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Management practices, and not surrounding habitats, drive bird and arthropod biodiversity within vineyards

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ABSTRACT

Agrochemical use and habitat loss associated with agriculture are drivers of biodiversity loss worldwide, and biodiversity-friendly farming practices, including organic management, are increasingly promoted by policy and industry in an attempt to offset this. Grapes are an important perennial crop globally, and in the UK, viticulture is the fastest growing agricultural sector and sustainable vineyard management is promoted by the Sustainable Wines of Great Britain 'SWGB' scheme. Here, we performed the first assessment of the simultaneous effects of surrounding habitats and vineyard management practices on bird and arthropod biodiversity across 22 English vineyards (10 certified-organic, 11 SWGB-accredited, and 3 both). We surveyed birds using point counts and arthropods with pitfall traps, and used linear mixed modelling to relate diversity and abundance to habitat and management predictors at landscape and local scales. We show that arthropod abundance is significantly higher on organic vineyards, whilst bird diversity is significantly lower on SWGB-accredited vineyards, but we find no other significant effects of organic certification or SWGB-accreditation on biodiversity. We also find no significant effects of the surrounding habitat structure on the biodiversity of birds and arthropods. Instead, we show that ecotoxicity scores derived from agrochemical use data have a significant negative impact on bird diversity, and on arthropod abundance and diversity. Organic status predicts a significant reduction in ecotoxicity scores, but only when application frequency is not considered, and contradictorily, SWGB-accredited vineyards have higher ecotoxicity scores than those without accreditation. Ground vegetation cover has a consistent, positive effect on bird and arthropod diversity, with model predicted diversity increasing 1.5 and 2.5-fold, respectively, in vineyards with the highest vegetation cover, and herbicide use has a negative effect on the vegetation cover. Our research demonstrates that individual management practices have a stronger effect on vineyard biodiversity than the habitat context, overall management regime or certification. Our study sets an important baseline for vineyard management and accreditation schemes and generates key recommendations for improvement. To benefit biodiversity within vineyards, we recommend that sustainability accreditation schemes include requirements to reduce the ecotoxicity of used agrochemicals, and promote higher ground vegetation cover and height by reducing herbicide use.

1. Introduction

Habitat loss, landscape simplification and increased chemical use associated with agricultural expansion and intensification are major causes of biodiversity loss globally (Pereira et al., 2012; Newbold et al., 2016). A recent Europe-wide analysis found agricultural intensification, and particularly the associated agrochemical use, to be the main driver of most bird population declines (Rigal et al., 2023). Similarly, a global study found terrestrial arthropod abundance and biomass to have been steadily declining by ~9% per decade, supporting strong declines in Europe and a negative relationship with land use change (van Klink et al., 2020). These declines not only risk extinctions of rare species, but also threaten the loss of key ecosystem functions that benefit agriculture (Hendershot et al., 2020).

Grapevines are an important global crop, with over 7 million hectares of land dedicated to their production, which accounts for about 5% of the global cover by perennial crops (Ritchie and Roser, 2013, Venkitasamy et al. 2019). At landscape scale, increased vineyard cover has

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been shown to have a detrimental effect on biodiversity, including on birds (Assandri et al., 2016; Pithon et al., 2016), bats (Rodríguez-San Pedro et al., 2019), and arthropods (Geldenhuys et al., 2022). However, maintaining habitat heterogeneity, through habitat retention and provision within vineyards can help offset these impacts (Winter et al., 2018; Paiola et al., 2020), particularly in more homogenous landscapes where resources are otherwise limited (Assandri et al., 2016; Martin et al., 2019). For example, retention of native woodlands and hedgerows within Swiss (Guyot et al., 2017), German (Rösch et al., 2023), and Chilean vineyards (Steel et al., 2017; Muñoz-Sáez et al., 2020) enhanced bird abundance and diversity, whilst wildflower mixes and reduced mowing that increased ground vegetation cover benefited arthropod and bird diversity in European (Puig-Montserrat et al., 2017; Griffiths-Lee et al., 2023) and South African vineyards (Geldenhuys et al., 2022). Studies from Europe (Brambilla et al., 2017), and South America (Muñoz-Sáez et al., 2020) have also demonstrated the potential for vineyards to support high abundances of threatened and endemic bird species.

The viticultural industry faces pressure to move towards more environmentally sustainable management (Merot et al., 2019; Barbaro et al., 2021), intensified by the new Global Biodiversity Framework, which sets a target to manage agricultural landscapes sustainably, '*including through a substantial increase of the application of biodiversity friendly practices*' (Keping, 2023). Managing agricultural landscapes in ways that are less detrimental to biodiversity is often encouraged through agri-environmental schemes and accreditations, and rewarded through compensation or higher product prices (Tscharntke et al., 2012; Boetzl et al., 2021). Globally, organic farming has been shown to enhance species richness on agricultural land by an average of 30%, though this positive effect is greater in more homogenous and intensively-managed agricultural landscapes (Tuck et al., 2014).

In the UK, the viticultural industry is the fastest growing agricultural sector, attributed to increasing summer temperatures making the climate increasingly comparable to other European wine-growing regions (Nesbitt et al., 2019). Due to the recent expansion, specific recommendations for UK viticulture are lacking, and research is limited, though a recent industry survey found heavy reliance on agrochemicals (Griffiths-Lee et al., 2022). To address this, a national sustainability scheme called Sustainable Wines of Great Britain (henceforth 'SWGB') has been formed and over 80 vineyards, accounting for over 55% of the UK's vineyard hectarage, are now members (WineGB, 2022). This scheme strives to ensure environmental, social, and economic sustainability within the industry through a process of continual improvement (WineGB, 2022). It lacks minimal requirements or specific targets, but rather provides broad recommendations such as *"create new habitats in order to increase biodiversity, such as hedges"*, and members commit to a continual cycle of improvement towards sustainability, with minimal agrochemical use and biodiversity conservation strongly encouraged.

By working in multiple English vineyards spanning a range of management practices, and also varying in the structure of their surrounding landscapes, we provide the first simultaneous assessment of the effects of surrounding semi-natural habitats and management on vineyard biodiversity. Our aims were: (1) to assess the relative impact of surrounding habitat structure and vineyard management on bird and arthropod abundance and diversity, and (2) to compare bird and arthropod abundance and diversity between certified-organic and non-organic vineyards, and based on SWGB accreditation status. Due to their differing mobility, we predict surrounding semi-natural habitats to have a stronger effect on birds than on arthropods, while in contrast we predict arthropods to be more strongly affected by vineyard management, including organic viticulture, which we expect to have an overall positive effect on biodiversity.

2. Materials and methods

2.1. Study vineyards

This study took place within 22 English vineyards from across the UK's key wine-growing regions (Fig. 1). Sites were chosen to represent the broader English vineyard industry, with 10 sites being certified organic and 12 non-organic, and half being accredited through the SWGB scheme (three sites were accredited through both). Sites were selected using an objective site selection protocol following Gillespie et al. (2017) to maximise landscape structure and management gradients. Study sites were selected from a list of candidate sites, ensuring that they: (i) are within UK's key viticultural regions (Fig. 1), (ii) grow the key UK grape varieties (WineGB, 2022), (iii) are established vineyards (planted before 2018), and (iv) are >1 ha in size to enable sufficient sampling. We characterised the landscape complexity and configuration of the candidate sites and selected sites that contrasted the most in surrounding habitats complexity, as well as in management (e.g. organic versus non-organic), whilst controlling for vineyard size. Full details are given in Appendix A.

2.2. Biodiversity sampling

We sampled bird and arthropod communities in 2021 and 2022, repeating surveys three times each year, with sampling seasons aligning with the key stages of the vine lifecycle ('budding': early to mid-April; 'flowering': late June-mid-July; 'harvest': mid-September to mid-October).

2.2.1. Birds

We performed 10-minute point counts across 44 locations during each survey season (average 1.81 $\pm$ 0.18 SE point counts per site, range 1–4, depending on vineyard size), and in total, we performed 222-point counts across the six sampling periods. We aimed to conduct a survey at all 44 locations each sampling period, but this was limited by poor weather and access restrictions, thus we performed between 35 and 40 point counts per period. Point count locations were placed in vine fields, at least 50 m from boundary habitats, and minimum 250 m from other point count locations (see Appendix B for a diagram). Surveys were conducted between 05:00 – 09:00 and within 3 hours of sunrise, which varied between sampling seasons. Surveys only took place on dry and still days (Bibby et al., 2000), and were performed by the same observer.

2.2.2. Arthropods

Arthropod communities were sampled using pitfall traps along 79 transects across 21 vineyards each survey season (average of 3.85 $\pm$ 1.89 SD transects, range 2–10). We deployed pitfall traps along transects running perpendicular to the field boundary (see Appendix B), with traps placed directly underneath vines at 20-meter intervals. Transects varied in length depending on field size and were 40 (3 traps, n=10 transects), 60 (4 traps, n=37) or 80 (5 traps, n=30) meters in length. Transects ran from 3 distinct boundary types: woodland (n=32 transects), hedgerow (n=32), or open boundary lacking any features (n=15). We aimed to evenly distribute transects of different lengths and from different boundary types between certified-organic and non-organic, and between SWGB-accredited and non-SWGB vineyards. This was possible for hedge and woodland transects up to 60 m in length, but out of the 16 open-boundary transects, 15 were in non-organic and 14 in SWGB-accredited vineyards, whilst of the 30 80-meter transects, 25 were in non-organic and 23 in non-SWGB vineyards.

