Ref. Ares(2024)7694179 - 29/10/2024

Framework to define operational biodiversity targets developed, implemented and tested in three contrasting European farming systems

Deliverable D5 (D1.5)

24 October 2024

David Kleijn[1], Jeroen Scheper[1], Flaviu Bodea[2], Cristina Costache[2], José Herrera[3,4], Denis Isidro Medinas[3], Vasco Mendes[3], Silvia Pereira Barreiro[3], Razvan Popa[2], Laszlo Rakosy[2], Vanesa Rivera[3], Kati Häfner[5]

[1]Wageningen University (WU), Wageningen, The Netherlands, [2]Babeș-Bolyai University, Cluj-Napoca, Romania; [3]University of Evora, Evora, Portugal,[4]University of Cadiz, Cadiz, Spain,[5]Leibniz Centre for Agricultural Landscape Research (ZALF), Müncheberg, Germany

Prepared under contract from the European Commission

SHOWCASE

SHOWCASing synergies between agriculture, biodiversity and Ecosystem services to help farmers capitalising on native biodiversity

This project receives funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 862480.

Grant agreement No. 862480
EU Horizon 2020 Research and Innovation action

Project acronym:	**SHOWCASE**
Project full title:	**SHOWCASing synergies between agriculture, biodiversity and Ecosystem services to help farmers capitalising on native biodiversity**
Start of the project:	November 2020
Duration:	60 months
Project coordinator:	Prof. David Kleijn Wageningen University
Deliverable title:	Framework to define operational biodiversity targets developed, implemented and tested in three contrasting European farming systems
Deliverable n°:	D1.5 (D5)
Nature of the deliverable:	Report
Dissemination level:	Public
WP responsible:	WP1
Lead beneficiary:	WU
Citation:	Kleijn, D., Scheper, J., Bodea, F., Costache, C., Herrera, J., Isidro Medinas, D., Pereira Barreiro, S., Popa, R., Rakosy, L., Rivera, V. & Häfner, K. (2024). *Framework to define operational biodiversity targets developed, implemented and tested in three contrasting European farming systems*. Deliverable D1.5 (D5) EU Horizon 2020 SHOWCASE Project, Grant agreement No 862480.
Due date of deliverable:	Month n° 36
Actual submission date:	Month n° 36

Deliverable status:

Version	Status	Date	Author(s)
1.0	Final	14 11 2023	Kleijn, D., Bodea, F., Costache, C., Häfner, K., Herrera, J., Isidro Medinas, D., Pereira Barreiro, S., Popa, R., Rakosy, L., Rivera, V. & Scheper, J.
2.0	Revised	24 10 2024	Kleijn, D., Scheper, J., Bodea, F., Costache, C., Herrera, J., Isidro Medinas, D., Mendes, V., Pereira Barreiro, S., Popa, R., Rakosy, L., Rivera, V. & Häfner, K.

Table of contents

Summary

Policy makers set ever more ambitious conservation targets in attempts to halt biodiversity loss. There is overall agreement on *what* needs to be done to halt biodiversity loss, but it is generally unclear *how* this should be achieved. There is a lack of tailored instruments with which the achievement of conservation targets can be realized, monitored and evaluated. Here we propose a framework for the operationalization of biodiversity targets that is based on a set of easily quantifiable Key Performance Indicators for biodiversity conservation (KPI_bs). These KPI_bs characterize land use and landscape composition for which relationships with biodiversity can be empirically established.

Focussing on farmed landscapes and using plants, bees and spiders as biodiversity indicators, we test this framework in three contrasting European countries and show that the KPI_b 'percentage semi-natural area' consistently predicts farm-level biodiversity in all three countries and farming systems. Additional KPI_bs were less consistently related with farm-level biodiversity, with the exception of the KPI_b 'percentage woody landscape elements' in Romania which showed a positive relation with biodiversity as well. KPI_bs that were strong predictors of farm-level biodiversity all had in common that they were based on the non-productive parts of the farm. Relations between farm-level biodiversity and KPI_bs based on productive land-use types (e.g. crop diversity, field size) were weak or absent.

In a next step we investigated the acceptance of the KPI_bs among farmers via a survey in the three countries using econometric methods. Based on the selected KPI_bs, we designed a hypothetical future business solution to be implemented on the whole farm and asked farmers about their willingness to participate under different scenarios. In all three countries the willingness to accept business models was generally high for KPI_bs that were strongly related to farm-level biodiversity, notably the cover of semi-natural areas, while this was much lower for KPI_bs that did not predict biodiversity well, such as crop diversity, field size and scattered oak trees in olive orchards. By targeting these indicators, society or private organisations interested in fighting biodiversity loss on farmland, would get more value for money. Additionally, the results suggest that more farmers would be willing to participate in business models or payment for ecosystem services schemes that target these indicators which would increase impact.

The general approach outlined in this paper can greatly help with the monitoring and evaluation of the progress made towards national and international conservation targets. A strength is that it quantifies the relationships with landscape variables that can be relatively easily monitored across large spatial scales for example through remote sensing approaches The in-situ establishment of relationships between biodiversity and land-use in combination with landscape characteristics allows for validated predictions of biodiversity change that is based on the contribution of all parts of the landscape, not just the sites with conservation actions.

The framework summarizes the results in a single farm-level biodiversity estimate. It can therefore be used to formulate unequivocal and evidence-based targets for conservation policies. The farm-level biodiversity estimates can additionally be used as a basis to financially incentivize biodiversity conservation by farmers. For example, large companies are increasingly interested in rewarding farmers for biodiversity-friendly management but this hinges on the possibility to reliably quantify and justify the impact of interventions on biodiversity. This study shows that straight-forward and easy to quantify KPI_bs can act as reliable evidence for the biodiversity effects of farmer efforts. They can therefore be the basis for much-needed biodiversity-based business models that provide the economic rationale that many farmers need to integrate biodiversity into farm-management.

1 Introduction

Biodiversity is increasingly valued for the myriad contributions it makes to the demands of a growing world population (Diaz et al., 2019). This is reflected in ever more ambitious conservation targets set by global policy makers, such as having restored at least 30 per cent of degraded ecosystems by 2030 or ensuring sustainable management of areas under agriculture, aquaculture, fisheries and forestry (CBD, 2022). Whether this will be enough to halt biodiversity decline is open to debate. Most of the global Aichi biodiversity targets that were set for 2020 have not been achieved and have therefore failed to bend the curve of biodiversity loss (Diaz et al., 2019). This may in part have been due to the fact that countries were not obligated to report on them (Anonymous, 2020) but also because the targets were often not very realistic and difficult to quantify (Green et al., 2019).

There is overall agreement on *what* needs to be done to halt biodiversity loss, but it is generally unclear *how* this should be achieved. For example, a key conclusion of the recent IPBES global assessment report was that a fundamental, system-wide reorganization across technological, economic, and social factors is needed that makes sustainability the norm rather than the altruistic exception (Diaz et al., 2019). Such a transformative change is difficult to operationalize and translate into concrete actions that can directly be implemented. Even for more straight-forward conservation objectives it is difficult to see how they should be achieved. The EC Biodiversity Strategy for 2030 states that there is an urgent need to bring back at least 10% of agricultural area under high-diversity landscape features like hedges, buffer strips and other semi-natural habitats (EC, 2020). On arable farms in the Netherlands, for example, the average cover of non-productive semi-natural areas such as ditch banks, road verges and woodlots is around 2% (Manhoudt & de Snoo, 2003). How the gap between 2 and 10% high-diversity landscape features should be closed is unclear.

Furthermore, we essentially do not know how much conservation effort is needed to halt biodiversity decline. There is no scientific framework that connects conservation actions on the ground, through their effects on biodiversity, to the formulated policy objectives (Kleijn, Rundlof, Scheper, Smith, & Tscharntke, 2011). Except for some charismatic species (Baker, Freeman, Grice, & Siriwardena, 2012; Donald et al., 2007; Geldmann et al., 2013; Xu et al., 2022)), we are currently incapable of translating local conservation outcomes to biodiversity estimates or trends at higher spatial scales. Most studies that evaluate conservation policies examine responses that relate the number of species or individuals per area to conservation actions such as protected area designation (Wauchope et al., 2022) or the implementation of wildlife-friendly management (Pywell et al., 2012). While this provides valuable information on the local impacts of conservation actions, it misses critical dimensions that are needed to estimate how this contributes to biodiversity at the scale for which conservation targets are set.

The area on which conservation is being implemented, and therefore which part of biodiversity can benefit from it, is generally ignored in studies evaluating conservation impact (Kleijn et al., 2018). The suitability as wildlife habitat of the area outside protected areas is often ignored probably because it is used for other purposes such as food production or housing and therefore considered unsuitable for wild species of plants and animals. However, terrestrial protected areas cover less than 17% of the planet's land surface area (UNEP-WCMC & IUCN, 2021) so that a large part of biodiversity still depends on what is going on outside protected areas. To connect conservation actions to biodiversity levels and trends, studies need to take into account how land-use outside protected areas affects overall biodiversity.

This highlights another important issue. By and large, land-use is determined by pressures originating from economic forces to produce food, feed, materials and goods at the lowest possible costs. Because biodiversity is a public good that lacks monetary value it is poorly captured by markets (Turner & Daily, 2008) which generally results in decisions being made at the detriment of it (Polasky, Nelson, Pennington, & Johnson, 2011). If we want to achieve the conservation objectives we have set for ourselves, we need to avoid situations where socio-economic developments lead to environmental degradation on non-conservation sites that are nullifying the results of efforts on conservation sites (Butchart et al., 2010; Leclère et al., 2020). The conservation achievements of land owners outside protected areas should therefore ideally be connected to their business models (Kleijn et al., 2020) as a form of payment for ecosystem services (Daily et al., 2009; Turner & Daily, 2008). That way the conservation outcomes will be internalized in all land-management decisions.

Here we propose a framework for operationalizing biodiversity targets that is based on a set of easily quantifiable indicators for which evidence-based relationships with biodiversity can be obtained using straight-forward standardized methods (Figure 1). We use an indicator-based framework because in-situ surveying of biodiversity is labourious and time-consuming, and therefore an unrealistic approach to operationalize and evaluate farm- or landscape-level biodiversity targets. Proposals for large-scale monitoring programs to assess the effectiveness of conservation policies have been made before (Geijzendorffer et al., 2016; Pereira & Cooper, 2006) but have so far not resulted in actual monitoring programs. The indicators can function as Key Performance Indicators for biodiversity conservation (KPI_bs) and can be used as a proxy to monitor biodiversity trends at relatively large spatial scales (Figure 1). They can also be used by society and businesses as easily quantifiable indicators for result-based financial

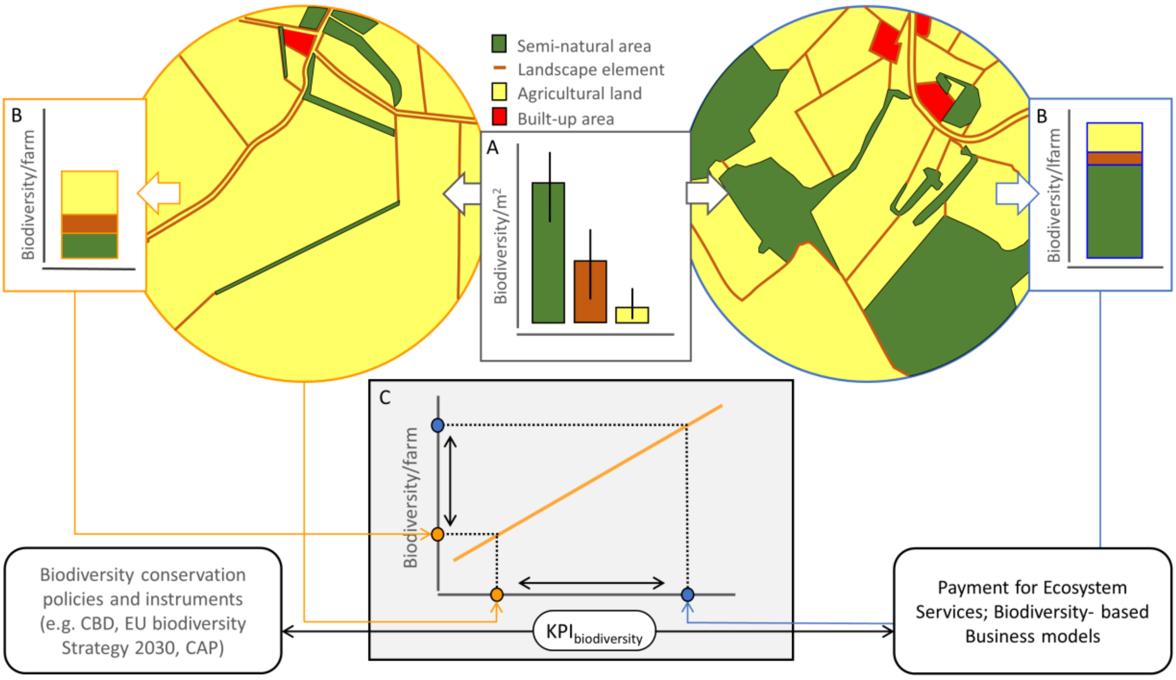

Figure 1: A general framework to connect conservation policy objectives to in-situ biodiversity data and biodiversity-based business models **A** Biodiversity levels per area are empirically established *in-situ* using standard area-based methods in the main habitat or land use types (e.g. Brook et al. 2008, Scheper et al. 2015). **B** Total biodiversity at the farm, landscape or regional level is estimated by extrapolation through multiplying the biodiversity levels per area with the total area of each habitat or land-use type (Kleijn et al. 2018). **C** Analyses reveal which (set of) landscape characteristics best explain biodiversity, which then form the basis for Key Performance Indicators for biodiversity (KPI_b). The relationships between biodiversity and KPI_bs can be used to link changes in land-use due to conservation efforts within the context of inherent economic developments to changes in biodiversity. Because KPI_bs can be easily monitored over large scales this approach allows for large scale monitoring of biodiversity trends.

incentives that reward land owners for enhancing biodiversity on their land (Kleijn et al., 2020). The KPI_bs thus contribute to an integrated approach that links biodiversity to key socio-economic drivers of biodiversity change.

To demonstrate and test the performance of the framework under different conditions we selected three countries with contrasting farming systems: The Netherlands, Portugal and Romania. In each country we selected two types of land use that are known to play important roles in supporting biodiversity: landscape elements such as hedges, field boundaries and isolated trees (Li et al., 2020; Loos et al., 2014; Rossetti et al., 2015) and semi-natural areas such as road verges, extensive grasslands or forest (Habel et al., 2013). We furthermore selected the main form of agricultural land-use, which generally supports very little biodiversity. As biodiversity indicators, we surveyed plants, bees and spiders, based on which we calculated a single biodiversity-index that reflected the multi-trophic components of biodiversity of these three indicators. We surveyed the biodiversity indicators in 342 sites to quantify the relationships between local biodiversity and the type and extent of the three main land-use types. We then used these relationships to predict biodiversity at the farm level based on a number of simple characteristics describing land use on the farm, which can be used as Key Performance Indicators for biodiversity conservation (KPI_bs).

Additionally, we investigated the acceptance of the KPI_bs among farmers via a survey in the three countries using econometric methods. It has been shown that result-based payments for biodiversity improvements have the potential to be more widely accepted by farmers than action-based (prescriptive) approaches, because they allow farmers to use their knowledge and experience, give them the freedom to adapt the practices to their farm management, and also because of perceived lower administrative costs (Pe'er et al., 2022; Šumrada, Japelj, Verbič, & Erjavec, 2022). Based on the selected KPI_bs, we designed a hypothetical future business solution to be implemented on the whole farm and asked farmers about their willingness to participate under different scenarios/conditions.

2 Methods

2.1 Study areas

2.1.1 The Netherlands

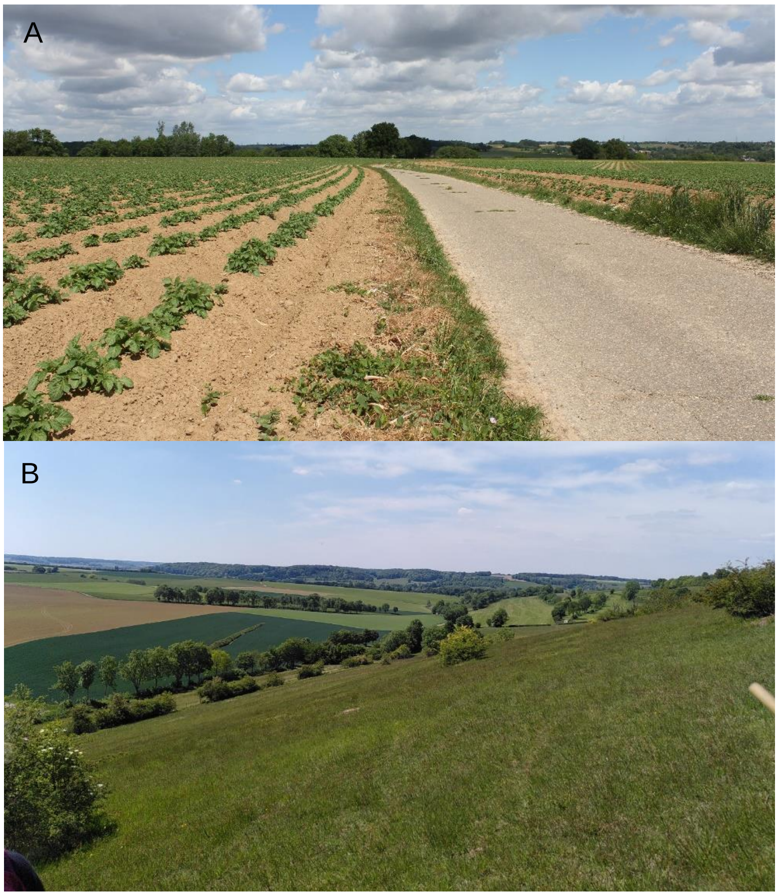

Figure 2: An illustration of the landscape composition of the Dutch study area in Zuid-Limburg which combines (A) areas with high-intensity arable farming and (B) areas with a high proportion of Natura2000 calcareous grasslands.

The Dutch study area is characterized by mixed farming, with a combination of intensively managed arable land (mainly cereals, potatoes, sugar beets and maize) and grasslands for dairy farming, orchards (mostly apple and pear) and protected areas. The study area contains one of the largest Natura2000 areas of the Netherlands, which targets amongst other calcareous grasslands (Fig 2B). This means that for Dutch standards the proportion of protected areas is relatively high. The area has been farmed very intensively for decades. Currently, the main threats for ongoing biodiversity loss related to farm management are the steady disappearance of (linear) landscape elements, such as field boundaries, hedges and roadside verges (Fig 2A), the poor management of the remaining semi-natural landscape elements, the uniformity of landscape and management (the same few crops being cultivated in the same way over vast areas), and nitrogen deposition and leaching which have indirect negative effects on biodiversity.

