influenced by climatic factors (Devita *et al* 2007, Wozinak, 2014), nutrient inputs (Pearsons *et al* 2022) and crop rotations (Lopez-Bellido *et al* 1998). However, these studies measure the influence of individual practices, and we believe that the combination of increased legumes in the rotation (Lopez-Bellido *et al* 1998), organic matter inputs (Pearsons *et al* 2022) and reduced tillage (Darguza and Gaile, 2019) in the Regenerative fields may result in an increase in micronutrients in crop yields. For example, Montgomery *et al* (2022) showed increases in vitamins (e.g. B1 and B2), minerals (Na and P), and phytochemicals (phenolics, phytosterols, and carotenoids) in regeneratively managed crops, with a similar combination of regenerative practices, compared to conventional management. It is likely that crop nutrient content will respond rapidly to these changes, and with our detailed information on inputs, soil analysis and grain content we will also be able to track changes in nitrogen use efficiency across the transition as farms move towards reduced inputs.

3.2. Measuring the transition to regenerative agriculture

Despite many reported benefits of regenerative agriculture, adoption of a core combination of regenerative farming practices by UK farmers remains limited (Jaworski *et al* 2023a). The farmer interviews conducted in our experiment will provide an understanding of farmers' motivations for transitioning to regenerative farming, as well as the barriers to adoption. This farmers' perspective is fundamental to refining the definition of regenerative agricultural practices within a broader political landscape (e.g. comparisons with organic farming systems, and different approaches to certification; MacMillan and Benton 2014, Beacham *et al* 2023, Jaworski *et al* 2023a). Interviews with policymakers will complete the picture by providing an analysis of the volitile policy environment within which our research participants are following their uncertain path towards the adoption of regenerative agriculture practices.

Intensive practices, such as deep tillage and application of pesticides, may have long-lasting effects, even on land that has been recently converted to regenerative techniques (Crotty *et al* 2016, Beaumelle *et al* 2023). Regenerative farms in the experiment will have begun their regenerative journey at different times and may be at different points in the transition towards a fully regenerative agricultural system (as defined by our list of selected practices) (section 2.2). Similarly, the Change farms will be implementing different subsets of the nine practices. However, we expect that regenerative farms will show marked differences in soil properties and other biodiversity measures, compared to the Control fields (section 3.1). Over time, the change group is expected to become more like the regenerative group, as it adopts regenerative practices (figure 4(A)). This kind of time-series data can be one of the most effective measures of impacts of interventions in ecology (Wauchope *et al* 2021).

The transition to regenerative agriculture is inherently multifaceted, and understanding the changes resulting from the transition to regenerative agriculture requires analytical methods that consider the variation in implementation of practices both between and within farms. While previous studies have focused on the influence of single practices, the adaptability and flexibility in the implementation of different practices is one key strength of regenerative agriculture, but also of 'on farm experiments' through co-produced research (Lacoste *et al* 2022). Past research suggests that soil health and biodiversity outcomes can be influenced by the combination of practices, with some practices likely to act synergistically (e.g. transition to no-tillage after leys may be more effective in regenerating earthworm populations and soil aggregation than simply adopting no-tillage in continuous arable cropping). Others can be complementary, and reduce negative effects associated with a change of practice (e.g. nutrient addition reduces the yield loss related to no-till; Pittelkow *et al* (2015), but might also be achieved by direct drilling legumes as the first crop after leys). Finally, not all practice combinations are realistic to implement for farmers (Jaworski *et al* 2023a). Teasing out this nuance will be an important aspect of our research agenda (figure 5).

Therefore, an analysis of the impacts of regenerative practices must consider not just the presence of regenerative practices, but the combination of practices, the duration and the consistency with which they are implemented. One analytical method would be to generate a 'regenerative score' that can categorise the variation in practices and map the changes in implementation of practices over time (e.g. Fenster *et al* 2021, Jaworski *et al* 2023a). This score could then be used as an explanatory variable to measure the impact of a transition towards regenerative agriculture, rather than the initial categorical approach associated with the BACI design (section 2.2.). Similar to Jaworski *et al* (2023a), a regenerative score could be achieved by a point-based system that awards higher scores for the adoption of multiple regenerative practices that represent all five principles, with the most regenerative farms being those that implement all nine practices in our study (figure 4(B)). To better align the score with potential environmental benefits, it could also capture the quality of the practice (e.g. no tillage should be attributed a higher score than minimum tillage along the regenerative farming transition; Haddaway *et al* 2017), and the frequency of use of the practice, since real farming constraints may prevent the use of some practices in some years (Blanco-Canqui and Wortmann 2020), especially within the early transition period.