We used pitfall traps to sample arthropods (following methodology from Brown and Matthews, 2016), deploying 1713 traps across the six sampling periods (average 291 per sampling period, range: 237–316; totalling 786 in 2021, and 927 in 2022). Clear plastic cups with 10 cm diameter were placed in the ground with the cup lip flush with the soil surface, and covered with a metal mesh square (0.8 cm mesh size) to

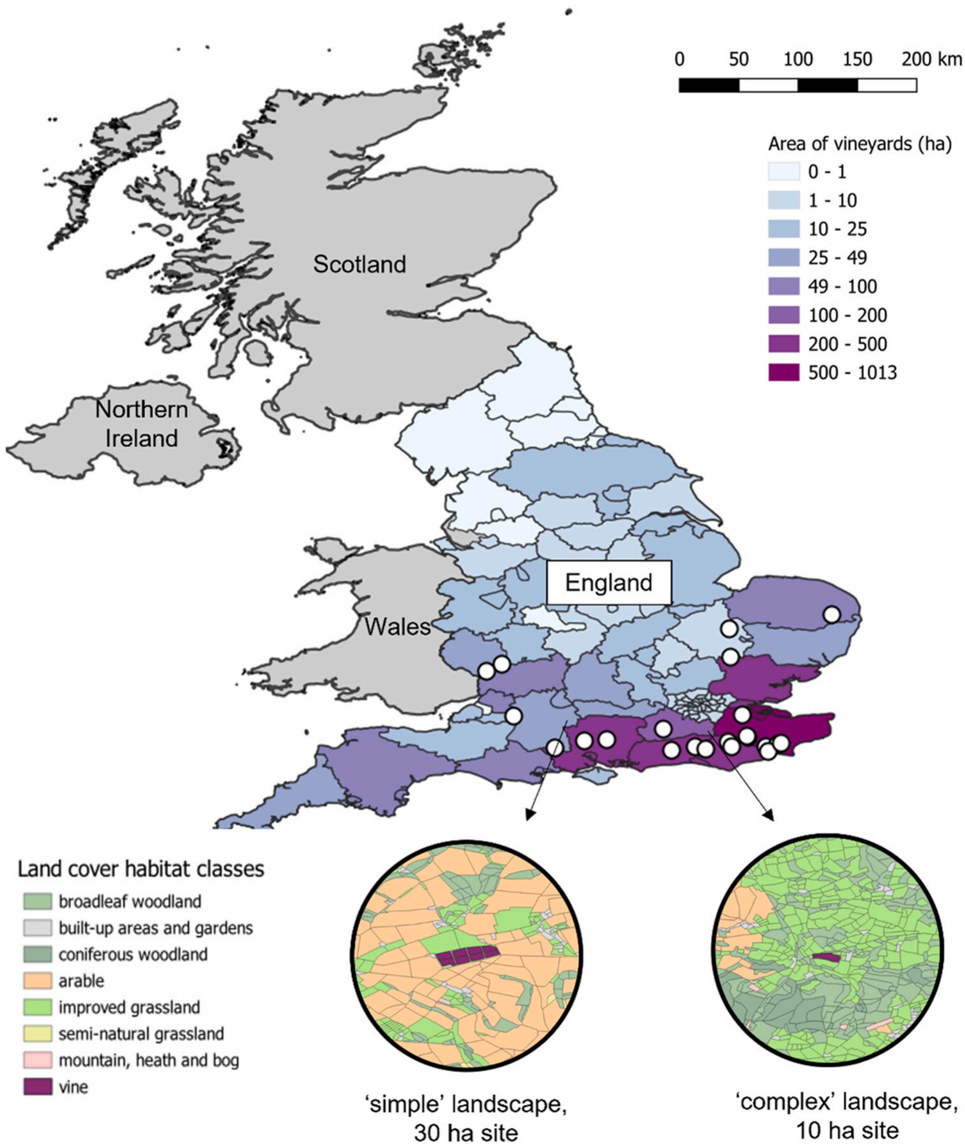

Fig. 1. Map of the UK showing the area, in hectares, of commercial vineyards per county in England. White circles indicate study sites. Examples of two 2.5 km landscape buffers from contrasting landscapes are shown. Hectarage data were from EnglishWine.com, compiled by Stephen Skelton.

reduce by-catch. Traps were filled with ~50 ml of water with organic unscented washing up detergent (10 ml detergent per 5 L water) and left for 24 hours after which the catch was drained, and any earthworms and slugs were discarded. Collected specimens were stored in 75% ethanol. Across the sampling seasons, 34 traps were damaged or destroyed, and catch from these was discarded. All arthropods were identified to order level.

2.3. Characterising landscape structure and vineyard management

2.3.1. Habitat characteristics

We mapped the landcover habitats using the CEH Land cover 2021 map (Marston et al., 2022) at two spatial scales. First, we used a 'landscape-scale' buffer of 2.5 km around the central coordinates of each site, which was large enough to encompass the whole of our largest site, whilst minimising the overlap of buffers between sites. We used this size buffer as the availability of semi-natural habitats in the wider landscape can be an important driver of vineyard bird communities (e.g. Guyot et al., 2017; Muñoz-Sáez et al., 2020). Except for two site pairs, the landscape buffers did not overlap and were spatially independent. Secondly, we used a 'local-scale' buffer of 200 m around bird point count locations, and 100 m around arthropod transects, which was informed by similar studies (Caprio et al., 2015; Barbaro et al., 2021). At both scales, we calculated the cover by woodland (combining coniferous and deciduous), semi-natural grassland and agricultural (including improved grassland) areas, as well as total vine area, the average field size, and edge density. The presence of freshwater bodies was limited (at most constituting 0.02% of the landscape buffer), so instead, we calculated the length of '*linear water features*', which included rivers and streams, though this predictor was only calculated at the landscape-scale, as rivers and streams were absent from the local scale. At the local scale, we calculated the length of '*linear wooded features'* which included hedgerows and tree lines within vineyards, as these may harbour biodiversity or be used as stepping stones for species using the vine fields. We performed this in QGIS (3.30.00).

The amount of semi-natural habitats around our study sites ranged between <1% and 42% in a 2.5 km radius buffer (Fig. 1) and the vineyards also varied in vine hectarage, which ranged between 1 and 182 ha (mean 24.39 ha; covering 0.05–9.27% of each buffer). There were a few significant differences in vineyard size and the surrounding habitats between organic and non-organic, and between SWGB-accredited and non-SWGB sites, with the mean field size being significantly larger at

SWGB-accredited (4.50 ha ± 0.02) than non-SWGB sites (4.16 ha ± 0.04, $t(19) = 2.617$, $p = 0.017$), and the length of linear wooded features being higher in non-organic (1496.4 m ± 5.41) than organic sites (1150.4 m ± 7.13, $t(42) = -3.664$, $p < 0.001$), and higher in SWGB-accredited (1487.0 m ± 4.62) than non-SWGB sites (1157.4 m ± 9.21, $t(42) = 3.463$, $p < 0.001$; Appendix C).

2.3.2. Management

Ground vegetation cover across English vineyards varies across the vine lifecycle, between sites and within the vine rows (see Appendix D for examples). To measure this variation, at 0 and 40 m along one arthropod transect (see Appendix B) in each sampling field, we measured sward height across the alleyway between two vine rows (18–22 measurements per alleyway, depending on its width), and we also estimated proportion of bare ground, to the nearest 5%, using a randomly placed 50×50 cm quadrat. We computed a crude *ground vegetation cover* metric from these measurements: *veg cover = μ [ground vegetation height] * (1 – proportion bare ground),* where μ is the average across the transect. For landscape scale analyses, ground vegetation cover was averaged per site, giving an indication of the general management at each site, whilst for local scale analyses, we used the value measured at the given survey location and time.

Through a vineyard management survey completed by site managers, we collected information on chemical inputs and vineyard management practices across our study sites. First, by using the reported lists of chemical inputs from each vineyard, we calculated a measure of ecotoxicity by obtaining environmental toxicity information for individual active ingredients from the Pesticide and Bio-pesticide Properties Databases (Lewis et al., 2016; see Appendix E for details), which was also used by similar recent studies (e.g. Möth et al., 2021). The databases include environmental toxicity (ecotoxicity) assessments for active ingredients, which are calculated based on the substances' impact (as indicated by LC50 and LD50 values) on several organism groups (e.g. mammals, birds, honeybees), and these are aggregated to give a single categorical rating (low, moderate and high; Lewis et al., 2016) per active ingredient. We translated these ratings to numerical scores (low = 1, moderate = 2, high = 3) and as our sites used standard concentrations of agrochemicals, we summed the scores for all active ingredients used to derive an overall '*Ecotoxicity score'* (calculated for each of 21 sites that provided the necessary information). Higher *Ecotoxicity scores* indicated either the use of a higher number of agrochemicals, which could affect more species groups, and/or of agrochemicals with higher environmental toxicity. Recent research has shown that the frequency of agrochemical application is an important driver of biodiversity responses to toxicity (Möth et al., 2021), so to account for this, we additionally created an '*Ecotoxicity frequency score'* by multiplying each active ingredient's *Ecotoxicity score* by its number of annual applications and summed these values (for each of 17 sites that provided the necessary information). Higher *Ecotoxicity frequency scores* indicated high rates of agrochemical applications, which increase the exposure of organisms to the substances and may increase the likelihood of supplying a lethal dose.

We rated vineyard management practices in terms of their potential benefits to or detrimental impacts on biodiversity. We did this using the Conservation Evidence database (ConservationEvidence.com, 2023), which provides effectiveness categories for management practices (such as sowing cover crops) intended to conserve biodiversity. These categories are derived from an expert elicitation process in which a panel of independent experts evaluates the available scientific evidence linked to each action, for effectiveness, certainty and harms. Final scores, following an iterative Delphi process, are converted to one of five effectiveness categories, as described in Sutherland et al. (2021), and these are reported on the website. We translated the Conservation Evidence categories to quantitative scores as follows: '*beneficial'* = 2, *'likely to be beneficial'* = 1, *'trade-offs between benefits & harms'*, '*unknown effectiveness'* = 0, '*unlikely to be beneficial'* = −1, and '*likely to be ineffective or harmful*' = −2. Then, for each site, we summed the scores for all vineyard management practices employed to calculate a '*Practice score'* for each study site. Further details are provided in Appendix E.

We tested for collinearity between our predictors and we found a strong negative correlation between woodland and agricultural habitats cover ($r(21) = -0.876$, t-value = −9.305, p-value<0.001) and strong positive correlations between: total vine area and length of linear wooded features ($r(21) = 0.880$, t-value = 9.342, p-value<0.001), average field size and total vine area ($r(21) = 0.815$, t-value = 20.701, p-value<0.001), total edge density and woodland cover ($r(21) = 0.775$, t-value = 16.482, $p<0.001$) and so, we only included woodland cover and average field size, which we considered more ecologically meaningful predictors, in subsequent landscape-scale analyses. We also found moderate correlations between the *Ecotoxicity score* and woodland cover ($r(20) = -0.417$, t-value = −6.701, $p<0.001$), the *Ecotoxicity frequency score* and woodland cover ($r(16) = -0.637$, t-value = −11.131, $p<0.001$), and the *Practice score* and length of linear water features ($r(20) = 0.538$, t-value = 9.324, $p<0.001$), but retained all variables in models as the correlations were below 0.7, thus unlikely to distort model estimations (Dormann et al., 2013).

