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Research article

Assessment of determinants of high nature value (HNV) farmland at plot scale in Western Pyrenees

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A B S T R A C T

The concept of High Nature Value (HNV) farming hinges on the causality between agroecosystems with low intensity of management and the corresponding environmental outcomes, including high levels of biodiversity and the presence of semi-natural habitats. Although European strategies for rural development and biodiversity conservation have long recognized the importance of HNV farmlands, many of those areas are currently threatened by intensification and land abandonment. A variety of approaches have been developed for identifying HNV areas and measuring changes in their distribution and extent at landscape scales. In contrast, quantitative approaches for evaluating differences in HNV among the most basic units of management (farms and farm plots) are scarce and almost exclusively based on biodiversity indicators. This gap limits our ability to derive existing gradients of HNV at fine scale and the underlying cause conducive to HNV. Hence, we implemented an index to capture multiple facets of HNV based on expert knowledge criteria and field surveys performed at the finest scale of management (plot). First, we computed this index for hundreds of grasslands located across the Western Pyrenees. Then, we analysed the relationship between the nature value of plots and environmental, management and socio-economic variables. Our results evidence a gradient between low diversity and intensively used agricultural plots and HNV grasslands in the Western Pyrenees. Highest nature values were significantly related to the occurrence of plots in meadows located in steep areas within the Natura 2000 network, whereas lowest values were related to recently opened areas and the number of treatments per year. Importantly, this index, which could be adapted to other farming areas, provides quantitative information to support the implementation of result-based schemes, including eco-schemes and agri-environment-climate interventions of the new CAP (2023–2027).

1. Introduction

Much of terrestrial biodiversity is associated with agricultural landscapes with low intensity of management and a significant presence of semi-natural vegetation (Baldock et al., 1993; Bignal and McCracken, 2000). These agricultural areas are known as High Nature Value (HNV) farmlands in the European Union (EU), where they cover about 30% of all agricultural land of the territory (Oppermann et al., 2012). The concept of HNV farming links the conservation of biodiversity to the (1) maintenance of low-intensity farming practices on semi-natural agricultural lands characterized by the presence of both natural and semi-naturals habitats, (2) the existence of linear elements such as hedgerows, stone walls, patches of woodland or scrub, ponds, etc. that provide ecological niches for wildlife, and (3) the presence of rare species (Andersen et al., 2003). This concept is applicable at multiple scales, from landscape units to individual farm plots (EENRD, 2010). Although HNV farmlands play a pivotal role in European strategies for rural development and biodiversity conservation for more than two decades (e.g. Hodgson et al., 2005), there is evidence of decline in some regions (Anderson and Mammides, 2020; Schmitz et al., 2021). Socio-economic shift in rural areas and market developments have led to intensification of livestock farming practices in lowlands and more favourable valley areas and abandonment in remote areas such as high mountain grasslands in Europe (McGinlay et al., 2017; Moran et al., 2021; van Vliet et al., 2015). Due to their natural constrains HNV farmlands require labour intensive practices, generate relatively low outputs (Keenleyside

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et al., 2014; O'Rourke et al., 2016; Pienkowski, 2011), and hence, they are not competitive in the market (Dax et al., 2021). For that reason, the maintenance of HNV farming systems is particularly dependent on the public funding, mostly provided by the Common Agricultural Policy (CAP) (Lomba et al., 2020). At the same time, the concept of HNV has been central to assess the performance of the CAP since 2014. The presence and extent of HNV farmlands were included among the indicators of the Common Monitoring and Evaluation Framework (CMEF) of the CAP 2014–2020 to describe the situation in which the policy was implemented (context indicator) as well as to assess the impact of policy changes in rural development programs regarding CAP objectives (impact indicator) (European Commission, 2017). In the CAP 2023–2027, the CMEF has been replaced by the Performance Monitoring and Evaluation Framework to support the shift in policy focus from compliance with rules to performance and results (Regulation, 2021/2115). Within this new CAP, HNV farming is an indicator to monitor the progress of EU Member States towards the goals of the Green Deal, and in particular, those included in the Farm to Fork Strategy and the updated Biodiversity Strategy to 2030. In addition, HNV farming is recognized as an environment-friendly farming practice that could be supported by direct payments through eco-schemes of the new CAP (European Commission, 2021).

The development of indicators to measure, monitor and report the extent of HNV farmlands has long been mandatory for all EU Member States. However no common methodology exists yet. Member States and regions have addressed these tasks in different ways, by using land cover, land use, landscape mosaics and biodiversity data (Peppiette, 2011 and references herein; Lomba et al., 2015; Brunbjerg et al., 2016; Kikas et al., 2017), and giving priority to best available information sources and most cost-efficient data collection methods (Benedetti, 2017). Importantly, most national-level indicators have been conceived to identify HNV farmlands at landscape scale, and as a result, they are not applicable at farm or plot level (see Brunbjerg et al., 2016), which constitute the most basic units at which management decision are taken and sustainable practices can be identified and supported by policy incentives (Boyle et al., 2015). Developing HNV indicators at management scale is therefore needed in the context of land use policy to design and implement results-oriented measures for biodiversity (Stolze et al., 2015), such as eco-schemes and agri-environment-climate interventions under the new CAP.

Most biodiversity indicators are based on taxonomic abundance and richness (mainly plants, invertebrates and birds; Benedetti, 2017; Kok et al., 2020). However, this kind of indicators are not always positively associated with HNV farmlands. Mäkeläinen et al. (2019) found that patterns of species richness of butterflies in Finland were spatially associated to HNV, whereas those of birds were not. In contrast, Morelli et al. (2014) found support for the use of birds as indicators of HNV farmlands. Furthermore, biodiversity indicators alone are insufficient surrogates for the ecological condition of the habitat and nature-friendly management practices that embraces the concept of HNV (Tasser et al., 2019; Kok et al., 2020). The scoring system developed in the Results-Based Agri-Environment Pilot Scheme (RBAPS) project (https://rbaps.eu/) was devoted to bridge this gap by combining a set of indicators related to the quality of farmlands in terms of biodiversity, the ecological integrity of the habitat they harbour and the presence of fine-scale landscapes structures that support biodiversity while accounting for management practices (Maher et al., 2015). Each indicator was measured on a poor-to-excellent scale, representing the nature value of three pilot habitats (semi-natural grasslands suitable for rare marsh fritillary butterfly and flower-rich hay meadows in Ireland and mosaic of permanent crops in Navarre. Here, we adapted that methodology to assess the nature value of farmlands in the Western Pyrenees (Navarre, Spain). Specifically, the aims of this study were to: (1) compute a nature value index combining field data and expert knowledge to capture the gradient between low diversity and intensively used farming areas and HNV areas at plot scale and; (2) test the relationship of the index with management practices, socio-economic and environmental conditions widely linked to HNV farmlands. The suitability of this index in the context of result-oriented agri-environmental schemes aiming at promoting HNV farmlands is discussed.

2. Material and methods

2.1. Study area

This study was conducted in the Western Pyrenees (Navarre, Spain; Fig. 1), a sparsely populated and topographically complex area. The climate corresponds to a marine west coast, warm summer climate (Köppen classification: Cfb): precipitation decreases in the north-south direction (from mountains in the north to valleys in the south), while temperature increases, and coupled with decrease in temperatures with elevation (http://meteo.navarra.es/). Due to the complex topography, the biogeographical characteristics and the historical land use (characterized by the predominance of extensive farming), the biodiversity and extension of semi-natural habitats is remarkable (ca. 80% of the study area; Iragui et al., 2010), as underpinned by the large number of Natura2000 sites designed there (13 Special Areas of Conservation and 8 Special Protection Areas). In this study, we mainly focused on semi-natural grasslands, including those harvested predominantly by mowing (meadows), grazing (pastures) or both (i.e. mown and grazed meadows), and dry-grasslands (*sensu* Peeters et al., 2014).

We performed analyses at the scale of management plots, defined as the land unit or set of adjacent land units declared in the CAP by the same farmer and managed under the same regime. According to the CAP declaration, there were 338 livestock farms in the study area, from which we selected 30 for analyses using a stratified random approach to ensure representation of all livestock types in the area. These farms comprised a total of 1.179 declared land units, which were merged into 569 management plots. We performed a stratified random sampling to capture the existing heterogeneity in the type of grazing animal(s) among plots of the same farm. In this case, the selection was constrained to include up to 5 plots per farm. Selected plot (N = 138) were located along an elevational gradient from 480 to 1182 m a.s.l. Field surveys of selected plots were conducted from May and September 2019.