2.1.2 Portugal

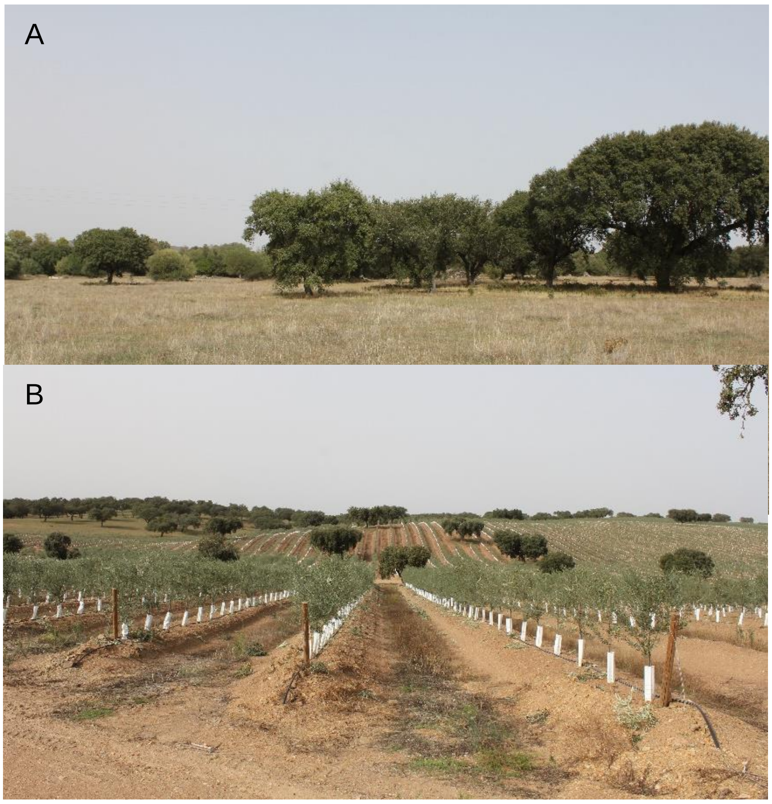

Figure 3: An illustration of the change from (A) the Montado system to (B), intensive short-lived irrigated olive orchards. Note a remnant of the Montado in the back, and the legally protected holm and cork oak trees inside the olive orchard.

In Portugal the study was carried out in the Alentejo region, south of the city of Evora. The traditional farming system here is *Montado*, an agro-silvo-pastoral system with scattered cork and holm oak trees in grazed grasslands and some cereal fields (Fig. 3A). Traditionally, the cork oaks were used for cork production, the acorns were fed to pigs for pork production and the herbaceous vegetation between and underneath trees was used for cattle and sheep grazing. *Montado*'s are high-biodiversity systems. For example, two large Natura2000 *Montado* areas each harbour approximately 100 species of European importance, including the globally threatened Iberian lynx and Spanish imperial eagle. Originally, olive orchards only existed in the direct vicinity of houses and settlements. Such orchards have trees that are often centuries old and are spaced approximately 10m apart in a regular grid. The system is now changing rapidly. To a large extent this has been made possible by the completion in 2002 of the Alqueva dam in the Guadiana river which created the largest reservoir of Western Europe and allowed for the expansion of irrigated agriculture. In specific areas, the edges of *Montado* are now being transformed into intensive olive orchards, and more recently almond orchards. Intensive olive production occurs in rows, with shrub-like plants that are drip-irrigated and do not become much older than 10-15 years (Fig. 3B). Management of intensive olive orchards is virtually all Integrated Pest Management (IPM). Under IPM, farmers are not allowed to cultivate the soil. In between rows there is therefore semi-natural vegetation. Only when farmers are seeding plant species, for example legumes like *Lupinus* spp. or *Trifolium* spp. is it allowed to cultivate the soil. This vegetation flowers early in the growing season but under conventional management is cut before plants set seed. This means that the vegetation is changing in composition with many species disappearing and only species remaining that can cope with these specific conditions. Because it is prohibited by law to cut down holm oak or cork oak trees, many olive orchards still have trees scattered throughout the plantations (Fig. 3B).

2.1.3 Romania

In Romania, the study was carried out in the Natura 2000 area of the hills north-eastern from Cluj (Fig. 4). In this area, high biodiversity is mainly associated with semi-natural areas that comprise species-rich grasslands, which are amongst the most biodiverse in the world. The grasslands require management to maintain their species-richness. The optimal management regimes are (1) mowing with removal of cuttings once a year after mid-July; if necessary a second time around mid-September, or (2) low intensity grazing with sheep, cattle or ideally a combination of sheep and goats at grazing densities of about 0.3-1 Life Stock Units per hectare. The main biodiversity related threats in the area are related to agricultural abandonment leading to shrub and tree encroachment, over-grazing or conversion of grasslands into arable land.

Figure 4: An impression of the type of landscape in the study area with (A) a continuous area of semi-natural grassland with a roaming sheep herd and (B) expanding arable fields surrounded by permanent grasslands that are partly overgrown by shrubs.

2.2 Study design

In each country, we selected 40 study sites that each comprised a sampling location in a semi-natural area, a landscape element and on cultivated land (Figure 5). We selected the three habitat or land-use types that were most widespread and occupied the largest area and that together covered most of the landscape.

The study area in **the Netherlands** is densely populated and intensively used with very little semi-natural habitats outside protected areas. Here we sampled semi-natural areas comprising of road verges (n=17), woodlots (10), forest edges (5) and miscellaneous other habitats (8) that were located in the agricultural matrix but never directly bordered arable fields; field boundaries as the dominant landscape elements which mostly consisted of the permanent vegetation of tracks or lanes (n=26), hedges (9), fences (4) and stream banks (1) that bordered arable fields and; the *arable fields* themselves which were cropped with barley (n=7), maize (8), potato (8), sugar beet (7), winter wheat (9). One of the fields was fallow.

In **Portugal** we sampled semi-natural areas that mostly consisted of *Montado* (n=24), riparian vegetation alongside streams and irrigation channels (9), road verges (3) and extensive grasslands (3); *Landscape elements* that consisted of scattered holm and cork oak trees in olive orchards (n=26) and in *Montado* (n=7). Agricultural land was invariably intensive drip-irrigated *olive orchards* (n=30) that are planted in rows and are generally removed after 10-15 years. All sampled orchards were under Integrated Pest Management (IPM) which in this system, amongst other things, prohibits farmers to cultivating the soil in between the olive rows so that this generally contains semi-natural vegetation.

In **Romania** we sampled semi-natural areas that consisted of extensively managed permanent grasslands. Although these grasslands are farmed, they are generally unfertilized, unfenced and grazed by roaming sheep herds, and acknowledged biodiversity hotspots. We therefore did not consider them agricultural land. Sampling locations were located in grasslands classified as sub-Pannonic steppic grasslands (habitat code 6240; n=4), semi-natural dry grasslands and scrubland facies on calcareous substrates (6210, n=5), grasslands representing a mix of the previous two types (6240_6210, n=10), *Molinia* meadows on calcareous, peaty or clayey-silt-laden soils (6410, n=4), lowland hay meadows (6510, n=11) and unclassified grasslands outside the Natura2000 area (n=6). Additionally we sampled woody landscape elements that comprise patches of shrubs and trees bordering arable fields (n=25) and in permanent grasslands (15). Agricultural land was represented by arable fields that were cultivated with the cereals maize (n=9),wheat (7), barley (3) and oats (3) and the insect-pollinated crops alfalfa (7), sunflower (6), oilseed rape (2), red clover (2) and soybean (1).

Across countries, the average <u>maximum</u> distance between sampling locations within study sites was 335 ± 23.1m (mean±se). The average <u>minimum</u> distance between sampling locations from different study sites was 1311±138.7. In Portugal some of the land use types were not available in the direct vicinity of the other land use types and were therefore not sampled there so that we ended up with a data set containing biodiversity data from 342 sampling locations (NL:120, PT:102, RO: 120).

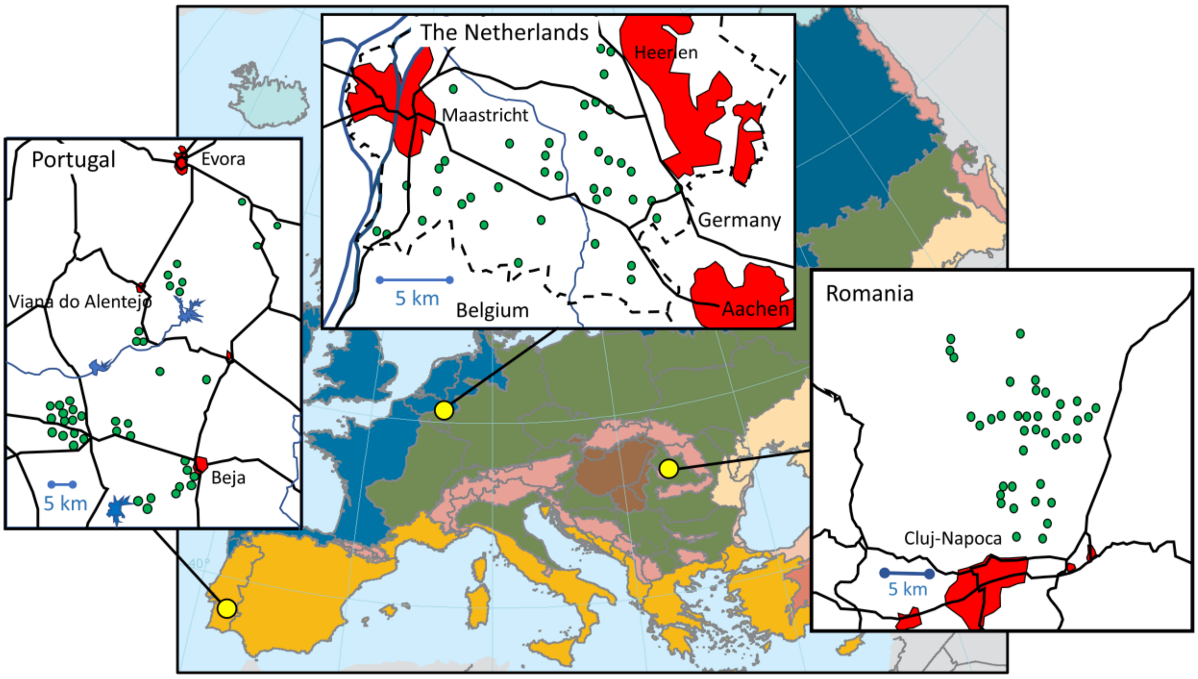

Figure 5: An overview of the locations of the study areas and study sites.

2.3 Biodiversity indicators

We used three species groups representing three different trophic levels as biodiversity indicators: vascular plants, wild bees and spiders. These species groups are diverse and occur throughout the continent, are known to be sensitive to land-use and management changes and can be observed on both productive and non-productive land. Because we were interested in assessing the relationships between biodiversity and the main land-use types and not in obtaining a complete species list per site we maximized the number of sampling locations rather than the number of samples per site. Each species group was therefore sampled once during a period with peak activity. The assumption underlying this approach is that patterns that are found for the subset of biodiversity that is captured in a single visit will be representative for overall biodiversity.

Vascular plants are at the basis of all food webs. Plant diversity or richness is particularly sensitive to specific field management, but also to the presence of pollinators or seed dispersers. Therefore, they are good bioindicators of agricultural management and practices, and they are widely studied and well documented (Herzog et al., 2012; Kleijn et al., 2009). Plants were surveyed once during the season in each sampling location by means of four quadrats of 0.5*0.5 m that were regularly distributed alongside the bee transects (see below) and were therefore typically spaced 10-50 m apart in a representative location of the area. Cover of all vascular plant species in each quadrat was estimated to the nearest percentage. Species without noticeable cover were given a cover of 0.01%.

Wild bees are functionally important as key pollinators of both crops and wild plants (Herbertsson et al., 2021; Rader et al., 2016) but also include many threatened nationally red-listed species. They are therefore relevant for conservation as well as ecosystem service provision (Klein, Boreux, Fornoff, Mupepele, & Pufal, 2018; Rasmont, Devalez, Pauly, Michez, & Radchenko, 2017). Wild bees generally demonstrate consistent relationships with environmental drivers (Hendrickx et al., 2007; Schweiger et al., 2005). Wild bees were surveyed once per season following (J. Scheper et al., 2015) using transect surveys in transects of 150 m^2 during 15 minutes net sampling time (i.e. excluding handling the specimens). Transects were usually 150*1 m, but depending on site dimensions could also be 75*2 m or 50*3 m. Because canopy cover of scattered trees was less than 150 m^2 in the Portuguese study area, the transects were subdivided in three sub-transects of 50 m^2 that were located under three separate trees. Specimens that could not be identified on the wing, were caught and brought to the lab for identification.

Spiders are a species-rich group of predators, with several of them preying on agricultural pest insects and thereby reducing crop damages. Sensitive to farming practices, vegetation composition and structure (Diehl, Mader, Wolters, & Birkhofer, 2013; Schweiger et al., 2005), they are good indicators of management at the plot level. Spiders were sampled by means of suction sampling (Brook, Woodcock, Sinka, & Vanbergen, 2008) using a modified vacuum shredder (e.g. Stihl SH 86-D, Andreas Stihl AG & Co. KG or a local equivalent). Per sampling location, eight rings of 0.357 m internal diameter (equivalent to 0.1 m^2) were placed in the vegetation on both sides of the vegetation quadrats and all spiders were sucked out of the vegetation, pooled, stored in alcohol and brought to the lab for identification.

2.4 Quantification of landscape level land-use

To be able to assess whether the examined land-use types in a study area also indirectly influenced the sampled local abundances of the studied species groups through presence in the surrounding landscape (e.g. (Gabriel, Thies, & Tscharntke, 2005; Mei et al., 2023; Steffan-Dewenter, Munzenberg, Burger, Thies, & Tscharntke, 2002) we determined the cover of the three land-use types (and the number of solitary trees in case of Portugal) in 500m buffers (e.g. (Galle et al., 2022; Steffan-Dewenter et al., 2002; Toivonen, Herzon, & Kuussaari, 2015) around each sampling location. This was done using satellite imagery (Google Earth) and subsequent ground-truthing during field work and quantified using ArcGIS pro Version 2.9.5 (ESRI, 2022) and QGIS (QGIS.org, 2023).

2.5 KPI selection

In each country we selected two or three KPI_bs as potential biodiversity indicators (Table 1) using three selection criteria: (i) there is scientific support for the importance of the selected indicator for biodiversity, (ii) can they be easily quantified based on in-situ site visits or remote sensing techniques (e.g. aerial or satellite photographs; no need for collection of management data which is complicated in many parts of Europe) and (iii) can relationships with biodiversity be empirically established by means of data collected from the three main land-use types.

Table 1: An overview of the selected KPI_bs used per country in the study.

KPI$_b$	**Evidence for links with biodiversity**	**KPI_b quantification**	**Thresholds used for extrapolating effects on biodiversity and farmer acceptance**
The Netherlands			
Percentage semi-natural area	(Cormont et al., 2016; Dainese et al., 2019; Fijen, Scheper, Boekelo, Raemakers, & Kleijn, 2019; Kleijn & van Langevelde, 2006)	Percentage cover of woodlots, road verges, hollow ways, railway embankments and field boundaries.	No restrictions, 4%, 7%
Average arable field size	(Batary et al., 2017; Clough, Kirchweger, & Kantelhardt, 2020; Hass et al., 2018)	The average size of all arable fields that are bordered by permanent non-productive structures supporting semi-natural vegetation and that are for at least 50% of their size in the 500 m buffer.	No restrictions, 5.0, 2.0 ha.
Crop diversity	(Raderschall, Bommarco, Lindstrom, & Lundin, 2021; Redlich, Martin, & Steffan-Dewenter, 2018; Tamburini et al., 2020)	The number of different crops cultivated on agricultural land, including grass leys and orchards.	No restrictions, 5, 6 crops per farm.
Portugal			
Percentage semi-natural area	(Simonson, Allen, Parham, Santos, & Hotham, 2018)	Percentage cover of Montado, semi-natural grasslands, riparian vegetation and Mediterranean forest.	No restrictions, 5%, 15%.
The number of solitary holm or cork oak trees	(Rossetti et al., 2015)	The combined total number of solitary holm oak and cork oak trees.	No restrictions, 5, 10 trees per ha.
Romania			
Percentage semi-natural area	(Petermann & Buzhdygan, 2021)	Percentage cover of natural or extensively managed, mostly grazed grasslands that receive little or no external inputs.	10%, 50%, 100%
Percentage woody landscape elements	(Loos et al., 2014)	Percentage cover of shrubs, thickets, hedgerows, trees, tree lines and woodlots.	0%, 20%, 50%
Average arable field size	(Batary et al., 2017; Clough et al., 2020; Hass et al., 2018)	The average size of all arable fields that are bordered by permanent non-productive structures supporting semi-natural vegetation and that are for at least 50% of their size in the 500 m buffer.	50, 20, 0.1 ha

The KPI_b's 'percentage semi-natural area', 'crop diversity', 'the number of solitary trees' and 'percentage woody landscape elements' were quantified directly from satellite images followed by ground-truthing and based on a buffer of 500 m around a sampling location. For the KPI_b 'field size' in the Netherlands and Romania we assumed an average field boundary width of 1 m and empirically determined the relationships between field size and circumference (NL - circumference (km) = -0.0008*(field size in ha $)^2$ + 0.1209*(field size in ha) + 0.3697, R^2 = 0.94; RO - circumference (m) = -0.0003*(field size in ha$)^2$ + 0.0953*(field size in ha) + 0.5851, R^2 = 0.88) to assess the relative proportion of biodiversity in the land-use types arable field and field boundary with increasing field size.

In consultation with local stakeholders (e.g. farmer cooperation responsible for implementation of agri-environment schemes), we established thresholds for each KPI that were subsequently used to quantify effects on biodiversity and farmer acceptance (Table 1; Table S1).

2.6 Assessing farmer acceptance of KPI$_b$s

For the assessment of the acceptance of the selected KPI$_b$s among farmers we developed a questionnaire. For The Netherlands and Portugal, where access to farmers was relatively easy and response rates were expected to be suitable for this, we used discrete choice experiments (DCEs). DCEs are a survey-based stated preference method commonly used for nonmarket valuation in controlled experimental settings, suitable for e.g. pre-testing new or hypothetical policy instruments (Colen et al., 2016; Schulze et al., 2024).

The theoretical foundations of DCEs are based on Lancaster's theory of demand, which states that "The total utility gained from a product or service is the sum of the individual utilities provided by the attributes of that good" (Lancaster, 1966). Combinations of these attributes and their levels are used to construct alternatives to be evaluated by respondents. A series of choice tasks, each usually containing two or three alternatives, is then presented to participants, who are asked to trade-off and select their preferred option (see example in Fig. 6) (Colen et al., 2016). The advantage of this approach is that each alternative is evaluated as a whole, and the choices can be modelled as a function of the attributes of the alternatives (McFadden, 1974).

Based on the selected KPI$_b$s, we designed a hypothetical future business solution to be implemented on the whole farm and asked farmers about their willingness to participate under different conditions. The choice tasks of the DCE were created with a D-efficient design via DCEtool in R, which is based on Fedorov modified algorithm to generate an optimal design (https://danielpereztr.github.io/posts/DCEtool/). For the design we set priors for the attribute levels and the opt-out option to acknowledge for likely preference patterns (i.e. the higher the restriction, the higher the necessary payment). In that way the attributes levels were combined in a balanced way and in a manner that resulted in realistic choice tasks that were not too easy, nor too difficult to be answered. In total, each respondent was presented 12 choice tasks, consisting of two different contract designs, and an opt-out option (no participation).

(1/12) For which of the following options would you choose to sign a contract for your entire farm?

	Option 1	Option 2	None
Minimum share of semi-natural habitat of the farm	At least 5 %	no prescription	None of these options
Management between rows	Only late cut before harvest	Only late cut before harvest	
Minimum number of solitary trees	no prescription	min. 10 trees/ha	
additional payment per l of olive oil	€1.00	€5.00	
	○	○	○

Figure 6: An example of a choice task from the DCE in Portugal.