Alternatively, we could use multivariate distance-based ordination methods to compare the practices implemented for each site through time, and map the changes in a multidimensional space, composed of the farming practices, and possibly of other characteristics of the farming systems (e.g. soil type; figure 4(C)). The resulting distance-based matrix could then be compared to the outcomes using matrix-correlation methods (e.g. canonical correlation and multiple correspondence analyses; Adachi 2016). Advantages are the possibility to analyse all metrics simultaneously in a non-hierarchical way, but also to incorporate qualitative metrics (e.g. how successful a cover crop has been perceived to be by a farmer; Donaires *et al* 2023). This method may allow us to capture potential interactions between different types of regenerative practices, and

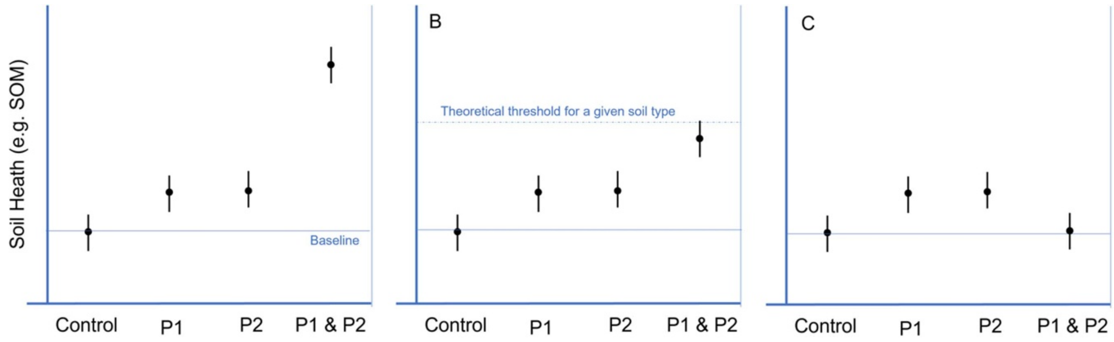

Figure 5. Theoretical interactions between implementation of regenerative practices ('P1′ and 'P2′) and their effect on the soil health outcomes. (A) Practices implemented together may act synergistically to produce significantly better outcomes than the sum of effects from individual practices (B) Alternatively, practices may be synergistic but there may be a limiting threshold that reduces the impact of additional practices, e.g. soil organic matter may be limited by the capacity of a given soil type (Page *et al* 2020). (C) Practices that have positive effects individually may also have antagonistic effects when implemented together. While this has been demonstrated for IPM (Stenberg 2017), there is no evidence of this for regenerative practices.

between practice combinations and outcomes (figure 5). For example, Nunes *et al* (2018) found that benefits of no-till practices on soil health were enhanced with addition of cover crops, across a range of indicators, and we expect this to be further enhanced by introducing 2–3 year leys in arable rotations.

Finally, the relative effectiveness of in-field practices may be challenging to separate from changes occurring outside fields due to the adoption of agri-environment schemes (Scheper *et al* 2013). Farms from all three groups in our experiment have existing agreements or will adopt new agri-environment practices, alongside their transition to regenerative agriculture. The adoption of agri-environment schemes in areas out of production has been shown to have a larger impact on biodiversity than in-field schemes (Bátáry *et al* 2015), although this may vary depending on the local context, such as the proportion of semi-natural habitat areas (Jaworski *et al* 2023b). By characterising both in-field practices and semi-natural habitat management including agri-environment scheme practices, we can use multivariate modelling techniques to distinguish between the relative impact of in-field practices versus landscape-scale changes to habitat availability (figure 4(D)).

4. Discussion

We have presented the approach and methodology for our landscape-scale, transdisciplinary experiment aiming to quantify the benefits of regenerative agriculture for soil health, crop quality, and biodiversity, and to understand the experience of farmers in transitions to regenerative farming. Our experiment will bring novel scientific, quantitative evidence on the effects of regenerative agriculture on soil quality and carbon sequestration, crop yield quality, and bird and arthropod biodiversity in farmlands. By drawing from numerous disciplines, we will provide an integrative vision on the intrinsically complex nature of food system research (Holmes *et al* 2018, Kallio and Houtbeckers 2020, Jackson *et al* 2021).