2.2. Nature value index

Following the methodology proposed in the RBAPS project (Maher

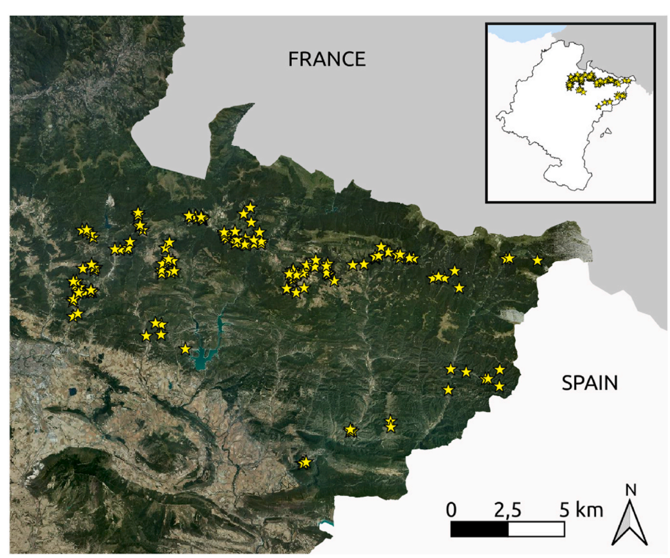

Fig. 1. Location of the studied farmlands plots in the Western Pyreenes (Navarre, Spain).

et al., 2018), the nature value index was based on aggregation of individual indicators related to three concepts: the agro-ecological integrity of grasslands, their threats, pressures and future prospects, and the presence of structural elements that foster biodiversity (e.g. hedgerows, isolated trees, …). The relative importance (scores) of each of these concepts to measure the nature value, and subsequently, the weight of each individual indicator was elicited following a three stage validation process in three pilot areas (two in Ireland and one in Navarre, Spain; Fig. 2): first, Technical committees of each pilot area prepared a proposal of indicators, thresholds and scores. Each of these committees were comprised of experts on fauna, flora, habitats, agri-environment payment schemes and agronomy from the academia, NGOs and local administration. Proposals of each Technical group were then scrutinized by the Steering group, which was made up by several members of Technical groups, plus two experts in HNV systems from the European Forum on Nature Conservation and Pastoralism. There were several rounds of examination before the Steering group agreed on indicators, thresholds and scores. Next, a second validation stage was performed by the Technical advisory group (mainly, stakeholders, consultants and representatives of Farmers Associations), and their feedback was used to refine proposals. Finally, some external evaluators and members of the Technical Committees performed field testing and validation of indicators at each location. A more detailed description of the process and

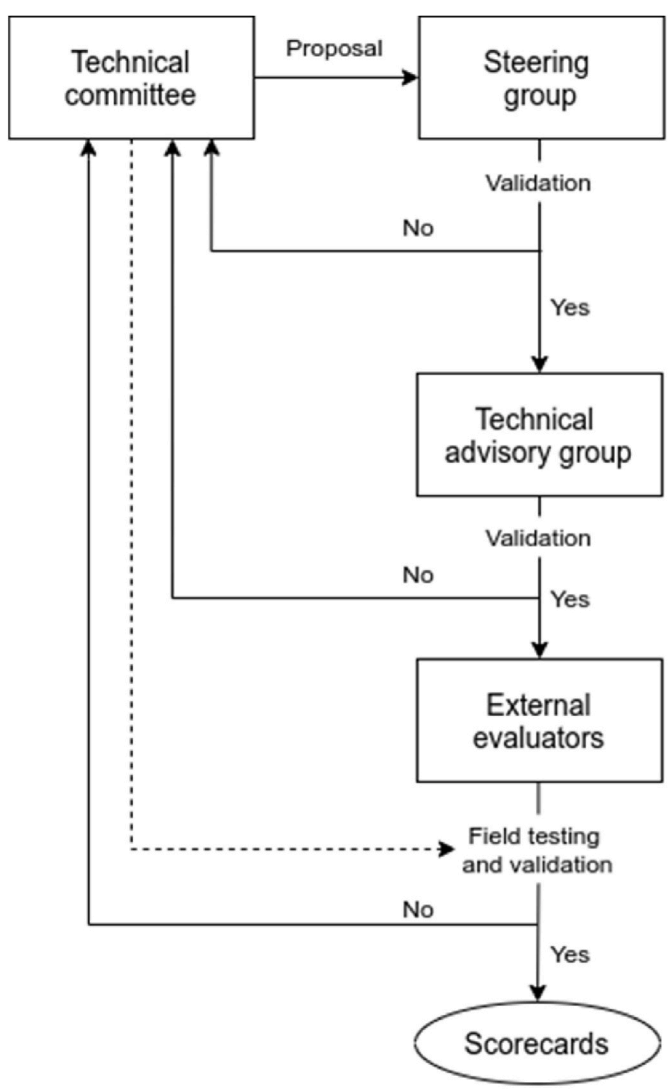

Fig. 2. Validation workflow of indicators, thresholds and scores of the nature value index based on expert knowledge.

the final scorecards (i.e. list of indicators and their thresholds) are in Maher et al. (2018).

In this study, we adapted the scorecard proposed in RBAPS (e.g. Berastegi et al., 2018) to characteristics of Pyrenean grasslands. Individual indicator and their grading scales are explained below and in Table 1. In order to built the nature value index, scores (positive and negative) of individual indicators were added and the sum was bound to 0–100 range (i.e. negative index scores were converted to 0, and scores above 100 to 100) for the ease of interpretation.

2.2.1. Indicators of agro-ecological integrity

Ecological integrity was originally defined by Karr and Dudley (1981) as 'the capacity of an ecosystem to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of reference grasslands in the region'. In this study we assessed the agro-ecological integrity of grasslands based on the quality of plant community, which is positively related to wider biodiversity and ecosystem benefits (e.g. Knops et al., 1999). Selected indicators were based on the richness and cover of positive and negative plant indicators associated to agricultural practices (Table 1). Positive indicators were defined as those species representative of semi-natural grasslands with limited agricultural improvement and/or habitats included in the Annex I of the Habitats Directive (Council Directive 92/43/EEC) occurring in the study area. Plants listed on the Annex II, IV or V of the Habitats Directive and/or national and regional catalogues of threatened species were also considered as positive indicators due to their conservation interest (see complete list of species in Appendix A). The number and cover of positive plant indicators are a reliable surrogate of wider biodiversity, including butterfly richness and overall plant diversity (EFNCP, 2018), Negative indicators included perennial species with no forage value such as thistles, docks (*Rumex* sp.) and nettle (*Urtica dioica*), that spread due to overgrazing and/or excess available soil nitrogen and indicate intensification. Importantly, this indicator can provide early warning signals of deterioration of species-rich grasslands (Maher et al., 2018). A list of positive and negative species potentially present in the study area was retained from plant occurrence records in regional databases. The number of positive indicators was recorded along a transect laid across the longest diagonal of each plot, whereas the cover of (i) areas without positive indicators, (ii) dense patches with low diversity and (iii) negative indicator plants were estimated visually by walking the full plot (see Table 1).

2.2.2. Indicators of threats, pressures and future prospects

As in many other European mountains, the Pyrenees have experienced a reduction in human population and livestock densities and substantial changes in land use since the mid-20th century, which have led to land abandonment of many mountain areas, and intensification of more accessible and productive ones (Dax et al., 2021; García-Ruiz et al., 1996). At the early stages of abandonment (i.e. in absence of grazing and mowing activities), litter tends to accumulate on the ground reducing plant richness in grasslands (Tilman et al., 2001). In contrast, overgrazing is a significant problem in more accessible areas, which can foster the dominance of grazing-resistant species (often low palatable and unpalatable species), as well as an increase the levels of nitrification. These changes can lead to a simplification of plant communities and a biodiversity loss. Other practices relatively frequent in the study area with negative impacts on biodiversity are drainage of land, enhanced soil erosion and burning. To account for these potential impacts, we used a second set of indicators based on the ground surface covered by litter, the cover of species indicative of grazing abandonment (e.g. *Brachypodium rupestre*, *Pteridium aquilinum*) and magnitude of shrub encroachment, overgrazing and/or nitrification (evidenced by the abundance of negative species), another damaging activities regarding vegetation, soil and water (Table 1).

