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RESEARCH ARTICLE

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Agroecological farming promotes yield and biodiversity but may require subsidy to be profitable

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Abstract

1. Intensive arable agriculture uses agrochemicals to replace ecosystem services (e.g. pest control and soil health) while simultaneously degrading others (e.g. pollination). Agroecological farming aims to reduce this reliance on agrochemicals. Whether these practices maintain yields at a scale relevant to farm business viability is unclear.
2. In a 4-year replicated study across 17 English farms we assessed the ability of farmer co-designed agroecological systems to support regulating services, beneficial invertebrates, crop yield and profitability. We test three management systems: (1) 'business-as-usual (BAU)' control; (2) 'enhancing-ES' supporting beneficial invertebrates with wildflower field margins and protecting soils with cover crops; (3) 'maximising ES' with the further addition of soil organic matter and in-field strips to bring beneficial invertebrates into the crop.
3. Soil carbon stocks were highest in the maximising-ES system. Predation and pollination ecosystem services were higher in the enhancing-ES and maximising-ES systems, as were earthworms and other populations of beneficial predatory and pollinating invertebrates. Pest snail biomass was also lowest in the enhancing-ES and maximising-ES systems, although aphid numbers were higher.
4. The enhancing-ES and maximising-ES systems increase yields of cereals and oilseed rape. However, the loss of productive agricultural land and establishment costs exceeded the value of increased yields. Only enhancing-ES breaks even with agri-environmental subsidies.
5. *Synthesis and applications*. These results highlight that while evidence for the role of ecosystem services in supporting crop yield can be found, overcoming economic constraints within conventional farming systems is likely to be a key barrier to widespread uptake. Agri-environmental subsidy payments can offset these costs, but only for moderate interventions. Transition to more sustainable farming systems needs to overcome these economic constraints with new policy interventions.

KEYWORDS
agroecological, arable, biodiversity, ecosystem services, farming, profit, sustainable intensification, yield

1 | INTRODUCTION

The intensification of global agriculture has devolved reliance on natural ecosystem services to agrochemicals and crop breeding, while simultaneously losing semi-natural habitats (Priyadarshana et al., 2024; Pywell et al., 2015; Stoate et al., 2009). In the short term this has increased yields at a level inconceivable in previous centuries but has come with negative externalities like biodiversity loss, nutrient losses and pollution burdens (Defra, 2023; Stoate et al., 2009; Tscharntke et al., 2012; Wan et al., 2025; Woodcock, Isaac, et al., 2016). Further, the reliance on agrochemicals may have resulted in ‘intensification traps’ whereby declines in biodiversity and linked ecosystem services drive ever greater dependence on high input intensive agriculture (Burian et al., 2024). This may over the long term threaten food production through evolution of pesticide resistance (Hawkins et al., 2019) and reduced resilience to climate change (Kane et al., 2021). Increased public awareness of the unsustainable nature of these systems has prompted both ground up interest (e.g. organic and regenerative farming movements) and top-down policy approaches to elicit system change, for example, the EU Farm-to Fork strategy or the UK Sustainable Farming Incentives (Cusworth et al., 2021; Jaworski et al., 2024).

Although intensive management systems still predominate, interest from farmers and policy makers in developing agricultural systems with a focus on longer-term sustainability is becoming increasingly widespread (Heller et al., 2024; Pywell et al., 2015; Wezel et al., 2020). Such agroecological systems can enhance regulating and supporting ecosystem services by increasing soil organic matter (Heller et al., 2024), reducing periods of bare soil (e.g. with post harvest temproary cover crops; Hufnagel et al., 2020; Heller et al., 2024) as well as creating semi-natural habitat to increase natural pest control and pollination (Batary et al., 2015; Priyadarshana et al., 2024; Pywell et al., 2015). However, evidence reviews have suggested that these effects can be variable. For example, Albrecht et al. (2020) demonstrated that the benefits of flower strips were seen for insect pollinators but not those providing natural pest control, while Martin et al. (2019) identified how the configuration of crop and semi-natural habitat interventions alters their effectiveness in supporting beneficial taxa. These approaches can be supported by agri-environmental scheme payments dictated by policy, as well as through industry, natural capital markets or farmer-led initiatives (Batary et al., 2015; Hufnagel et al., 2020; Pywell et al., 2015).

Considerable research effort has attempted to quantify the benefits of agroecological farming practices in supporting biodiversity, beneficial invertebrates and soil functions (Batary et al., 2015; de Graaff et al., 2019; Heller et al., 2024; Pywell et al., 2015; Tscharntke et al., 2012). Some studies have identified benefits for regulating ecosystem services (e.g. pest control and pollination) and yield (de Graaff et al., 2019; Pywell et al., 2015; Woodcock, Bullock, et al., 2016). However, the extent to which yield gains offset economic losses associated with establishment costs and land lost from productive agriculture (e.g. wildflower field margins don't directly produce crops) is rarely considered (Pywell et al., 2015). The dynamic characteristics of both economic and biological systems make such assessments complex, particularly when the contribution of regulating and supporting ecosystem services has high variability. Understanding these impacts over multi-year timescales is necessary to quantify how the cost of interventions and their associated benefits alter through time. Such research needs to be undertaken across the real-world heterogeneity of commercial farms to understand how viable agroecological farming practices are in practice (DeLonge et al., 2016; Pywell et al., 2015). This evidence is crucial for farmer decisions to adopt sustainable agroecological systems (Goulet et al., 2023).

Here we address these issues through a multi-site experiment across a 4-year rotation of 17 English arable farms. Superimposed over normal management practices for each farm were three field scale management systems: (i) ‘business as usual’ (BAU) considered as a conventional management control; (ii) ‘enhancement of ecosystem services’ (enhanced-ES), a simple agroecological farming system incorporating wildflower strips at field margins to promote populations of beneficial insects and cover crops to reduce soil erosion over the winter; (iii) ‘maximization of ecosystem services’ (maximise-ES), which adds to the enhancing-ES management system in-field flower strips to reduce field sizes and promote spill-over of beneficial insects in to the crop with addition of organic matter to soils. These systems were co-developed with farmers to be compatible with conventional arable farms. We considered the impacts of these systems on beneficial invertebrates, regulating ecosystem services (pollination and pest control), supporting services (soil carbon), and ultimately yield and profitability. We tested the hypotheses: (H1) Agroecological management practices enhance beneficial invertebrates and associated regulating ecosystem services; (H2) Improving these regulating services led to increased crop yields; (H3) Increases in yield are sufficiently high to offset land lost from production and management costs leading to a net benefit for farm profitability.