2.4. Data analyses

2.4.1. Response variables

For birds, the response variables were: abundance, species richness and Shannon diversity. We calculated these separately for each point count location and survey across the sampling periods (n=222). All observed species were included in analyses, including non-native species such as common pheasant (*Phasianus colchicus*), due to their potential role in contributing to vineyard functions such as pest control or grape damage (anecdotal reports from vineyard managers).

For arthropods, the response variables were total abundance, and for samples containing individuals from more than one order (n=813), we calculated order Shannon diversity. We calculated these for each pitfall trap separately across all sampling locations and periods (n=1679).

2.4.2. Models

We performed a set of general(ized) linear mixed models (GLMMs) to test the relative effects of habitat characteristics and vineyard management on bird and arthropod biodiversity. All analyses were performed in R 4.3.0 (R Core Team, 2021), and all models were fitted using the *spaMM* package (Rousset, 2018). We included a spatial autocorrelation function, called the *Matérn* term, which uses the latitude and longitude of survey locations to calculate distance between every point pair and then calculates similarity between these along a distance gradient, accounting for spatial autocorrelation between survey locations (Rousset and Ferdy, 2014). This term replaces the need for random effects of survey locations (e.g. transects or point counts at each site) to be included in models.

Firstly, we related each of the five response variables described above to certified organic and SWGB-accreditation statuses, to understand whether the overall management regime is indicative of biodiversity. Secondly, to identify the relative importance of semi-natural habitats and finer-scale management predictors on biodiversity, we related each response variable to the habitat and management predictors described above, including an interaction between ground vegetation cover and season to account for vegetation cover varying across the sampling seasons. These models were repeated for predictors calculated at the landscape and local scales. All models also included survey season and year as fixed effects to account for temporal non-independence of samples. We did not run a single model containing all habitat and management predictors, as well as the organic and SWGB-statuses, to avoid model overparameterization (Harrison et al., 2018). If season showed a significant effect, we undertook post-hoc contrast analysis to see which seasons significantly varied from each other, using the *emmeans* package (Lenth et al., 2023).

The normality of residuals was tested using Shapiro-Wilk tests, and Gaussian error distributions were employed in models with normally distributed residuals, which included all bird models. Arthropod data were zero-inflated and therefore we performed hurdle models for these responses, which is a two-step modelling approach consisting of a presence-absence model followed by a truncated model excluding zeros (Potts and Elith, 2006). The presence-absence model was fitted using a binomial error distribution with a *clog* link function. The arthropod abundance truncated model was fitted using a negative binomial error distribution as this could not be normalised through transformation and showed overdispersion with a Poisson distribution, whereas arthropod order Shannon diversity met assumptions of normality and was therefore fitted using a Gaussian distribution. We inspected the distribution of residuals, dispersion and checked for influential points using the *DHARMa* package (Hartig, 2022). For the arthropod abundance model, we removed one outlier, where arthropod abundance was over double the next highest value, and 91% of individuals were ants, likely indicating a nearby nest. Model outputs with the retained outlier are reported in Appendix F.

Finally, given the influence of ground vegetation cover on bird and arthropod diversity (see Sections 3.1. and 3.2), we fitted a Gaussian GLM with ground vegetation cover (logged term) as a response variable, and mowing and cultivation frequency per year, herbicide use, sowing of cover crops or wildflower mixes and average field size as predictors.

We used full models, which we did not seek to simplify (following Shutt et al., 2018) and we interpreted predictor significance based on whether the model estimates with 95% confidence intervals passed zero, and if $p < 0.05$. We used full models as our choice of predictors was hypothesis-driven, and non-significant predictors were retained in models as they still hold some explanatory power and may be important confounder variables (Smith, 2018; Sutherland et al., 2023). All response and predictor variables, model structure, error terms and link functions are summarised in Appendix F.

3. Results

3.1. Birds

Across 222-point count surveys, we recorded 6853 individuals belonging to 61 species, including 15 Red-listed species of conservation concern in the UK (Stanbury et al., 2021; see species list in Appendix G). Bird abundance did not vary significantly between organic and non-organic sites (GLMM: t-value=-0.691, p>0.05), nor between SWGB-accredited and non-SWGB vineyards (t-value=-0.723, p-value>0.05, Fig. 2a). Bird abundance was significantly higher at flowering (Tukey t-value=4.132, p-value<0.001) and harvest (t-value=17.31, p-value<0.001) than at budding, and significantly higher at harvest than flowering (t-value=3.017, p-value<0.001, Fig. 2a). Landscape and local scale semi-natural habitats cover and management did not have a significant effect on bird abundance (Fig. 3).

Bird species richness did not significantly vary between organic and non-organic sites (t-value=-0.751, p-value>0.05), but it was significantly lower at SWGB-accredited than non-SWGB vineyards (t-value=-2.196, p-value=0.036, Fig. 2a). Species richness was significantly higher at harvest than at budding (t-value=2.102, p-value=0.015), but the differences between other seasons were not significant (Tukey p-value>0.05, Fig. 2a). Bird species richness was not significantly affected by landscape or local scale semi-natural habitats cover or by vineyard management predictors (Fig. 3).

Bird Shannon diversity did not significantly vary between organic and non-organic sites (t-value=0.157, p-value>0.05), and it was significantly lower at SWGB-accredited than non-SWGB vineyards (t-

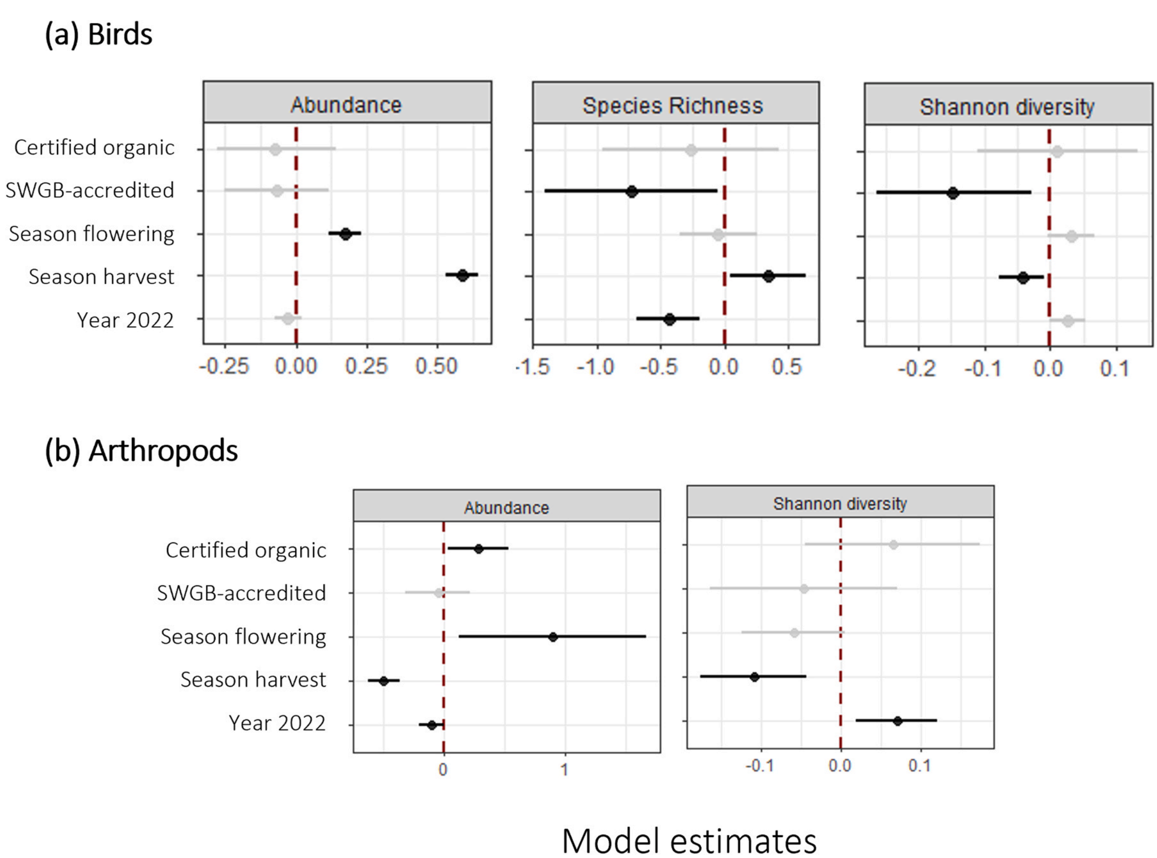

Fig. 2. Estimates and 95% confidence intervals from GLMMs comparing bird **(a)** and arthropod **(b)** response variables between vineyards (n=22) accredited through different schemes. For birds, the modelled response variables were: abundance (marginal R^2=0.42), species richness (R^2=0.16) and Shannon diversity (R^2=0.27); for arthropods, the response variables were: abundance (R^2=0.20) and order Shannon diversity (R^2=0.12).