Table 1
Individual indicators used to compute the nature value index in Pyrenean grasslands. Indicator were proposed according to the literature and validated through a decision process based on expert knowledge and in situ trialling.

Concept	Indicator	Thresholds and scores
Agro-ecological integrity (−30 to 85 points)	Number of positive indicator plants (0–35 points; weight = 7)	<8 plants (0 points); 8–9 plants (5 points.); 10–11 plants (10 points.); 12–13 plants (15 points.); 14–15 plants (20 points.); 16–17 plants (25 points.); 18–19 plants (30 points); >19 plants (35 points)
	Plot area without positive indicators plants (−10 to 30 points; weight = 6)	>90% (−10 points); 75–90% (0 points); 50–75% (10 points); 25–50% (20 points); <25% (30 points)
	Plot area covered by dense patches with low diversity (−10 to 15 points; weight = 3)	>90% (−10 points); 75–90% (−5 points); 50–75% (5 points); 25–50% (10 points); <25% (15 points)
	Plot area of negative indicator plants (−10 to 5 points; weight = 1)	>5% (−10 points); 2–5% (0 points); <2% (5 points)
Threats, pressures and future prospects (−80 to 15 points)	Cover of litter (−20 to 5 points; weight = 1)	>50% (−20 points); 25–50% (−10 points); 10–25% (−5 points); <10% (5 points)
	Cover of species indicative of grazing abandonment, overgrazing or nitrification (−20 to 5 points; weight = 1)	>50% (−20 points); 25–50% (−10 points); 10–25% (−5 points); <10% (5 points)
	Magnitude of detrimental practices on vegetation, including land drainage, soil erosion, trampling, waste dumping, …(−40 to 5 points; weight = 1)	High (−40 points); Medium-High (−20 points); Low-Medium (−10 points); Low or non-existent (5 points)
Landscape structures that support biodiversity (0–20 points)	Natural structures (0–10 points; weight = 2)	The hedgerow total length is less than 25% of the plot perimeter or structures are lacking (0 points); The hedgerow total length is between 25 and 50% of the plot perimeter and/or diameter of isolated trees is 0.5–1 m, and/or wetland size is between 25 and 50 m^2 (5 points); Thehedgerow total length is larger than 50% of the plot perimeter, and/or diameter of isolated trees is > 1 m, and/or wetland size is larger than 50 m^2, and/or microhabitats cover more than 100 m^2 (10 points)
	Artificial structures (0–10 points; weight = 2)	Absent or unsuitable to harbour fauna and flora (0 points); The stone hut's roof or walls are damaged, and/or at least one stone wall is between 25 and 50 m length (5 points); The stone hut is not damaged, and/or at least one stone wall is longer than 50 m (10 points)

2.2.3. *Indicators of landscape structures that support biodiversity*

We positively scored the presence of both natural (hedgerows, isolated trees, old trees and wetlands) and artificial structures (dry stone walls and huts) that provide shelter, food and/or breeding sites and corridors for wildlife. Hedgerows are line arrows of trees, shrubs and herbaceous vegetation around or within a plot. They offer fundamental habitat for wildlife (specially birds, but also invertebrates and plants), as well as numerous ecosystem services (see Collier, 2021 and references herein). Overall, hedge size is positively linked with species richness and abundance of breeding birds (Hinsley and Bellamy, 2000). In addition, large hedgerows can also provide ecological corridors (Collier, 2021). We therefore gave higher scores to largest hedgerows (Table 1). Isolated trees (>50 cm of diameter) were also positively scored because their known ecological roles in carbon storage and provision of wildlife habitat (Lindenmayer, 2017). In the case of stouts and/or old trees, they were given the highest score regardless of their diameter (Table 1).

The presence of small wetlands such as springs and ponds provides microhabitat for aquatic plants, amphibians, birds and several groups of invertebrates, that otherwise would not be present in grasslands (Oertli et al., 2002). Given that larger wetlands support more species (some of which are absent in smaller wetlands; Oertli et al., 2002), Those elements that covered an area (*in situ* estimated) greater than 50 m^2 (the minimum size of ecologically meaningful wetlands on agricultural soil studied in NE Spain; Moreno-Mateos et al., 2010) or that harbour distinctive species (as confirmed during the field work) were awarded maximum score (Table 1). Other microhabitats, including rocks and dense herbaceous vegetation around grasslands were also positively scored (Table 1), because they can provide important habitat for arthropods, and in particular, predators and parasitoids involved in biological control (MacLeod et al., 2004; Clem and Harmon-Threatt, 2021).

Regarding artificial structures, dry stone walls used frequently as boundaries constitute prominent habitat for lichens, mosses and a number of invertebrates (Ruas et al., 2022). If they are long enough they can act as a corridor. As corroborated during field testing performed in pillot areas, some bird species can use dry stone walls and huts as nest sites. These artificial structures were assessed according to their capacity to accommodate fauna and flora species (Table 1).

2.2.4. *Sensitivity of the nature value index to indicators' weighting*

Each indicator received a different weight according to their relevance for nature value (and overall biodiversity) based on expert consensus (Table 1). Nonetheless, different weightings can lead to different outcomes. To assess whether the use of different weightings had an effect on the scores of the nature value index across plots, we performed a sensitivity analysis by comparing values of the index to those obtained using equal and random weightings (between −2 and 2 for each original weighting; 10,000 iterations). The sensitivity of nature value scores to weightings was measured according to the Spearman's rank correlation coefficient. Correlation was high when plots have a similar rank when comparing nature values obtained with two different weightings.

2.3. *Environmental, management and socio-economic variables*

We collected information of variables frequently assessed in the literature of HNV (Oppermann et al., 2012), as well as those related to management practices found in the study area (Table 2). As crop rotation and the addition of liquid manure were rare practices in studied grasslands, their effect was not evaluated. Environmental data was retrieved from existing regional cartography and a Digital Terrain Model (5 m resolution) using GIS, whereas information on management practices of grasslands and socio-economic conditions of farms (i.e. the same for all plot of the same landowner; Table 2) was gathered through interviews with farmers in the year 2020 (see questionnaire in Appendix A).

Table 2
Management, environmental and socio-economic categorical (Cat.) and numeric (Num.) variables evaluated in relation to scores of the nature value index in grasslands of the Western Pyrenees. Categorical variables were grouped according to clusters found using a hierarchical partitioning. To avoid redundancy between variables, only one variable per group (denoted with an asterisk) was used for subsequent analyses.

Cluster	Type	Variable	Source	Categories/measurement unit
1	Cat.	Property regime	Interview	Lease; commonage; property
1*	Cat.	Historical land-use (HLU)	Interview	Arable land; forest; grasslands
2*	Cat.	Liming	Interview	Yes; no
3*	Cat.	Number of treatments (including mineral fertilisation, manure addition or liming) per year	Interview	None; 1; 2; 3
3	Cat.	Mineral fertilisation	Interview	Yes; no
4*	Cat.	Accessibility	Cartography	Good (access through roads, tracks or trails in good condition); poor (access through poor trails); no access
5*	Cat.	Sowing	Interview	Yes; no
5	Cat.	Solid manure	Interview	Yes; no
5	Cat.	Years since last sowing	Interview	<5 yrs; 5–10 yrs; >10 yrs; no sowing
6*	Cat.	Plot location relative to Natura2000 network	Cartography	Inside; outside
7*	Cat.	Current land-use (CLU)	Field survey	Dry grassland; pasture; meadow; mowed and grazed meadow
8*	Cat.	Type of livestock	Interview	None; cattle; horses; sheep; mixed herd (sheep plus any other animal including goats)
	Num.	Slope	Digital Terrain Model	Mean plot slope (%)
	Num.	Elevation	Digital Terrain Model	Mean plot elevation (m)
	Num.	Area	Digital Terrain Model	Plot area (ha)
	Num.	Distance to barn	Cartography	Shortest distance to main farmyard (m)
	Num.	Distance to nearest plot	Cartography	Shortest distance to plots with different management (m)

2.4. Statistical analyses

In preliminary analyses, we found that some variables were strongly related to each other, which may cause multicollinearity issues, leading to inaccurate coefficient estimates and lower statistical power of the mixed model explained below (Zuur et al., 2009). A widely used approach to overcome this problem is to construct synthetic variables using a Principal Component Analysis (or a Multiple Correspondence Analysis in the case of categorical variables), but it comes at the cost of interpretability. A popular alternative is to drop collinear variables (dimension reduction) and keep just one (variable selection; Guyon and Elisseeff, 2003; Fop and Murphy, 2018). To eliminate the redundancy of categorical variables, we first arranged variables into groups of redundant variable (clusters) using a hierarchical clustering algorithm. The number of clusters was estimated based on the resulting dendrogram (Appendix A), and by using a bootstrap approach for identification of the most stable partition with functions included in the 'ClustOfVar' package (Chavent et al., 2012). Next, we selected one variable per group, giving priority to those variables easier to collect and interpret (see Table 2). Reduction of continuous variables was done based on the correlation matrix. From each pair of significantly correlated (Spearman correlation coefficient >0.6) variables, we selected the one with lower overall correlation (i.e. the one less correlated with all other variables; Table 2). The final set of continuous variables were scaled to unit variance and centred on their mean for analyses.

