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Aymeric Mondière [a], Michael S. Corson [a,*], Lou Valence [b], Loïs Morel [c,d], Hayo M.G. van der Werf [a]

[a] UMR SAS, INRAE, Institut Agro, Rennes, France
[b] ARTIFEX, Albi, France
[c] UMR DECOD (Ecosystem Dynamics and Sustainability), Institut Agro, INRAE, Rennes, France
[d] UMR LETG, CNRS, Université de Nantes, Nantes, France

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ABSTRACT

The loss of biodiversity in agricultural landscapes is caused mainly by intensive agricultural production, especially livestock farms. However, certain farms that favor biodiversity attempt to decrease this loss, and the biodiversity hosted by these farms needs to be assessed. To this end, certain methods assess integrated variables of the overall state of an ecosystem. Among them, the "rewilding score" is based on assessing human forcing (i.e. inputs used in the ecosystem and products exported from the ecosystem) and ecological integrity to consider short-term and long-term effects of human activities on ecosystems. This study aimed to adapt the rewilding score for cattle-oriented farms, apply the method, and compare its results to observed biodiversity in order to discuss the relevance of the adapted rewilding score for assessing the biodiversity potential of livestock farms. Two adapted rewilding scores were tested with seven farms by combining one assessment of human forcing with two approaches for estimating ecological integrity. Biodiversity indices based on bird species inventories were calculated and compared to the adapted rewilding scores. Their moderate correlations with the adapted rewilding scores ($r = 0.54$–0.69) supported using this score to assess one type of biodiversity potential of agroecosystems. Nonetheless, the correlation between the adapted rewilding score and biodiversity remains to be confirmed by using biodiversity indices for other taxonomic groups and for a larger set of farms. This method could be used as a decision aid by farmers or as a tool to help governments calculate subsidies by considering short-term and long-term effects of livestock farms on biodiversity.

1. Introduction

The loss of biodiversity in agricultural landscapes is caused mainly by intensive agricultural production (Rigal et al., 2023; Tscharntke et al., 2005), especially livestock farms (Leip et al., 2015). However, not all livestock farms cause biodiversity to decline (Duru et al., 2015). Agroecological livestock farms base their production on ecological processes to decrease their use of inputs such as fertilizers and pesticides by using ecosystem services provided by the agroecosystem (Kremen and Miles, 2012). By promoting ecological processes, they preserve the biodiversity (Bommarco et al., 2013) on which their production depends. Certain livestock farms go further by considering the reversal of biodiversity loss (for its intrinsic value and not for its function of agricultural production) as the primary objective (Mondière et al., 2022). In Europe, this is advocated, for example, by the French organization "Paysans de nature", whose member farmers consider their farms as nature reserves (Paysans de Nature, 2021) that they manage in order to leave room for "wild nature".

Another approach for supporting biodiversity aims to restore and or conserve an ecosystem, habitat, or species by stopping human activities (or nearly so) in order to restore ecological processes (Carver et al., 2021; Fernández et al., 2017). This vision corresponds to "rewilding", defined as "the process of rebuilding, following major human disturbance, a natural ecosystem by restoring natural processes and…the…food web…as a self-sustaining and resilient ecosystem with biota that would have been present had the disturbance not occurred" (Carver et al., 2021). Agricultural rewilding is an emerging form of land use that combines restoration of ecological processes with some degree of

* Corresponding author.
E-mail address: michael.corson@inrae.fr (M.S. Corson).

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agricultural production, most often of herbivores (Corson et al., 2022). In the present study, we considered farms in the "Paysans de nature" network and agricultural rewilding projects as "biodiversity-friendly farms".