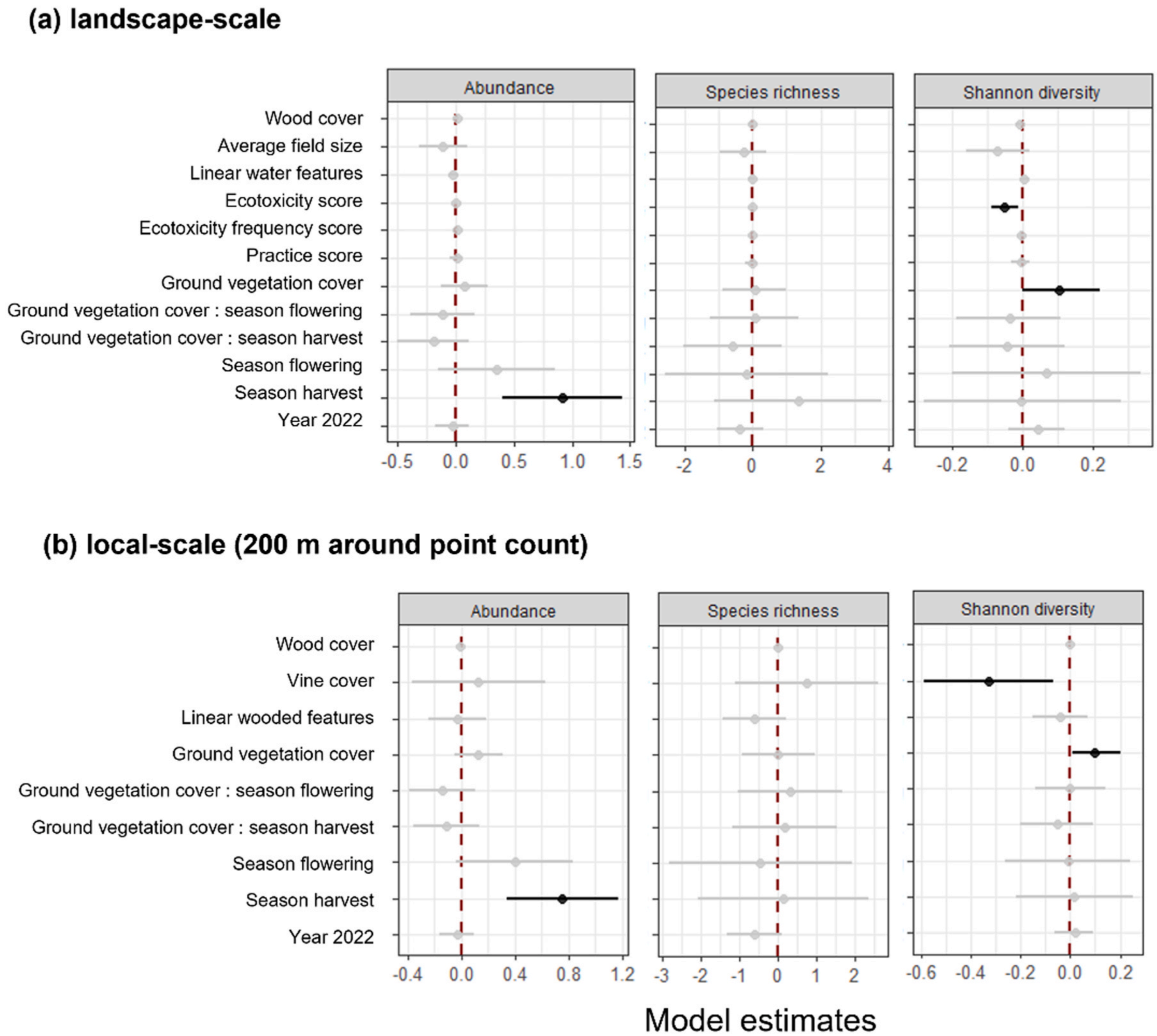

Fig. 3. Estimates and 95% confidence intervals from GLMMs at **(a)** the landscape-scale (2.5 km buffer around each site, n=22), and **(b)** the local-scale (200 m buffer around each point count survey, n=44) for the effects of habitat and management predictors on bird abundance (marginal $R^2 = 0.46$ at landscape scale, $R^2 = 0.27$ at local scale), species richness ($R^2 = 0.35$ and 0.03, respectively), and Shannon diversity ($R^2 = 0.63$ and 0.16, respectively). Estimates in black indicate predictors with supported effects (95% CI do not cross zero, and p<0.05).

value=-2.631, p=0.012, Fig. 2a). Shannon diversity was significantly lower at harvest than at budding (t-value=-2.123, p-value=0.002), but the differences between other seasons were not significant (Tukey p-value>0.05). At both, the landscape (t=2.010, p-value=0.026) and local scales (t=1.866, p-value=0.01), ground vegetation cover had a significant positive effect on Shannon diversity (Fig. 3). Model predicted Shannon diversity was 1.5 and 1.2-fold higher in vineyards with the highest vegetation cover compared to those with the lowest cover at the landscape (Fig. 4a) and local-scales (Fig. 4c), respectively. At the landscape scale, Shannon diversity declined significantly with increasing *Ecotoxicity score* (t=-2.662, p=0.019; Fig. 3a), and the model predicted Shannon diversity was 11% higher in vineyards with the lowest *Ecotoxicity score*, compared to at vineyards with the highest *Ecotoxicity score* (Fig. 4b). Shannon diversity was also negatively and significantly affected by vine cover at the local scale (t-value=-2.559, p-value=0.018, Fig. 3b), and the predicted Shannon diversity was 13% lower at point count locations surrounded by the highest vine cover in the study (85%) compared to the lowest (10%, Fig. 4d).

3.2. Arthropods

We deployed 1679 pitfall traps and arthropods were caught in 59% of the traps (988). The probability of arthropod presence was significantly higher in the flowering (t-value=5.296, p-value<0.001) and harvest seasons (t-value=7.189, p-value<0.001), compared to the budding season, and it was significantly lower in 2022 compared to 2021 (t-value=-5.697, p-value<0.001; Fig. 5a). The probability of arthropod presence decreased away from the field edge (t-value=-3.311, p-value<0.001), whilst the *Practice score* (t-value=5.558, p-value<0.001) and ground vegetation cover (t-value=5.523, p-value<0.001) had a significant positive effect on arthropod presence (Figs. 5a, 6a-c).

Altogether, we caught 8726 individuals belonging to 19 orders, with the most abundant orders being Araneae (n=2155), Coleoptera (n=2045) and Hymenoptera (n=1867, see Appendix G for the list of orders). Arthropod abundance was significantly higher at certified-organic than non-organic sites (GLMM: t-value=2.354, p-value=0.024), and it did not vary significantly between SWGB-accredited and non-SWGB sites (t-value=-0.315, p-value>0.05, Fig. 2b). Arthropod abundance was significantly higher at flowering than at budding (t-value=2.329, p-value<0.001) and harvest (t-value=1.891, p-value=0.021), and the abundance at harvest was also significantly lower than at budding (t-value=-3.211, p-value<0.001; Fig. 2b). The *Practice score* had a significantly positive effect on arthropod abundance (t-value=2.439, p-value=0.015, Fig. 5b), and the predicted abundance was 29% higher at vineyards with the highest *Practice score* compared to

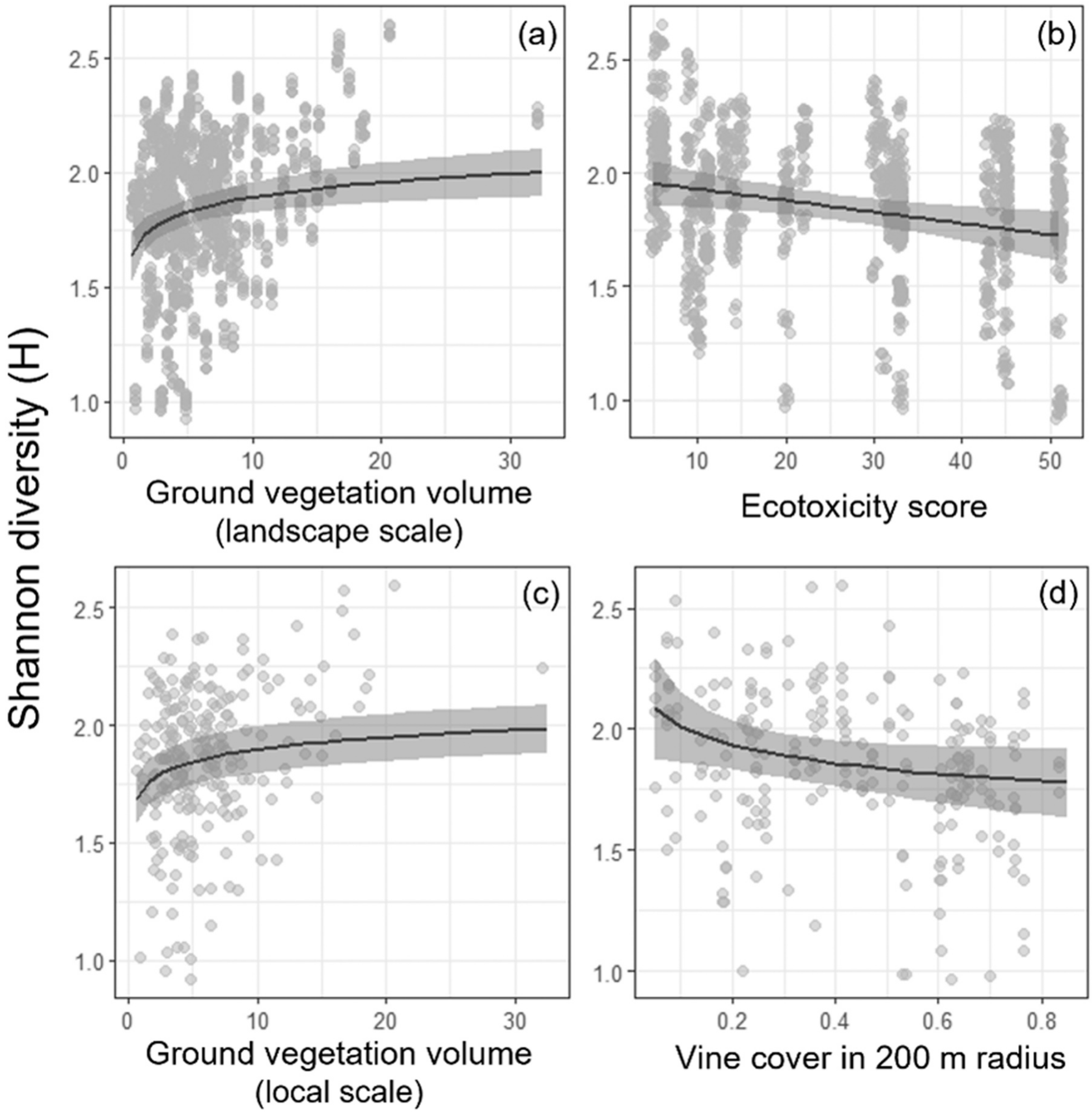

Fig. 4. Raw (circles, n=222) bird Shannon diversity in relation to significant predictors, with model predictions (black line with 95% confidence intervals in grey). Landscape scale refers to 2.5 km buffer around vineyards (n=22, **a-b**), and local scale to 200 m buffer around point count locations (n=44**, c-d)**.

sites with the lowest score (Fig. 6g). Ground vegetation cover had a significantly positive effect on arthropod abundance at both the landscape (t-value=2.217, p-value=0.033) and local scales (t-value=4.717, p-value<0.001, Fig. 5), and the predicted arthropod abundance was 33% higher at the highest vegetation cover compared to the lowest cover (Fig. 6d-e). The *Ecotoxicity score* had a significant negative effect on arthropod abundance (t-value=-2.026, p-value=0.043, Fig. 5b), and the predicted arthropod abundance at vineyards with the lowest *Ecotoxicity score* was 12% higher than in vineyards with the highest *Ecotoxicity score* (Fig. 6f).