The relationships between the nature value index and environmental, management and socio-economic variables were assessed using a mixed model with municipality as a random effect (intraclass correlation = 0.41). We initially fitted a model with all selected variables plus the interaction between slope and elevation (as a surrogate of terrain suitability for agricultural use) in the fixed term. A plot of residuals against fitted values indicated violations of the assumption of homoscedasticity. To deal with this, we included a residual exponential variance structure of 'distance to barn' variable following the procedure detailed in Zuur et al. (2009). Afterwards, we identified the optimal fixed term using a backward stepwise elimination procedure based on the Akaike Information Criteria. The final model was refitted with Restricted Maximum Likelihood (REML) to reduce bias in parameter estimation due to unbalanced categorical data. The absolute goodness-of-fit of the most parsimonious model was assessed by using the marginal R^2 (variance explained by the fixed effects alone) and the conditional R^2 (variance explained by both fixed and random effects) as formulated by Nakagawa and Schielzeth (2013).

Model validation indicated no problems and no influential points were detected. Variance Inflation Factors (<5 in all cases) evidenced no multicollinearity issues either. Visual inspection of model residuals revealed no signal of spatial correlation. According to the R^2 marginal value the fixed term explained 0.51 of the observed variance, and up to 0.71 including the random effect. Thus, indicating that the model was acceptable for the aim of this study (i.e. to assess the relation of the index with variables), though limited for performing accurate predictions of the nature value index outside the study area. Post-hoc pairwise comparisons of levels of significant categorical variables were performed using Tukey test.

All statistical analyses were performed in R (R Core Team, 2022) using the 'nlme' package (Pinheiro et al., 2022) for mixed models, 'buildmer' (Voeten, 2022) for model selection, and 'emmeans' (Lenth, 2022) for post-hoc pairwise comparisons.

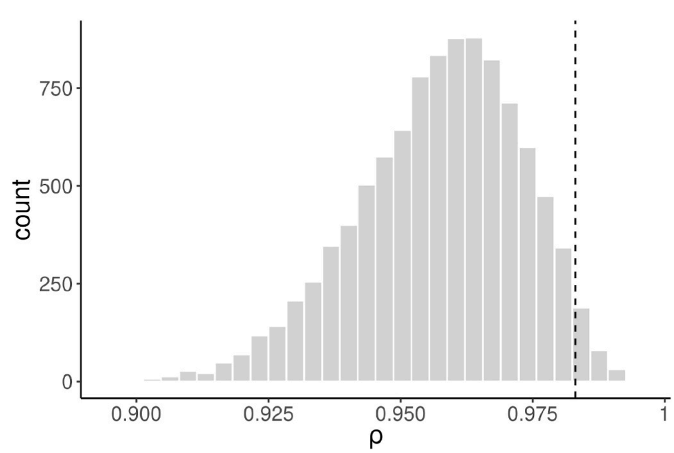

Fig. 3. Histogram of Spearman's rank correlation coefficient (ρ) between nature values of plots obtained using original weightings for indicators and random weightings. Dashed line indicates the correlation coefficient obtained without weighting indicators.

3. Results

3.1. Nature value in grasslands of the Western Pyrenees

We recorded a marked gradient of nature value in grasslands of the Western Pyrenees (mean: 59, SD: 34). Observed gradient was insensitive to weightings assigned to individual indicators, as denoted the high Spearman's correlation coefficients ($\rho_{median} = 0.96$) between nature values obtained with original weightings and those at random (Fig. 3). Higher nature values than the observed mean were found in less accessible areas, located within the Natura2000 network, under low-intensity management (neither treated nor sowed) and mainly grazed by mixed herds formed by sheep and any other type of livestock (Fig. 4). In contrast, plots with lower nature value than the mean laid in plots historically covered by forest and shrubs, which once opened were intensively managed with two and three treatments (liming and/or mineral fertilisation) per year and sowed to produce forage (Fig. 4).

3.2. Relation of the nature value index with environmental, management and socio-economic variables

The final model included six variables related to management practices (sowing, liming, the number of treatment, the type of livestock and the historical and current land use), the level of accessibility, distance to nearest plot, distance to barn, the protection status (inside or outside Natura 2000) and three topographical variables (slope, elevation and their interaction as proxy of area suitability for agricultural use).

Plots included within the Natura2000 network showed significantly higher nature values than those located outside (Table 3). Model coefficient also pointed out a significant negative effect of intensive management practices (number of treatments and sowing) on nature value of grasslands (Table 3). Post-hoc comparisons revealed that application of more than one treatment per year was detrimental in terms of nature value: . In particular, marginal mean of the nature index was ca. 50

Fig. 4. Numeric description of plots with lower and higher nature values than the observed mean. Variable names and their definition are given in Table 2.

Table 3
Estimated regression parameters, standard errors, t-values and P-values for the linear mixed used to assess the effect of management, environment and socio-economic variables on scores of the nature index. Reference categories were 'no' for liming, 'no grazed' for livestock species, 'no' for sowing, '0' for number of treatments per year, 'forest' for historical land-use (HLU), 'dry grassland' for current land-use verified in the field (LU-verified).

	Estimates	Std. Error	*t*-value	*P*-values
(Intercept)	38.07	14.02	2.72	0.008
Accessibility:none	6.98	6.23	1.12	0.265
Accessibility:poor	7.96	4.54	1.75	0.083
Elevation	8.80	3.70	2.37	0.020
Slope	5.85	2.49	2.35	0.021
Elevation*Slope	3.98	2.21	1.80	0.075
Liming:yes	−10.38	11.18	−0.92	0.357
Distance to nearest plot	3.65	2.04	1.79	0.077
Distance to barn	−1.58	2.71	−0.58	0.562
Plot area	−2.32	1.98	−1.18	0.243
Grazing:no grazed	−32.76	11.62	−2.82	0.006
Grazing:sheep	−14.17	9.45	−1.50	0.137
Grazing:cattle	−0.32	8.35	−3.82	<0.000
Grazing:mixed herd	−21.45	9.92	−2.16	0.033
Sowing:yes	-17.23	6.813	−2.53	0.013
Nb. of treatments:1	−11.97	6.74	−1.78	0.079
Nb. of treatments:2	−22.21	7.70	−2.89	0.005
Nb. of treatments:3	−49.42	14.64	−3.37	0.001
HLU:grassland	22.58	11.25	2.01	0.048
HLU:arable land	22.81	12.14	1.88	0.063
LU:pasture	16.50	8.88	1.85	0.067
LU:meadow	38.63	9.99	3.87	<0.000
LU:mowed and grazed meadow	25.69	9.52	2.70	0.008
Natura2000:yes	42.06	2.57	16.40	<0.001

points lower in intensively managed (with 3 treatments per year) than in untreated plots (Fig. 5). Another frequent management practice, liming, was also negatively related to the nature value index, though the effect was not significant.

Lack of sowing and the presence of grazing activities were related to high nature values (Table 3). The highest marginal mean of the nature value index was found in meadows, followed by mown and grazed meadows. Their nature value was between 39 and 26 points greater than in dry grasslands (Table 3 and Fig. 5). Regarding the land-use legacy, the only significant difference was found between plots recently cleared (i.e. previously covered by shrubs and forest) and those covered by grasslands that have remained stable over decades (Table 3 and Fig. 5).