2 | MATERIALS AND METHODS

2.1 | Experimental design

The experiment was undertaken over 4 years and replicated across 17 farms in England. Crops and their rotations varied between farms, but typically included winter wheat, other cereals (spring and winter oats and barley), oilseed rape and a break crop (e.g. beans; Table S3). All farms were managed using agrochemicals. Overall, 67.6% of crops were cereals (43.1% spring and 56.9% winter sown) with winter wheat being the most frequent of these (39.2%).

We identified agroecological farming practices in collaboration with our cohort of farmers that would be suitable for integration into their farming systems. This co-design process was part of an open discussion during start-up meetings with participating farmers and, where relevant, their agronomists, where we discussed the practical viability of implementing different field scale manipulations. Written consent for this process was obtained. These agroecological farming practices were: (1) *created non-crop semi-natural habitat*: These provide foraging resources, refuges and overwintering habitat for beneficial invertebrates (Pywell et al., 2015). We created two types of non-crop habitat interventions. The first was 6m wide wildflower strips sown on field edges, which have been widely demonstrated to support beneficial invertebrates (Batary et al., 2015; Pywell et al., 2015; Woodcock, Bullock, et al., 2016). The second was 6m wide in-field strips established every 96m (three spray boom widths) running the length of the field. These in-field strips were intended to promote movement and spill-over of beneficial invertebrates into the crop (Defra, 2023). Both were established by sowing 22 species of flowering forbs and grasses (see Table S1). (2) *Cover crops*: When spring-sown crops (e.g. spring barley) are sown, cover crops provide vegetative ground cover over the winter period, protecting soils prone to erosion, improving drainage and acting as green manure. Cover crops were established using low-cost agricultural varieties of black oats, radish and some flowering plants (see Table S2). (3) *Organic matter*: Farmyard manure from cattle, but also in some cases as green composted waste, was applied in the winter prior to the first experimental harvest year. This provided direct soil fertilisation and supported beneficial soil fauna (Pulleman et al., 2005).

Building again on farmer led co-design the three classes of management practices were combined into management systems along a hypothesised gradient of agroecological enhancement (Figure 1). Each management system was applied to a randomly selected field on that farm (three fields per farm, one corresponding to each of the management systems). Mean field sizes were 11.1ha (SE ± 0.53, range 5.33–22.1ha). Within a farm and for a given year the three fields were part of the same rotation, that is, growing the same crop. These were established at 15 of the farms in autumn 2017 (monitored over 4 harvest years from 2018 to 2021), with the remaining two farms established in autumn 2018 (monitored from 2019 to 2021; see Table S3). The three management systems were: *Business as usual (BAU) control*: here the conventional crop specific management practices typical to a given farm were in operation. Soil fertility and pest control depended primarily on inorganic fertilisers and pesticides; *enhancement of ecosystem services (enhancing-ES)*: normal crop management practices continued, however, wildflower field margins were along ≥50% of the perimeter of the field (see Table S4). Cover crops were sown preceding spring crops; *maximisation of ecosystem services (maximising-ES)*: In addition to practices used in the enhancing-ES system between 1 and 3 in-field strips were established depending on the size of the field (Table S4). In addition, 30 tonnes ha^{-1} of organic matter was added in the winter before the first sampling year. Farmers participating in this study applied pesticides consistently across all three treatments.

2.2 | Quantifying crop regulating ecosystem services

We focused all assessments on wheat, barley, oats (spring or winter sown crops) and winter oilseed rape. These represent the largest and most economically important crops in the United Kingdom. Sampling was undertaken along four transects in each field at 12, 24, 48 and 96m intervals from the crop boundary (16 in-crop sampling points per field; Figure 1). The following assessments were undertaken at each sampling point (Supporting Information, Methods for full detail). *Slug predation*: Slug predation by ground active predatory beetles with cutting mandibles was assessed using six artificial slugs (3 small at 15 × 3mm and 3 large at 5 × 30mm) made from non-toxic plasticine placed out in May and June. Visual predators, like ground beetles and rove beetles, will attack these fake slugs (Howe et al., 2009). The average number of artificial slugs with bites at each sampling point was quantified. *Aphid predation*: At each sampling point, a 2 × 10cm piece of card with five live *Sitobion avenae* aphids glued to it was attached to a wheat tiller. These were left in place for 24h and the average number of aphids eaten was assessed at the field scale (Winqvist et al., 2011). These assessments were undertaken in May and June on winter sown cereals only. *Aphid parasitism*: Ten cereal stems or 10 racemes of oilseed rape were hand searched in April and June for parasitized aphid mummies. The total mean abundance per sampling point was determined. *Pollination services*: For oilseed rape, two plants of similar size were identified at each sampling point during the pre-flowering phase in March. Insect pollinators were excluded from one plant with a fine mesh net bag. The other plant was the control exposed to insect pollinators. Pollination attributable to pollinators was determined as the average yield of the control plant minus that of the bagged plant in a field. *Soil carbon stocks*: Soil organic carbon stocks (g · cm^{-3}) were estimated as bulk density × percentage soil carbon determined from in-field soil cores and accounting for inorganic carbonates. This was assessed once at the end of the study after the final harvest (winter 2021).