For Romania, we used a different method to assess the acceptance of the KPI_bs, because few farmers regularly use internet and access to farmer networks is more difficult. We therefore expected a relatively small survey sample, necessitating a more simple approach. We ran a pricing study based on Van Westendorp's Price Sensitivity Meter (van Westendorp, 1976), a method developed to assess consumer price perceptions and price sensitivities to calculate the acceptable price range. The method uses a set of basic questions which are answered by the respondent referring to a scale of prices. As we are not assessing willingness to pay (WTP) of consumers for a product, but willingness to accept (WTA) to implement KPI_bs, we adapted the questions to our setting in the following way. We first presented the potential business solution based on KPI_bs and described each of the KPI_bs with its potential levels. Then respondents were asked 1) at which payment level they would start thinking about participating in the scheme and complying with the KPI_b level, and 2) at what level of payment they would think that participating would be a good deal for them.

We conducted the survey between March and November 2023 in The Netherlands (NL), mainly in South Limburg, the region where the biodiversity sampling was conducted, in Portugal (P) among farmers that manage olive yards, and in Romania (RO) in two communities, Saschiz and Bunesti. The survey was conducted online in NL and P, while in Romania respondents filled out a paper-based version in a workshop-like environment. The recruitment differed across countries. In NL and P farmers were invited to the online survey via emails (direct, or as members of collectives and farmer networks), which was accompanied by social media campaigns on X (formerly twitter), an informative article in a farmer magazine announcing the survey, personal phone calls, and few personal farm visits. In RO those farmers who were guests of an information workshop on agri-environmental schemes, were asked to participate.

Table 2: Descriptive statistics on respondents farms characteristics.

	The Netherlands *n = 67*	**Portugal** *n = 111*	**Romania** *n = 30*
Average farm size [ha]	82	546	65
Median farm size	65	40	45
Min farm size	1	1	0.3
Max farm size	250	24350	290
Average area with permanent crop [ha]	-	179	-
Median area of permanent crop		25	
Maximum area of permanent crop		7600	
Average share land owned [%]	58	80	40
Median share land owned [%]	56	100	28
Conventional (intensive) [%]	67	8	17
Conventional with low inputs [%]	31	25	33
Organic [%]	12	40	3
Organic in conversion [%]	0	18	40
Farm focus: livestock farming [%]	18	3	47
Farm focus: arable [%]	53	0	0
Farm focus: mixed [%]	21	17	50
Farm focus: permanent crops [%]	0	76	3
Farm focus: other [%]	9	4	0
Full-time [%]	75	67	77
Part-time [%]	25	33	23

Additionally, no farmer with a farm focus on arable farming answered the survey. However, also for Romania the respondents of our survey had comparably large farms with an average of 65 ha (90% of Romanian farms are smaller than 5 ha).

In total, we collected 208 responses on the acceptance of the KPI_bs (Table 2). For The Netherlands we have a rather representative sample of respondents with mainly conventional, arable farmers with an average farm size of 82 ha, which is above the average in The Netherlands (48 ha). In Portugal, as we focus on olive orchards, we surveyed the acceptance of the business model based on KPI_bs only among farmers managing olive orchards. Therefore, the farm focus of our respondents is mainly on permanent crops. We observe a bias towards ecological farms, but still cover a very wide range of farm sizes. The average farm size of our Portuguese respondents is considerably higher than the average of Portugal (14 ha). For Romania we have only a small dataset with a strong bias towards organic farms (in conversion), compared to the average share of organic farming in Romania (around 2 % of farming area).

2.7 Analysis

2.7.1 Calculating the biodiversity index

Responses to land-use and landscape composition may vary greatly among different taxonomic groups (Karp et al., 2018; Martin et al., 2019) which makes it difficult to develop general environmental policies that benefit overall biodiversity. To be able to demonstrate and evaluate the performance of the KPI_b framework for biodiversity as a whole, we combined the three biodiversity indicators (vascular plants, bees, spiders) into an integrated, single biodiversity index which combines both species richness and number of individuals (Fig. 7). Incorporating abundance into biodiversity indicators that assess conservation achievements is important because species richness is constrained by the regional species pool. Consequently, conservation efforts in areas that already have high biodiversity levels tend to have less pronounced effects on species richness than conservation efforts in areas with low biodiversity levels (Tscharntke et al., 2012) even though the effects on the number of individuals may be the same (Hammers, Muskens, van Kats, Teunissen, & Kleijn, 2015). Furthermore, all else being equal, abundance, or population size, is expected to positively correlate with genetic diversity (Leffler et al., 2012), a key component of biodiversity, and is important for the provision of biodiversity-based ecosystem services (Winfree, Fox, Williams, Reilly, & Cariveau, 2015). For bees and spiders, our abundance estimates were expressed as number of individuals per 150 m^2 (the sampling area of the bees) by multiplying the number of spiders with 150/0.8. Since it is impossible to determine the number of plants in the field, plant cover was used as an estimate of the abundance of plants and was likewise expressed per 150 m^2 by multiplying the cover estimates with 150.

In a first step, we constructed for each species group, in each country and land-use type separately, species accumulation curves using the R package “iNEXT” and extrapolated the species richness to the asymptotes (Fig 7A; Fig. S1). Species accumulation curves for wild plants were based on sampling-unit based incidence data (1 m^2 pooled quadrat area); species accumulation curves for wild bees and spiders were based on individual-based abundance data. We considered the asymptotes proxies for the regional species pool size in each land-use type. We then weighted the abundance data of each species group in a land-use type (Fig. 7B) based on the relative size of the regional species pool in that land-use type where the weighting factor of semi-natural area was set at 1. The weighting factor of the landscape elements and agricultural land was then expressed proportional to the regional species pool size in the semi-natural area (e.g. regional species pool size in agricultural land/regional species pool size in semi-natural area; Fig. 7A). Using species-richness weighted abundance means that when a particular land-use type supports a higher density of a species group, the number of species it can support will also represent a larger proportion of the estimated species pool size for that land-use type (i.e. shift right along the species accumulation curve; Fig. 7B)

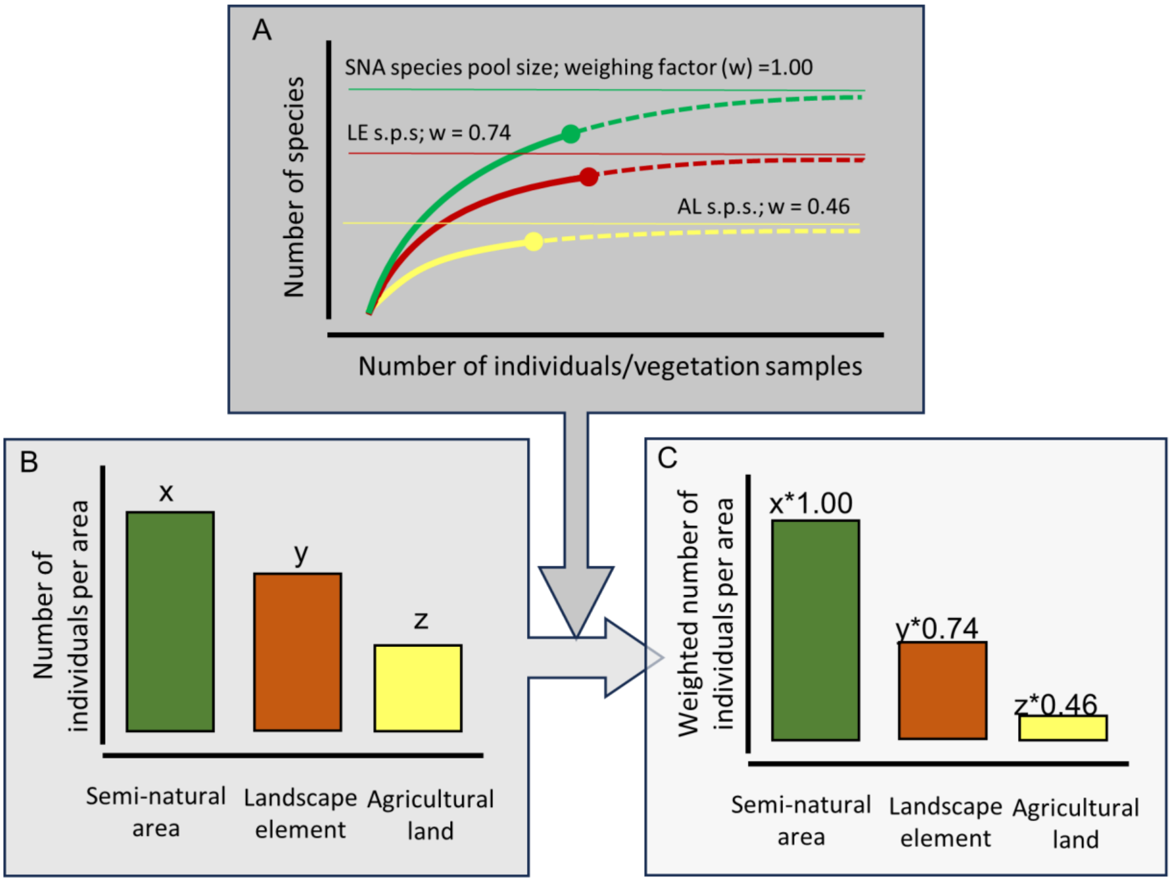

Figure 7: Calculating weighted species abundance estimates that incorporate differences in the regional species pool per land use type. The weighted abundance estimates reflect the fact that in land-use types that can support more species (i.e. have larger regional species pools), species richness increases faster with increasing number of individuals than in land-use types supporting less species. (A) Assessing the regional species pool size based on extrapolated species accumulation curves and determined for each species group per country and land use type (solid lines reflect the empirically assessed range; dashed lines the extrapolations). (B) Assessing the mean number of individuals per area per land use type based on field data. (C) The mean number of individuals per area per land-use type weighted for the number of species each land use type can support.

In a second step, we calculated a biodiversity index for each sampling location by standardizing each weighted biodiversity indicator relative to each country-level mean, and averaging the standardized biodiversity indicators:

$$Biodiversity\ index_i = \left(\frac{wb_i}{wb_m} + \frac{sp_i}{sp_m} + \frac{pl_i}{pl_m}\right) / 3$$

where wb_i is the weighted abundance of wild bees, sp_i is the weighted abundance of spiders and pl_i is the weighted cover of vascular wild plants in each sampling location i, and wb_m, sp_m and pl_m are their respective means across all sampling locations in a particular country. This way each species group contributed equally to the biodiversity index despite the inherent differences in species richness of the three groups. The biodiversity index reflects the relative importance of a sampling location compared to the national mean.

2.7.2 Analysing biodiversity indicators and index

Analyses were performed for each country separately. In a first set of analyses, we used linear mixed models and an information theoretic approach to examine whether and how the local biodiversity index was affected by land-use type and the landscape-level KPI_bs in a buffer of 500 m around the sampling locations. To this end, we constructed a model set consisting of all possible combinations (including an intercept-only model) of the variables land-use type, the KPI_bs, and all two-way interactions between land-use type and the KPI_bs. The KPI_bs included in the analyses were in The Netherlands percentage semi-natural area (log-transformed to reduce positive skew), average arable field size (log-transformed) and crop diversity; in Portugal percentage semi-natural area and the number of solitary holm or cork oak trees (log-transformed) and in Romania percentage permanent grassland, percentage woody landscape elements and average arable field size (log-transformed). The biodiversity index was square-root transformed in all analyses to improve normality and homoscedasticity of residuals. Study site was included as random factor in the analyses to account for non-independence of sampling locations within study sites. Variance inflation factor values indicated no problems with multicollinearity in any of the analyses (all VIFs < 3). We ranked all models in the model set based on their corrected Akaike's Information Criterion (AIC_c), and restricted our candidate model set to models within Δ AICc < 6 (Harrison et al., 2018). For each variable we calculated the variable weight by summing the Akaike weights of all models in the candidate set that include that particular variable. Statistical inference was based on models that included variables with a variable weight > 0.7. To illustrate how the patterns for overall biodiversity were reflecting the patterns for the individual biodiversity indicators, we also analysed whether and how weighted wild bee abundance, spider abundance and plant cover were each affected by land-use type and the landscape-level KPI_bs. These analyses followed the same approach as above. Analyses of the Dutch dataset were performed on square-root transformed weighted plant cover, square-root transformed weighted spider abundance and log-transformed weighted bee abundance; All response variables of the Portuguese dataset were square-root transformed; for the Romanian dataset weighted bee abundance was log-transformed, weighted spider abundance was square-root transformed and weighted plant cover required no transformation to meet model assumptions. All analyses were performed using R version 4.2.1 (Team, 2022).

Next, we used the results obtained in the analyses above to explore how scoring higher on KPI_bs (i.e. different configurations of land-use) translates into overall biodiversity at the farm-level. Based on the best supported model in each country, we first predicted the local biodiversity index values in each land-use type (i.e. Fig. 1a). When relevant, i.e. when local biodiversity was affected by one or more landscape-level KPI_bs, we predicted the local biodiversity at different levels of the KPI_bs in the surrounding landscape. Next, assuming linear relations between area and the weighted abundance estimates of the biodiversity index (Kleijn et al., 2018; Taki, Murao, Mitai, & Yamaura, 2018) we extrapolated the local biodiversity index values to farm-level total biodiversity index values by multiplying the model-estimated local biodiversity values in each land-use type by the total area of the land-use type on the farm, and summing the resulting total values for the different land-use types (i.e. Fig. 1b). In this process, for simplicity, we assumed a contiguous farm with a size of 78.5 ha (i.e. a landscape buffer with a radius of 500 m). We then calculated farm-level biodiversity for different KPI_bs scores (i.e. Fig. 1c), which effectively represent different scenarios of the farm composition. For these biodiversity scenario's we used the range in KPI_b scores that was similar to the KPIb thresholds used in the study assessing farmer acceptance of KPI_bs (Table 1). In the scenarios for the Netherlands, we varied the percentage of semi-natural area between 0 and 10% at average arable field size of 2.5, 5.0 or 7.5 ha (which determines the total area of field boundary at the farm, see Section 2.5: KPI selection). Because the third Dutch $KPI_{b,}$ crop diversity, was not significantly related to the biodiversity index or any of the individual biodiversity indicators, we did not consider this KPI_b in our extrapolations. For Portugal, the cover of semi-natural area was varied between 0 and 20% at <1, 5 or 10 solitary trees per ha. Lastly, for Romania, we let the cover of permanent grassland vary between 10 and 75%, at levels of woody landscape element cover of 10, 35 and 75%

2.7.3 Analysing KPI_b acceptance

The analysis of the DCE is based on the random utility theory of (McFadden, 1974), where an individual's utility depends on a deterministic and random utility component. The overall utility (U_{ij}) that a person *i* derives from a chosen alternative *j* in a given choice set contains a deterministic component (V_{ij}) and a stochastic component (ε_{ij}):

$$U_{ij} = V_{ij} + \varepsilon_{ij}$$

Individuals will choose the alternative that yields the highest utility for them and will choose alternative *j* over any other alternative *k* only if the utility U_{ij} is greater than the utility of any alternative U_{ik}. The deterministic component is a function of *n* attributes (x_1,…, x_n), which describe each alternative, while the parameters β_n represent individual preferences for each attribute.

The deterministic component is estimated as:

$$V_j = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \cdots + \beta_n x_n$$

where β_0 represents the alternative specific constant (ASC) for the opt-out option. The stochastic component ε_{ij} cannot be measured. Therefore, some assumptions must be applied regarding its distribution. The conditional logit model assumes that ε_{ij} is independently and identically distributed (IID) according to a Gumbel extreme value type-I distribution.

The specification of the conditional logit model is given by:

$$P(j|k) = \frac{e^{V_{ij}}}{\sum_{k=1}^{K} e^{V_{ik}}}$$

meaning the probability *P* of choosing alternative *j* over the alternatives *k* is equal to the exponent of the deterministic utility *V* of alternative *j* divided by the sum of the exponents of the deterministic utilities of all *k* alternatives in the choice task. In our model, each attribute level was dummy coded (0 = absent and 1 = present), except for the payment attribute, and the minimum levels of all attributes were included as the reference category.

Because the data of Romania were of medium to low quality, we analysed the data by simple descriptive statistics of the price levels named. All data analysis was performed in the statistical software Stata (StataCorp, version 17). The DCE models were estimated using clogit and lclogitml2 command.

3 Results and discussion

In the Netherlands we observed 151 species of wild plants, 47 species (380 individuals) of wild bees and 124 species (5432 individuals) of spiders. In the other two countries not all specimens could be identified to species level, nevertheless, in Portugal, 236 unique plant taxa, 120 (885 individuals) unique wild bee taxa and 112 (4866 individuals) unique spider taxa were observed. In Romania 369 plant taxa, 88 (671 individuals) wild bee taxa and 150 (2297 individuals) spider taxa were observed.

3.1 Relations between area-based biodiversity estimates and main land-use types

In all three countries, variation in the local biodiversity index was best explained by land-use type alone and there was little support for relationships with landscape-level variables (Fig. 8, Table S2). This may seem surprising since numerous previous studies have found important effects of landscape complexity or composition on local species richness or abundance (Chaplin-Kramer, O'Rourke, Blitzer, & Kremen, 2011; Dainese et al., 2019; Gabriel et al., 2005; Hass et al., 2018; Martin et al., 2019). However, these studies generally sampled a single land-use type (mostly crops) and therefore did not consider the difference in biodiversity levels *between* land-use types. Studies that did consider effects of both land-use type and landscape composition have found stronger relationships with land-use type than with landscape composition (Andersson et al., 2022; Holzschuh et al., 2016; Neumüller, Burger, Krausch, Blüthgen, & Ayasse, 2020; Tobisch et al., 2023). Biodiversity is almost invariably lower on productive land than in semi-natural habitats (Alison et al., 2016), which is perhaps an obvious result but the relative difference in biodiversity levels between productive and non-productive habitats is important to consider when we want to quantify farm- or landscape-scale biodiversity levels. Furthermore, relationships with landscape complexity or composition are generally most pronounced when the response is considered of species groups displaying similar ecological and life history traits (e.g. (Flick, Feagan, & Fahrig, 2012; Gabriel et al., 2005; Hass et al., 2018)). Studies that examine responses of species groups displaying more diverse traits (such as our biodiversity index) generally show less consistent responses (Karp et al., 2018). In line with this, landscape characteristics such as the proportion of semi-natural area or mean field sizes did feature in the best models for some of the individual biodiversity indicators: for weighted plant cover in the Netherlands and for all three weighted biodiversity indicators in Romania (Table S3, S5). However unlike the differences between land-use types, these relationships were not consistent enough across all three biodiversity indicators to feature in the best model explaining the biodiversity index. Agricultural land in particular supported lower biodiversity levels than the other land-use types in all three countries, although the difference with scattered trees in Portugal was not significant (Fig. 8, Table S2). The difference in biodiversity levels between productive and non-productive land-use types seemed considerably larger in the very intensively farmed Dutch landscapes than in the more extensive landscapes in Portugal and Romania. In the Netherlands and Portugal, semi-natural areas supported significantly higher local biodiversity levels than the landscape elements (field boundaries and scattered trees in olive orchards; both directly exposed to agricultural land). In Romania, the opposite pattern was observed with woody landscape elements (mostly located within the semi-natural areas) supporting significantly higher biodiversity levels than semi-natural areas (Fig. 8). Overall, these findings indicate that attempts to enhance local biodiversity levels on productive land, for example through increasing crop diversity, will be ecologically much less effective than efforts to enhance semi-natural areas and landscape elements in between productive lands.