Arthropod order Shannon diversity did not differ significantly between certified-organic than non-organic sites (t-value=1.251, p-value>0.05; Fig. 2b), nor between SWGB-accredited and non-SWGB sites (t-value=-0.789, p-value>0.05; Fig. 2b). Shannon diversity of arthropod orders was significantly lower at harvest than at budding (t-value=-3.214, p-value=0.003, Fig. 2b), but it did not vary significantly between the other seasons (Tukey p-value>0.05). Ground vegetation cover had a significantly positive effect on arthropod Shannon diversity at both the landscape (t-value=2.610, p-value<0.001) and local scales (t-value=6.067, p-value<0.001; Fig. 5), and the predicted Shannon diversity was 50% higher at sites with the highest vegetation cover compared to the lowest (Fig. 6h-i). Shannon diversity declined significantly with increasing *Ecotoxicity score* (t-value=-2.415, p-value<0.001), and increasing *Ecotoxicity frequency score* (t-value=-3.987, p-value<0.001, Fig. 5b). The predicted Shannon diversity in vineyards with the lowest *Ecotoxicity and Ecotoxicity frequency scores* was 12% and 38% higher than in vineyards with highest scores*,* respectively (Fig. 6j-k).

3.3. Management differences between vineyards

Ecotoxicity scores were significantly lower for certified-organic (12.33 ± 2.69 SE) than for non-organic vineyards (33.17 ± 2.69; t (19)=-4.758, p<0.001; Fig. 7a), whilst the *Ecotoxicity score* was significantly lower at non-SWGB (17.20 ± 2.78) than SWGB-accredited vineyards (30.64 ± 4.74; t(19)=2.385, p<0.05; Fig. 7b). The *Ecotoxicity frequency scores* did not vary based on vineyard management (certified-organic vs non-organic: t(15)=0.005, p-value>0.05; SWGB-

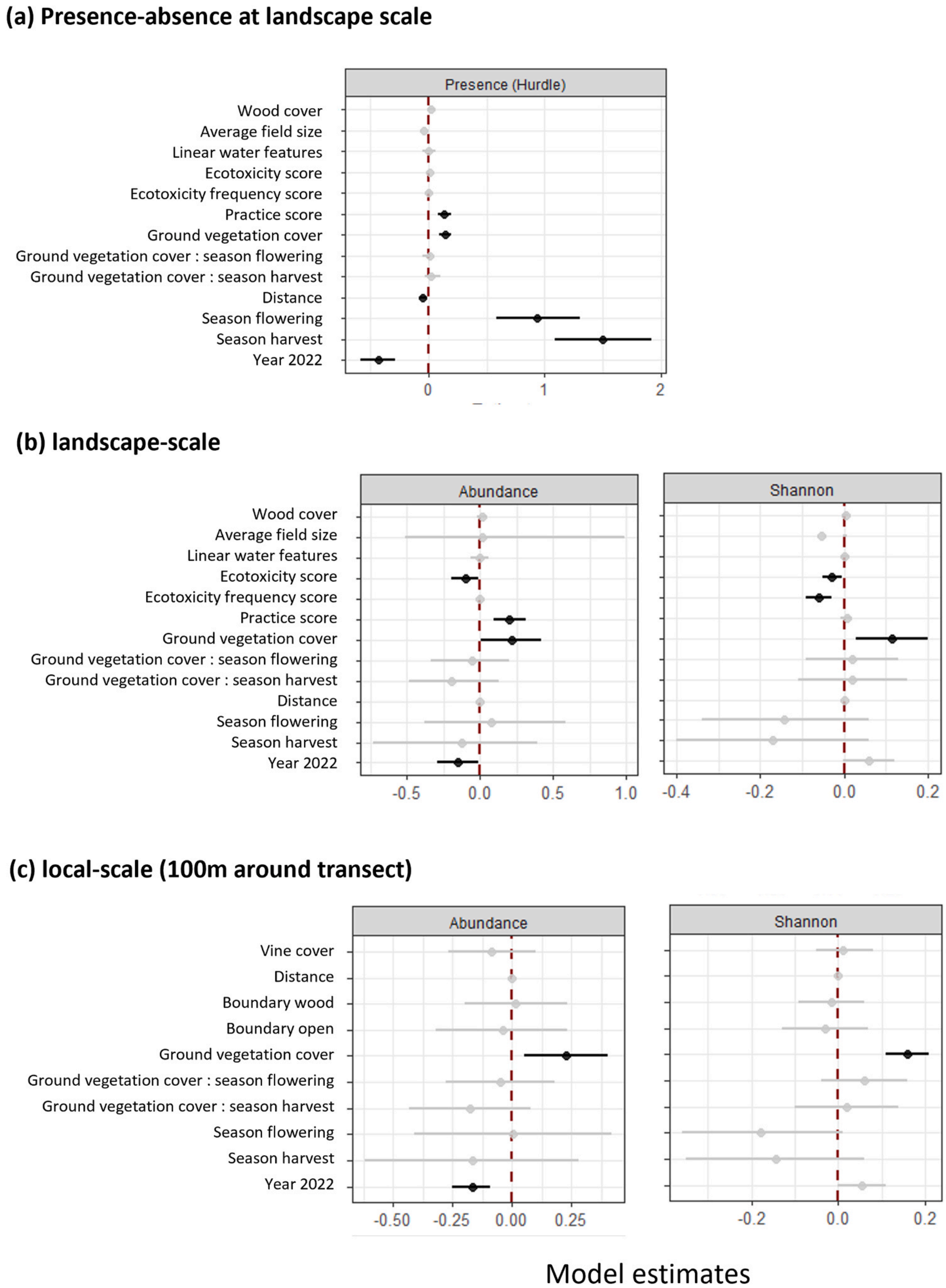

Fig. 5. Estimates and 95% confidence intervals from hurdle GLMMs at **(a, b)** the landscape-scale (2.5 km buffer around each site, n=22), and **(c)** the local-scale (100 m buffer around each transect, n=79) for the effects of habitat and management predictors on arthropod presence (**a,** marginal $R^2 = 0.31$), abundance ($R^2 = 0.56$ at the landscape, $R^2 = 0.18$ at the local-scale) and order Shannon diversity ($R^2 = 0.19$ and $R^2 = 0.15$, respectively; **b, c)**. Estimates in black indicate predictors with supported effects (95% CI do not cross zero, and p<0.05).

accredited vs non-SWGB: t(15)=1.319, p-value>0.05).

The *Practice scores* were significantly higher at non-SWGB than at SWGB-accredited vineyards (0.89 ± 0.82 and −1.25 ± 0.66 respectively; t(19)=2.040, p-value>0.05; Fig. 7d), but did not vary between certified-organic and non-organic vineyards (non-organic: −1.45 ± 0.74; certified-organic: 0.90 ± 0.67; t(19)=1.823, p-value>0.05; Fig. 7c). There were no significant differences between vineyards under different management in mowing (certified-organic vs. non-organic: t (19)=-0.265, p<0.05; SWGB-accredited vs. non-SWGB: t(19)=0.087, p-value>0.05), cultivation frequency (certified-organic vs. non-organic: t (19)=-1.170, p-value>0.05; SWGB-accredited vs. non-SWGB: t(19)= 1.748, p-value>0.05), or overall ground vegetation cover (certified-organic vs. non-organic: t(19)=0.974, p-value>0.05; SWGB-accredited vs. non-SWGB: t(19)=-0.804, p-value>0.05; see Appendix I).

Ground vegetation cover was significantly higher in vineyards that did not use herbicides than those that did (t-value=2.760, p-value=0.015), and it significantly decreased with field size (t-value=-2.784, p-value=0.014; Fig. 8a-c).

Full model results are reported in Appendix H.

4. Discussion

We found vineyard management practices, rather than the surrounding semi-natural habitats, to be the key drivers of differences in bird and arthropod biodiversity across English vineyards. We show that neither organic certification nor a wine industry sustainability

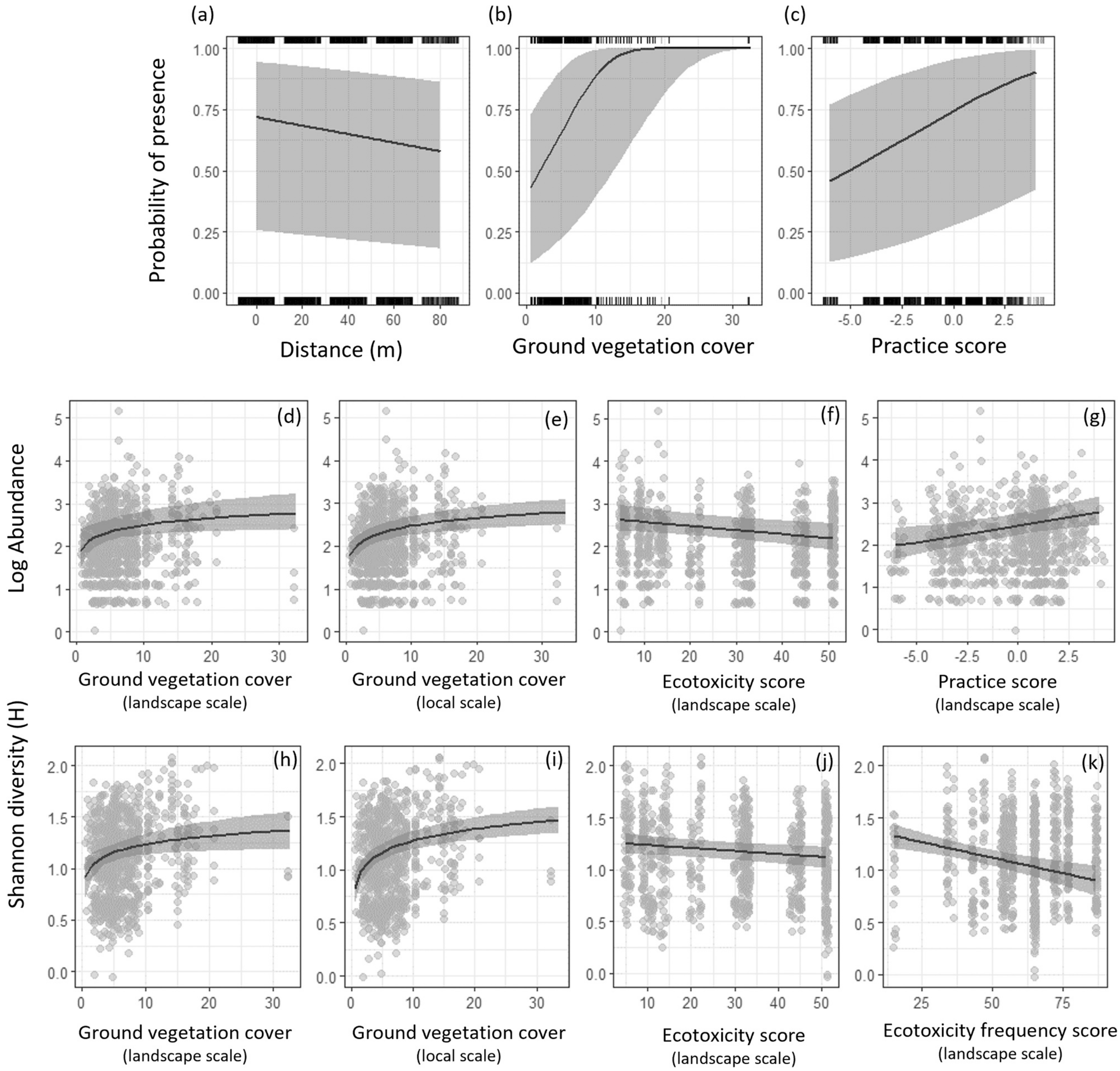

Fig. 6. Raw (circles) arthropod occurrence **(a-c)**, abundance **(d-g)** and order-level Shannon diversity **(h-k)** in relation to significant predictors with model predictions (black line with 95% confidence intervals in grey). Landscape scale refers to 2.5 km buffer around vineyards (n=22), and local scale to 100 m buffer around transects (n=79).