The integrative co-production approach developed here is relevant to the adoption and spread of regenerative agriculture practices. This is because it starts with farmers' definition of regenerative agriculture in two UK landscapes, and their efforts to constantly match scientific needs with applicable questions of interest to real-world farming. Secondly, the experiment builds on a large UK stakeholder network beyond the two farmer clusters, including the Agricultural Development and Advisory Service, the Agriculture and Horticulture Development Board, and the Soil Association, which will be key in disseminating results. The experiment should also drive farmers to seek scientific evidence of the outcomes of regenerative farming by themselves and adopt a scientific approach whenever useful to measure such outcomes. For instance, this experiment includes farmer workshops demonstrating simple, but scientifically informative soil tests that can be performed by farmers. This could help farmers connect with their soils and relate their farming practices to soil health, more than sending their soil samples to laboratories for analyses (Jaworski *et al* 2023a). No matter if scientific evidence confirms or disproves benefits of regenerative agriculture, the current co-production research will assess the outcome of regenerative agriculture practices, and this should help farmers define and optimize their implementation of regenerative agriculture practices in the UK and beyond. For instance, data collected will potentially help identify specific practices which should not be used in regenerative farming combinations, due to harmful environmental impact. Similarly, it could help identify combinations of practices that bring the highest environmental benefits (Jaworski *et al* 2023a), and any potential trade-offs among these benefits, such as profitability, soil health and biodiversity.

This task is particularly urgent given contestations around the trajectory of UK agriculture. Until recently, UK agri-food policy has relied on 'cheap' (in terms of price) imports of food to focus more heavily on producing environmental 'goods' (i.e. ecosystem services) domestically rather than increasing agricultural productivity at all costs, a paradigm broadly characterised as post-productivist (Ward 1993, Beacham *et al* 2023). Yet the increased challenges associated with cross-border trade and the development of new UK agricultural policy post-Brexit has led to the notable resurgence of narratives around self-sufficiency whilst also maintaining the focus on environmental goods (Helm 2017, de Boon *et al* 2022, Beacham *et al* 2023, Defra 2023). In this context, regenerative agriculture has emerged as a favourable model for sustainable production ((Beacham *et al* 2023). The UK Department for Environment, Food and Rural Affairs (Defra) has developed ELMs as a tapered replacement to the European Union Common Agricultural Policy (CAP) in 2024, and SFI within ELMs aims to incentivise farmers to reach new goals partially aligned to regenerative farming, and notably sustainable soil management (Defra 2023). ELMs payments are planned to be based on management, not outcomes. Our experiment can help refine what practices are more likely to contribute to achieving environmental goals by providing evidence of the link between practices and outcomes. For instance, the scientific methodology adopted here to measure soil carbon sequestration could guide the selection of appropriate soil test requirements driving payments under the SFI.

Efforts to create a certification scheme for regenerative agriculture (e.g. A Greener World 2020) may not be desirable to farmers that value the flexibility of regenerative approaches (Beacham *et al* 2023), and may be hampered by logistic constraints, and similarity with other types of low-input approaches that have similar goals. However, political recognition of the links between practices and outcomes in our study will be an important step towards certification (Elrick *et al* 2022). Beyond certification, bringing traceability to the consumers along with guarantees of environmentally-friendly agriculture, new routes for recognition of the environmental benefits of regenerative agriculture have emerged. For instance, there is an emerging market of carbon credits, where farmers can receive income from carbon-emitting companies to protect their agricultural soil (Newton *et al* 2020, Jackson Hammond *et al* 2021). Here too, however, a regulatory framework is urgently required to ensure that carbon credits, and therefore companies' environmental footprints, are associated with real, additional outcomes (i.e. the actual sequestration of soil carbon; and reduced net greenhouse gas fluxes e.g. of N_2O; Newton *et al* 2020, Jackson Hammond *et al* 2021).

Our landscape experiment originally aimed to follow a BACI design, which brings high statistical power by accounting for local environmental variability and specifically quantifying the changes in measured metrics through time (Osenberg *et al* 2006, Christie *et al* 2019). However, the co-production process resulted in a change of approach due to the logistical constraints of real farming (Lacoste *et al* 2022). This was due in part to the discrepancies in defining regenerative farming between clusters and farmers, which required moving away from clearly defined distinct treatments to focus on a more incremental, gradual, and multi-dimensional approach. Similarly, it was not ethically acceptable to ask farmers to transition to regenerative agriculture at the same pace and following the same trajectory, and indeed some of them had started transitioning before the baseline year (Year 1). While these changes may partially reduce statistical power, we advocate that similar approaches increasing the flexibility of ideal designs is necessary to produce scientific evidence strongly anchored in reality. Also, and for the same need to increase realism in scientific production (Gascuel-Odoux *et al* 2022), landscape-scale experiments are essential. This is true for their ecological dimension, where a number of ecosystem services are affected at spatial scales larger than the field (Rusch *et al* 2016, Harrison *et al* 2019, Jaworski *et al* 2022), and for which field-scale experiments cannot bring evidence on large scale spatio-temporal dynamics. This is also true for the social dimension, where only farmer networks can lead to the communication necessary for large-scale coordinated action. Such action can be useful to further enhance ecosystem services (e.g. landscape-scale crop mosaics promoting diverse communities of pests' natural enemies; Jaworski *et al* 2022, 2023b), but is also absolutely crucial to induce food system changes and to make local outlets available (Meynard *et al* 2017, Thomine *et al* 2022).