The type of livestock had also a significant effect on the nature value of grasslands (Table 3). Horse grazing led to significantly higher nature value than cattle grazing or no grazing (Fig. 5). Nature value of plots grazed by mixed flock were also higher than in those grazed by cattle (Fig. 5).

Finally, model results indicate that slope and elevation significantly contribute to increase the nature value of plots (Table 3). The greater the value of these variables, the higher the nature value. However, their effect was inferior than those variables related to management practices. Other variables that deemed a certain degree of isolation such as distance to other plots and barn had not significant effect on studied grasslands.

4. Discussion

Measurement of HNV farmlands have been traditionally carried out at landscape scale (e.g. Peppiette, 2011; Lomba et al., 2015; Brunbjerg et al., 2016; Kikas et al., 2017), but the link between low intensity agriculture, biodiversity and natural and structural elements is cross-scale, from landscape to plot (EENRD, 2010). Linking these scales into an indicator has been attempted before (Boyle et al., 2015; Brunbjerg et al., 2016; Tasser et al., 2019). Similarly, here we demonstrate how indicators of biodiversity, intensification and natural and structural elements can be combined into a single index to assess the nature value of grasslands at plot scale. In this study, in addition, we determine the factors contributing most to underpin high nature value farmland. Hence, the methodology here presented does not only provide a more comprehensive picture of biodiversity than most commonly used indicators (e.g. species richness) (Kok et al., 2020), but can also guide management of the grasslands to enhance biodiversity and their natural value. The simplicity of the proposed field survey method, moreover,

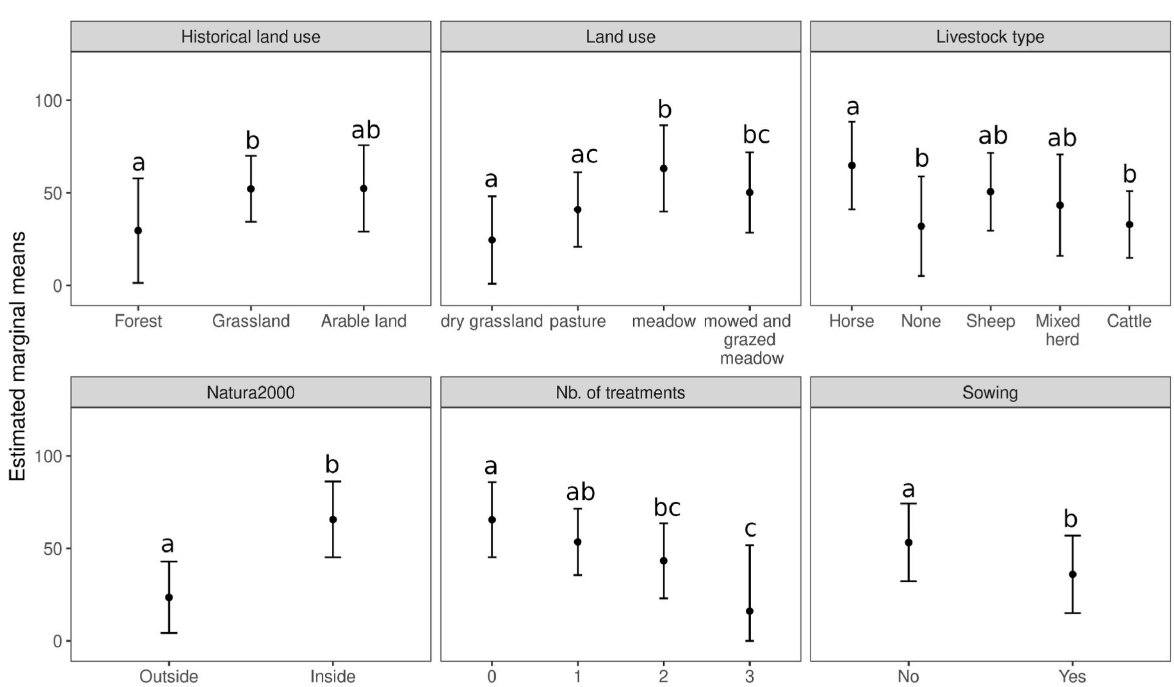

Fig. 5. Post-hoc comparison among level of categorical variables with significant effect on HNV scores according to the fit of the linear mixed model at plot scale. Significant pairwise differences were tested using a Tukey correction and are denoted by different letters. Error bands stand for 95% CI.

makes the index suitable for monitoring success of the subsidized practices in biodiversity-rich grasslands. Although the nature value index was specifically tested for Pyrenean grasslands, it can be easily adapted to other regions by modifying species lists or particular practices to suit the local context (Maher et al., 2015).

Our results are coherent with the essence of the HNV concept, scoring low values to low biodiversity, intensively used grasslands and high values to extensive and high biodiversity grasslands. With this methodology, moreover, we can derive the specific factors and management practices that determine the "extensive management". Previous studies associate HNV with extensive land use in generic terms (Bartolini and Brunori, 2014) or with a limited set of farming practices (Boyle et al., 2015; Brunbjerg et al., 2016; Tasser et al., 2019) that hardly deal with management at plot level. The approach here suggested is similar to that from e.g. Boyle et al. (2015) or Brunbjerg et al. (2016) or Tasser et al. (2019), yet we here provide further understanding of the impact of individual practices and combinations of those. This could help providing guidance and advice to practitioners at field level.

As shown in this study, plots with highest nature values were those extensively mowed and/or grazed by horses and mixed herds. Moreover, no fertilisation treatments, liming or sowing were applied to these meadows. Despite the high biodiversity and cultural values of meadows are long recognized in Europe (Veen et al., 2009), they are becoming a rare agricultural practice (Janišová et al., 2023). In the Pyrenees, many traditional meadows have been transformed to pastures (Ascaso et al., 2020), which as shown in this study, have substantially lower nature values (see Fig. 5). Therefore, reverting this situation should be priority.

The lower nature values found in recently open areas (i.e. in those historically covered by forest and shrublands) suggests that site-specific land-use legacies can constrain HNV condition. Although positive plant indicators (some of which are specialist) can rapidly colonize open areas, at least if they are present in surrounding areas or the seed bank, it may take decades until they reach reference conditions (abundance and distribution) (Waldén and Lindborg, 2016). Therefore, priority should be given to protect and conserve farmland plots with already HNV. This finding stresses that policies should not incentivize the creation of semi-natural habitats or new open areas for livestock farming, but the conservation of farm plots with already HNV and conversion of existing intensively managed low nature value plots to HNV plots. Thus, it would be desirable to design (or reinforce) interventions preventing abandonment of HNV and limiting intensification of farming.

According to our analysis, the type of livestock has a strong effect on the nature value of grasslands. All else being equal, the contribution of horse grazing to nature value of grasslands was substantially greater than that of cattle grazing. Yet, this finding should be handled with care. We cannot discard that observed differences were mediated by differences in stocking rates rather than by the type of livestock. López-i-Gelats et al. (2015), for instance, explain that horses are managed more extensively than cattle in the Pyrenees and many other mountain areas in Europe. The grazing pressure by livestock is usually determined dividing the livestock units by the amount of land utilized (own and rented) by the farm. Such estimation, however, is not accurate enough to be applied at plot level. According to Strohbach et al. (2015), livestock densities cannot be linked to pastures, grassland type and usage when not sufficiently recorded. Grazing has been reported to both benefit and damage biodiversity (Kok et al., 2020) depending on the purpose of the grazing, the intensity and type of management (Kun et al., 2021), or the livestock species because of their different grazing behaviour (Kok et al., 2020; O'Rourke et al., 2016). Hence, we consider that further studies are required to properly capture the different dimensions of grazing in such indicator at plot scale. Under current circumstances, and whatever the reason is, our results suggest that horse grazing is a potential indicator of HNV.