In the Netherlands, the significant differences in the local biodiversity index between semi-natural areas, field boundaries and arable fields (Fig. 8) resulted from weighted plant cover and weighted bee and spider abundance all showing clear and consistent differences between the three land-use types, being highest in semi-natural areas of road verges and woodlots and lowest on arable fields where biodiversity was mainly determined by a small set of spiders (Fig. S2; Table S3). Dutch agriculture is very intensive and arable field are generally devoid of wild plants (Fig S2A, close to zero cover) and, because none of the crops in the arable fields were insect-pollinated, support very few wild bees. Few species can persist under such conditions as indicated by the regional species pool size of arable fields that was significantly lower than those of field boundaries and semi-natural areas (Fig. S1; confidence intervals of arable fields not overlapping the mean of the other two land-use types). The regional species pool size of field boundaries and semi-natural areas were similar but tended to be higher in the latter land-use type. The difference between land-use types in weighted plant cover was influenced by semi-natural area cover in the landscape, but the moderating effect was small compared to the main effect of land-use type (Fig.S2A). In sum, landscape-level biodiversity in Dutch arable landscapes seems to be mainly a function of the area occupied by semi-natural habitats and field boundaries.

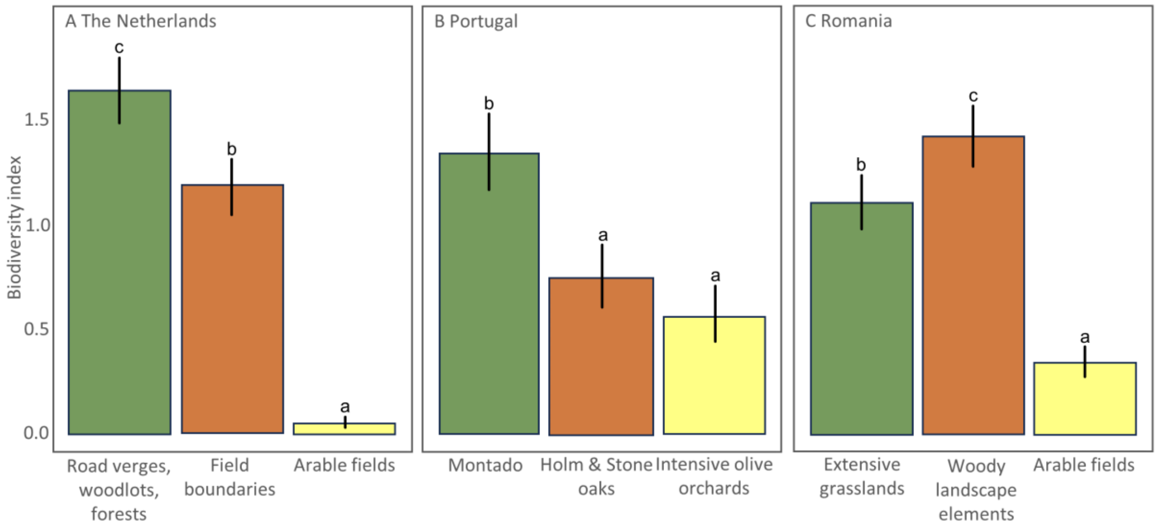

Figure 8: Differences in local biodiversity levels between semi-natural areas (green), landscape elements (brown) and agricultural land (yellow) in farming systems in **A** the Netherlands, **B** Portugal and **C** Romania. Results represent predictions (means ± standard errors) of the models from the candidate set of best models (Table S2) for which there was most support.

In Portugal, differences in biodiversity levels supported by the three land-use types were less pronounced than in the Netherlands, most likely because the examined olive orchards were all part of an Integrated Pest Management (IPM) program which does not allow removal of native vegetation between the crop rows. This probably also explains the relatively large species pools in orchards of plants and spiders (Fig. S1). Because this vegetation is regularly mowed for easy access of farm machinery to the orchards, the vegetation flowers less abundantly (J. Herrera, pers. obs.) than semi-natural areas, which may explain why the species pool of wild bees within the orchards was much smaller. We could only sample the area in between the rows of olive trees, and not the bare soil underneath the olive trees and our results therefore probably overestimate the biodiversity levels of the olive orchards. Nevertheless, the weighted abundance of plants, bees and spiders were all higher in the semi-natural *Montados* (Fig. S3; Table S4) resulting in a significantly higher local biodiversity index compared to the olive orchards and holm and cork oak trees (Fig. 8). Oak trees had biodiversity levels similar to the olive orchards in which they grow and actually had significantly lower plant cover. This may have been the result of shading by the trees although farmers also often use the area underneath the tree canopy to turn or store farm machinery (J. Herrera, pers. obs.) which may negatively affect vegetation cover. This in turn may have offered extra nesting opportunities for wild bees which, in orchards, can often be found nesting in the bare soil at the base of fruit trees (Fountain et al., 2023). Our study indeed found significantly higher weighted abundances of bees underneath the oak trees than in the olive orchard proper.

In Romania, woody landscape elements supported significantly higher biodiversity levels than extensively managed grasslands which in turn supported significantly higher biodiversity levels than arable fields (Fig. 8). Partly in line with previous studies (Bartholomée, Aullo, Becquet, Vannier, & Lavorel, 2020; Bennett & Isaacs, 2014; Schoch, Tschumi, Lutter, Ramseier, & Zingg, 2022), landscape characteristics played an important role in driving all three weighted three biodiversity indicator groups in Romania. However, landscape characteristics showed different and sometimes contrasting relationships with these groups. For example, wild bees and plants were both positively related, while spiders were negatively related to the proportion of extensively managed grasslands within 500m of the sampling locations (Fig. S4; Table S5). As a result, in Romania, as in the other two countries, the model with only land-use type best explained differences in the local biodiversity index (Tables S1, S4). The significantly higher biodiversity levels in woody landscape elements compared to extensively managed grasslands was mainly driven by the response of the spiders (Fig S4E) most likely because spiders benefit from structural complexity of the vegetation (Garratt, 2017; Mestre et al., 2018). The biodiversity levels of woody landscape elements may have been overestimated somewhat. It is impossible to survey inside the woody landscape elements that generally consist of thorny species such as hawthorn (*Crataegus monogyna*) or blackthorn (*Prunus spinosa*). Surveys were therefore done at the edge of the landscape elements where, because the elements were primarily located in the extensively managed grasslands, many grassland species managed to persist. The regional species pool size of plants and bees were indeed very similar in these two habitats, with only spiders having a significantly larger regional species pool in woody landscape features than in extensive grasslands (Fig. S1). The regional species pool of wild bees was surprisingly high in arable fields and similar to those of the other two land-use types. Although being cultivated, the sampled Romanian fields may actually have provided resources to a wide range of species because four of the eight cultivated crops were insect-pollinated and average wild plant cover of the arable fields was relatively high (Fig. S4) indicative of a rich weed flora.

3.2 Extrapolated relations between KPI_b scores and farm-level biodiversity

To illustrate the relationships between KPI_b scores (i.e. easily quantifiable farm characteristics) and farm-level biodiversity we used the outcomes of the analyses on local biodiversity in the three main land-use types to predict farm-level biodiversity (for a virtual circular farm of 78.5 ha). For each country we focused on the KPI_bs for which there was most support of them being related to local biodiversity (Table S2). The predictions are illustrated in Fig. 9 and show that in all three countries the KPI_b's describing the cover of semi-natural areas are strongly related to farm-level biodiversity. In Romania, the KPI_b woody landscape element cover likewise was a strong predictor for farm-level biodiversity levels but in the other countries the KPI_b's describing landscape elements or agricultural land showed weak or no relationships with farm-level biodiversity. For example, in the Netherlands, increasing the cover of semi-natural areas (at a field size of 7.5 ha and a standard four-crop rotation) results in a 31.5% increase in farm-level biodiversity while reducing field size from 7.5 to 5 ha on farms with 2% semi-natural area cover and a four-crop rotation results in a 2.9% increase in biodiversity (Fig. 9A). In Portugal, the weak relations between biodiversity and the KPI_bs that describe management or characteristics of productive land are caused by the much lower biodiversity levels supported by this land-use type (Fig. 8). Agriculture provides suitable habitat conditions and resources for only a small set of wild species (Kleijn et al., 2015; Mei et al., 2023) and, apart from short periods when crops may provide a short flux of massive resources (e.g. mass-flowering crops) most of the species that do occur, do so at relatively low densities. KPI_bs describing landscape elements can be good indicators of farm-level biodiversity in farming systems where these landscape elements support high biodiversity levels, such as in Romania but less so when

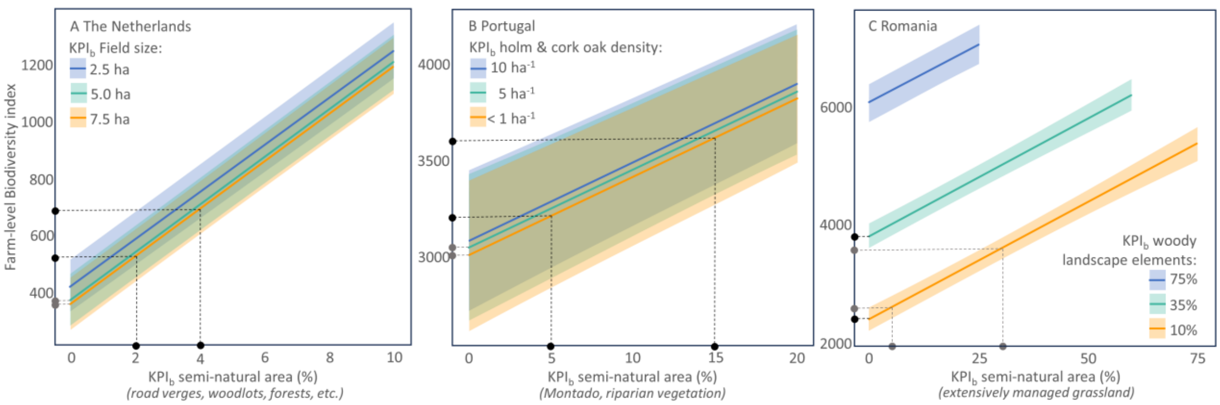

Figure 9: The relationships between the extrapolated farm-level biodiversity index and different combinations of the two Key Performance Indicators for biodiversity (KPI_b) that were most strongly related to local area-based biodiversity estimates in three countries with contrasting farming systems. Farm-level biodiversity estimates were assessed for a hypothetical farm of 78.5 ha (500 m radius) by extrapolating local densities using the most parsimonious models in the best candidate model sets. Dots and dashed lines indicate shifts in the biodiversity index between two KPI_b levels for which farmers were asked their preference in the discrete choice experiment. Black indicates the ecologically most effective option and grey the second best option.

this is not the case, as in Portugal where local biodiversity in holm and cork oaks were as low as the biodiversity levels of the intensive olive orchards in which most of them were located (Fig. 8). The semi-natural areas in all three countries and woody landscape features in Romania all have in common that they are not intensively farmed and not readily exposed to intensive farming practices. KPI_bs that are based on the main form of non-productive land therefore seems to be effective indicators of farm-level biodiversity. The KPI_b field size is partially based on non-productive land-use types as it describes the amount of crop edge or field boundary habitat where local biodiversity levels are generally higher than in the field center (Hass et al., 2018). However, field boundary habitat makes up a very small proportion of arable fields (e.g. in the Netherlands 1.7% of the average 7.1 ha large field) so that at the farm-level reducing field size does not result in meaningful biodiversity increases.

3.3 Acceptance of KPI_bs

3.3.1 The Netherlands

In general, 50 out of the 67 respondents in the Netherlands indicated they would be willing to participate in the presented business solution based on KPI_bs (= 75%). The models analysing farmers' acceptance of the KPI_bs developed in T1.5 show very clear results for the Netherlands (Table 3). The higher the KPI_b levels (which farmers are likely to perceive as restrictions), the more negative their willingness to participate in the business solution based on KPI_bs.

The prescription of a **minimum share of semi-natural area on the farm** has the highest acceptance amongst Dutch farmers. Acceptance of the 4 % level is not even significantly different from the reference level (i.e. no requirement for a minimum percentage of SNA on the farm). At the time of the survey, the regulations of the new EU Common Agricultural Policy still included the greening requirement that arable farmers had to have at least 4% non-productive features and areas on their land. This could explain why the farmers who responded to our survey did not demand any additional compensation for having 4% of their land devoted to semi-natural areas. In the meantime this requirement has been omitted and the response of Dutch farmers may be different if the same survey would have been carried out in 2024. The relatively high acceptance of a prescribed percentage of a semi-natural area may be explained by two factors according to (Alif, Thoyer, Preget, & Šumrada, in prep.). First, SNAs can be established with a comparatively lower effort and/or lower costs than the two other KPI_b. For example, setting one possibly less productive land parcel aside for establishing an extensive meadow comes with low efforts, and comparably low income forgone. Second, if farmers have cattle, the production of hay or low quality feed from semi-natural areas such as extensive meadows can still present a meaningful source of feed, and therefore be of use on the farm. However, other SNAs, such as landscape features can require significantly higher efforts in terms of management. The form, quality and or character of the SNAs to be enhanced was not pre-scribed in our potential business model.

The **restriction of the maximum field size** is the least preferred KPI_b. Reducing field size reduces the work efficiency of farmers and leads to lower profits and higher costs (Tscharntke, Grass, Wanger, Westphal, & Batary, 2021), which could explain why farmers are generally reluctant to reduce field size. More generally, (Kleijn et al., 2019) found that biodiversity-enhancing measures that may interfere with farming practices (e.g. undersown spring cereals, cover crops) are less popular amongst farmers than measures that do not interfere directly with farming (e.g. hedge maintenance, ditch management). Even though there are some agro-ecological benefits of having smaller fields, such as reduced risks of pest outbreaks (Larsen & Noack, 2017) which could lower insecticide costs, farmers apparently perceive that such potential benefits do not compensate for the negative aspects. To overcome the economic-ecological trade-offs farmers face, technological innovation could reduce the costs associated with small field sizes. Cutting-edge technology adapted to small fields could work independently of rectangular shapes, adapting the management configuration to the soil quality, by leaving areas out of cultivation, where productivity is very low locally, or annually (Clough et al., 2020) However, this technology is still developing and regulations are slowing its adoption (Lowenberg-DeBoer et al., 2022), but mainstreaming its use in the future may increase farmers' acceptance of this KPI_b.

Table 3: Conditional logit model results of the Discrete Choice Experiment on the acceptance of KPI_b levels at the farm level in The Netherlands (n=67). Conservation efficiency is estimated as the biodiversity increase divided by the change in farmer acceptance, both relative to the reference situation, and gives an indication of the relative cost-effectiveness of the use of a KPI_b level for biodiversity conservation. The Biodiversity Index reference level was set at 2.1% SNA (Manhoudt & De Snoo 2003), a four crop rotation and 7.5 ha field size. The payment level coefficient multplied by the coefficient of each KPI-level provides the amounts in euro farmers indicated at the time of the survey. SNA: semi-natural areas.

		Acceptance by Farmers				**Conservation efficiency**	
		Coefficient	*SE*	*z*	*P*	*Biodiversity Index*	*Efficiency* (ΔBI/ΔCoeff)
	ASC (opt-out option)	0.48	0.19	2.47	0.014	535	
Min. % of semi-natural area	No prescription (reference)						
	Min. 4 % SNA	-0.23	0.15	-1.54	0.123	692	221
	Min. 7 % SNA	-0.68	0.17	-3.98	0	940	349
No. crops in rotation	No prescription (reference)						
	Min. 5 crops	-0.74	0.14	-5.13	0	535	0
	Min. 6 crops	-0.91	0.16	-5.69	0	535	0
Max. arable field size	No prescription (reference)						
	Max. 5 ha	-0.73	0.14	-5.29	0	550	12
	Max. 2.5 ha	-1.64	0.18	-8.97	0	594	28
Payment [€/ha*a]		0.004	0	8.01	0		

The acceptance of business solutions based on increasing the **number of crops in the rotation** was intermediate to the other two KPI_bs. Interestingly, farmers had a slightly higher acceptance of having 7% semi-natural area on their farm than to add a new crop to the typical Dutch four-crop rotation. The reluctance of adding crops to the rotation can be explained by the barriers to crop diversification, such as access to inputs and knowledge, many uncertainties involved in growing a new crop, including how to cultivate and process it effectively, how to market it, where to sell the product and what price can be expected (Revoyron et al., 2022). Farmers who are motivated by agronomic and market related “better” performance of crops (e.g. margins, sale prices) generally show low or slow crop diversification patterns. In contrast, farmers with a strong or steady increase in the number of crops are motivated by short distribution channels or direct contacts with industries, and have lots of personal (or network related) trials, personal expertise or are supported by downstream actors buying the crops (ibid). These are factors not addressed in our survey.

3.3.2 Portugal

About 91 of the 111 respondents managing olive orchards in Portugal stated they would in general be willing to implement a potential business solution for nature-friendly olive farming based on the KPI_bs (= 82 %).

The results for the acceptance of each of the KPI_bs are a bit ambiguous (Table 4). Only the condition to have at least 10 oak trees per ha on the entire farm significantly lowered the willingness of responding farmers to participate in the business solution based on KPI_bs. The other two KPI_bs, minimum share of semi-natural area of farm and the time of cutting the vegetation between rows, did not significantly influence the willingness to participate. However, based on the confidence intervals, we observe a large variance in the perceptions of the respondents. This could be explained by the existing farm management. Firstly, only 14 farms have less than 5% of SNH on their farm, and 26 farms have between 5 and 15% of semi-natural area. All the others have 15% semi-natural area or more. Also, about half of the respondents, who mow the vegetation between the rows (N=69), do so late in the season just before harvest. About a third do this late in spring, before summer (e.g. to prevent fire). In addition, only 7 respondents reported having no oak trees on their farm,

Table 4: Conditional logit model results of the Discrete Choice Experiment on the acceptance of KPIb levels at the farm level in Portugal (n=111). Conservation efficiency is estimated as the biodiversity increase divided by the change in farmer acceptance, both relative to the reference situation, and gives an indication of the relative cost-effectiveness of the use of a KPIb level for biodiversity conservation. The Biodiversity Index reference level was determined at 0% SNA, less than 1 oak tree per ha and no restrictions regarding mowing of vegetation in between olive rows. The payment level coefficient multplied by the coefficient of each KPI-level provides the amounts in euro farmers indicated at the time of the survey. SNA: semi-natural areas.

		Acceptance by Farmers				**Conservation efficiency**	
		Coefficient	*SE*	*z*	*P*	*Biodiversity Index*	*Efficiency (ΔBI/ΔCoeff)*
	ASC (opt-out option)	0.45	0.13	3.38	0.001	3005	
Min. % of semi-natural area	No prescription (reference)						
	Min. 5 % SNA	0.02	0.1	0.23	0.819	3209	474
	Min. 15 % SNA	0.15	0.1	1.43	0.151	3616	2036
Timing vegetation cutting between olive rows	No prescription (reference)						
	Late cut in the season	0.12	0.08	1.58	0.115	NA	NA
No. of holm and cork oak trees	No prescription (reference)						
	Min. 5 trees/ha	-0.03	0.1	-0.26	0.793	3044	81
	Min. 10 trees/ha	-0.43	0.11	-4.01	0	3083	89
Payment [€/l]		0.27	0.03	8.11	0		

32 had 1 - 5 trees/ha, 22 had 5 - 10 trees/ha, and 35 had even more than 10 trees/ha on their farm. In fact, about half of the survey respondents already meet or are close to meeting the proposed KPI_b levels. Therefore, for the respondents to our survey, the need for additional funding is limited (on average).