Many indicators and methods exist to assess biodiversity on livestock farms or effects of livestock on biodiversity (FAO, 2015; Kok et al., 2020). In a review of effects of livestock on biodiversity in Europe, Kok et al. (2020) classified studies by the context (i.e., country), scale (e.g., genetic, field, farm, landscape or larger), species, function of livestock assessed (i.e., food production, conservation, or both), approach (i.e., measured, modelled, or qualitative scoring), and indicators used. The many differences among studies reflect the difficulty of assessing biodiversity on livestock farms in a general manner. The most appropriate assessment method depends on the conservation objective (e.g., biodiversity for its own sake or to supply ecosystem services) (Kleijn et al., 2011), and despite the variety of methods available, it can be difficult to find one completely adapted to a given objective (Bockstaller et al., 2011; LEAP, 2015). Because of these factors, each of the many methods developed to date can assess only one part of the biodiversity of livestock farms (Kok et al., 2020).

Instead of assessing biodiversity itself, certain methods assess integrated variables of the overall state of ecosystems, such as their "ecological potential" (Delzons et al., 2021), "health" (Quinn et al., 2012), "restoration" (Gann et al., 2019), "naturalness" (Guetté et al., 2018), "ecological integrity" (Carter et al., 2019), or the "damages" that they experience (Huijbregts et al., 2016). These methods describe the state of ecosystems as a function of the human disturbances that they have experienced and can be based, among others, on indicators that relate agricultural practices to biodiversity (Manneville et al., 2014) or on models that predict potential impacts of pollutant emissions and resource use on ecosystems (Huijbregts et al., 2016).

We sought a relatively simple and operational method to assess the potential of extensive livestock farms, as a function of their practices and land use, to host overall biodiversity. Several existing methods had a conceptual framework and/or indicators that seemed to meet this objective, including life cycle assessment (LCA), the index of ecological potential (IPE) (Delzons et al., 2021), BIOTEX (Manneville et al., 2014), and the framework of Torres et al. (2018). To distinguish relative impacts of small-scale practices on biodiversity, the FAO (2015) suggests using indicators of the Pressure-State-Response framework rather than LCA, which aggregates all inputs and emissions associated with production of a given product. LCA also seems less adapted for assessing conservation objectives, in which food production is less relevant (Kok et al., 2020). IPE, developed by the French Natural History Museum, assesses the biodiversity and ecological functionality of human-impacted sites by combining data on species abundance and diversity, as well as indicators of ecological functionality (Delzons et al., 2021). BIOTEX, developed by the French Livestock Institute, assesses the biodiversity hosted by mixed crop-livestock farms based on indicators of farming practices and landscape composition (Manneville et al., 2014).

Torres et al. (2018) developed a conceptual framework that quantifies a "rewilding score" that assesses the degree of rewilding of an ecosystem and positions it along a gradient of naturalness. The framework combines two dimensions: (i) human forcing (defined as the inputs used in the ecosystem and products exported from the ecosystem), considered as a proxy of current effects of human management on the ecosystem, and (ii) ecological integrity (defined as a function of the "naturalness of disturbances and stochastic events, …the connectivity of terrestrial and aquatic systems and…the composition and complexity of the trophic network" (Torres et al., 2018)), considered as a proxy of past effects of human management the ecosystem.

We were interested in LCA's ability to aggregate processes and their environmental burdens, the ability of IPE and BIOTEX to assess biodiversity, and the inclusion of human forcing in the framework of Torres et al. (2018), but each method on its own was not exactly what we sought. In particular, BIOTEX had difficulty distinguishing the biodiversity potential of extremely extensive farms and, except for grasslands, did not consider the intensity of land use. Fortunately, the framework of Torres et al. (2018) is designed to be adapted to new systems by refining, changing, and/or adding indicators. We thus focused on adapting their framework into a method to quantify restoration of ecological processes in livestock systems without explicitly considering biodiversity, and hypothesized that restoring these processes favors biodiversity. To this end, we replaced some of the framework's indicators with pressure and state indicators from LCA, IPE, and BIOTEX. Using the adapted rewilding score (ARS) as a proxy of the biodiversity potential, we adapted the framework to obtain an operational assessment of biodiversity potential that requires few specific or complex skills to apply, which is a key criterion for adoption by stakeholders (Dardonville et al., 2022; Jeanneret et al., 2014).