The significantly higher HNV scores of plots located inside Natura 2000 site is another finding that deserves further research. This variable is a recurrent indicator of HNV at regional, national and European scale (Lomba et al., 2014), and as shown in this study, at local scale too. However, the causality between high HNV scores and the Natura2000 network is unclear. Some studies have coupled implementation of the Habitat Directive (Council Directive 92/43/EEC) to extensification of agricultural land use and persistence of semi-natural habitats in farmlands (Anderson and Mammides, 2020; Levin et al., 2018). Others argue that HNV farmland within Natura2000 may be benefited from the protection status and prevent conversion into artificial surfaces (Anderson and Mammides, 2020). In contrast, in many EU countries, HNV farmland within Natura 2000 is threatened by encroachment and forest succession due to land abandonment (Anderson and Mammides, 2020). However, whether policies and management decision implemented inside Natura2000 sites actually contributed to foster or maintain habitat condition in studied plots cannot be supported from our data. This is because our focus on field level, whereas trends in land use change are usually monitored at larger scale. Moreover, it is likely habitat condition in the plots inside Natura2000 sites was already good at the time the Habitat Directive was implemented (which probably motivated their inclusion within the network). Hence, there is a need to strengthen regulations within Natura 2000 (to conserve what is left), but also to expand Natura 2000 to farmland that can still qualify under HNV (Anderson and Mammides, 2020), as well as to strive for more effective use of the measures available through the Common Agricultural Policy (Pe'er et al., 2017).

The baseline information of this index is collected on the field, which may be considered a limitation because it is time consuming and might be difficult to operationalise at large spatial extents (e.g. at national scale). However, on the one hand, observations on the field might be inevitable. When it comes to biodiversity monitoring, for instance, remote sensing approaches still benefit or even require field surveys for validation and completeness (Chandler et al., 2017). On the other hand, field surveys for this type of studies and indicators do not require extensive effort and expertise (compared to e.g. sampling of whole plant communities; Boyle et al., 2015). Therefore, surveys at field level could be conducted by farmers themselves (Tasser et al., 2019) or an extended community in collaboration with the scientists (Danielsen et al., 2022). Involving this extended community is usually coined as community monitoring or citizen science and is developing fast in science and in practice (Danielsen et al., 2022) Involving the community, and particularly the farmers, to monitoring would allow applying the index over significant portions of area, and more importantly can lead to far reaching benefits, such as making farmers more aware of the relationship between management and biodiversity outcomes, incentivising farmers to shift to more biodiversity friendly management, or recognising their efforts towards environmental goals (Tasser et al., 2019; Stolze et al., 2015), as well as building confidence between farmers and stakeholders, empowering farmers by learning about the features they are managing, and engaging in a dialogue with scientists and regulation bodies (Stolze et al., 2015; Birge et al., 2017; van de Gevel et al., 2020; Danielsen et al., 2022). In line with previous studies (Stolze et al., 2015; O'Rourke et al., 2016; Tasser et al., 2019), we consider that this study and the approach here discussed contribute to enhance definitions and better identification of HNV farmland, transforming biodiversity and other ecosystems services objectives into practical support measures for HNV farmland, focus on practices and farming systems that deliver the desired environmental results, and help to be designed in a local context with the involvement and recognition of the relevant stakeholders.

5. Conclusions

We presented an index to assess the nature value of grasslands at plot scale based on field surveys. As shown in this study, this index can be effectively used to identify gradients between low diversity and intensively used agricultural areas and HNV areas at the most basic scale of management. Therefore, it could be useful to guide payment-by-results schemes such as the eco-schemes and the environmental and climate related interventions of the new CAP (2023–2027), The index is based in multiple indicators, which together provide a more comprehensive biodiversity assessment than do conventional indicators such as species richness and related index. Once thresholds of each indicator is defined according to expert knowledge, data could be easily collected in the field, which brings the opportunity to engage non-specialist (e.g. farmers) in the assessment and monitoring of the nature value of grasslands. Importantly, this approach could be extended to any other farming area by slightly adapting the indicators to local context. The results of this study allowed us to identify key management practices that are needed to generate and maintain desired biodiversity achievements in grasslands of the Western Pyrenees. The type of livestock arises as an important factor determining the nature value of grasslands, however, future studies should investigate underlying causes such as grazing intensity and timing and duration of grazing to improve our predictive ability of HNV.

CRediT author statement

Iker Pardo: Methodology, Formal Analyses, Writing – original draft preparation. Silvia Zabalza: Methodology, Data Collection, Data Curation. Asun Berastegi: Methodology, Data Collection. Raimon Ripoll-Bosch: Writing – original draft preparation. Carlos Astrain: Conceptualization, Methodology, Funding adquisition, Project administration.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2023.119516.

References

Andersen, E., Baldock, D., Bennett, H., Beaufoy, G., Bignal, E., Brouwer, F., Elbersen, B., Eiden, G., Godeschalk, F., Jones, G., McCracken, D., Nieuwenhuizen, W., van Eupen, Mi, Hennekens, S., Zervas, G., 2003. Developing a High Nature Value Farming Area Indicator. Internal report for the European Environment Agency. IEEP.

Anderson, E., Mammides, C., 2020. Changes in land-cover within high nature value farmlands inside and outside Natura 2000 sites in Europe: a preliminary assessment. Ambio 49, 1958–1971. https://doi.org/10.1007/s13280-020-01330-y.

Ascaso, J., Reiné, R., Barrantes, O., 2020. Evolution of hay meadows between 1956, 1986, and 2016 and its relation to the characteristics and location of the plots in the valley of the river Esera (Pyrenees, Spain). Agronomy 10, 329. https://doi.org/10.3390/agronomy10030329.

Baldock, D., Beaufoy, G., Bennett, G., Clark, J., 1993. Nature Conservation and New Directions in the EC Common Agricultural Policy: the Potential Role of EC Policies in Maintaining Farming and Management Systems of High Nature Value in the Community. Institute for European Environmental Policy.

Bartolini, F., Brunori, G., 2014. Understanding linkages between common agricultural policy and High Nature Value (HNV) farmland provision: an empirical analysis in Tuscany Region. Agric. Food Econ. 2, 13. https://doi.org/10.1186/s40100-014-0013-2.

Benedetti, Y., 2017. Trends in High Nature Value farmland studies: a systematic review. Eur. J. Ecol. 3, 19–32. https://doi.org/10.1515/eje-2017-0012.

Berastegi, A., López, F., Torres, J., Alzaga, V., Astrain, C., Campión, D., Iragui, U., Clavería, V., Moreno, S., Oroz, J., 2018. Results-based Agri-Environmental Payments Scoring Guidelines. Perennial Crops in the Mediterranean Mosaic Landscape. Report prepared for the European Commission, DG Environment, Agreement No. 07.027722/2014/697042/SUB/B2.

Bignal, E.M., McCracken, D.I., 2000. The nature conservation value of European traditional farming systems. Environ. Rev. 8, 149–171.

Birge, T., Toivonen, M., Kaljonen, M., Herzon, I., 2017. Probing the grounds: developing a payment-by-results agri-environment scheme in Finland. Land Use Pol. 61, 302–315. https://doi.org/10.1016/j.landusepol.2016.11.028.

Boyle, P., Hayes, M., Gormally, M., Sullivan, C., Moran, J., 2015. Development of a nature value index for pastoral farmland—a rapid farm-level assessment. Ecol. Indicat. 56, 31–40. https://doi.org/10.1016/j.ecolind.2015.03.011.

Brunbjerg, A.K., Bladt, J., Brink, M., Fredshavn, J., Mikkelsen, P., Moeslund, J.E., Nygaard, B., Skov, F., Ejrnæs, R., 2016. Development and implementation of a high nature value (HNV) farming indicator for Denmark. Ecol. Indicat. 61, 274–281. https://doi.org/10.1016/j.ecolind.2015.09.027.

Chandler, M., See, L., Copas, K., Bonde, A.M.Z., López, B.C., Danielsen, F., Legind, J.K., Masinde, S., Miller-Rushing, A.J., Newman, G., Rosemartin, A., Turak, E., 2017. Contribution of citizen science towards international biodiversity monitoring. Biol. Conserv. 213, 280–295. https://doi.org/10.1016/j.biocon.2016.09.004.

Chavent, M., Kuentz-Simonet, V., Liquet, B., Saracco, J., 2012. ClustOfVar: an R package for the clustering of variables. J. Stat. Software 50, 1–16. https://doi.org/10.18637/jss.v050.i13.