2.3 | Quantifying beneficial invertebrates supporting regulating services

The average sampling point abundance of beneficial invertebrates for each field was annually assessed (Supporting Information, Methods). *Parasitic wasps in the crop*: A vortis suctions sampler was used to collect parasitic wasps within the crop associated with (i) wheat crop pests and (ii) oilseed rape pests in April and June each year. *Ground active predators*: Abundance (activity density) of ground beetles, rove beetles and spiders was assessed using pitfall traps at each

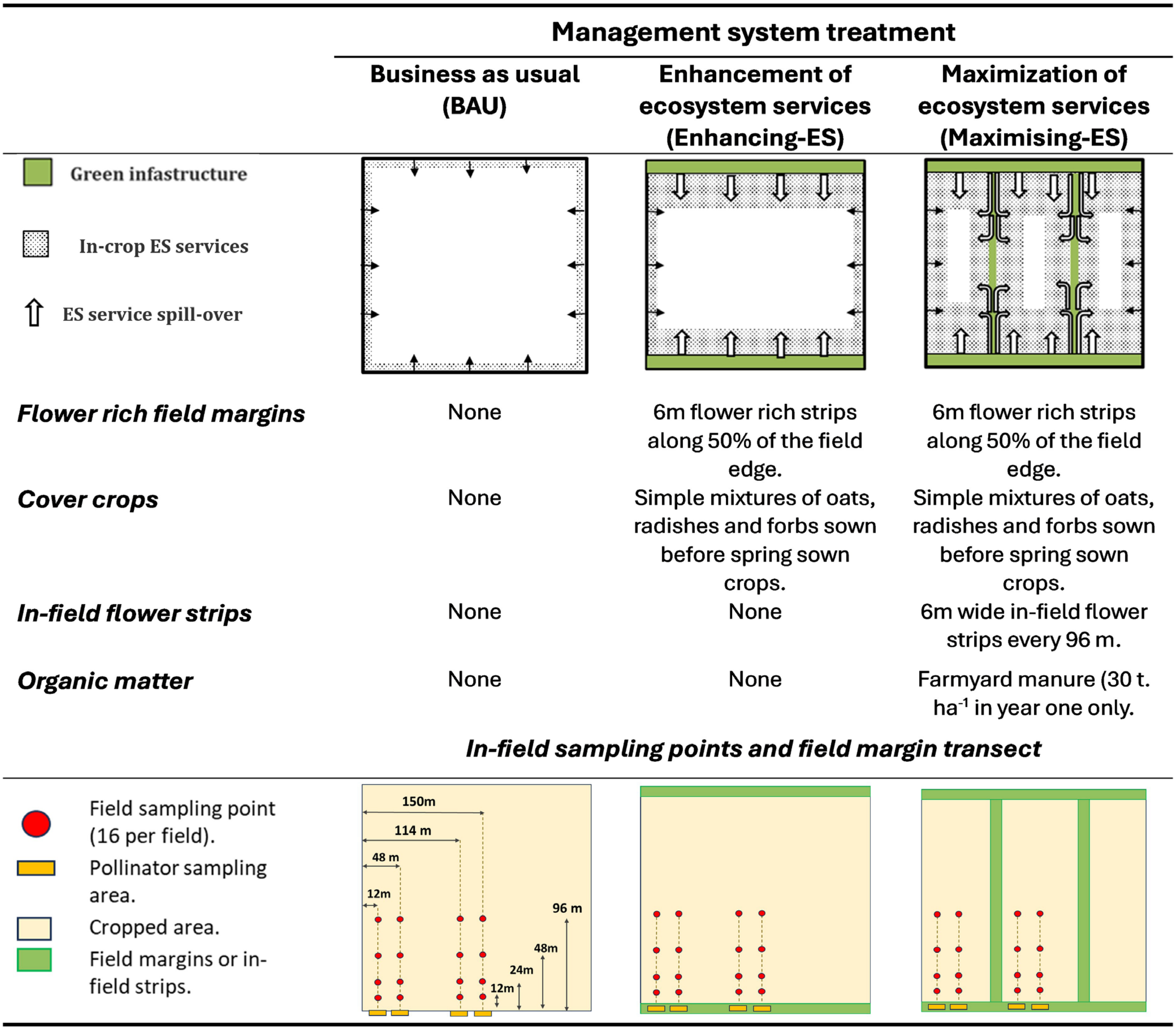

FIGURE 1 Pictorial representation of the composite management systems applied to three fields in each of 17-replicate UK farms, including a visualisation of the sampling strategy.

sampling point in April and June (20 days annual sampling). *Canopy active predators*: In April and June, inspection of cereal tillers or oilseed racemes was used to count the summed abundance of predatory invertebrates, principally aphidophagous hoverflies, lacewings, ladybirds and spiders. *Crop earthworm counts*: Eight 20 × 20 × 20 cm soil monoliths were extracted in October and hand sorted, counting deep burrowing anecic earthworms. *Pollinator communities in field margin areas*: In June and July, 10 × 2 m off-crop transects running along the field edge were used to sample bees, hoverflies and parasitic wasps, following restrictions for weather given by Pollard and Yates (1993) (Figure 1). The sampling of all invertebrates was from improved agricultural land and collected species that do not fall under the UK Animals (Scientific Procedures) Act 1986 and Animals in Scientific Procedures Act 1982, so separate ethical approval at the time of sampling was not required under our institutional guidelines.

While permission to sample on farmland was sought, no official licences or permits were required to undertake this work.

2.4 | Quantifying occurrence of crop pest species

Aphids: All aphids were counted during the inspection of cereal tillers and oilseed racemes when quantifying canopy active predators. *Snails and slugs*: Wheat mash baited saucer traps were used in May of each year to sample slugs and snails. Average biomass for each group was derived. *Arable Weeds*: A 0.5 × 0.5 m quadrat was placed at each in-crop sampling point in June, and counts of all economically important arable weed plants were recorded. We also counted tillers of Black grass (*Alopecurus myosuroides*) as a major pest in wheat.

2.5 | Measuring crop yield

Yield was directly measured in each field for cereals and oilseed rape at each of the 16 within-field sampling locations by hand harvesting the crop from 0.5×0.5 m quadrats. These were hand processed to remove chaff before calculating mean yield (tonnes ha^{-1}). We used precision yield from GPS-linked combine harvesters to validate these estimates. This was available from 36 field and year combinations, (ca. 24% of the fields monitored across years) and showed a strong correlation with in-field quadrat measurements (see Figure S1, *Precision yield* = 0.93 + 0.77 × *quadrat yield*, $F_{1,34}=46.7$, $p<0.001$, $R^2=0.58$). As the quadrat-based approach underestimated yield relative to the precision agriculture yields, this equation was used as a correction factor. This was done to avoid systematic under-estimation of field productivity.