Our experiment cannot directly quantify the effect of single practices on soil quality and biodiversity metrics, due to the systemic approach adopted here, and due to the large amount of social and environmental variation, although this is partially compensated by the relatively large scale (25 sites of 60 ha; Olson *et al* 2014, Alesso *et al* 2019). The large scale also prevents high precision on some metrics measured—for instance, soil properties are only measured twice—due to logistical constraints inherent to wide collaborative efforts. Similarly, the relatively short time scale of four years—determined by the duration of the majority of research funding in the UK—does not allow a complete measure of the transition to regenerative farming. Such transition likely occurs over a longer time in real farms, and benefits both for soil health and biodiversity may increase over a much longer time scale (Mondal and Chakraborty 2022). In addition, we could only measure a selection of soil, ecological and economic indicators, a trade-off with more precise and complete measures at a smaller spatial scale and in a more controlled environment (controlling for crop type and rotation synchronisation, soil type, topography, climatic conditions, etc). Nonetheless, the scale and innovative approach of our experiment will allow us to develop more generalisable claims around the benefits of regenerative agriculture while also identifying some of the challenges that the farmers face. Further, the social scientific dimension of this research allows a better understanding of farmers' perspectives on potential transitions to regenerative agriculture as they are formed within a wider constellation of social, political, economic and cultural conditions (Krzywoszynska 2019, Beacham *et al* 2023).

Our research collects unique, precise information on inputs used in farms included in our experiment; therefore, it has the potential to accurately measure the reliance on inputs and in particular pesticides, a key and flexible aspect of regenerative agriculture (Newton *et al* 2020, Schreefel *et al* 2020, Giller *et al* 2021). For instance, while some farmers consider the use of synthetic insecticides incompatible with regenerative farming, the use of synthetic herbicides remains common and may actually increase in regenerative farming to reduce weed infestation rather than through tilling practices, and to suppress leys and plant cover before drilling in the next crop (Beacham *et al* 2023, Jaworski *et al* 2023a). Our experiment can investigate the relationships between biodiversity decline and agrochemical use, a major driver of insect declines despite scarce evidence on long-term trends (Ewald *et al* 2015, 2016, Raven and Wagner 2021). Another major question that our experiment has the potential to answer is the relative contribution of landscape (composition and configuration of semi-natural habitats and of agricultural land) versus farming practices on biodiversity (McHugh *et al* 2022, Scheper *et al* 2013, Bátáry *et al* 2015). This is made possible since all ecological measures are accurately geolocated, and we work across a gradient of farming systems in two contrasted and heterogeneous UK landscapes, differing in the quality and quantity of semi-natural habitats, and in the use of agri-environment schemes.

Our approach aims to foster new transdisciplinary research integrating social and natural science dimensions in food system research and provide an evidence-based vision for informing beneficial changes in the field of agroecology from field-to landscape scales. Our research provides a demonstration of the benefits of co-production in experimental designs to measure and monitor change in real-world farming practices and ecology.

Data availability statement

No new data were created or analysed in this study.

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Author contribution statement

Funding: L V D, P J, D C.

Experimental design: C C J, K B, L V D, J R L, H3F C.

Farmers coordination: C C J, K B, H3F C.

Social science protocols: A K, P J, J D B.

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Identification and lab work protocols: C C J, K B, L C, N M M, K J, L V D.

Theoretical development: K B, C C J, L V D

Figures: K B, C C J

Writing: C C J, K B

Manuscript editing: All.

ORCID iDs

Katherine Berthon https://orcid.org/0000-0001-8224-2828
Coline C Jaworski https://orcid.org/0000-0002-6136-8656
Jonathan D Beacham https://orcid.org/0000-0001-8684-3719

Peter Jackson https://orcid.org/0000-0002-3654-1891
Jonathan Leake https://orcid.org/0000-0001-8364-7616
Niamh M McHugh https://orcid.org/0000-0001-6659-7266
Tim Daniell https://orcid.org/0000-0003-0435-4343
Anna Krzywoszynska https://orcid.org/0000-0002-8304-0440
Duncan Cameron https://orcid.org/0000-0002-5439-6544
John Holland https://orcid.org/0000-0003-3742-7035
Sue Hartley https://orcid.org/0000-0002-5117-687X
Kelly Jowett https://orcid.org/0000-0002-3672-7813
Lynn V Dicks https://orcid.org/0000-0002-8304-4468

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