Clem, C.S., Harmon-Threatt, A.N., 2021. Field borders provide winter refuge for beneficial predators and parasitoids: a case study on organic farms. J. Insect Sci. 21, 1–6. https://doi.org/10.1093/jisesa/ieab027.

Collier, M.J., 2021. Are field boundary hedgerows the earliest example of a nature-based solution? Environ. Sci. Pol. 120, 73–80. https://doi.org/10.1016/j.envsci.2021.02.008.

Moreno-Mateos, D., Pedrocchi, C., Comín, F.A., 2010. Effects of wetland construction on water quality in a semi-arid catchment degraded by intensive agricultural use. Ecol. Eng. 36, 631–639. https://doi.org/10.1016/j.ecoleng.2009.02.003. Carbon, nutrient and metal retention in wetlands in a restoration context.

Danielsen, F., Eicken, H., Funder, M., Johnson, N., Lee, O., Theilade, I., Argyriou, D., Burgess, N.D., 2022. Community monitoring of natural resource systems and the environment. Annu. Rev. Environ. Resour. 47, 637–670.

Dax, T., Schroll, K., Machold, I., Derszniak-Noirjean, M., Schuh, B., Gaupp-Berghausen, M., 2021. Land abandonment in mountain areas of the EU: an inevitable side effect of farming modernization and neglected threat to sustainable land use. Land 10, 591. https://doi.org/10.3390/land10060591.

European Commission, 2017. Technical Handbook on the Monitoring and Evaluation Framework of the Common Agricultural Policy 2014 – 2020. Directorate-General for Agriculture and Rural Development.

European Commission, 2021. List of Potential Agricultural Practices that Eco-Schemes Could Support. Directorate-General for Agriculture and Rural Development.

European Evaluation Network for Rural Development (EENRD), 2010. Working Paper on Approaches for Assessing the Impacts of the Rural Development Programmes in the Context of Multiple Intervening Factors. European Commission, Agriculture and Rural Development. Brussels, Belgium. https://enrd.ec.europa.eu/sites/default/files/eb43a527-c292-f36c-fc51-9ea5b47cedae.pdf. accessed on May 2022.

Fop, M., Murphy, T.B., 2018. Variable selection methods for model-based clustering. Stat. Surv. 12, 18–65. https://doi.org/10.1214/18-SS119.

García-Ruiz, J.M., Lasanta, T., Ruiz-Flano, P., Ortigosa, L., White, S., González, C., Martí, C., 1996. Land-use changes and sustainable development in mountain areas: a case study in the Spanish Pyrenees. Landsc. Ecol. 11, 267–277. https://doi.org/10.1007/BF02059854.

Guyon, I., Elisseeff, A., 2003. An introduction to variable and feature selection. J. Mach. Learn. Res. 3, 1157–1182. https://doi.org/10.1162/153244303322753616.

Hinsley, S.A., Bellamy, P.E., 2000. The influence of hedge structure, management and landscape context on the value of hedgerows to birds: a review. J. Environ. Manag. 60, 33–49. https://doi.org/10.1006/jema.2000.0360.

Hodgson, J.G., Grime, J.P., Wilson, P.J., Thompson, K., Band, S.R., 2005. The impacts of agricultural change (1963–2003) on the grassland flora of Central England: processes and prospects. Basic Appl. Ecol. 6, 107–118. https://doi.org/10.1016/j.baae.2005.01.009.

Iragui, U., Astrain, C., Beaufoy, G., 2010. Sistemas agrarios y forestales de alto valor natural en Navarra. Identificación y monitorización. Gobierno de Navarra, p. 110.

Janišová, M., Bojko, I., Ivascu, C.M., Iuga, A., Biro, A.-S., Magnes, M., 2023. Grazing hay meadows: history, distribution, and ecological context. Appl. Veg. Sci. 26, e12723 https://doi.org/10.1111/avsc.12723.

Karr, J., Dudley, D., 1981. Ecological perspective on water quality goals. Environ. Manag. 5, 55–68. https://doi.org/10.1007/BF01866609.

Keenleyside, C., Beaufoy, G., Tucker, G., Jones, G., 2014. High nature value farming throughout EU-27 and its financial support under the CAP: final report. Report Prepared for DG Environment, Contract No ENV B.1/ETU/2012/0035. Institute for European Environmental Policy, London.

Kikas, T., Bunce, R.G.H., Kull, A., Sepp, K., 2017. New high nature value map of Estonian agricultural land: application of an expert system to integrate biodiversity, landscape