2.6 | Economics of agroecological farming systems

The following summarises the economic assessment, with full detail provided in the Appendix S1 and Table S5. To assess the annual balance of profit and loss for each farming system (cereal and oilseed rape), a baseline predicted profit per field (GBP £ per field) was determined from the measured yield and crop prices (4-year mean of published commodity prices 2018–2021). We deducted the costs associated with the loss of productive land (e.g. the value of foregone crop production), and consumable and management costs for the enhancing-ES and maximising-ES systems for each field and year. For the UK governmental agri-environmental scheme (AES) payments are available for field margins, infield strips and cover crops established in the enhancing-ES and maximising-ES systems (Defra, 2023). We therefore also derived profit (GBP £ ha^{-1}) accounting for these subsidies (Table S5).

2.7 | Analysis

We assessed the response of yield (crop tonnes ha^{-1}) and profit (GBP £ ha^{-1}) with and without AES government subsidies to the management system (BAU, enhancing-ES and maximising-ES), year since establishment, crop type (spring and winter sown wheat, barley and oats as well as oilseed rape) and all interactions using general linear mixed models within lme4 in the R Statistical Environment (Bates et al., 2010; R Core Development Team, 2023). Random effects for the intercepts were specified as year since establishment nested within farm. Using the same generalised linear mixed model structure, regulating service provision (i.e. slug predation, aphid predation and oilseed rape pollination; soil carbon), beneficial invertebrate mean abundance (i.e. earthworms, ground beetles, rove beetles, spiders, hoverflies, bees and parasitoids) and crop pests (e.g. slug and snail biomass or weed counts) were assessed in response to management system, year and their interaction. The response metrics were all continuous (e.g. average plot abundance) and so modelled initially using a Gaussian distribution with identity link. Assessment of these initial models was undertaken using the DHARMa package (Hartig, 2022). In most cases deviations identified by this process were addressed using a $\log_e(N+1)$ transformation of the response. For continuous but overdispersed data a Tweedie distribution with log link was used within glmmTMB (Brooks et al., 2017). Models were simplified using deletion of least significant effects using either likelihood ratio tests *F*-test (normally distributed) or χ^2 (Tweedie distributed). Data were excluded for three occasions where the crops in a field for a particular year failed to establish. Some analyses were restricted to crops where that data was collected (i.e. pollination and oilseed pest parasitoid abundance to oilseed rape, cereal parasitoid abundance to cereal crops and aphid predation to winter wheat).

3 | RESULTS

3.1 | Supporting and regulating ecosystem services

Supporting and regulating ecosystem services benefited from the establishment of agroecological farming systems. Soil carbon stocks were found to differ between the three management systems ($F_{2,33.2}=13.8$, $p<0.001$; Figure 2A), being significantly higher within the maximising-ES treatment than the BAU control ($t_{30}=3.14$, $p<0.001$). Aphid predation was significantly higher in the enhancing-ES and maximising-ES treatments ($F_{2,51.4}=9.79$, $p<0.01$, Figure 2B). Neither slug predation rates ($F_{2,93.3}=1.99$, $p>0.05$; Figure 2C) nor counts of mummified aphids ($\chi^2_1=4.99$, $p<0.1$; Figure 2D) differed significantly between the management systems. However, in both cases there was a non-significant trend for these to be higher in the maximising-ES treatment. Seed set of oilseed rape attributable to insect pollination differed between the management systems ($F_{2,9}=6.19$, $p=0.02$; Figure 2E) with the enhancing-ES system being marginally higher than BAU ($t_9=2.13$, $p=0.06$) and significantly higher for the maximising-ES treatment ($t_9=3.49$, $p<0.01$). Overall effects of years since establishment were identified for aphid parasitism ($\chi^2_1=24.9$, $p<0.001$) and aphid predation ($F_{3,23.7}=5.11$, $p<0.01$). This was not a continuing temporal trend but varied in magnitude between years. No year and management system interactions were found ($p<0.05$).

3.2 | Populations of beneficial invertebrates

The provisions of within crop ecosystem services are underpinned by components of biodiversity present within the crops and immediate surrounding habitat. There was evidence that the agroecological management systems increased population sizes for anecic earthworms (treatment×year: $F_{6,97.6}=2.49$, $p=0.03$; Figure 2F), spiders ($F_{2,96.5}=7.36$, $p<0.01$; Figure 2N), crop-canopy predators ($\chi^2_2=8.03$, $p=0.02$; Figure 2K), bees (treatment×year: $\chi^2_6=14.2$, $p=0.03$; Figure 2M) and hoverflies (Treatment×year: $\chi^2_6=13.7$, $p=0.03$; Figure 2N). For these, the maximising-ES systems supported higher populations than