The subsequent use of a latent class model allowed us to take into account that there seemed to be structural differences in the group of respondents with associated differences in their preferences, and facilitated the identification of two distinct groups with diverging preferences (Table 5). The first group has a low preference for implementing a KPI_b business solution, based on their overall positive perception of the opt-out option. Two of the KPI_bs are significantly influencing the willingness to participate: the minimum share of SNH and the minimum number of single trees per hectare. The higher the KPI_b levels (perceived as restrictions by farmers), the less willing they are to participate. The second group of respondents has a contrasting preference pattern, with a high preference for participating in the KPI_b business solution. This is based on their overall rejection of the opt-out option, i.e. much more often selecting option 1 or 2 of the choice task, regardless of the compensation payment or KPI_b levels. Only a prescribed number of at least 10 trees/ha significantly influences the willingness to participate. Similarly, other studies have documented considerable variation in willingness to participate. For instance, (Salazar-Ordóñez, Rodríguez-Entrena, & Villanueva, 2021) also identified notable discrepancies in this regard between high-yield and low-yield farms and found that compensation demanded for biodiversity-related management prescriptions was lower for low-yield farmers than for high-

Table 5: Model output of the Latent Class Model with 2 classes applied to the data on the acceptance of KPI_b levels for the entire farm in Portugal (N=111).

		Acceptance by Farmers				Conservation efficiency	
		Coefficient	*SE*	*z*	*P*	*Biodiversity Index*	*Efficiency (ΔBI/ΔCoeff)*
Class 1 (30%)							
	ASC (opt-out option)	1.79	0.49	3.68	0	3005	
Min. % of semi-natural area	No prescription (reference)						
	Min. 5 % SNA	-0.75	0.35	-2.14	0.033	3209	80
	Min. 15 % SNA	-1.33	0.44	-3.05	0.002	3616	196
Timing vegetation cutting between olive rows	No prescription (reference)						
	Late cut in the season	0.04	0.34	0.13	0.899	NA	
No. of holm and cork oak trees	No prescription (reference)						
	Min. 5 trees/ha	-0.86	0.37	-2.35	0.019	3044	15
	Min. 10 trees/ha	-1.95	0.5	-3.92	0	3083	21
Payment		0.24	0.11	2.16	0.031		
Class 2 (70%)							
	ASC (opt-out option)	-1.04	0.19	-5.46	0	3005	
Min. % of semi-natural area	No prescription (reference)						
	Min. 5 % SNH	0.05	0.11	0.47	0.64	3209	187
	Min. 15 % SNH	0.17	0.12	1.43	0.152	3616	505
Timing vegetation cutting between olive rows	No prescription (reference)						
	Late cut in the season	0.03	0.09	0.32	0.753	NA	
No. of holm and cork oak trees	No prescription (reference)						
	Min. 5 trees / ha	0.02	0.12	0.15	0.884	3044	37
	Min. 10 trees / ha	-0.45	0.13	-3.51	0	3083	132
Payment		0.42	0.05	9.17	0		

yield farmers. (Sardaro et al., 2016) likewise found opposing patterns in willingness to accept, with high-profit farms being unwilling to take part in a conservation programme for biodiversity in olive orchards while family farms, especially managed by older farmers, were generally willing to take part in the programme. Such a difference in willingness to participate between farmers that seemingly belong to the same group (e.g. olive growers) suggest that differentiation in compensation payments is required to make biodiversity management attractive to the majority of the farmers. This seriously complicates the development of targeted programmes or business models to integrate biodiversity into farm management because on one hand care must be taken to avoid the free riders while the one hand programmes need to be made appealing to the majority of farmers.

In conclusion, some patterns are stable across the diverse farmers' responses. The KPI_b of single trees in the landscape is perceived as the most influential. Compared to the other farming restrictions, the condition to have at least 10 oak trees per ha is strongly rejected and is the least accepted KPI_b. Additionally, a late cut in the season of the vegetation between rows did not have a significant influence on the acceptance of the business solution, and is rather accepted without additional compensation.

3.3.3 Romania

The analysis of the Romanian dataset must to be treated with caution because the survey was completed by only 30 farmers, a sample size that is generally considered too small to draw reliable conclusions. Furthermore, the survey was completed at a workshop in Romania on paper, which allowed for the filling out of illogical responses. For instance, respondents indicated that they would not participate in a measure, but then provided information on the amount of money they would demand for participating. The interpretation of such contradictory responses is ambiguous. In online versions of the survey, contradictory responses could be prevented through the use of filter questions. Furthermore, not all questions were answered by all respondents and it is unknown why they chose not to answer these questions. These reasons could include a lack of interest in participating in the measure, a lack of understanding of the questions, or a lack of time to answer. Again, such inconsistencies could be prevented in an online version through the use of forced responses or the implementation of reminders, requesting that respondents provide answers to the questions posed. Thirdly, it is not entirely clear, how well the questions were understood. For example, some farmers indicated an arable field size on their farm that was either larger than or equal to their entire farm size.

Twenty of the 30 respondents stated that, in general, they would be willing to implement a new business solution based on KPI_bs, while four respondents stated they would be not willing to participate. However, even these 4 respondents partially filled in some payment levels under which they would be willing to implement some of the KPI_bs. Another six respondents did not answer this question on the general willingness to participate, but stated necessary payments for the implementation of KPI_b levels.

For the analysis, we distinguish between a) where farmers would start to think about entering the scheme, and the payment level, where they would perceive they made a good deal. Keeping the data quality in mind only a few trends can be observed regarding the acceptance of the three investigated KPI_bs and their respective levels. If the KPI_bs would need to be implemented on the entire farm, the KPI_b field size has the overall lowest acceptance among the farmers. There, a high number of respondents answered they would not participate in the scheme, if the maximum field size was prescribed, whatever the payment level would be (Table 6, indicated by “not considering”). In contrast, the KPI_b permanent grassland has the highest acceptance among farmers, probably because most surveyed farmers already met the required KPI_b levels (Table 7). All farmers stated they would be willing meet this requirement for a specific amount of money (even those farmers who just stated before they would not be willing to participate the business solution in general). On average, if the contract requires the farmer to maintain at least 10 % of the land on their farm as permanent grassland, they would be willing to participate for 1643 RON/ha (equals about 329 €/ha). It is noteworthy that the Romanian respondents indicated they needed relatively high payment levels for being willing to meet a requirement that, on average, they indicated most of them already had on their farms (Table 7).

Table 6: Descriptive statistics of payment levels at which Romanian farmers would consider entering the KPI_b business solution, or perceive it as a good deal, and count how many respondents would not consider it, whatever the payment level. NA: not available. Biodiversity Index reference levels set at 0% semi-natural areas, 10% woody landscape elements and arable fields size 5 ha, except for the KPI_b requirement of 100% when other KPI_bs are set at zero.

	Acceptance by Farmers			Conservation efficiency	
KPI_b requirement	*No. responses*	*Mean (€)*	*Range*	*Biodiversity Index*	*Efficiency (ΔBI/€)*
Permanent grassland min. 10 % of farm					
Not considering	0			2373	
Start [€/ha for entire farm]	30	329	20-1000	2773	1.21
Deal [€/ha for entire farm]	29	371	20-900	2773	1.08
Permanent grassland min. 50 % of farm					
Not considering	1			2373	
Start [€/ha for entire farm]	27	492	100-1000	4368	4.05
Deal [€/ha for entire farm]	27	563	120-1400	4368	3.54
Permanent grassland min. 100 % of farm					
Not considering	2			2373	
Start [€/ha for entire farm]	26	708	120-2000	5797	4.84
Deal [€/ha for entire farm]	24	734	120-2000	5797	4.66
Remove all landscape elements on the farm					
Not considering	4			2373	
Start [€/ha for entire farm]	23	730	20-2000	2008	-0.50
Deal [€/ha for entire farm]	23	796	160-2000	2008	-0.46
Landscape elements on 20 % of the land on farm					
Not considering	0			2373	
Start [€/ha for entire farm]	27	616	20-1600	3139	1.24
Deal [€/ha for entire farm]	26	685	40-1800	3139	1.12
Landscape elements on 50 % of the land on farm					
Not considering	3			2373	
Start [€/ha for entire farm]	25	533	20-2000	4835	4.62
Deal [€/ha for entire farm]	24	604	60-2000	4835	4.08
Field size of all the arable fields on farm < 0.1 ha					
Not considering	13			2373	
Start [€/ha for entire farm]	15	399	20-1000	2373	0.00
Deal [€/ha for entire farm]	15	457	20-1000	2373	0.00
Field size of all the arable fields on farm < 20 ha					
Not considering	8			2373	
Start [€/ha for entire farm]	19	238	20-600	2373	0.00
Deal [€/ha for entire farm]	19	307	20-1400	2373	0.00
Field size of all the arable fields on farm < 50 ha					
Not considering	9			2373	
Start [€/ha for entire farm]	18	227	20-600	2373	0.00
Deal [€/ha for entire farm]	18	251	20-600	2373	0.00

Table 7: Descriptive statistics of farm characteristics from Romanian respondents.

Farm characteristic	*No. Responses*	*Mean*	*Range*
Share of permanent grassland on farm [%]	57	26	5-100
Share of land of the farm with landscape features [%]	23	27	3-100
Arable field size [ha]	24	17	0.5-100

3.4 Linking biodiversity effects to farmer acceptance

Linking the estimated changes in the biodiversity index to the changes in farmer acceptance provides an estimate of the relative efficiency with which biodiversity increases can be achieved on farmland through financial incentives that use KPI_b levels as payment criteria. In all three countries, the willingness to accept business models was generally high for KPI_bs that were relatively good indicators of biodiversity such as increasing the cover semi-natural areas like road verges and woodlots in the Netherlands, *Montado* and riparian habitats in Portugal and permanent grasslands in Romania. Farmer acceptance of KPI_bs that performed poorly as biodiversity indicators, such as crop diversity and especially field size (in the Netherlands and Romania) was very low. This resulted in indicators that target the percentage non-productive, semi-natural areas on farms having for example at least an eight-fold higher conservation efficiency than the other KPI_bs in the Netherlands (Table 3) and a five-fold higher conservation efficiency in Portugal (Table 4). By targeting these indicators, society or private organisations interested in fighting biodiversity loss on farmland, would get more value for money. Additionally, the results suggest that more farmers would be willing to participate in business models or payment for ecosystem services schemes that target these indicators.

Our results furthermore suggest that targeting higher KPI_b levels generally results in higher conservation efficiency than lower KPI_b levels. Because we modelled linear relationships between the farm-level biodiversity index and KPI_b levels this must be caused by farmers having relatively higher acceptance levels for higher KPi_b levels once a certain financial threshold has passed that makes them accept the incentive scheme.

4 Conclusions

This study shows that it is possible to obtain reliable estimates of farm-level biodiversity by means of one or two easily quantifiable indicators describing land-use. Our framework considers abundance and richness of contrasting species groups integrally on productive and non-productive land-use types and takes into account their relative cover in the landscape. It therefore presents an integrated assessment of the level of biodiversity that each land use type supports and the area that each land use type occupies. The advantage of this approach when used in monitoring is that it can capture changes over time in the quality of each land use type, for example because of ongoing agricultural intensification. It additionally captures changes in the relative proportion of different land use types (e.g. ongoing landscape simplification).

We used vascular plants, wild bees and spiders as biodiversity indicators and our choice for these groups most likely influenced the outcome. Different species groups are known to respond differently to land use or landscape composition (Mei et al., 2023). Although we are convinced that these three groups represent high-quality indicators of biodiversity in general, we do not suggest that these groups should be applied generally everywhere. This study provides a proof of concept and illustrates an approach that can be adopted flexibly depending on the spatiotemporal context or the objective of the study. For example, species or species groups can be selected that are relevant from a conservation point of view (red-listed species), that are particularly important functionally (pollinators) or that can be linked easily to larger scale and longer-term monitoring efforts (e.g. butterflies, birds).

The approach outlined in this study does not directly consider the effect of management on biodiversity, even though agricultural management is an important driver of farmland biodiversity (Ekroos et al., 2020; Guzman et al., 2024; Kleijn et al., 2009). Collecting detailed management information is complicated and time consuming and therefore almost impossible to achieve across large areas and for many farms. Furthermore, relationships with individual management variables are difficult to interpret because with increasing intensity of farming these variables may change simultaneously (e.g. agro-chemical input, mechanization, field size, crop diversity) making it difficult to disentangle their effects. Our approach bypasses this issue by sampling biodiversity on a representative sample of sites. The biodiversity levels should then reflect the effects of all management that was implemented on these sites so that indirectly management is taken into account.

Our results provide an evidence base for the generally held conviction amongst conservation scientists that semi-natural areas and landscape elements are central for biodiversity in European agricultural landscapes (Pe'er et al., 2022) and suggest that schemes targeting these land-use types are more acceptable to farmers and produce more value for money. Our result suggest that the larger area of productive land does not compensate for the lower biodiversity levels or conservation effectiveness (Batary, Dicks, Kleijn, & Sutherland, 2015). It underscores the importance of conservation policy targets aimed at non-productive land use types such as the 10% high-diversity landscape features that is part of the EU biodiversity strategy 2030. At the same time it suggests that other strategies, such as enhancing crop diversity or reducing field size are far less (cost-)efficient in promoting biodiversity.

The general approach outlined in this paper can greatly help with the monitoring and evaluation of the progress made towards national and international conservation targets. A strength is that it quantifies the relationships with landscape variables that can be relatively easily monitored across large spatial scales for example through remote sensing approaches (Corbane et al., 2015; Lucas, Bouten, Koma, Kissling, & Seijmonsbergen, 2019). In some countries, detailed and regularly updated maps of linear landscape elements or semi-natural habitats are already available (CBS & WUR, 2022; EEA, 2014). The in-situ establishment of relationships between biodiversity and land-use in combination with landscape characteristics (i.e. Fig. 7) can be used to make validated predictions of biodiversity change that is based on the contribution of all parts of the landscape, not just the sites with conservation actions (e.g. protected areas; (Wauchope et al., 2022)). However, the relations between biodiversity and land-use will inevitably change over time because of land-use and climate change. It will therefore be important to regularly (e.g. every 5-10 years) validate these relationships by re-assessing biodiversity levels in different types of land use and in landscapes with different composition.

Last but not least, because our framework summarizes the results in a single farm-level biodiversity estimate it can be used to formulate unequivocal and evidence-based targets for conservation policies. Furthermore, large companies are increasingly required to report on the impacts they have on climate and the environment (e.g. Corporate Sustainability Reporting Directive - CSRD; (Fiege Vos de Wael, Reygers, & Csengő, 2024). This has boosted their interest in interest in rewarding farmers for biodiversity-friendly management if the impact this has on biodiversity can be reliably justified and quantified. This study shows that straight-forward and easy to quantify KPI_bs can act as reliable evidence for biodiversity effects of efforts farmers make. They can therefore be the basis for much-needed biodiversity-based business models that provide the economic rationale that many farmers need to integrate biodiversity into farm-management (Jeroen Scheper et al., 2023).

5 Acknowledgements

We are extremely grateful to Alessandro Albanese, Viktoria Gräser, Wouter Oe, Mauricio Puzzovivo, Cas de Ruiter and Zarah Tomesen and Vasco de Sousa Mendes for their help and good cheer with data collection.

6 References

Alif, Z., Thoyer, S., Preget, R., & Šumrada, T. (in prep.). Establishing 10% of semi-natural habitat on farmland for biodiversity: recommendations for cost-effective policy. Preprint available at SSRN: https://ssrn.com/abstract=4927933

Alison, J., Duffield, S. J., van Noordwijk, C. G. E., Morecroft, M. D., Marrs, R. H., Saccheri, I. J., & Hodgson, J. A. (2016). Spatial targeting of habitat creation has the potential to improve agri-environment scheme outcomes for macro-moths. *Journal of Applied Ecology, 53*(6), 1814-1822. doi:10.1111/1365-2664.12750

Andersson, G. K. S., Boke-Olén, N., Roger, F., Ekroos, J., Smith, H. G., & Clough, Y. (2022). Landscape-scale diversity of plants, bumblebees and butterflies in mixed farm-forest landscapes of Northern Europe: Clear-cuts do not compensate for the negative effects of plantation forest cover. *Biological Conservation, 274*, 109728. doi:https://doi.org/10.1016/j.biocon.2022.109728

Anonymous. (2020). The United Nations must get its new biodiversity targets right. *Nature, 578*, 337-338.

Baker, D. J., Freeman, S. N., Grice, P. V., & Siriwardena, G. M. (2012). Landscape-scale responses of birds to agri-environment management: a test of the English Environmental Stewardship scheme. *Journal of Applied Ecology, 49*(4), 871-882. doi:10.1111/j.1365-2664.2012.02161.x

Bartholomée, O., Aullo, A., Becquet, J., Vannier, C., & Lavorel, S. (2020). Pollinator presence in orchards depends on landscape-scale habitats more than in-field flower resources. *Agriculture Ecosystems & Environment, 293*. doi:10.1016/j.agee.2019.106806

Batary, P., Dicks, L. V., Kleijn, D., & Sutherland, W. J. (2015). The role of agri-environment schemes in conservation and environmental management. *Conservation Biology, 29*(4), 1006-1016. doi:10.1111/cobi.12536

Batary, P., Galle, R., Riesch, F., Fischer, C., Dormann, C. F., Musshoff, O., . . . Tscharntke, T. (2017). The former Iron Curtain still drives biodiversity-profit trade-offs in German agriculture. *Nature Ecology & Evolution, 1*(9), 1279-1284. doi:10.1038/s41559-017-0272-x

Bennett, A. B., & Isaacs, R. (2014). Landscape composition influences pollinators and pollination services in perennial biofuel plantings. *Agriculture Ecosystems & Environment, 193*, 1-8. doi:10.1016/j.agee.2014.04.016

Brook, A. J., Woodcock, B. A., Sinka, M., & Vanbergen, A. J. (2008). Experimental verification of suction sampler capture efficiency in grasslands of differing vegetation height and structure. *Journal of Applied Ecology, 45*(5), 1357-1363. doi:10.1111/j.1365-2664.2008.01530.x

Butchart, S. H. M., Walpole, M., Collen, B., van Strien, A., Scharlemann, J. P. W., Almond, R. E. A., . . . Watson, R. (2010). Global Biodiversity: Indicators of Recent Declines. *Science, 328*(5982), 1164-1168. doi:10.1126/science.1187512

CBD. (2022). Kunming-Montreal Global Biodiversity Framework. CBD/COP/15/L25. www.cbd.int.