and land use management indicators. Ecol. Indicat. 94, 87–98. https://doi.org/10.1016/j.ecolind.2017.02.008.
Knops, J.M.H., Tilman, D., Haddad, N.M., Naeem, S., Mitchell, C.E., Haarstad, J., Ritchie, M.E., Howe, K.M., Reich, P.B., Siemann, E., Groth, J., 1999. Effects of plant species richness on invasion dynamics, disease outbreaks, insect abundances and diversity. Ecol. Lett. 2, 286–293. https://doi.org/10.1046/j.1461-0248.1999.00083.x.
Kok, A., de Olde, E.M., de Boer, I.J.M., Ripoll-Bosch, R., 2020. European biodiversity assessments in livestock science: a review of research characteristics and indicators. Ecol. Indicat. 112, 105902 https://doi.org/10.1016/j.ecolind.2019.105902.
Kun, R., Babai, D., Csathó, A.I., Vadász, C., Kálmán, N., Máté, A., Malatinszky, Á., 2021. Simplicity or complexity? Important aspects of high nature value grassland management in nature conservation. Biodivers. Conserv. 30, 3563–3583. https://doi.org/10.1007/s10531-021-02262-z.
Lenth, R.V., 2022. Emmeans: Estimated Marginal Means, Aka Least-Squares Means.
Levin, G., Frederiksen, P., Nainggolan, D., 2018. Land-use trends related to Natura2000 sites in Denmark. Land Use Pol. 76, 432–441. https://doi.org/10.1016/j.landusepol.2018.02.018.
Lindenmayer, D.B., 2017. Conserving large old trees as small natural features. Biological Conservation. Small Nat. Featur. 211, 51–59. https://doi.org/10.1016/j.biocon.2016.11.012.
Lomba, A., Guerra, C., Alonso, J., Honrado, J.P., Jongman, R., McCracken, D., 2014. Mapping and monitoring high nature value farmlands: challenges in European landscapes. J. Environ. Manag. 143, 140–150. https://doi.org/10.1016/j.jenvman.2014.04.029.
Lomba, A., Alves, P., Jongman, R.H.G., McCracken, D.I., 2015. Reconciling nature conservation and traditional farming practices: a spatially explicit framework to assess the extent of High Nature Value farmlands in the European countryside. Ecol. Evol. 5, 1031–1044. https://doi.org/10.1002/ece3.1415.
Lomba, A., Moreira, F., Klimek, S., Jongman, R.H., Sullivan, C., Moran, J., Poux, X., Honrado, J.P., Pinto-Correia, T., Plieninger, T., McCracken, D.I., 2020. Back to the future: rethinking socioecological systems underlying high nature value farmlands. Front. Ecol. Environ. 18, 36–42. https://doi.org/10.1002/fee.2116.
López-i-Gelats, F., Rivera-Ferre, M.G., Madruga-Andreu, C., Bartolomé-Filella, J., 2015. Is multifunctionality the future of mountain pastoralism? Lessons from the management of semi-natural grasslands in the Pyrenees. Spanish J. Agric. Res. 13 (4) https://doi.org/10.5424/sjar/2015134-6960MacLeod e0307-e0307.
MacLeod, A., Wratten, S.D., Sotherton, N.W., Thomas, M.B., 2004. 'Beetle banks' as refuges for beneficial arthropods in farmland: long-term changes in predator communities and habitat. Agric. For. Entomol. 6, 147–154. https://doi.org/10.1111/j.1461-9563.2004.00215.x.
Maher, C., Alzaga, V., Astrain, C., Beaufoy, G., Berastegi, A., Bleasdale, A., Byrne, D., Clavería, V., Donaghy, A., Finney, K., Gones, G., Moran, J., O'Donoghue, B., Torres, J., 2015. Developing result-based agri-environment payment schemes for the conservation of species and habitats in Ireland and Spain. In: Ó hUallacháin, D., Finn, J.A. (Eds.), Farmland Conservation with 2020 Vision. Teagasc, Wexford, pp. 28–29. ISBN 978-1-84170-620-7.
Maher, C., Moran, J., Beaufoy, G., Berastegi, A., Bleasdale, A., Byrne, D., Copland, A., Dunford, B., Edge, R., Finney, K., Iragui Yoldi, U., Jones, G., Kelly, S., Lopez Rodriguez, F., McLoughlin, D., O'Donoghue, B., 2018. Results-based Agri-Environmental Payments General Guidance Handbook. Step-by-step Guide to Designing a Results-Based Payments Scheme: Lessons from ireland and spain (Technical report, Report prepared for the European Union, Agreement No. 07.027722/2014/697042/SUB/B2).
Mäkeläinen, S., Harlio, A., Heikkinen, R.K., Herzon, I., Kuussaari, M., Lepikkö, K., Maier, A., Seimola, T., Tiainen, J., Arponen, A., 2019. Coincidence of High Nature Value farmlands with bird and butterfly diversity. Agric. Ecosyst. Environ. 269, 224–233. https://doi.org/10.1016/j.agee.2018.09.030.
McGinlay, J., Gowing, D.J.G., Budds, J., 2017. The threat of abandonment in socio-ecological landscapes: farmers' motivations and perspectives on high nature value grassland conservation. Environ. Sci. Pol. 69, 39–49. https://doi.org/10.1016/j.envsci.2016.12.007.
Moran, J., Byrne, D., Carlier, J., Dunford, B., Finn, J., Ó hUallacháin, D., Sullivan, C., 2021. Management of high nature value farmland in the Republic of Ireland: 25 years evolving toward locally adapted results-orientated solutions and payments. Ecol. Soc. 26 https://doi.org/10.5751/ES-12180-260120.
Morelli, F., Jerzak, L., Tryjanowski, P., 2014. Birds as useful indicators of high nature value (HNV) farmland in Central Italy. Ecol. Indicat. 38, 236–242. https://doi.org/10.1016/j.ecolind.2013.11.016Moreno-Mateos.
Nakagawa, S., Schielzeth, H., 2013. A general and simple method for obtaining R2 from generalized linear mixed-effects models. Methods Ecol. Evol. 4, 133–142. https://doi.org/10.1111/j.2041-210x.2012.00261.x.
Oertli, B., Joye, D.A., Castella, E., Juge, R., Cambin, D., Lachavanne, J.-B., 2002. Does size matter? The relationship between pond area and biodiversity. Biol. Conserv. 104, 59–70. https://doi.org/10.1016/S0006-3207(01)00154-9.
Oppermann, R., Beaufoy, G., Jones, G., 2012. High Nature Value Farming in Europe, vol. 2012. Ubstadt-Weiher, Germany, p. 544. Verlag, Regionalkultur.
O'Rourke, E., Charbonneau, M., Poinsot, Y., 2016. High nature value mountain farming systems in Europe: case studies from the Atlantic Pyrenees, France and the Kerry Uplands, Ireland. J. Rural Stud. 46, 47–59. https://doi.org/10.1016/j.jrurstud.2016.05.010.
European Forum on Nature Conservation and Pastoralism (EFNCP), 2018. Pilot results-based agri-environment measures in Ireland and Navarra. Action: call for proposals-Pilot on-farm projects to test result-based remuneration schemes for the enhancement of biodiversity. End of Project Technical Synthesis Report.
Peeters, A., Beaufoy, G., Canals, R.M., De Vliegher, A., Huyghe, C., Isselstein, J., Jones, G., Kessler, W., Kirilov, A., Mosquera-Losada, M.R., Nilsdotter-Linde, N., Parente, G., Peyraud, J.-L., Pickert, L., Plantureux, S., Porqueddu, C., Rataj, D., Stypinski, P., Tonn, B., van den Pol – van Dasselaar, A., Vintu, V., Wilkins, R.J., 2014. Grassland term definitions and classifications adapted to the diversity of European grassland-based systems. In: Grassland Science in Europe. Presented at the EGF at 50. the Future of European Grasslands.
Peppiette, Z. (Ed.), 2011. The Challenge of Environmental Monitoring: the Example of HNV Farmland. https://doi.org/10.22004/ag.econ.99586.
Pe'er, G., Lakner, S., Müller, R., Passoni, G., Bontzorlos, V., Clough, D., et al., 2017. Is the CAP Fit for Purpose? an Evidence-Based Fitness-Check Assessment. German Centre for Integrative Biodiversity Research (iDiv) Halle-Jena-Leipzig.
Pinheiro, J., Bates, D., R Core Team, 2022. Nlme: Linear and Nonlinear Mixed Effects Models.
R Core Team, 2022. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.
Regulation (EU), 2021. 2115 of the European Parliament and of the Council of 2 December 2021 establishing rules on support for strategic plans to be drawn up by Member States under the common agricultural policy (CAP strategic plans) and financed by the European agricultural guarantee fund (EAGF) and by the European agricultural fund for rural development (EAFRD) and repealing regulations (EU) No 1305/2013 and (EU) No 1307/2013. Orkesterjournalen L 435, 1–186, 6.12.2021.
Ruas, S., Rotchés-Ribalta, R., Ó hUallacháin, D., Volpato, A., Gormally, M., White, B., Moran, J., 2022. Assessing stone walls habitat quality – which factors affect bryophytes and macrolichens on farmland stone walls in Ireland? Ecol. Indicat. 139, 108948 https://doi.org/10.1016/j.ecolind.2022.108948.
Schmitz, M.F., Arnaiz-Schmitz, C., Sarmiento-Mateos, P., 2021. High nature value farming systems and protected areas: conservation opportunities or land abandonment? A study case in the Madrid region (Spain). Land 10, 721. https://doi.org/10.3390/land10070721.
Stolze, M., Frick, R., Schmid, O., Stöckli, S., Bogner, D., Chevillat, V., Dubbert, M., Fleury, P., Neuner, S., Nitsch, H., Plaikner, M., Schramek, J., Tasser, E., Vincent, A., Wezel, A., 2015. Result-oriented Measures for Biodiversity in Mountain Farming – A Policy Handbook. Frick: Research Institute of Organic Agriculture (FiBL).
Strohbach, M.W., Kohler, M.L., Dauber, J., Klimek, S., 2015. High Nature Value farming: from indication to conservation. Ecol. Indicat. 57, 557–563. https://doi.org/10.1016/j.ecolind.2015.05.021.
Tasser, E., Rüdisser, J., Plaikner, M., Wezel, A., Stöckli, S., Vincent, A., Nitsch, H., Dubbert, M., Moos, V., Walde, J., Bogner, D., 2019. A simple biodiversity assessment scheme supporting nature-friendly farm management. Ecol. Indicat. 107, 105649 https://doi.org/10.1016/j.ecolind.2019.105649.
Tilman, D., Reich, P.B., Knops, J.M.H., Wedin, D.A., Mielke, T., Lehman, C., 2001. Diversity and productivity in a long-term grassland experiment. Science 294, 843–845. https://doi.org/10.1126/science.1060391.
van de Gevel, J., van Etten, J., Deterding, S., 2020. Citizen science breathes new life into participatory agricultural research. A review. Agron. Sustain. Dev. 40, 35. https://doi.org/10.1007/s13593-020-00636-1.
van Vliet, J., de Groot, H.L.F., Rietveld, P., Verburg, P.H., 2015. Manifestations and underlying drivers of agricultural land use change in Europe. Landsc. Urban Plann. 133, 24–36. https://doi.org/10.1016/j.landurbplan.2014.09.001.
Veen, P., Jefferson, R., de Smidt, J., van der Straaten, J. (Eds.), 2009. Grasslands in Europe of High Nature Value. KKNV, Deen Haag, The Netherlands.
Voeten, C.C., 2022. Buildmer: Stepwise Elimination and Term Reordering for Mixed-Effects Regression.
Waldén, E., Lindborg, R., 2016. Long term positive effect of grassland restoration on plant diversity - success or not? PLoS One 11, e0155836. https://doi.org/10.1371/journal.pone.0155836.
Zuur, A.F., Ieno, E.N., Walker, N., Saveliev, A.A., Smith, G.M., 2009. Mixed Effects Models and Extensions in Ecology with R, Statistics for Biology and Health. Springer New York, New York, NY. https://doi.org/10.1007/978-0-387-87458-.