accreditation scheme are currently indicative of higher biodiversity, as the only positive impact on vineyard biodiversity was seen for arthropod abundance in organic vineyards, whilst we found lower bird diversity in SWGB-accredited vineyards. Across Europe, mixed effects of organic viticulture on bird diversity have been noted, as some studies, like ours, failed to detect an effect (Assandri et al., 2016, 2017a), whilst others reported positive effects (Puig-Montserrat et al., 2017; Rollan et al., 2019; Barbaro et al., 2021; Beaumelle et al., 2023). We may have found stronger effects if we included plant diversity in our comparisons, as plants have been shown to benefit from organic viticulture more than mobile organisms (Fuller et al., 2005; Assandri et al., 2016; Ostandie et al., 2021). Furthermore, the benefits resulting from organic, or otherwise sustainable management, may be stronger in landscapes that are more homogenous and intensively managed than those in our study (Tuck et al., 2014; Rollan et al., 2019). The species found across English vineyard landscapes may also be generalists and less sensitive to farming and management practices, as farmland biodiversity across lowland England has been strongly altered since mid-20th Century by agricultural intensification (Robinson and Sutherland, 2002).

SWGB-accredited vineyards had significantly higher *Ecotoxicity scores*, which were negatively related to bird and arthropod diversity, and arthropod abundance. A direct negative effect of insecticide and fungicide use on biodiversity was previously demonstrated across European farmland (Geiger et al., 2010; Rigal et al., 2023). Whilst organic vineyards used fewer chemicals, which resulted in significantly lower *Ecotoxicity scores*, the frequency with which chemicals were applied was higher, leading to the *Ecotoxicity frequency score* not differing significantly between organic and non-organic sites. Across Europe, the application frequency of organic agrochemicals, such as copper and sulfur used to treat fungal diseases, is higher compared to synthetic fungicides due to their lower efficacy (Möth et al., 2021; Reiff et al., 2021). This is an important distinction, as we found the *Ecotoxicity frequency score* to have a stronger negative effect on arthropod diversity than the *Ecotoxicity score* alone. This aligns with similar findings from

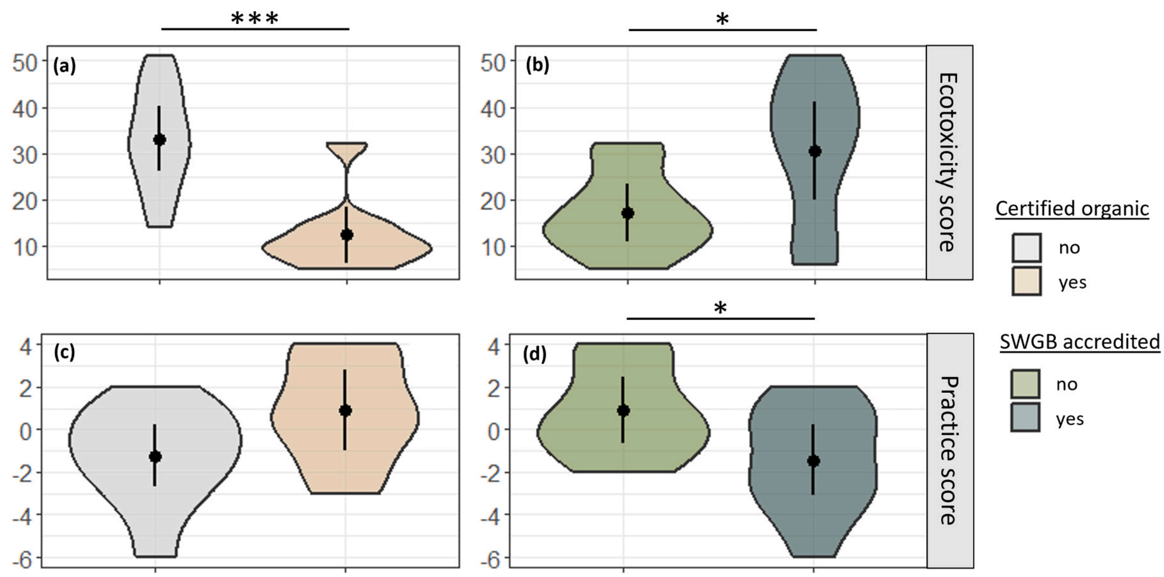

Fig. 7. Comparison of Ecotoxicity (a-b) and *Practice scores* (c-d) between vineyards that were certified organic (n=10/22) and non-certified organic (a and c), as well as those that were Sustainable Wines of Great Britain (SWGB) accredited (n=11/22) and those without the accreditation (b and d). The mean and 95% confidence intervals are indicated in black. Significance of differences was tested with two-sample t-tests and significant results are indicated with asterisks (* p<0.05, *** p<0.001).

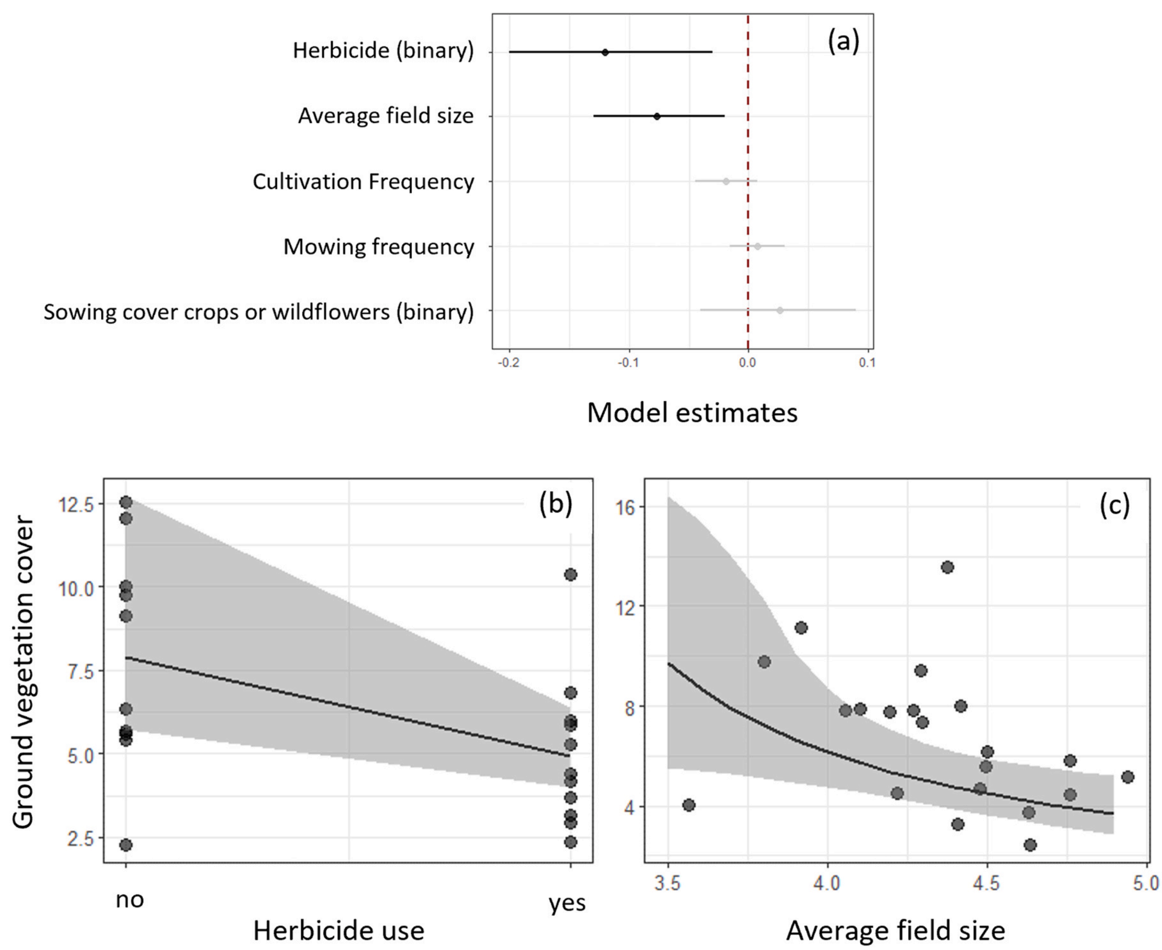

Fig. 8. Estimates and 95% confidence intervals from a GLM relating ground vegetation cover to predictors (marginal R^2=0.76, **a**) across our study vineyards (n=22), and the raw values (circles) and model predictions in black line with 95% confidence intervals in grey for significant predictors **(b,c)**.

European vineyards where, for example, higher rates of pesticide applications in organic vineyards were linked to lower predatory phytoseiid mite densities compared to conventional vineyards (Möth et al., 2021). This means that organic management without efforts to minimise application frequency may not be sufficient to support biodiversity on farms, especially as organically certified agrochemicals such as copper and sulfur have detrimental effects for biodiversity and microbial activity (Karimi et al., 2021). The need for repeated chemical applications may be having additional impacts, with increased levels of disturbance arising from management linked to higher incidence of bird nest abandonment (Assandri et al., 2017b), whilst repeated machinery use may increase soil compaction, especially in vineyards with smaller fields (Clough et al., 2020), which organic vineyards in our study had.