CBS, & WUR. (2022). *Natural Capital Accounting in the Netherlands – Technical report. Statistics Netherlands (CBS) and Wageningen University and Research (WUR)*. Retrieved from

Chaplin-Kramer, R., O'Rourke, M. E., Blitzer, E. J., & Kremen, C. (2011). A meta-analysis of crop pest and natural enemy response to landscape complexity. *Ecology Letters, 14*(9), 922-932. doi:10.1111/j.1461-0248.2011.01642.x

Clough, Y., Kirchweger, S., & Kantelhardt, J. (2020). Field sizes and the future of farmland biodiversity in European landscapes. *Conservation Letters, 13*(6), 12. doi:10.1111/conl.12752

Colen, L., Paloma, S. G. Y., Latacz-Lohmann, U., Lefebvre, M., Préget, R., & Thoyer, S. (2016). Economic Experiments as a Tool for Agricultural Policy Evaluation: Insights from the European CAP. *Canadian Journal of Agricultural Economics-Revue Canadienne D Agroeconomie, 64*(4), 667-694. doi:10.1111/cjag.12107

Corbane, C., Lang, S., Pipkins, K., Alleaume, S., Deshayes, M., García Millán, V. E., . . . Michael, F. (2015). Remote sensing for mapping natural habitats and their conservation status – New opportunities and challenges. *International Journal of Applied Earth Observation and Geoinformation, 37*, 7-16. doi:https://doi.org/10.1016/j.jag.2014.11.005

Cormont, A., Siepel, H., Clement, J., Melman, T. C. P., WallisDeVries, M. F., van Turnhout, C. A. M., . . . de Snoo, G. R. (2016). Landscape complexity and farmland biodiversity: Evaluating the CAP target on natural elements. *Journal for Nature Conservation, 30*, 19-26. doi:10.1016/j.jnc.2015.12.006

Daily, G. C., Polasky, S., Goldstein, J., Kareiva, P. M., Mooney, H. A., Pejchar, L., . . . Shallenberger, R. (2009). Ecosystem services in decision making: time to deliver. *Frontiers in Ecology and the Environment, 7*(1), 21-28. doi:10.1890/080025

Dainese, M., Martin, E. A., Aizen, M. A., Albrecht, M., Bartomeus, I., Bommarco, R., . . . Steffan-Dewenter, I. (2019). A global synthesis reveals biodiversity-mediated benefits for crop production. *Science Advances, 5*(10). doi:10.1126/sciadv.aax0121

Diaz, S., Settele, J., Brondizio, E. S., Ngo, H. T., Agard, J., Arneth, A., . . . Zayas, C. N. (2019). Pervasive human-driven decline of life on Earth points to the need for transformative change. *Science, 366*(6471), 1327-+. doi:10.1126/science.aax3100

Diehl, E., Mader, V. L., Wolters, V., & Birkhofer, K. (2013). Management intensity and vegetation complexity affect web-building spiders and their prey. *Oecologia, 173*(2), 579-589. doi:10.1007/s00442-013-2634-7

Donald, P. F., Sanderson, F. J., Burfield, I. J., Bierman, S. M., Gregory, R. D., & Waliczky, Z. (2007). International conservation policy delivers benefits for birds in Europe. *Science, 317*(5839), 810-813. doi:10.1126/science.1146002

EC. (2020). EU Biodiversity Strategy for 2030 - Bringing nature back into our lives. COM(2020) 380 final. European Commission, Brussels.

EEA. (2014). *Terrestrial habitat mapping in Europe – An overview*: Publications Office.

Ekroos, J., Kleijn, D., Batary, P., Albrecht, M., Baldi, A., Bluthgen, N., . . . Smith, H. G. (2020). High land-use intensity in grasslands constrains wild bee species richness in Europe. *Biological Conservation, 241*, 8. doi:10.1016/j.biocon.2019.108255

ESRI. (2022). ArcGIS pro Version 2.9.5. Redlands, CA: Environmental Systems Research Institute. .

Fiege Vos de Wael, T., Reygers, T., & Csengő, Z. (2024). Business models based on supply chain incentives in The Netherlands, Portugal and Romania. Deliverable D2.9 EU Horizon 2020 SHOWCASE Project, Grant agreement No 862480. .

Fijen, T. P. M., Scheper, J. A., Boekelo, B., Raemakers, I., & Kleijn, D. (2019). Effects of landscape complexity on pollinators are moderated by pollinators' association with mass-flowering crops. *Proceedings of the Royal Society B-Biological Sciences, 286*(1900), 9. doi:10.1098/rspb.2019.0387

Flick, T., Feagan, S., & Fahrig, L. (2012). Effects of landscape structure on butterfly species richness and abundance in agricultural landscapes in eastern Ontario, Canada. *Agriculture, Ecosystems & Environment, 156*, 123-133. doi:https://doi.org/10.1016/j.agee.2012.05.006

Fountain, M. T., Tsiolis, K., Silva, C. X., Deakin, G., Garratt, M. P. D., O'Connor, R., . . . Potts, S. G. (2023). Location and Creation of Nest Sites for Ground-Nesting Bees in Apple Orchards. *Insects, 14*(6). doi:10.3390/insects14060490

Gabriel, D., Thies, C., & Tscharntke, T. (2005). Local diversity of arable weeds increases with landscape complexity. *Perspectives in Plant Ecology Evolution and Systematics, 7*(2), 85-93. doi:10.1016/j.ppees.2005.04.001

Galle, R., Tolgyesi, C., Torma, A., Batori, Z., Lorinczi, G., Szilassi, P., . . . Batary, P. (2022). Matrix quality and habitat type drive the diversity pattern of forest steppe fragments. *Perspectives in Ecology and Conservation, 20*(1), 60-68. doi:10.1016/j.pecon.2021.11.004

Garratt, M. P. D., Senapathi, D., Coston, D.J., Mortimer, S.R. & Potts, S.G. (2017). The benefits of hedgerows for pollinators and natural enemies depends on hedge quality and landscape context. . *Agriculture Ecosystems Environment, 247*, 363–370.

Geijzendorffer, I. R., Targetti, S., Schneider, M. K., Brus, D. J., Jeanneret, P., Jongman, R. H. G., . . . Herzog, F. (2016). How much would it cost to monitor farmland biodiversity in Europe? *Journal of Applied Ecology, 53*(1), 140-149. doi:10.1111/1365-2664.12552

Geldmann, J., Barnes, M., Coad, L., Craigie, I. D., Hockings, M., & Burgess, N. D. (2013). Effectiveness of terrestrial protected areas in reducing habitat loss and population declines. *Biological Conservation, 161*, 230-238. doi:10.1016/j.biocon.2013.02.018

Green, E. J., Buchanan, G. M., Butchart, S. H. M., Chandler, G. M., Burgess, N. D., Hill, S. L. L., & Gregory, R. D. (2019). Relating characteristics of global biodiversity targets to reported progress. *Conservation Biology, 33*(6), 1360-1369. doi:10.1111/cobi.13322

Guzman, L. M., Elle, E., Morandin, L. A., Cobb, N. S., Chesshire, P. R., McCabe, L. M., . . . M'Gonigle, L. K. (2024). Impact of pesticide use on wild bee distributions across the United States. *Nature Sustainability*. doi:10.1038/s41893-024-01413-8

Habel, J. C., Dengler, J., Janisova, M., Torok, P., Wellstein, C., & Wiezik, M. (2013). European grassland ecosystems: threatened hotspots of biodiversity. *Biodiversity and Conservation, 22*(10), 2131-2138. doi:10.1007/s10531-013-0537-x

Hammers, M., Muskens, G., van Kats, R. J. M., Teunissen, W. A., & Kleijn, D. (2015). Ecological contrasts drive responses of wintering farmland birds to conservation management. *Ecography, 38*(8), 813-821. doi:10.1111/ecog.01060

Harrison, X. A., Donaldson, L., Correa-Cano, M. E., Evans, J., Fisher, D. N., Goodwin, C. E., . . . Inger, R. (2018). A brief introduction to mixed effects modelling and multi-model inference in ecology. *Peerj, 6*. doi:10.7717/peerj.4794

Hass, A. L., Kormann, U. G., Tscharntke, T., Clough, Y., Baillod, A. B., Sirami, C., . . . Batary, P. (2018). Landscape configurational heterogeneity by small-scale agriculture, not crop diversity, maintains pollinators and plant reproduction in western Europe. *Proceedings of the Royal Society B-Biological Sciences, 285*(1872), 10. doi:10.1098/rspb.2017.2242

Hendrickx, F., Maelfait, J. P., Van Wingerden, W., Schweiger, O., Speelmans, M., Aviron, S., . . . Bugter, R. (2007). How landscape structure, land-use intensity and habitat diversity affect components of total arthropod diversity in agricultural landscapes. *Journal of Applied Ecology, 44*(2), 340-351. doi:10.1111/j.1365-2664.2006.01270.x

Herbertsson, L., Ekroos, J., Albrecht, M., Bartomeus, I., Batary, P., Bommarco, R., . . . Smith, H. G. (2021). Bees increase seed set of wild plants while the proportion of arable land has a variable effect on pollination in European agricultural landscapes. *Plant Ecology and Evolution, 154*(3), 341-350. doi:10.5091/plecevo.2021.1884

Herzog, F., Balázs, K., Dennis, P., Friedel, J. K., Geijzendorffer, I., Jeanneret, P., . . . Pointereau, P. (2012). Biodiversity Indicators for European Farming Systems - A Guidebook. Forschungsanstalt Agroscope Reckenholz, Zürich.

Holzschuh, A., Dainese, M., Gonzalez-Varo, J. P., Mudri-Stojnic, S., Riedinger, V., Rundlof, M., . . . Steffan-Dewenter, I. (2016). Mass-flowering crops dilute pollinator abundance in agricultural landscapes across Europe. *Ecology Letters, 19*(10), 1228-1236. doi:10.1111/ele.12657

Karp, D. S., Chaplin-Kramer, R., Meehan, T. D., Martin, E. A., DeClerck, F., Grab, H., . . . Zou, Y. (2018). Crop pests and predators exhibit inconsistent responses to surrounding landscape composition. *Proceedings of the National Academy of Sciences of the United States of America, 115*(33), E7863-E7870. doi:10.1073/pnas.1800042115

Kleijn, D., Biesmeijer, K. J. C., Klaassen, R. H. G., Oerlemans, N., Raemakers, I., Scheper, J., & Vet, L. E. M. (2020). Integrating biodiversity conservation in wider landscape management: Necessity, implementation and evaluation. In D. A. Bohan & A. J. Vanbergen (Eds.), *Future of Agricultural Landscapes, Pt I* (Vol. 63, pp. 127-159).

Kleijn, D., Bommarco, R., Fijen, T. P. M., Garibaldi, L. A., Potts, S. G., & van der Putten, W. H. (2019). Ecological Intensification: Bridging the Gap between Science and Practice. *Trends in Ecology & Evolution, 34*(2), 154-166. doi:10.1016/j.tree.2018.11.002

Kleijn, D., Kohler, F., Baldi, A., Batary, P., Concepcion, E. D., Clough, Y., . . . Verhulst, J. (2009). On the relationship between farmland biodiversity and land-use intensity in Europe. *Proceedings of the Royal Society B-Biological Sciences, 276*(1658), 903-909. doi:10.1098/rspb.2008.1509

Kleijn, D., Linders, T. E. W., Stip, A., Biesmeijer, J. C., Wackers, F. L., & Bukovinszky, T. (2018). Scaling up effects of measures mitigating pollinator loss from local- to landscape-level population responses. *Methods in Ecology and Evolution, 9*(7), 1727-1738. doi:10.1111/2041-210x.13017

Kleijn, D., Rundlof, M., Scheper, J., Smith, H. G., & Tscharntke, T. (2011). Does conservation on farmland contribute to halting the biodiversity decline? *Trends in Ecology & Evolution, 26*(9), 474-481. doi:10.1016/j.tree.2011.05.009

Kleijn, D., & van Langevelde, F. (2006). Interacting effects of landscape context and habitat quality on flower visiting insects in agricultural landscapes. *Basic and Applied Ecology, 7*(3), 201-214. doi:10.1016/j.baae.2005.07.011

Kleijn, D., Winfree, R., Bartomeus, I., Carvalheiro, L. G., Henry, M., Isaacs, R., . . . Potts, S. G. (2015). Delivery of crop pollination services is an insufficient argument for wild pollinator conservation. *Nature Communications, 6*. doi:10.1038/ncomms8414

Klein, A. M., Boreux, V., Fornoff, F., Mupepele, A. C., & Pufal, G. (2018). Relevance of wild and managed bees for human well-being. *Current Opinion in Insect Science, 26*, 82-88. doi:10.1016/j.cois.2018.02.011

Lancaster, K. J. (1966). NEW APPROACH TO CONSUMER THEORY. *Journal of Political Economy, 74*(2), 132-157. doi:10.1086/259131

Larsen, A. E., & Noack, F. (2017). Identifying the landscape drivers of agricultural insecticide use leveraging evidence from 100,000 fields. *Proceedings of the National Academy of Sciences of the United States of America, 114*(21), 5473-5478. doi:10.1073/pnas.1620674114

Leclère, D., Obersteiner, M., Barrett, M., Butchart, S. H. M., Chaudhary, A., De Palma, A., . . . Young, L. (2020). Bending the curve of terrestrial biodiversity needs an integrated strategy. *Nature, 585*(7826), 551-556. doi:10.1038/s41586-020-2705-y

Leffler, E. M., Bullaughey, K., Matute, D. R., Meyer, W. K., Ségurel, L., Venkat, A., . . . Przeworski, M. (2012). Revisiting an Old Riddle: What Determines Genetic Diversity Levels within Species? *Plos Biology, 10*(9). doi:10.1371/journal.pbio.1001388

Li, P. Y., Kleijn, D., Badenhausser, I., Zaragoza-Trello, C., Gross, N., Raemakers, I., & Scheper, J. (2020). The relative importance of green infrastructure as refuge habitat for pollinators increases with local land-use intensity. *Journal of Applied Ecology, 57*(8), 1494-1503. doi:10.1111/1365-2664.13658

Loos, J., Dorresteijn, I., Hanspach, J., Fust, P., Rakosy, L., & Fischer, J. (2014). Low-Intensity Agricultural Landscapes in Transylvania Support High Butterfly Diversity: Implications for Conservation. *Plos One, 9*(7). doi:10.1371/journal.pone.0103256

Lowenberg-DeBoer, J., Behrendt, K., Ehlers, M. H., Dillon, C., Gabriel, A., Huang, I. Y., . . . Rose, D. (2022). Lessons to be learned in adoption of autonomous equipment for field crops. *Applied Economic Perspectives and Policy, 44*(2), 848-864. doi:10.1002/aepp.13177

Lucas, C., Bouten, W., Koma, Z., Kissling, W. D., & Seijmonsbergen, A. C. (2019). Identification of Linear Vegetation Elements in a Rural Landscape Using LiDAR Point Clouds. *Remote Sensing, 11*(3), 292. Retrieved from https://www.mdpi.com/2072-4292/11/3/292

Manhoudt, A. G. E., & de Snoo, G. R. (2003). A quantitative survey of semi-natural habitats on Dutch arable farms. *Agriculture Ecosystems & Environment, 97*(1-3), 235-240. doi:10.1016/s0167-8809(03)00123-3

Martin, E. A., Dainese, M., Clough, Y., Baldi, A., Bommarco, R., Gagic, V., . . . Steffan-Dewenter, I. (2019). The interplay of landscape composition and configuration: new pathways to manage functional biodiversity and agroecosystem services across Europe. *Ecology Letters, 22*(7), 1083-1094. doi:10.1111/ele.13265

McFadden, D. (1974). Conditional logit analysis of qualitative choice behavior. P 105:142 in P. Zarembka (Ed.), Frontiers in Econometrics, Academic Press, New York.. .

Mei, Z. L., Scheper, J., Bommarco, R., de Groot, G. A., Garratt, M. P. D., Hedlund, K., . . . Kleijn, D. (2023). Inconsistent responses of carabid beetles and spiders to land-use intensity and landscape complexity in north-western Europe. *Biological Conservation, 283*. doi:10.1016/j.biocon.2023.110128

Mestre, L., Schirmel, J., Hetz, J., Kolb, S., Pfister, S. C., Amato, M., . . . Entling, M. H. (2018). Both woody and herbaceous semi-natural habitats are essential for spider overwintering in European farmland. *Agriculture Ecosystems & Environment, 267*, 141-146. doi:10.1016/j.agee.2018.08.018

Neumüller, U., Burger, H., Krausch, S., Blüthgen, N., & Ayasse, M. (2020). Interactions of local habitat type, landscape composition and flower availability moderate wild bee communities. *Landscape Ecology, 35*(10), 2209-2224. doi:10.1007/s10980-020-01096-4

Pe'er, G., Finn, J. A., Díaz, M., Birkenstock, M., Lakner, S., Röder, N., . . . Guyomard, H. (2022). How can the European Common Agricultural Policy help halt biodiversity loss? Recommendations by over 300 experts. *Conservation Letters, 15*(6), e12901. doi:https://doi.org/10.1111/conl.12901

Pereira, H. M., & Cooper, H. D. (2006). Towards the global monitoring of biodiversity change. *Trends in Ecology & Evolution, 21*(3), 123-129. doi:10.1016/j.tree.2005.10.015

Petermann, J. S., & Buzhdygan, O. Y. (2021). Grassland biodiversity. *Current Biology, 31*(19), R1195-R1201. doi:10.1016/j.cub.2021.06.060

Polasky, S., Nelson, E., Pennington, D., & Johnson, K. A. (2011). The Impact of Land-Use Change on Ecosystem Services, Biodiversity and Returns to Landowners: A Case Study in the State of Minnesota. *Environmental & Resource Economics, 48*(2), 219-242. doi:10.1007/s10640-010-9407-0

Pywell, R. F., Heard, M. S., Bradbury, R. B., Hinsley, S., Nowakowski, M., Walker, K. J., & Bullock, J. M. (2012). Wildlife-friendly farming benefits rare birds, bees and plants. *Biology Letters, 8*(5), 772-775. doi:10.1098/rsbl.2012.0367

QGIS.org. (2023). QGIS Geographic Information System. Open Source Geospatial Foundation Project. http://qgis.org.

Rader, R., Bartomeus, I., Garibaldi, L. A., Garratt, M. P. D., Howlett, B. G., Winfree, R., . . . Woyciechowski, M. (2016). Non-bee insects are important contributors to global crop pollination. *Proceedings of the National Academy of Sciences of the United States of America, 113*(1), 146-151. doi:10.1073/pnas.1517092112

Raderschall, C. A., Bommarco, R., Lindstrom, S. A. M., & Lundin, O. (2021). Landscape crop diversity and semi-natural habitat affect crop pollinators, pollination benefit and yield. *Agriculture Ecosystems & Environment, 306*. doi:10.1016/j.agee.2020.107189

Rasmont, P., Devalez, J., Pauly, A., Michez, D., & Radchenko, V. G. (2017). Addition to the checklist of IUCN European wild bees (Hymenoptera: Apoidea). *Annales de La Société Entomologique de France, 53*, 17-32.