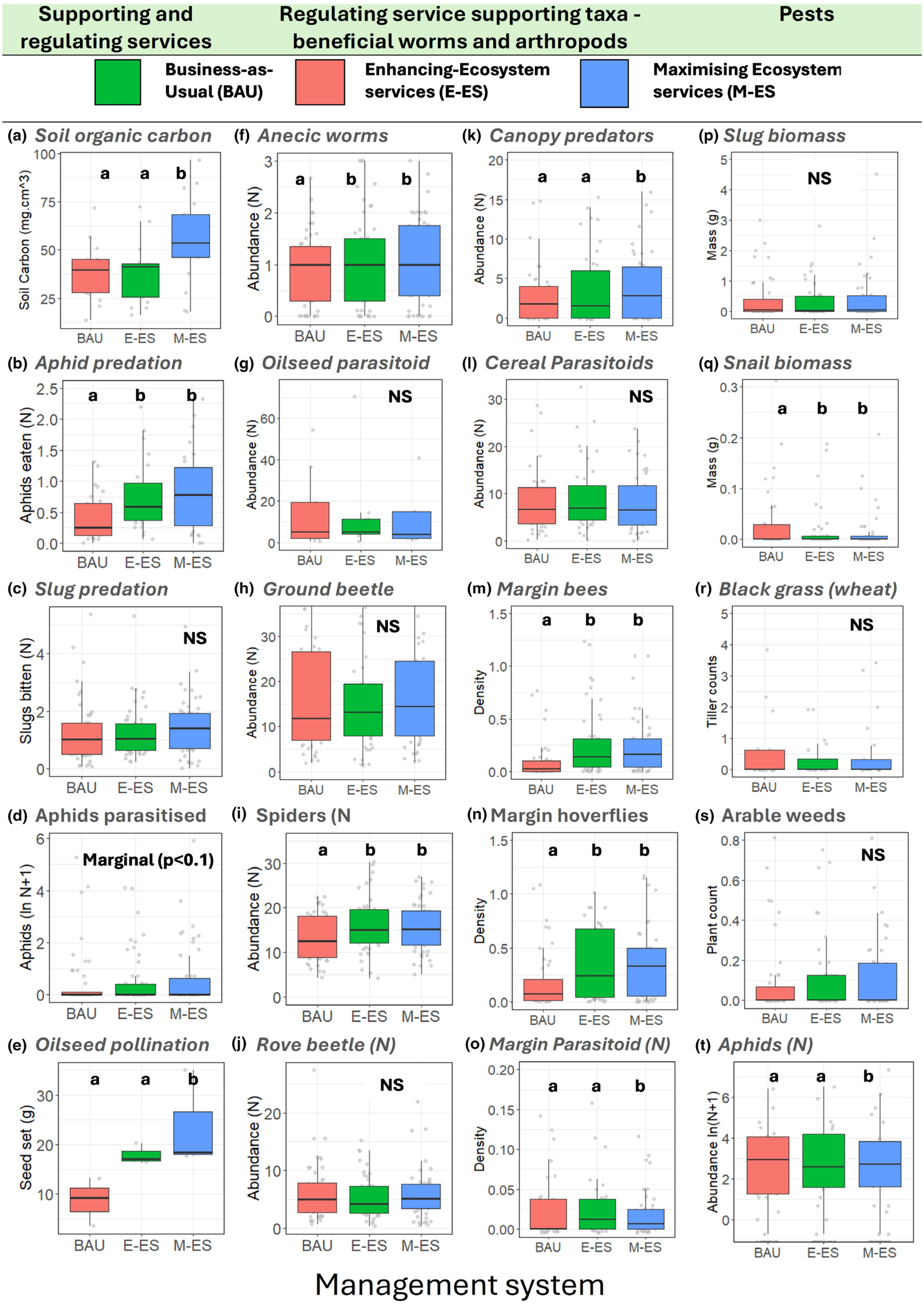

FIGURE 2 Boxplots showing the effect of agroecological farming systems across the 17 farms and 4 years on supporting (a–e) and regulating ecosystem services (f–o), as well as pest populations (p–t). Letters indicate significant differences from the BAU control. For each boxplot, the central line represents the median, with the box spanning the 25th–75th interquartile range.

the BAU control, with the enhancing-ES system supporting greater populations than the control only for earthworms, spiders, bees and hoverflies. For the earthworms and hoverflies, the significant year and management system interaction showed that the first year of monitoring had lower population sizes. Management system had no significant effect on within crop abundance of ground beetles ($F_{2,95.1}=0.54$, $p>0.05$; Figure 2H), rove beetles ($F_{2,96.9}=1.90$, $p>0.05$; Figure 2J), cereal pest parasitoids ($F_{2,78.2}=1.62$, $p>0.05$), oilseed rape parasitoids ($F_{2,18}=0.08$, $p>0.05$; Figure 2G) or the off crop field margin parasitoid density ($F_{2,100.2}=0.18$, $p>0.05$; Figure 2O). There was an unexpected significant negative effect of management treatment of margin-active parasitoids ($\chi^2_2=0.04$, $p=0.04$; Tweedie distribution; Figure 2O) indicating abundances were lower in general in the maximising-ES treatment than the BAU ($z=2.58$, $p<0.01$). Significant year effects were seen for earthworms ($F_{3,37.0}=6.80$, $p<0.001$), crop-canopy predators ($\chi^2_3=8.91$, $p=0.03$), oilseed pest parasitoids ($F_{3,9.0}=6.80$, $p=0.01$) and cereal pest parasitoids ($F_{3,28.1}=2.96$, $p=0.04$) (see Figure S2). No other significant effects were found ($p>0.05$).

3.3 | Pest populations within the crop

Creating farming systems that may benefit beneficial invertebrates runs the risks of also promoting pest populations. Aphid numbers responded to management system ($\chi^2_2=6.00$, $p=0.05$; Figure 2S) with the maximising-ES treatment having slightly higher numbers than the BAU control ($z=2.14$, $p=0.03$). Snail biomass was affected by an interaction between management system and year ($\chi^2_6=15.7$, $p=0.02$; Tweedie distributed; Figure 2Q), although the overall trend was for lower biomass in the enhancing-ES ($z=-3.10$, $p=0.02$) and maximising-ES treatments ($z=-3.16$, $p<0.001$) relative to the control. Management system had no effect on slug biomass ($\chi^2_2=1.07$, $p>0.05$; Figure 2P) and arable weed abundance ($\chi^2_2=0.23$, $p>0.05$; Figure 1S). While Black Grass tiller abundance was not significantly affected by management system, there was a trend of it being lower in the enhancing-ES and maximising-ES systems ($\chi^2_2=0.89$, $p>0.05$; Figure 2R). Aphids ($\chi^2_3=19.9$, $p<0.001$), slug biomass ($\chi^2_3=17.7$, $p<0.001$) and arable weeds ($\chi^2_3=19.4$, $p<0.001$) showed interannual variation (see Figure S3). No other significant effects were identified ($p>0.05$).

3.4 | Yield and profitability

Yield represents how effective our agroecological management systems have been in promoting key ecosystem services. Crop yield differed between the management systems ($F_{2,97.6}=6.01$, $p<0.01$), with higher yields within the cropped area seen for both the enhancing-ES ($t_{97}=2.87$, $p<0.01$) and maximising-ES ($t_{97}=3.14$, $p<0.01$) management systems relative to BAU control (Figure 3A). Yield was also affected by crop type ($F_{6,47.1}=29.5$, $p<0.001$) but not by an interaction between crop and management system ($p>0.05$). Model estimates suggested that relative to the BAU control, the enhancing-ES treatment increased yields by 0.30 (SE ± 0.11) tonnes ha^{-1} across all crops and by 0.32 (SE ± 0.10) tonnes ha^{-1} for the maximising-ES system. Yield was significantly affected by year, although there was no clear temporal trend, with the second and fourth years being higher yielding ($F_{3,38.7}=6.36$, $p<0.01$).