In line with previous vineyard research (Paiola et al., 2020; Winter et al., 2018), we found strong positive effects of ground vegetation cover on biodiversity. This is not surprising as ground vegetation provides shelter and more stable conditions for invertebrates, and food for both invertebrates and birds (Arlettaz et al., 2012; Winter et al., 2018). We found herbicide applications to decrease vegetation cover, whilst increasing a site's *Ecotoxicity score*, and detrimental effects of herbicide use on biodiversity have been shown in other European vineyards (Nascimbene et al., 2012; Duarte et al., 2014; Winter et al., 2018). Ground vegetation cover was also lower in vineyards with larger fields, which may be related to more intensive management methods, such as the use of heavier machinery and increased ground disturbance and trampling by vineyard workers (Cabodevilla et al., 2021).

Vine cover reduced Shannon diversity of the bird community at the local scale, supporting the observed negative effect of increasing vine cover in other European vineyards (Pithon et al., 2016; Rösch et al., 2023). This is likely because alternative surrounding habitats, such as hedgerows and woodland patches that are abundant across English vineyards, provide important habitats for birds (Rösch et al., 2023). These habitats may also be important for supporting arthropod communities, as arthropod presence decreased away from field edges. However, contrary to previous findings (Paiola et al., 2020; Barbaro et al., 2021; Rösch et al., 2023) and our predictions, we found no other effects of surrounding semi-natural habitat area on vineyard biodiversity. This could be because we measured these effects at larger spatial scales than those considered by other studies (e.g. Rösch et al., 2023), or that the species inhabiting English vineyards may be generalists and well-adapted to agricultural conditions, as shown in previous research for birds (Robinson and Sutherland, 2002) and arthropods (Geldenhuys et al., 2022), and so, may be less reliant on the surrounding habitats. Alternatively, responses to semi-natural habitats could vary between organism groups or taxa (Beaumelle et al., 2023).

A caveat to our findings is the coarseness of the ecotoxicity measurements used to calculate our ecotoxicity scores, as well as of only considering arthropod biodiversity at the order level. Firstly, the response of organisms to agrochemicals depends on environmental factors beyond those tested in laboratory studies on which ecotoxicity measures are based (Niederlehner et al., 1990; Chapman et al., 1998), whilst the synergistic and antagonistic interactions of co-applied agrochemicals are largely unknown (Hernández et al., 2017). Secondly, earlier research has found varying responses to semi-natural habitats and management of different arthropod groups in vineyards. For example, whilst overall arthropod diversity responded positively to organic management in Italian and French vineyards, there were significant differences in responses between arthropod orders (Beaumelle et al., 2023), with organic farming benefitting arachnids but not carabids, with further differences in the responses between guilds (Caprio et al., 2015). Similarly, vineyard fungicides were found to harm non-target predatory mites more than pest mites (Reiff et al., 2021), and so, a finer taxonomic resolution should be considered in future studies, particularly when focussing on ecosystem functions.

4.1. Synthesis, management and policy implications

Across English vineyards, we found that individual management practices had a stronger influence on vineyard biodiversity than the overall management regime or the surrounding habitats. Farming certifications have a lot of governmental and societal support and their uptake is increasing (European Commission, 2023; Gomes et al., 2023), but our research demonstrates that these may not reliably predict the expected higher biodiversity. It is therefore important to look beyond farming certifications or industry-accreditations; for example, across the UK viticultural industry, non-organic vineyards can be low-input and have reduced mowing and harbour more biodiversity than organic and SWGB-accredited vineyards. We identified key drivers of biodiversity in vineyards, which we use to make management recommendations for supporting biodiversity, and which should be used by those involved in the regulation of organic or 'sustainable' farming. Firstly, the types of agrochemicals used, and the frequency of application should be carefully managed to reduce detrimental impacts. Moving to organic management alone may not achieve this, as agrochemicals permitted within organic certifications, for example copper, can both have high ecotoxicity scores and lower efficacy, thus requiring repeated applications. Instead, vineyard managers should be encouraged to consult the open-access PBPD database (Lewis et al., 2016) to assess the environmental ecotoxicity scores and opt for chemicals with lower scores, a strategy already shown to benefit arthropod diversity in South African vineyards (Geldenhuys et al., 2022). Our research supports the recommendations of other recent studies (Assandri et al., 2017b; Möth et al., 2021; Reiff et al., 2023) that minimising agrochemical application frequency should be a priority for biodiversity-friendly viticulture. The second recommendation is to increase vegetation cover in vineyards, and eliminating herbicide applications should be a priority here, especially as it is associated with high environmental toxicity. Ground vegetation can be diversified and its cover increased by sowing cover crops and wildflowers, which has been shown to effectively increase biodiversity in vineyards, and yield further benefits through enhanced natural pest control (Winter et al., 2018; Brambilla and Gatti, 2022; Griffiths-Lee et al., 2023).

The English viticultural industry has a strong focus on sustainability, evidenced by the wide uptake of the SWGB-accreditation. However, we found SWGB-accredited vineyards to host lower bird diversity, and have higher *Ecotoxicity scores* than non-accredited vineyards. At present the scheme's accreditation does not appear to be indicative of higher biodiversity or the use of management practices expected to be positive for biodiversity. Our results suggest that herbicide use is not compatible with biodiversity-friendly vineyard management, and so we suggest that SWGB could introduce a minimal requirement of no herbicides. Nonetheless, the results presented here should not undermine the value of the scheme, which only started in 2020, as members commit to continual improvement and annual reporting of management across many areas, including biodiversity conservation. However, this also showcases the importance of studying and identifying the drivers of biodiversity across novel agro-ecosystems to enable accreditation schemes to make evidence-based recommendations and set minimal requirements that ensure their objectives are realised.

Declaration of Competing Interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Natalia Barbara Zielonka reports financial support was provided by Biotechnology and Biological Sciences Research Council.

Data availability

Data will be made available on request.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2024.108982.