Redlich, S., Martin, E. A., & Steffan-Dewenter, I. (2018). Landscape-level crop diversity benefits biological pest control. *Journal of Applied Ecology, 55*(5), 2419-2428. doi:10.1111/1365-2664.13126

Revoyron, E., Bail, M. L., Meynard, J. M., Gunnarsson, A., Seghetti, M., & Colombo, L. (2022). Diversity and drivers of crop diversification pathways of European farms. *Agricultural Systems, 201*. doi:10.1016/j.agsy.2022.103439

Rossetti, I., Bagella, S., Cappai, C., Caria, M. C., Lai, R., Roggero, P. P., . . . Seddaiu, G. (2015). Isolated cork oak trees affect soil properties and biodiversity in a Mediterranean wooded grassland. *Agriculture Ecosystems & Environment, 202*, 203-216. doi:10.1016/j.agee.2015.01.008

Salazar-Ordóñez, M., Rodríguez-Entrena, M., & Villanueva, A. J. (2021). Exploring the commodification of biodiversity using olive oil producers' willingness to accept. *Land Use Policy, 107*, 104348. doi:https://doi.org/10.1016/j.landusepol.2019.104348

Sardaro, R., Girone, S., Acciani, C., Bozzo, F., Petrontino, A., & Fucilli, V. (2016). Agro-biodiversity of Mediterranean crops: farmers' preferences in support of a conservation programme for olive landraces. *Biological Conservation, 201*, 210-219. doi:10.1016/j.biocon.2016.06.033

Scheper, J., Badenhausser, I., Kantelhardt, J., Kirchweger, S., Bartomeus, I., Bretagnolle, V., . . . Kleijn, D. (2023). Biodiversity and pollination benefits trade off against profit in an intensive farming system. *Proceedings of the National Academy of Sciences, 120*(28), e2212124120. doi:10.1073/pnas.2212124120

Scheper, J., Bommarco, R., Holzschuh, A., Potts, S. G., Riedinger, V., Roberts, S. P. M., . . . Kleijn, D. (2015). Local and landscape-level floral resources explain effects of wildflower strips on wild bees across four European countries. *Journal of Applied Ecology, 52*(5), 1165-1175. doi:10.1111/1365-2664.12479

Schoch, K., Tschumi, M., Lutter, S., Ramseier, H., & Zingg, S. (2022). Competition and Facilitation Effects of Semi-Natural Habitats Drive Total Insect and Pollinator Abundance in Flower Strips. *Frontiers in Ecology and Evolution, 10*. doi:10.3389/fevo.2022.854058

Schulze, C., Zagórska, K., Häfner, K., Markiewicz, O., Czajkowski, M., & Matzdorf, B. (2024). Using farmers' ex ante preferences to design agri-environmental contracts: A systematic review. *Journal of Agricultural Economics, 75*(1), 44-83. doi:https://doi.org/10.1111/1477-9552.12570

Schweiger, O., Maelfait, J. P., Van Wingerden, W., Hendrickx, F., Billeter, R., Speelmans, M., . . . Bugter, R. (2005). Quantifying the impact of environmental factors on arthropod communities in agricultural landscapes across organizational levels and spatial scales. *Journal of Applied Ecology, 42*(6), 1129-1139. doi:10.1111/j.1365-2664.2005.01085.x

Simonson, W. D., Allen, H. D., Parham, E., Santos, E. D. E., & Hotham, P. (2018). Modelling biodiversity trends in the montado (wood pasture) landscapes of the Alentejo, Portugal. *Landscape Ecology, 33*(5), 811-827. doi:10.1007/s10980-018-0627-y

Steffan-Dewenter, I., Munzenberg, U., Burger, C., Thies, C., & Tscharntke, T. (2002). Scale-dependent effects of landscape context on three pollinator guilds. *Ecology, 83*(5), 1421-1432. doi:10.2307/3071954

Šumrada, T., Japelj, A., Verbič, M., & Erjavec, E. (2022). Farmers' preferences for result-based schemes for grassland conservation in Slovenia. *Journal for Nature Conservation, 66*, 126143. doi:https://doi.org/10.1016/j.jnc.2022.126143

Taki, H., Murao, R., Mitai, K., & Yamaura, Y. (2018). The species richness/abundance-area relationship of bees in an early successional tree plantation. *Basic and Applied Ecology, 26*, 64-70. doi:10.1016/j.baae.2017.09.002

Tamburini, G., Bommarco, R., Wanger, T. C., Kremen, C., van der Heijden, M. G. A., Liebman, M., & Hallin, S. (2020). Agricultural diversification promotes multiple ecosystem services without compromising yield. *Science Advances, 6*(45), 8. doi:10.1126/sciadv.aba1715

Team, R. C. (2022). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Tobisch, C., Rojas-Botero, S., Uhler, J., Kollmann, J., Müller, J., Moning, C., . . . Ewald, J. (2023). Conservation-relevant plant species indicate arthropod richness across trophic levels: Habitat quality is more important than habitat amount. *Ecological Indicators, 148*. doi:10.1016/j.ecolind.2023.110039

Toivonen, M., Herzon, I., & Kuussaari, M. (2015). Differing effects of fallow type and landscape structure on the occurrence of plants, pollinators and birds on environmental fallows in Finland. *Biological Conservation, 181*, 36-43. doi:10.1016/j.biocon.2014.10.034

Tscharntke, T., Grass, I., Wanger, T. C., Westphal, C., & Batary, P. (2021). Beyond organic farming - harnessing biodiversity-friendly landscapes. *Trends in Ecology & Evolution, 36*(10), 919-930. doi:10.1016/j.tree.2021.06.010

Tscharntke, T., Tylianakis, J. M., Rand, T. A., Didham, R. K., Fahrig, L., Batary, P., . . . Westphal, C. (2012). Landscape moderation of biodiversity patterns and processes - eight hypotheses. *Biological Reviews, 87*(3), 661-685. doi:10.1111/j.1469-185X.2011.00216.x

Turner, R. K., & Daily, G. C. (2008). The ecosystem services framework and natural capital conservation. *Environmental & Resource Economics, 39*(1), 25-35. doi:10.1007/s10640-007-9176-6

UNEP-WCMC, & IUCN. (2021). *Protected Planet Report 2020*. Retrieved from

van Westendorp, P. (1976). NSS-Price Sensitivity Meter (PSM) – A new approach to study consumer perception of price. *Proceedings of the 29th ESOMAR Congress,*, 139–167.

Wauchope, H. S., Jones, J. P. G., Geldmann, J., Simmons, B. I., Amano, T., Blanco, D. E., . . . Sutherland, W. J. (2022). Protected areas have a mixed impact on waterbirds, but management helps. *Nature, 605*(7908), 103-+. doi:10.1038/s41586-022-04617-0

Winfree, R., Fox, J. W., Williams, N. M., Reilly, J. R., & Cariveau, D. P. (2015). Abundance of common species, not species richness, drives delivery of a real-world ecosystem service. *Ecology Letters, 18*(7), 626-635. doi:10.1111/ele.12424

Xu, Y., Yang, B., Dai, Q., Pan, H., Zhong, X., Ran, J. H., . . . Zhang, Z. J. (2022). Landscape-scale giant panda conservation based on metapopulations within China's national park system. *Science Advances, 8*(30). doi:10.1126/sciadv.abl8637

7 Supplementary Information

Table S1. Attributes, levels and justifications of the KPIbs used in (a) the Netherlands, (b) Portugal and (c) Romania.

a) The Netherlands

Attributes/KPI Definition	Levels	Notes on setting of levels
Minimum percentage of semi-natural habitat on your farm		
You need to maintain a minimum percentage of permanent semi-natural habitat on your farm. Semi-natural habitats are: hedges, woodlots, isolated trees, ponds, ditches and ditch banks, unpaved roads, field boundaries (permanent vegetation between two agricultural fields), permanent buffer strips, hay meadows. The percentage you need to maintain determines the height of the bonus.	no prescription min 4% SNA min 7% SNA	current level of SNH for arable farms: 2.1% (De Snoo & Manhoudt (2002)); 2022 Draft CAP Strategic Plan: Good Agricultural and Environmental Conditions (GAEC) and ecoscheme subsidies is to have 4% of the usable arable land in non-productive habitats; 7% therefore as realistic target
Crop diversity		
You need to grow a minimum number of crops in your rotation. Cover crops are excluded when counting the number of crops in a rotation.	no prescriptio Min. 5 crops Min. 6 crops	Dutch GAECs in draft strategic plans indicates that in 2024: three different crops in their rotation; Most common crop rotation for Dutch arable farmers is potato – spring barley – sugar beet – winter wheat (Smit &Jager 2018), so that is already four crops in the rotation --> reference for base level; business model should inventivice improvement
Maximum field size		
The average size of all the fields on your farm needs to remain smaller than a certain maximum. A field is defined as a cultivated parcel that is being surrounded on all sides by permanent vegetation.	no prescriptio Max. 5 ha Max. 2.5 ha	Average field size in the Netherlands = ~5 ha (https://www.cbs.nl/nl-nl/maatwerk/2022/08/gemiddelde-oppervlakte-percelen-akkerbouwgewassen-2017-2021); In province of Limburg, approximately 3.3 ha; However, probably refers to the size of the area on which a single crop was grown and doesn't necessary reflect the size of the field that is surrounded on all sides by permanent vegetation (the ecologists definition of a field) --> in reality, the sizes of the fields are probably larger
Bonus payment		
For meeting the thresholds of the above described KPIs you receive an annual additional payment per ha of your entire farm. (Example: If a farmer participates with a farm of 50 ha and would receive 100€/ha for scoring sufficiently high on the set of KPIs, then he/she would receive 5.000€ per year in total.)	50, 100, 180, 250, 350, 450 €/ha for the entire farm	calculating the approximate levels based on income forgone plus 20%; the estimated mean profit a farmer makes from a four crop rotation = €2863/ha (based on standard data from https://digitaal.kwin.nl/ and https://www.agrimatie.nl/binternet.aspx?ID=4&bedrijfstype=11) multiplied by 1.2 = in €3435/ha; --> 2% SNH: 0.02*3435=€69/ha; 4% SNH: 0.04*3435 = €137/ha; 7% SNH: 0.07*3435=€240/ha ; price levels varying between the extremes; discussed with stakeholders of farmer collective Natuurrijk Limburg and approved

(b) Portugal

Attributes/KPI Definition	Levels	Notes on setting of levels
Minimum percentage of semi-natural habitat on your farm		
You need to provide a specific amount of semi-natural habitat on your total farming area. Semi-natural habitats are: Montado grassland, semi-natural or improved grassland, riparian vegetation, and mediterranean forest. The amount you need to provide can range.	no prescription min. 5 % SNA min. 15 % SNA	0% are realistic for very intense systems with no ground vegetation; 15% is very likely the farmers' perceived maximum based on researchers experience
Management between the rows		
The management between the tree rows of your farm can be regulated. Either there is no prescription in the contract and you are free to manage the vegetation between the tree rows as you wish. Or the management is restricted and it is only allowed to cut the vegetation after the plants have set seed in the time just before the harvesting. Spraying the vegetation between the rows is not allowed either way.	no prescription Late cut in the season	discussed flower density (like 3 species/ha..) but this is hard to investigate in the ecological study, but would be nicely marketable/assessable via app/photo; cutting time -> no range (dichotom); missing level for traditional farms (they plough, mights spray, are not in the Integrated Management); Hence, timely variation, that is maybe hard to assess in the field in the ecological study; maybe only have 2 levels instead of three
Minimum number of living solitary trees per ha		
A minimum number of living solitary trees per ha is defined. These trees can either exist already, or you plant them new (they must survive the next 10 years; plant new if dead). The species must be native, such as holm oak or cork oak.	no prescription Min. 5 trees/ha Min. 10 trees/ha	costs are higher than the tree circumference, cause machinery needs to turn around etc.; 10x10 m equals 1% of a ha --> 10 trees make 10% of area (=income forgone)
Bonus payment		
For implementing the above described defined indicator levels you receive an additional payment per litre of olive oil (EVOO or VOO).	0.50, 1, 1.50, 2, 3, 5 €/l olive oil	international market prices for Portuguese olive oil ranged between 2.50€/kg olive oil (lampante) up to 5.40€/kg (extra virgin, organic) -> https://agriculture.ec.europa.eu/data-and-analysis/markets/price-data/price-monitoring-sector/olive-oil_en ; market prices in the retail trade for olive oils with label >Olivares Vivos< are ranging between 10 - 42 €/l with an average of 24 €/l -> own research in (online) shops; discussion: from environmental perspective the unit €/ha might be better, cause the target would be to bring as many ha as possible into the programm to receive as much payment as possible; €/l does result in maximisation of liter; but is easier to trade

(c) Romania

Attributes/KPI		
Definition	**Levels**	**Notes on setting of levels**
Minimum percentage of permanent grassland on your farm		
You need to maintain a minimum percentage of permanent grassland on your farm. The percentage you need to maintain determines the height of the bonus.	10%	
	50%	
	100%	
Percentage of landscape features on your farm		
You need to maintain a specified percentage of your farmland with landscape features. Landscape features are trees, shrubs, and margins.	remove all landscape features	Is too intense and would need to extensify a bit, keep more features to reach the optimum for biodiv
	20%	considered optimum
	50%	very extensive farm that needs to increase their management for optimum of biodiversity
Maximum arable field size on your farm		
If you have arable fields, the average size of all the arable fields on your farm needs to remain smaller than a certain maximum. An arable field is defined as a cultivated parcel that is being surrounded on all sides by permanent vegetation	max. 0.1 ha	
	max. 20 ha	
	max. 50 ha	
Bonus payment		
For meeting the thresholds of the above described KPIs you receive an annual additional payment per ha of your entire farm (Example: If a farmer participates with a farm of 50 ha and would receive 500 RON/ha for scoring sufficiently high on the set of KPIs, then he/she would receive 25 000 RON per year in total)	free entry of annual RON/ha for the entire farm	

Table S2. Results of the generalized linear mixed models analysing in **(a**) the Netherlands, **(b)** Portugal and **(c)** Romania the relationships between the Biodiversity Index and habitat type and whether this is influenced by landscape context. The tables present the candidate sets of best models (ΔAICc < 6). k represents the number of parameters in the model.

(a) the Netherlands

Model Nr	Land-use type	% Semi-Natural Area (in 500 m buffer)	Field size (in 500 m buffer)	Crop diversity (in 500 m buffer)	Land-use type × %Semi-natural area	Land-use type × Field size	Land-use type × Crop diversity	k	AICc	Delta AICc	Model weight
2	+							5	-36.9	0.00	0.211
20	+			-0.019			+	8	-36.1	0.77	0.144
4	+			-0.019				6	-35.6	1.24	0.114
6	+	0.014						6	-35.2	1.68	0.091
10	+		0.005					6	-34.7	2.14	0.073
24	+	0.012		-0.017			+	9	-34.2	2.72	0.054
28	+		0.008	-0.020			+	9	-34.0	2.93	0.049
8	+	0.012		-0.017				7	-33.8	3.11	0.045
12	+		0.008	-0.020				7	-33.6	3.32	0.040
14	+	0.018	0.011					7	-33.3	3.63	0.034
38	+	0.014			+			8	-32.9	3.95	0.029
32	+	0.016	0.013	-0.018			+	10	-32.2	4.69	0.020
16	+	0.016	0.013	-0.018				8	-31.9	4.99	0.017
40	+	0.012		-0.017	+			9	-31.4	5.46	0.014
56	+	0.012		-0.017	+		+	11	-31.1	5.75	0.012
46	+	0.018	0.011		+			9	-30.9	5.98	0.011
Parameter weight	**1.00**	**0.34**	**0.25**	**0.53**	**0.07**	**0.00**	**0.29**				
No. of models	**16**	**10**	**7**	**10**	**4**	**0**	**5**				

(b) Portugal

Model Nr	Land-use type	% Semi-Natural Area (in 500 m buffer)	Nr solitary trees (in 500 m buffer)	Land-use type × %Semi-natural area	Land-use type × Nr solitary trees	k	AICc	Delta AICc	Model weight
2	+					5	21.8	0.00	0.418
6	+	-0.033				6	22.7	0.93	0.263
4	+		-0.006			6	24.0	2.22	0.138
8	+	-0.042	0.016			7	24.8	2.98	0.094
22	+	-0.028		+		8	26.3	4.51	0.044
Parameter weight	**1.00**	**0.42**	**0.24**	**0.05**	**0.00**				
No. of models	**5**	**3**	**2**	**1**	**0**				

(c) Romania

Model Nr	Land-use type	% Semi-natural area (in 500 m buffer)	% Woody landscape features (in 500 m buffer)	Field size (in 500 m buffer)	Land-use type × % Semi-natural area	Land-use type × % Woody landscape features	Land-use type × Field size	k	AICc	Delta AICc	Model weight
2	+							5	-38.1	0.00	0.257
6	+		0.014					6	-36.3	1.79	0.105
20	+	0.003			+			8	-36.3	1.79	0.105
10	+			0.011				6	-36.2	1.94	0.097
4	+	0.003						6	-35.9	2.19	0.086
14	+		0.017	0.014				7	-34.5	3.61	0.042
24	+	0.002	0.014		+			9	-34.4	3.72	0.040
28	+	0.009		0.014	+			9	-34.4	3.74	0.040
8	+	0.002	0.014					7	-34.1	4.04	0.034
12	+	0.009		0.014				7	-34.1	4.06	0.034
38	+		0.014			+		8	-34.0	4.11	0.033
74	+			0.011			+	8	-33.0	5.16	0.019
32	+	0.008	0.016	0.017	+			10	-32.6	5.55	0.016
56	+	0.002	0.014		+	+		11	-32.4	5.67	0.015
16	+	0.008	0.016	0.017				8	-32.3	5.78	0.014
Parameter weight	**1.00**	**0.41**	**0.32**	**0.28**	**0.23**	**0.05**	**0.02**				
No. of models	**15**	**9**	**8**	**7**	**5**	**2**	**1**				

Table S3: Results of the generalized linear mixed models analysing the relationships **in the Netherlands** between plants, bees and spiders and habitat type and whether this is influenced by landscape context variables in a radius of 500m. The tables present the candidate sets of best models (ΔAICc < 6). k represents the number of parameters in the model.

Model Nr	Land-use type	% Semi-Natural Area (in 500 m buffer)	Field size (in 500 m buffer)	Crop diversity	Land-use type × %Semi-natural area	Land-use type × Field size	Land-use type × Crop diversity	k	AICc	Delta AICc	Model weight
Weighted plant cover (m ^{2})											
38	+	-0.068			+			8	236.7	0.00	0.436
46	+	-0.061	0.022		+			9	238.9	2.20	0.145
40	+	-0.066		0.012	+			9	239.0	2.29	0.138
110	+	-0.061	0.022		+	+		11	240.8	4.11	0.056
48	+	-0.060	0.021	0.010	+			10	241.3	4.55	0.045
2	+							5	241.4	4.64	0.043
6	+	-0.068						6	242.2	5.50	0.028
Parameter weight	**1.00**	**0.95**	**0.28**	**0.21**	**0.92**	**0.06**					
No. of models	**7**	**6**	**3**	**2**	**5**	**1**					
Weighted bee abundance											
20	+			-0.117			+	10	200.8	0.00	0.257
2	+							7	201.9	1.11	0.147
28	+		0.030	-0.121			+	11	202.7	1.89	0.100
24	+	-0.021		-0.120			+	11	203.0	2.16	0.087
4	+			-0.027				8	203.8	3.04	0.056
10	+		0.025					8	203.9	3.06	0.056
92	+		0.042	-0.122		+	+	13	203.9	3.14	0.054
6	+	-0.017						8	204.0	3.24	0.051
32	+	-0.013	0.026	-0.122			+	12	205.1	4.26	0.030
12	+		0.029	-0.031				9	205.7	4.91	0.022
74	+		0.026			+		10	205.7	4.91	0.022
8	+	-0.021		-0.030				9	205.9	5.14	0.020
14	+	-0.010	0.021					9	206.1	5.35	0.018
96	+	-0.014	0.038	-0.124		+	+	14	206.4	5.60	0.016
56	+	-0.017		-0.119	+		+	13	206.5	5.74	0.015
Parameter weight	**1.00**	**0.25**	**0.33**	**0.69**	**0.02**	**0.10**	**0.59**				
No. of models	**15**	**7**	**8**	**10**	**1**	**3**	**7**				
Weighted spider abundance											
6	+	5.174						6	1162.7	0.00	0.194
2	+							5	1163.4	0.76	0.133
38	+	5.175			+			8	1163.6	0.91	0.123
8	+	5.309		1.031				7	1164.8	2.14	0.067
14	+	5.059	-0.358					7	1164.9	2.24	0.063
24	+	5.309		1.032			+	9	1165.2	2.51	0.055
10	+		-1.993					6	1165.2	2.55	0.054
4	+			0.341				6	1165.6	2.96	0.044
40	+	5.309		1.032	+			9	1165.8	3.13	0.041
46	+	5.059	-0.358		+			9	1165.9	3.24	0.038
20	+			0.340			+	8	1165.9	3.26	0.038
56	+	5.309		1.033	+		+	11	1167.0	4.37	0.022
16	+	5.164	-0.465	1.074				8	1167.1	4.41	0.021
12	+		-2.072	0.611				7	1167.4	4.76	0.018
32	+	5.164	-0.465	1.074			+	10	1167.5	4.87	0.017
28	+		-2.074	0.613			+	9	1167.8	5.14	0.015
48	+	5.164	-0.465	1.073	+			10	1168.2	5.49	0.012
78	+	5.059	-0.358			+		9	1168.6	5.96	0.010
Parameter weight	**1.00**	**0.69**	**0.26**	**0.36**	**0.24**	**0.01**	**0.15**				
No. of models	**18**	**12**	**9**	**11**	**5**	**1**	**5**				

Table S4: Results of the generalized linear mixed models analysing **in Portugal** the relationships between plants, bees and spiders and habitat type and whether this is influenced by landscape context variables. The tables present the candidate sets of best models (ΔAICc < 6). k represents the number of parameters in the model.