Agroecological management comes at a cost in terms of the management to establish it as well as crop yields forgone to land converted to field margins and in-field strips (Table S4). Profit affects farmer attitudes to viability and adoption for agroecological farming systems. Profit was significantly affected by management system ($F_{2,95.0}=64.7$, $p<0.001$), year since establishment ($F_{3,32.1}=2.92$, $p<0.05$) and crop type ($F_{6,40.2}=2.92$, $p<0.05$). There was no interaction between crop and management system ($p>0.05$). The cost of establishing the agroecological farming systems relative to BAU meant that profit was significantly lower in the enhancing-ES ($t_{95}=-3.86$, $p<0.001$) and maximising-ES treatment ($t_{95}=-11.1$, $p<0.001$) (Figure 3B). Model estimates suggested that the enhancing-ES system and maximising-ES systems were respectively on average 83.9 (SE ± 21.7) and 240.6 (SE ± 21.7) GBP £ ha^{-1}year^{-1} less profitable the BAU control. This reduced profitability is in part due to establishing field margins, in-field strips and cover crops (see Figure S4). When AES subsidies are present, net profit is still significantly affected by management system ($F_{2,94.9}=29.0$, $p<0.001$), crop type ($F_{6,39.9}=11.33$, $p<0.001$) and year ($F_{3,32.0}=2.95$, $p<0.05$). However, the inclusion of AES subsidies meant that while the enhancing-ES system was no longer significantly different from BAU, the maximising-ES treatment remained significantly less profitable ($t_{95}=-6.95$, $p<0.001$) by on average 144.1 (SE ± 20.8) GBP £ ha^{-1}year^{-1} (Figure 3C).

4 | DISCUSSION

We have directly tested the efficacy of integrating agroecological measures into intensive arable agriculture both from the perspective of effects on proximate drivers, like regulating ecosystem services and beneficial invertebrates, as well as on end points of yield and profitability. Ultimately, farms are businesses and farmer decisions for adoption will always include consideration of economic viability (Sakrabani et al., 2023). Fostering agroecological farming requires evidence not just of its environmental effectiveness but also of its farm economic implications. This is critical for identifying economic drivers to elicit system change to promote greater sustainability, including government agri-environmental payments, commodity premiums for sustainable systems and natural capital markets.

4.1 | Regulating ecosystem services and beneficial invertebrates

Reintroducing key resources at local and landscape scales, often through the creation of semi-natural habitats, like wildflower field margins, has been proposed as a key approach to promote

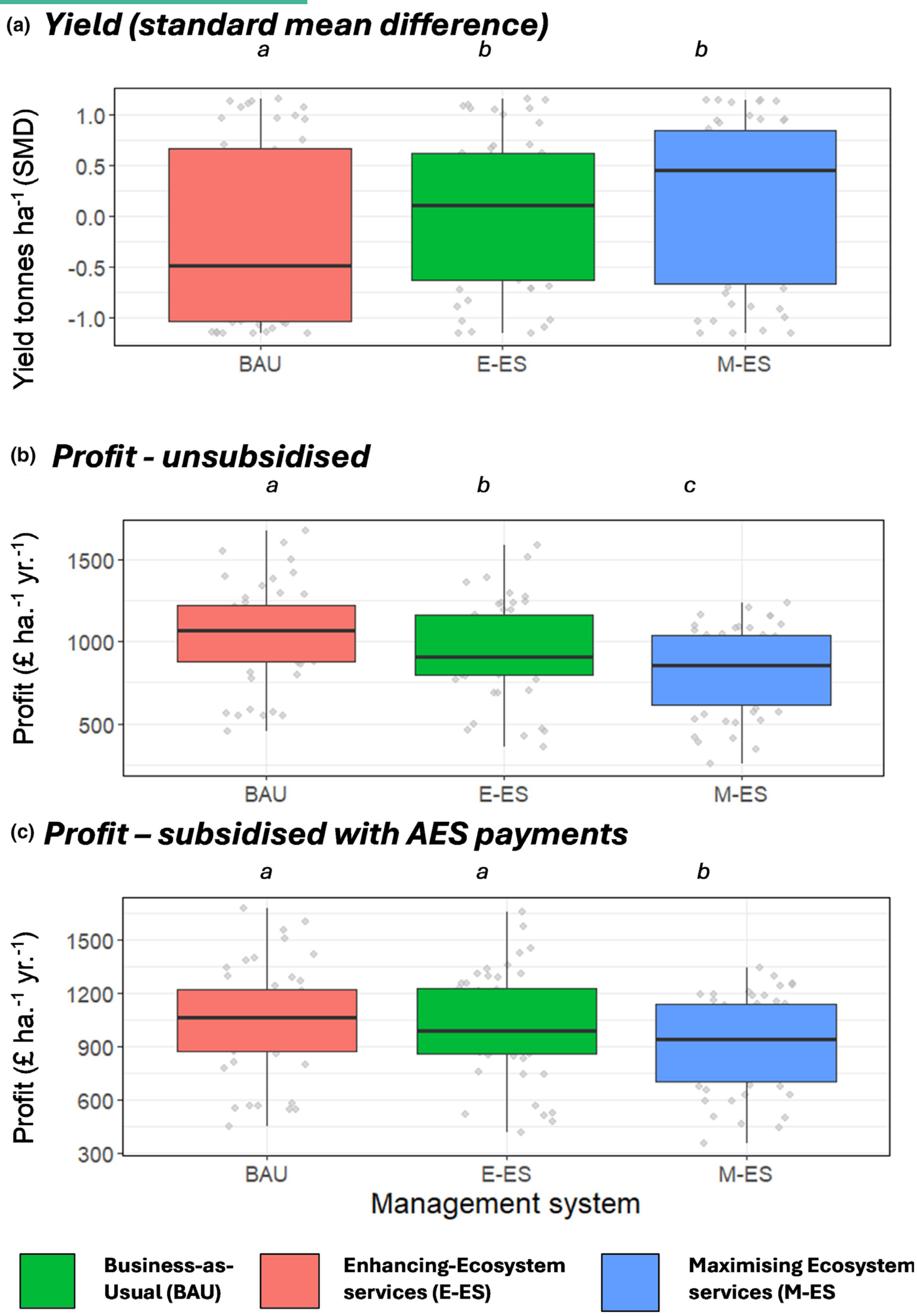

FIGURE 3 Boxplots comparing crop yield (a) and profit for the three farm management systems over 4 years. Profit is considered under econoimc conditions without (b) and with (c) English agri-environmental subsidies for the interventions within the Enhancing-Ecosystem Service (i.e. wild flower field-margins and cover crops) and Maximising-Ecosystem Service treatments (i.e. wild flower field margins, in-field strips and cover crops). The application of organic matter in the Maximising-Ecosystem Services treatment is unsubsidised in England. Yield is expressed as an effect size (standard mean difference) to account for between-crop type differences in average yield. Differences between the treatments are indicated by letters above the individual boxplots, where different letters indicate a significant difference in the treatment mean ($p < 0.05$).