References

Arlettaz, R., Maurer, M.L., Mosimann-Kampe, P., Nusslé, S., Abadi, F., et al., 2012. New vineyard cultivation practices create patchy ground vegetation, favouring Woodlarks. J. Ornithol. 153, 229–238.
Assandri, G., Bogliani, G., Pedrini, P., Brambilla, M., 2016. Diversity in the monotony? Habitat traits and management practices shape avian communities in intensive vineyards. Agric. Ecosyst. Environ. 223, 250–260.
Assandri, G., Bogliani, G., Pedrini, P., Brambilla, M., 2017a. Assessing common birds' ecological requirements to address nature conservation in permanent crops: Lessons from Italian vineyards. J. Environ. Manag. 191, 145–154.
Assandri, G., Giacomazzo, M., Brambilla, M., Griggio, M., Pedrini, P., 2017b. Nest density, nest-site selection, and breeding success of birds in vineyards: management implications for conservation in a highly intensive farming system. Biol. Conserv. 205, 23–33.
Barbaro, L., Assandri, G., Brambilla, M., Castagneyrol, B., Froidevaux, J., et al., 2021. Organic management and landscape heterogeneity combine to sustain multifunctional bird communities in European vineyards. J. Appl. Ecol. https://doi.org/10.1111/1365-2664.13885.
Beaumelle, L., Giffard, B., Tolle, P., Winter, S., Entling, M.H., et al., 2023. Biodiversity conservation, ecosystem services and organic viticulture: a glass half-full. Agric. Ecosyst. Environ. 351, 108474.
Bibby, C.J., Burgess, N.D., Hill, D.A. & Mustoe, S.H. 2000. Bird census techniques. Academic Press, London.
Boetzl, F.A., Krauss, J., Heinze, J., Hoffmann, H., Juffa, J., et al., 2021. A multitaxa assessment of the effectiveness of agri-environmental schemes for biodiversity management. Proc. Natl. Acad. Sci. 118.
Brambilla, M., Gatti, F., 2022. No more silent (and uncoloured) springs in vineyards? Experimental evidence for positive impact of alternate inter-row management on birds and butterflies. J. Appl. Ecol. 59, 2166–2178.
Brambilla, M., Ilahiane, L., Assandri, G., Ronchi, S., Bogliani, G., 2017. Combining habitat requirements of endemic bird species and other ecosystem services may synergistically enhance conservation efforts. Sci. Total Environ. 586, 206–214.
Brown, G.R., Matthews, I.M., 2016. A review of extensive variation in the design of pitfall traps and a proposal for a standard pitfall trap design for monitoring ground-active arthropod biodiversity. Ecol. Evol. 6, 3953–3964.
Cabodevilla, X., Arroyo, B., Wright, A.D., Salguero, A.J., Mougeot, F., 2021. Vineyard modernization drives changes in bird and mammal occurrence in vineyard plots in dry farmland. Agric. Ecosyst. Environ. 315, 107448.
Caprio, E., Nervo, B., Isaia, M., Allegro, G., Rolando, A., 2015. Organic versus conventional systems in viticulture: comparative effects on spiders and carabids in vineyards and adjacent forests. Agric. Syst. 136, 61–69.
Chapman, P.M., Fairbrother, A., Brown, D., 1998. A critical evaluation of safety (uncertainty) factors for ecological risk assessment. Environ. Toxicol. Chem. 17, 99–108.
Clough, Y., Kirchweger, S., Kantelhardt, J., 2020. Field sizes and the future of farmland biodiversity in European landscapes. Conserv. Lett. 13, e12752.
Dormann, C.F., Elith, J., Bacher, S., Buchmann, C., Carl, G., et al., 2013. Collinearity: a review of methods to deal with it and a simulation study evaluating their performance. Ecography 36, 27–46.
Duarte, J., Farfán, M.A., Fa, J.E., Vargas, J.M., 2014. Soil conservation techniques in vineyards increase passerine diversity and crop use by insectivorous birds. Bird. Study 61, 193–203.
, 2023European Commission Organic action plan https://agriculture.ec.europa.eu/farming/organic-farming/organic-action-plan_en.
Fuller, R. j, Norton, L. r, Feber, R. e, Johnson, P. j, Chamberlain, D. e, et al., 2005. Benefits of organic farming to biodiversity vary among taxa. Biol. Lett. 1, 431–434.
Geiger, F., Bengtsson, J., Berendse, F., Weisser, W.W., Emmerson, M., et al., 2010. Persistent negative effects of pesticides on biodiversity and biological control potential on European farmland. Basic Appl. Ecol. 11, 97–105.
Geldenhuys, M., Gaigher, R., Pryke, J.S., Samways, M.J., 2022. Interplay between amount of natural vegetation and effective vineyard management practices benefits local arthropod diversity. Biodivers. Conserv. 31, 3039–3053.
Gillespie, M.A.K., Baude, M., Biesmeijer, J., Boatman, N., Budge, G.E., et al., 2017. A method for the objective selection of landscape-scale study regions and sites at the national level. Methods Ecol. Evol. 8, 1468–1476.
Gomes, S., Lopes, J.M., Nogueira, S., 2023. Willingness to pay more for green products: a critical challenge for Gen Z. J. Clean. Prod. 390, 136092.
Griffiths-Lee, J., Goulson, D., Nicholls, E., 2022. Grape Expectations: A Survey of British Vineyard Land Management Practices From an Environmental Perspective.Journal of Agriculture and Sustainability 15, 2.
Griffiths-Lee, J., Davenport, B., Foster, B., Nicholls, E., Goulson, D., 2023. Sown wildflowers between vines increase beneficial insect abundance and richness in a British vineyard. Agric. For. Entomol. 25, 139–151.
Guyot, C., Arlettaz, R., Korner, P., Jacot, A., 2017. Temporal and spatial scales matter: circannual habitat selection by bird communities in vineyards. PLOS ONE 12, e0170176.
Harrison, X.A., Donaldson, L., Correa-Cano, M.E., Evans, J., Fisher, D.N., et al., 2018. A brief introduction to mixed effects modelling and multi-model inference in ecology. PeerJ 6, e4794.
Hartig, F. 2022. DHARMa: Residual diagnostics for hierarchical (Multi-Level / Mixed) regression models. R package version 0.4.5.
Hendershot, J.N., Smith, J.R., Anderson, C.B., Letten, A.D., Frishkoff, L.O., et al., 2020. Intensive farming drives long-term shifts in avian community composition. Nature 579, 393–396.
Hernández, A.F., Gil, F., Lacasaña, M., 2017. Toxicological interactions of pesticide mixtures: an update. Arch. Toxicol. 91, 3211–3223.
Karimi, B., Masson, V., Guilland, C., Leroy, E., Pellegrinelli, S., et al., 2021. Ecotoxicity of copper input and accumulation for soil biodiversity in vineyards. Environ. Chem. Lett. 19, 2013–2030.
Keping, M., 2023. Kunming-Montreal Global Biodiversity Framework: an important global agenda for biodiversity conservation. Biodivers. Sci. 31, 23133.
van Klink, R., Bowler, D.E., Gongalsky, K.B., Swengel, A.B., Gentile, A., et al., 2020. Meta-analysis reveals declines in terrestrial but increases in freshwater insect abundances. Science 368, 417–420.
Lenth, R.V., Bolker, B., Buerkner, P., Giné-Vázquez, I., Herve, M., et al. 2023. emmeans: Estimated Marginal Means, aka Least-Squares Means.
Lewis, K.A., Tzilivakis, J., Warner, D.J., Green, A., 2016. An international database for pesticide risk assessments and management. Hum. Ecol. Risk Assess. Int. J. 22, 1050–1064.
Marston, C., Rowland, C., S., O'Neil, A., W & Morton, R., D.. 2022. Land Cover Map 2021 (25 m rasterised land parcels, GB).
Martin, E.A., Dainese, M., Clough, Y., Báldi, A., Bommarco, R., et al., 2019. The interplay of landscape composition and configuration: new pathways to manage functional biodiversity and agroecosystem services across Europe. Ecol. Lett. 22, 1083–1094.
Merot, A., Alonso Ugaglia, A., Barbier, J.-M., Del'homme, B., 2019. Diversity of conversion strategies for organic vineyards. Agron. Sustain. Dev. 39, 16.
Möth, S., Walzer, A., Redl, M., Petrović, B., Hoffmann, C., et al., 2021. Unexpected effects of local management and landscape composition on predatory mites and their food resources in vineyards. Insects 12, 180.
Muñoz-Sáez, A., Kitzes, J., Merenlender, A.M., 2020. Bird friendly wine country through diversified vineyards. Conservation Biology 35, 274–284. https://doi.org/10.1111/cobi.13567.
Nascimbene, J., Marini, L., Paoletti, M.G., 2012. Organic farming benefits local plant diversity in vineyard farms located in intensive agricultural landscapes. Environ. Manag. 49, 1054–1060.
Nesbitt, A., Dorling, S., Jones, R., 2019. Climate resilience in the United Kingdom wine production sector: crews-UK. BIO Web Conf. 15, 01011.
Newbold, T., Hudson, L.N., Arnell, A.P., Contu, S., De Palma, A., et al., 2016. Has land use pushed terrestrial biodiversity beyond the planetary boundary? A global assessment. Science 353, 288–291.
Niederlehner, B.R., Pontasch, K.W., Pratt, J.R., Cairns, J., 1990. Field evaluation of predictions of environmental effects from a multispecies-microcosm toxicity test. Arch. Environ. Contam. Toxicol. 19, 62–71.
Ostandie, N., Giffard, B., Bonnard, O., Joubard, B., Richart-Cervera, S., et al., 2021. Multi-community effects of organic and conventional farming practices in vineyards. Sci. Rep. 11, 11979.
Paiola, A., Assandri, G., Brambilla, M., Zottini, M., Pedrini, P., et al., 2020. Exploring the potential of vineyards for biodiversity conservation and delivery of biodiversity-mediated ecosystem services: a global-scale systematic review. Sci. Total Environ. 706, 135839.
Pereira, H.M., Navarro, L.M., Martins, I.S., 2012. Global biodiversity change: the bad, the good, and the unknown. Annu. Rev. Environ. Resour. 37, 25–50.
Pithon, J.A., Beaujouan, V., Daniel, H., Pain, G., Vallet, J., 2016. Are vineyards important habitats for birds at local or landscape scales? Basic Appl. Ecol. 17, 240–251.
Potts, J.M., Elith, J., 2006. Comparing species abundance models. Ecol. Model. 199, 153–163.
Puig-Montserrat, X., Stefanescu, C., Torre, I., Palet, J., Fàbregas, E., et al., 2017. Effects of organic and conventional crop management on vineyard biodiversity. Agric. Ecosyst. Environ. 243, 19–26.
R Core Team. 2021. R: A language and environment for statistical computing.
Reiff, J.M., Ehringer, M., Hoffmann, C., Entling, M.H., 2021. Fungicide reduction favors the control of phytophagous mites under both organic and conventional viticulture. Agric. Ecosyst. Environ. 305, 107172.

Reiff, J.M., Pennington, T., Kolb, S., Theiss, K., Alakina, E., et al., 2023. Consistent benefits of fungicide reduction on arthropod predators and predation rates in viticulture: a five-year experiment. BioControl 68, 495–509.
Rigal, S., Dakos, V., Alonso, H., Auniņš, A., Benkő, Z., et al., 2023. Farmland practices are driving bird population decline across Europe. Proc. Natl. Acad. Sci. 120, e2216573120.
Ritchie, H. & Roser, M. 2013. Land Use.Published online at OurWorldInData.org.
Robinson, R.A., Sutherland, W.J., 2002. Post-war changes in arable farming and biodiversity in Great Britain. J. Appl. Ecol. 39, 157–176.
Rodríguez-San Pedro, A., Rodríguez-Herbach, C., Allendes, J.L., Chaperon, P.N., Beltrán, C.A., et al., 2019. Responses of aerial insectivorous bats to landscape composition and heterogeneity in organic vineyards. Agric., Ecosyst. Environ. 277, 74–82.
Rollan, À., Hernández-Matías, A., Real, J., 2019. Organic farming favours bird communities and their resilience to climate change in mediterranean vineyards. Agric. Ecosyst. Environ. 269, 107–115.
Rösch, V., Hafner, G., Reiff, J.M., Entling, M.H., 2023. Increase in breeding bird abundance and diversity with semi-natural habitat in vineyard landscapes. PLOS ONE 18, e0284254.
Rousset, F. 2018. An introduction to the spaMM package for mixed models.
Rousset, F., Ferdy, J.-B., 2014. Testing environmental and genetic effects in the presence of spatial autocorrelation. Ecography 37, 781–790.
Shutt, J.D., Bolton, M., Cabello, I.B., Burgess, M.D., Phillimore, A.B., 2018. The effects of woodland habitat and biogeography on blue tit cyanistes caeruleus territory occupancy and productivity along a 220 km transect. Ecography 41, 1967–1978.
Smith, G., 2018. Step away from stepwise. J. Big Data 5, 32.
Stanbury, A., Eaton, M., Aebischer, N., Balmer, D., Brown, A., et al. 2021. The fifth review of Birds of Conservation Concern in the United Kingdom, Channel Islands and Isle of Man and second IUCN Red List assessment of extinction risk of birds for Great Britain.
Steel, Z.L., Steel, A.E., Williams, J.N., Viers, J.H., Marquet, P.A., et al., 2017. Patterns of bird diversity and habitat use in mixed vineyard-matorral landscapes of central Chile. Ecol. Indic. 73, 345–357.
Sutherland, C., Hare, D., Johnson, P.J., Linden, D.W., Montgomery, R.A., et al., 2023. Practical advice on variable selection and reporting using akaike information criterion. Proc. R. Soc. B: Biol. Sci. 290, 20231261.
Sutherland, W.J., Dicks, L.V., Petrovan, S.O., Smith, R.K., 2021. What Works in Conservation: 2021. Open Book Publishers.
Tscharntke, T., Clough, Y., Wanger, T.C., Jackson, L., Motzke, I., et al., 2012. Global food security, biodiversity conservation and the future of agricultural intensification. Biol. Conserv. 151, 53–59.
Tuck, S.L., Winqvist, C., Mota, F., Ahnström, J., Turnbull, L.A., et al., 2014. Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta-analysis. J. Appl. Ecol. 51, 746–755.
Venkitasamy, C., Zhao, L., Zhang, R., Pan, Z., 2019. Chapter 6 - Grapes. In: Pan, Z., Zhang, R., Zicari, S. (Eds.), Integrated Processing Technologies for Food and Agricultural By-Products. Academic Press, pp. 133–163.
WineGB. 2022. WineGB Industry Report 2021-2022. https://winegb.co.uk/wp-content/uploads/2022/09/WGB_Industry-Report_2021-2022FINAL.pdf.
Winter, S., Bauer, T., Strauss, P., Kratschmer, S., Paredes, D., et al., 2018. Effects of vegetation management intensity on biodiversity and ecosystem services in vineyards: a meta-analysis. J. Appl. Ecol. 55, 2484–2495.