Model Nr	Habitat type	% Semi-Natural Habitat	Nr solitary trees	Habitat type × %Semi-natural area	Habitat type × Nr solitary trees	k	AICc	Delta AICc	Model weight
		in 500 m buffer							
Weighted plant cover (m^2)									
2	+					5	341.8	0.00	0.394
4	+		0.181			6	342.4	0.59	0.292
6	+	0.085				6	343.7	1.92	0.151
8	+	-0.024	0.194			7	344.6	2.88	0.093
12	+		0.178		+	8	347.0	5.23	0.029
22	+	0.083		+		8	347.7	5.96	0.020
Parameter weight	**1.00**	**0.27**	**0.42**	**0.02**	**0.03**				
No. of models	**6**	**3**	**3**	**1**	**1**				
Weighted bee abundance									
6	+	-0.239				6	364.0	0.00	0.270
2	+					5	364.2	0.20	0.245
8	+	-0.365	0.228			7	364.6	0.61	0.199
4	+		0.031			6	366.4	2.41	0.081
22	+	-0.216		+		8	366.6	2.56	0.075
24	+	-0.350	0.251	+		9	366.9	2.95	0.062
16	+	-0.388	0.231		+	9	367.7	3.68	0.043
12	+		0.019		+	8	369.8	5.78	0.015
Parameter weight	**1.00**	**0.66**	**0.40**	**0.14**	**0.06**				
No. of models	**8**	**5**	**5**	**2**	**2**				
Weighted spider abundance									
2	+					5	986.3	0.00	0.311
4	+		-4.061			6	986.9	0.55	0.236
6	+	-1.675				6	988.3	1.98	0.116
12	+		-2.818		+	8	989.0	2.71	0.080
8	+	0.761	-4.475			7	989.1	2.81	0.076
3			-5.495			4	989.7	3.36	0.058
1						3	990.7	4.40	0.034
16	+	1.242	-3.498		+	9	991.3	5.00	0.026
7		0.760	-5.908			5	991.9	5.53	0.020
5		-2.501				4	992.3	5.93	0.016
22	+	-1.317		+		8	992.3	5.97	0.016
Parameter weight	**0.87**	**0.27**	**0.50**	**0.02**	**0.11**				
No. of models	**7**	**6**	**6**	**1**	**2**				

Table S5: Results of the generalized linear mixed models analysing **in Romania** the relationships between plants, bees and spiders and habitat type and whether this is influenced by landscape context variables. The tables present the candidate sets of best models (ΔAICc < 6). k represents the number of parameters in the model.

Model Nr	Land-use type	% Semi-natural area	% Woody landscape elements	Field size	Land-use type × % Semi-natural area	Land-use type × % Woody landscape elements	Land-use type × Field size	k	AICc	Delta AICc	Model weight
		in 500 m buffer									
Weighted plant cover (m^2)											
56	+	5.007	-2.835		+	+		11	1029.5	0.00	0.499
64	+	5.048	-2.817	0.116	+	+		12	1032.0	2.47	0.145
24	+	5.007	-2.835		+			9	1032.2	2.63	0.134
20	+	4.755			+			8	1033.3	3.72	0.078
32	+	5.048	-2.817	0.116	+			10	1034.6	5.01	0.041
28	+	4.979		0.611	+			9	1035.5	5.92	0.026
Parameter weigl	**1.00**	**1.00**	**0.89**	**0.23**	**1.00**	**0.70**	**0.00**				
No. of models	**6**	**6**	**4**	**3**	**6**	**2**	**0**				
Weighted bee abundance											
28	+	0.157		0.156	+			9	261.7	0.00	0.170
12	+	0.157		0.156				7	261.8	0.11	0.161
2	+							5	263.7	2.02	0.062
76	+	0.157		0.156			+	9	263.8	2.08	0.060
20	+	0.099			+			8	263.8	2.11	0.059
32	+	0.156	0.034	0.162	+			10	263.8	2.13	0.059
16	+	0.156	0.034	0.162				8	263.9	2.16	0.058
4	+	0.099						6	264.0	2.30	0.054
10	+			0.098				6	264.1	2.34	0.053
92	+	0.157		0.156	+		+	11	265.3	3.54	0.029
48	+	0.156	0.034	0.162		+		10	265.4	3.67	0.027
64	+	0.156	0.034	0.162	+	+		12	265.6	3.87	0.025
6	+		0.018					6	265.9	4.17	0.021
80	+	0.156	0.034	0.162			+	10	265.9	4.21	0.021
74	+			0.098			+	8	265.9	4.23	0.021
14	+		0.037	0.105				7	266.0	4.33	0.020
24	+	0.099	0.009		+			9	266.1	4.43	0.019
8	+	0.099	0.009					7	266.3	4.54	0.018
38	+		0.018			+		8	267.2	5.52	0.011
46	+		0.037	0.105		+		9	267.5	5.76	0.010
96	+	0.156	0.034	0.162	+		+	12	267.5	5.76	0.010
40	+	0.099	0.009			+		9	267.7	5.97	0.009
Parameter weigl	**1.00**	**0.80**	**0.31**	**0.74**	**0.38**	**0.08**	**0.14**				
No. of models	**22**	**15**	**13**	**14**	**7**	**5**	**5**				
Weighted spider abundance											
4	+	-3.898						6	1031.9	0.00	0.188
8	+	-4.064	1.858					7	1033.3	1.39	0.094
2	+							5	1033.3	1.43	0.092
40	+	-4.063	1.858			+		9	1033.5	1.62	0.083
12	+	-4.441		-1.481				7	1033.7	1.78	0.077
20	+	-3.898			+			8	1033.8	1.94	0.071
6	+		1.497					6	1035.0	3.13	0.039
56	+	-4.063	1.858		+	+		11	1035.0	3.14	0.039
38	+		1.497			+		8	1035.2	3.29	0.036
16	+	-4.483	1.676	-1.187				8	1035.3	3.38	0.035
24	+	-4.064	1.858		+			9	1035.3	3.41	0.034
10	+			0.149				6	1035.5	3.64	0.030
48	+	-4.482	1.676	-1.186		+		10	1035.6	3.70	0.030
28	+	-4.441		-1.481	+			9	1035.7	3.80	0.028
76	+	-4.441		-1.481			+	9	1036.6	4.75	0.017
92	+	-4.442		-1.481	+		+	11	1037.0	5.08	0.015
64	+	-4.483	1.676	-1.186	+	+		12	1037.2	5.30	0.013
14	+		1.578	0.440				7	1037.2	5.35	0.013
32	+	-4.482	1.676	-1.186	+			10	1037.4	5.49	0.012
46	+		1.578	0.440		+		9	1037.5	5.58	0.012
Parameter weigl	**1.00**	**0.77**	**0.46**	**0.29**	**0.22**	**0.22**	**0.03**				
No. of models	**20**	**14**	**12**	**11**	**7**	**6**	**2**				

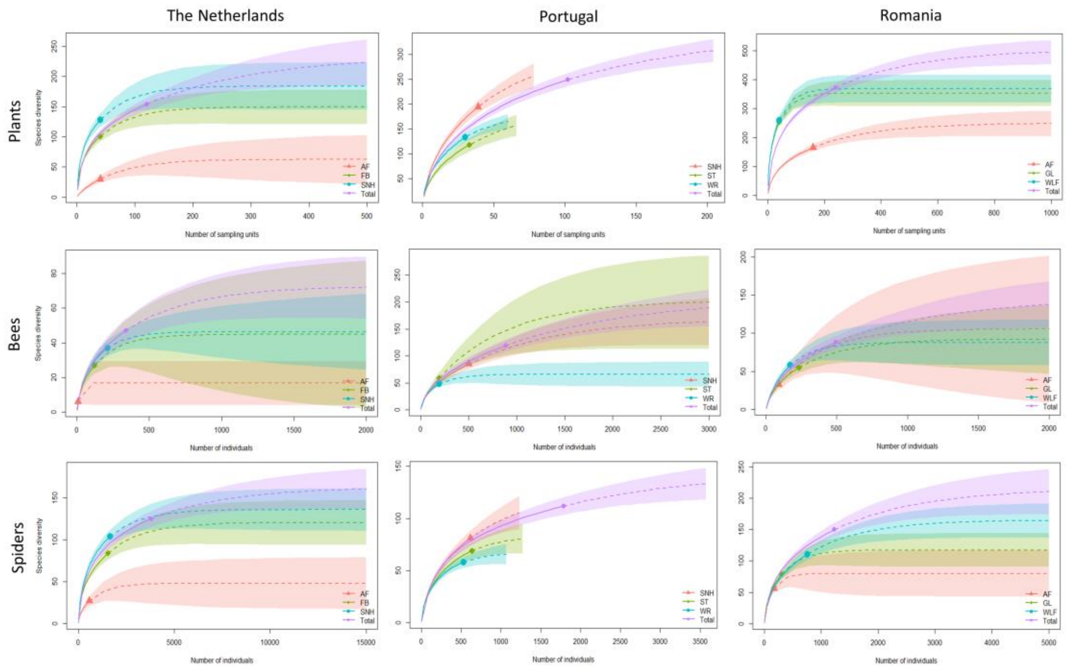

Figure S1: Species accumulation curves for plants, bees and spiders in different land use types in The Netherlands, Portugal and Romania. Species accumulation curves were calculated across the 40 locations of each land-use type in each country and are therefore indicative of the regional species pool size in those land-use types. Solid lines indicate calculated curves, dashed lines indicate extrapolated curves and shaded areas indicate 95% confidence intervals. Semi-natural areas: SNH (semi-natural areas) and GL (extensively managed grasslands); Landscape Elements: FB (field boundaries), ST (scattered trees in olive orchards), WLF (woody landscape features); Agricultural land: AF (arable fields) and WR (within rows of live orchards).

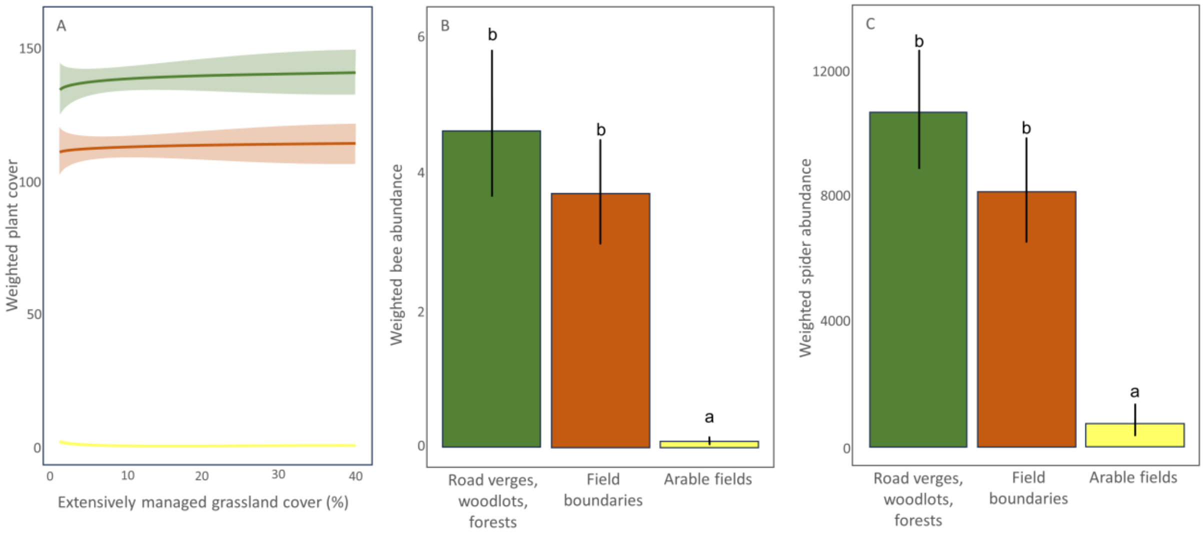

Figure S2: Illustrations of the relations between local biodiversity indicator groups and land-use types in agricultural landscapes **in the Netherlands** for (a) weighted plant cover m^{-2} (b) weighted bee abundance and (c) weighted spider abundance. Different colours indicate different land-use types: green - semi-natural areas, brown – landscape elements, yellow – agricultural land. Results represent predictions (means ± standard errors) of the models from the candidate set of best models (Table S3) for which there was most support. Shaded areas represent 95% confidence intervals.

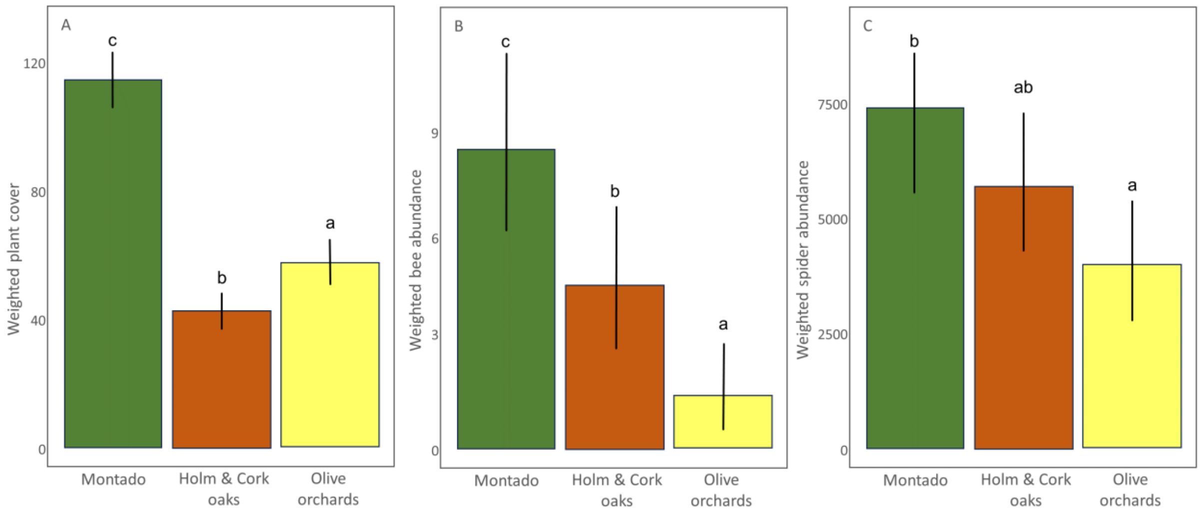

Figure S3: Illustrations of the relations between local biodiversity indicator groups and land-use types in agricultural landscapes **in Portugal** for (a) weighted plant cover m^{-2} (b) weighted bee abundance and (c) weighted spider abundance. Results represent predictions of the model (means ± se) from the candidate set of best models (Table S3) for which there was most support.

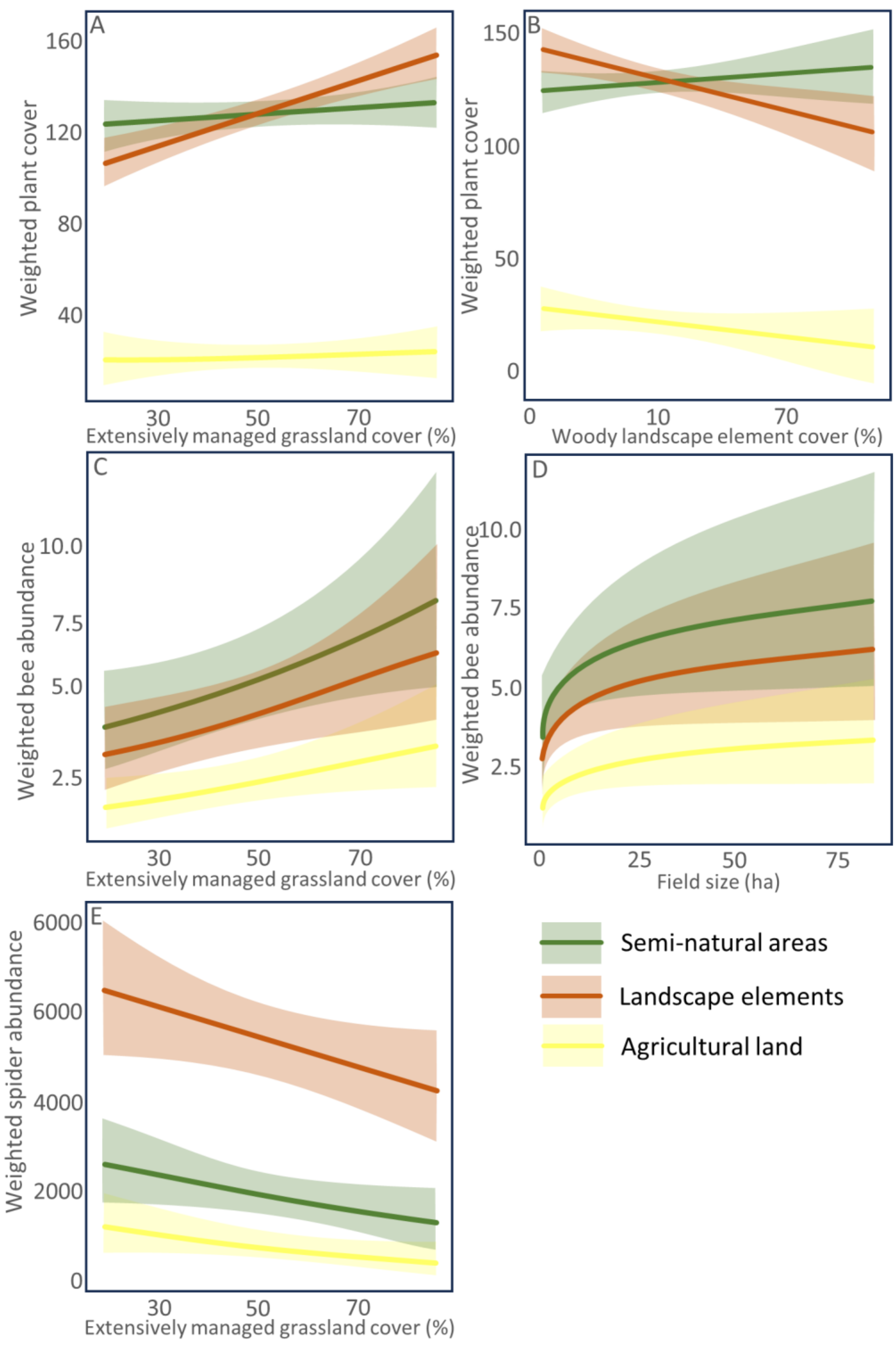

Figure S4: Illustrations of the relations between local biodiversity indicator groups and land-use types in agricultural landscapes **in Romania** for (a) weighted plant cover m^{-2} (b) weighted bee abundance and (c) weighted spider abundance. Results represent predictions of the model (means ± se) from the candidate set of best models (Table S3) for which there was most support.