populations of beneficial insects. The extent to which this is important for all taxa is unclear as it is based largely on a limited number of widely monitored groups, in particular insect pollinators and ground beetles (Albrecht et al., 2020; Batary et al., 2015). It is also possible that many of the reported responses of these taxa are due to concentration effects within these created habitats and so represent artefacts of sampling strategies (but see Carvell et al., 2017). Approaches like in-field strips are an extension of this concept that facilitate the spill-over of beneficial invertebrates from off-crop areas into the crop (Defra, 2023; Priyadarshana et al., 2024; Woodcock, Bullock, et al., 2016). These interventions underlie our enhancing-ES (field margins and cover crops) and maximising-ES (field margins, in-field strips, cover crops and organic matter) management systems. These were intended to provide critical foraging, breeding and overwintering resources for beneficial invertebrates, through which we show here positive effects on spiders, earthworms, sward active predatory invertebrates, as well as off-crop bees and hoverflies. Increases in the population of these beneficial invertebrates had knock-on benefits for some regulatory ecosystem services.

Predation rates of aphids were highest within the enhancing-ES and maximising-ES management systems. This corresponds with increased abundances of crop active predators, like ladybirds, hoverfly larvae and spiders. There was also a trend, albeit non-significant, of increasing rates of aphid parasitism by Hymenoptera under the maximising-ES management system. Interestingly, this contrasts with the finding of higher parasitoid numbers in the BAU treatment off-cropped areas. This trend was likely a result of the prevalence of umbellifers at the edges of most arable fields often frequented by parasitoids. However, these umbellifer-rich edges were only monitored in the BAU treatment, with the wildflower field margins which contained relatively few umbellifers being monitored in the enhancing-ES and maximising-ES management systems. Unlike aphid predation, rates of slug predation showed no response to the agroecological farming systems, a finding which was concordant with the absence under these management systems of population changes in ground or rove beetles. Why the creation of field margins and in-field strips had no apparent benefit on the abundance of crop ground and rove beetle populations is unclear. A meta-analytical review suggested that these taxa may be less sensitive to increased availability of semi-natural habitat than other invertebrate natural enemies (Shackelford et al., 2013) although it is possible that these communities are already quite diminished in homogeneous crop lands where source populations are scarce (Priyadarshana et al., 2024). However, by grouping these taxa together and ignoring individual species responses, we may not be quantifying community-level responses between the management systems (Jowett et al., 2019). It is likely that a focus on this metric may conceal specific responses for species with high pest attack or consumption rates (Greenop et al., 2020). This may explain why the biomass of snails under the enhancing-ES and maximising-ES treatments was lower than the BAU treatment, with this being driven by populations of specific species, for example, large *Pterostichus* spp. While pollination was assessed only for oilseed rape, we found strong evidence that seed set was highest in the maximising-ES management systems. This is consistent with the increased densities of bees and hoverflies within the sown field margin areas of this management system.

4.2 | Crop yield benefits from agroecological farming systems

While crop yield is only one aspect of agricultural production, with aspects of quality (e.g. oil or protein content) and aesthetics (e.g. blemishes on fruits) being important, it remains a major concern to farmers. When we focused on the economically most important crops of cereals (wheat, barley and oats) and oilseed rape, we found that yield responded positively to both agroecological farming systems. Model predictions suggested an increase of ca. 0.3 tonnes ha^{-1} across all crops for the enhancing-ES and maximising-ES, although average individual yields of crops differed. Outside of increases in beneficial invertebrates and the ecosystem services they provide, the addition of organic matter (principally farmyard manure) applied in the initial year of the maximising-ES management is also likely to have increased yields relative to the enhancing-ES system. However, while farmyard manure provides nitrogen, phosphorus and potassium, it has a low bioavailability with 10% of these nutrients available for the first crop and 5% for the following (AHDB, 2023). Certainly, a recent meta-analysis suggests that organic amendments may have largely substitutive (as opposed to additive) effects on yield when analysed with synthetic fertilisers (MacLaren et al., 2022). Applied only before the first year of the study, its contribution to the overall trends in yields may therefore have been relatively small. In the medium term, adding organic matter improves soil structure, nutrient supply and microbial activity which may have accumulating positive yield impacts. In the longer term, it enhances soil organic matter (as found in this study) and so helps sequester carbon (Johnston et al., 2009). In the enhancing-ES where no farmyard manure was applied, as well as more generally for the maximising-ES treatment over the 4 years of the rotation, yield differences are therefore likely to have been linked to a large part with maximising other regulating ecosystem services like pollination and pest control.

4.3 | Conditions for economic viability

Profit margins are arguably more relevant than yield for farmer decisions as they integrate the hidden costs associated with different management systems. We have shown that changes in management that enhance beneficial invertebrates and regulate ecosystem services have a positive effect on yield. However, this increase comes at a cost in land forgone to production, as well as establishment and subsequent management. Profitability of the investigated agroecological farming systems is achieved only under certain circumstances. Critically, as management interventions move from trying to enhance ecosystem services (wildflower field margins and cover crops) to attempts to maximise them (wildflower field margins and in-field strips, cover crops and organic matter) the cost increases. This means that without agri-environmental scheme subsidies for wildflower field margins, in-field strips and cover crops, neither of the agroecological farming systems would have been profitable.

Even so, subsidies as they currently exist are only sufficient to make the enhancing-ES system comparable in profit to the BAU control. Without subsidies, there is a negative correlation between profit and the ratio of agri-environmental interventions (wildflower field margins and in-field strips) to cropped area (see Figure S4). Subsidising farmers for management that has societal benefits (e.g. biodiversity) may be critical to facilitate the transition to lower impact farming systems (Batary et al., 2015). The importance of such subsidies to ease the transition to agroecological farming therefore remains critical.

The unprofitability of the maximising-ES system, even with agri-environment subsidies, will likely act as a barrier to its adoption. This failure to be profitable is associated in part with an absence of subsidies for organic matter additions (Defra, 2023). As a result, while the addition of farm-yard manure increased soil carbon stocks, its use had an insufficient impact on yields to offset losses in profit. In addition, the cost of bulk organic matter products may be high relative to any short term (e.g. 4 years) capacity for them to increase yields. This issue is exacerbated by the polarisation of UK farming systems to either livestock or arable (often with a regional bias) (Stoate et al., 2009). On-farm composting may represent alternatives, albeit one with scaling issues, to providing organic matter sources that do not rely on livestock, while more circular local supply through municipal links (e.g. green composted waste) could also provide opportunities for low-cost organic matter sources. It is worth noting that results from the precision yield data alone (collected by certain types of combine harvesters and so available for only a limited number of sites) demonstrated that the maximising-ES system increased average yields by double that of the enhancing-ES system (ca. 0.6 tonnes ha^{-1}) (Figure S5). Precision yield data accounts for within-crop variability in yield (e.g. patches of high pest pressure) that may be missed by quadrats. Such an increase if typical of other fields would increase the economic viability of more intensive agroecological interventions. Future developments of the approaches proposed here could take advantage of ecologically informed precision conservation principles that target habitat creation (e.g. field margins and in-field strips) to the lowest yielding areas of the crop (Knapp et al., 2023). In so doing, the benefits of these interventions in promoting crop yield may come at the lowest cost in terms of loss of productive crop land.

5 | CONCLUSIONS

Our results provide some optimism for an economic basis to promote the adoption of complex agroecological farming systems. However, as the complexity of the agroecological interventions increases they become less profitable within the current economic environment, even with subsidies. In the absence of new financial drivers for adoption individual farmer attitudes will likely remain the ultimate limitation to complex agroecological systems uptake, particularly where these require significant investment of effort or deviation from experiential comfort zones (Burian et al., 2024; Comer et al., 1999; Follett et al., 2024). Even so, evidence that some agroecological farming systems can meet an economic breakeven point may encourage adoption. Attitudes within the farming community acknowledge the need to future proof farming systems, particularly in the face of declining soil health, pesticide resistance and future environmental stresses like climate change (Comer et al., 1999; Jaworski et al., 2024; Novickyte, 2019). As a result, farmer attitudes may well be shifting to increased acceptance to adopt such practices even at the risk of reduced short-term profit if it infers longer term farm business sustainability benefits. Supporting farmers to better understand how effective agroecological farming practices have been on a site-by-site basis may also be a critical step breaking farmers free from 'intensification traps' (Burian et al., 2024). Better training in agroecological management and advances in farmer led ecosystem services monitoring may further help to address uncertainty in these systems promoting adoption (McCracken et al., 2015; Shirali et al., 2024). Overall, enhancing ecosystem services creates an opportunity to reduce agrochemical inputs, saving costs and reducing environmental impacts.

AUTHOR CONTRIBUTIONS

The analysis and first draft by Benjamin A. Woodcock, with revisions by all authors. Experimental design and coordination by Benjamin A. Woodcock, Sam Cook, Matthew S. Heard, Marek Nowakowski, Jonathan Storkey, James M. Bullock, John Redhead and Richard F. Pywell. Field and lab work by Sarah Hulmes, Lucy Hulmes, Martin Torrance, Jennifer Swain, Jordan Rainey, Maico G. Weites, with data management from Richard Ostler. Project support by Claire Carvel.

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CONFLICT OF INTEREST STATEMENT

Nowakowski runs an agronomist consultancy supporting farmers in establishing agri-environmental interventions. Woodcock, Bullock and Pywell have received funding from Bayer and Syngenta to undertake research into field margins and the impacts of pesticides on bees.

DATA AVAILABILITY STATEMENT

Data and code is available at Zenodo https://doi.org/10.5281/zenodo.15366699 (Woodcock, 2025).

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Figure S1. Relationship between precision yields from farmers combine harvesters and yields measured manually using a quadrat.

Figure S2. Significant year effects were seen for earthworms ($F_{3,37.0}=6.80$, $p<0.001$), crop-canopy predators ($\chi^2_3=8.91$, $p=0.03$), oilseed pest parasitoids ($F_{3,9.0}=6.80$, $p=0.01$) and cereal pest parasitoids ($F_{3,28.1}=2.96$, $p=0.04$).

Figure S3. For pest aphids ($\chi^2_3=19.9$, $p<0.001$), slug biomass ($\chi^2_3=17.7$, $p<0.001$) and arable weeds ($\chi^2_3=19.4$, $p<0.001$) inter annual variation between years was found.

Figure S4. Effect of the ratio between agri-environmental scheme interventions (combined sown field margins and in-crop strips) and the cropped area in fields on yield, (a), profit without agri-environmental scheme payments (b) and profit with (c) agri-environment scheme (AES) payments.

Figure S5. The average (±SE) difference in yield between three farm management systems expressed as standard mean difference yield, for precision yield data derived from combine harvesters.

Table S1. Seed mix, sowing rates and average establishment success in field margins for plants used in the establishment of both field margins and in-field strips for the T2—enhancing ES and T3—maximising ES.

Table S2. Plant species mix and sowing rates used in the cover crop mixtures used in this study.

Table S3. Rotations on the 17 farms sampled during this experiment.

Table S4. Details of the individual fields sampled in this study, providing the area (cropped and total), the length of flower-rich field margins and in-crop strips established at each site, and Shannon-Winer land use diversity surround the fields to a radius of 1 km.

Table S5. Costs associated with farm operations needed to support sustainable management practices in the enhancing-ES and maximising-ES treatments include staff time, fuel, machinery purchase costs and longer-term depreciation of machinery.

Table S6. Significance tests for the response of beneficial invertebrates effect of crop management practices (cover crops, farmyard manure-FYM), field margin establishment (floral species richness and sward structure), the relative areas of field margins and infield strips to the cropped areas, and to landscape structure (Land use Shannon diversity of habitat types—H').

Appendix S1. Supplementary methods and results.

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