To this end, the objectives of this study were to (i) adapt the method for calculating the rewilding score to livestock farms, (ii) apply it to cattle-oriented farms that considered biodiversity to differing degrees in their production strategy, and (iii) compare the ARS to biodiversity observed on each of the farms. This approach allowed us to assess the relevance of using the ARS to estimate the biodiversity potential of cattle-oriented farms.

2. Materials and methods

2.1. Adapting the framework for calculating the rewilding score

We adapted the human forcing and ecological integrity dimensions of the framework of Torres et al. (2018) (hereafter "Torres method") using indicators from other methods.

2.1.1. Estimating human forcing

In estimating human forcing, Torres et al. (2018) were inspired by the "cultural energy" index of Anderson (1991), defined as the quantity of energy of human origin (e.g., fossil, nuclear, renewable) required to maintain the current functioning of an ecosystem. To simplify assessment by practitioners, Torres et al. (2018) quantified human forcing as a function of specific types of human inputs into rewilding projects that influence wildlife species (i.e., artificial feeding, population reinforcement) or their habitats (i.e., carrion or deadwood removal). They also considered specific human activities that export products from the area of these projects: hunting, fishing, agriculture, forestry, grassland production/grazing, and mining. To adapt the method to livestock systems, we reverted to the approach of Anderson (1991) by considering only the quantity of energy of human origin input into a livestock system.

We used LCA to estimate this energy as the indicator "cumulative energy demand" (Mondière et al., in review). It estimated the energy of human origin used directly (i.e., on the farm) and indirectly (i.e., to produce and supply inputs for the livestock system), expressed in MJ per ha of utilized agricultural area. We included the energy used directly to produce livestock or indirectly to produce crops fed to them but excluded the energy used to produce crops not fed to the animals (e.g., cash crops). See Mondière et al. (in review) for details about the LCA performed. To obtain a final score for human forcing from 0 (minimum) to 1 (maximum), cumulative energy demand was divided by a reference value that represented the maximum cumulative energy demand for the types of livestock systems subsequently analyzed.

2.1.2. Estimating ecological integrity

Since the Torres method focuses on relatively natural ecosystems at a large scale (hundreds to thousands of ha), it seemed necessary to adapt its estimation of ecological integrity to apply it to the smaller scale of livestock farms (tens to hundreds of ha). We developed two methods to estimate ecological integrity: (i) adapting the initial Torres method or (ii) adapting the IPE using a field study. Two people (A.M. and L.V.) collected data and implemented the two methods for the farms in 2021.

2.1.2.1. Adapting the ecological integrity of the Torres method. The Torres method estimates ecological integrity by calculating nine indicators that assess three ecological principles: the naturalness of disturbances and stochastic events, connectivity of terrestrial and aquatic systems, and complexity of the trophic network (Table 1). The indicators, calculated using qualitative and/or semi-quantitative approaches, are aggregated into a final score of ecological integrity that ranges from 0 (minimum) to 1 (maximum).

We adapted several of the indicators used to assess ecological integrity (Table 1). Among the four indicators of disturbance regimes, "natural avalanche and/or rock slide regimes" and "natural fire regimes" were not changed, since these regimes did not differ among the farms in the sample. In contrast, two indicators were adapted to consider agricultural practices explicitly in order to distinguish the farms: "natural hydrological regimes" considered the presence of drainage, irrigation, and ditch maintenance, while "natural pest regimes and mortality events" considered the proportion of the farm that did not receive pesticides.

Among the four indicators of landscape connectivity and composition, "terrestrial landscape fragmentation" was modified because livestock farms are smaller than the rewilding projects assessed by the original method, and thus their fragmentation by human infrastructure is more difficult to assess. To quantify terrestrial landscape connectivity, we used an indicator of the density and connectivity of semi-natural areas developed in BIOTEX. It is based on cartographic analysis of the distribution of semi-natural areas on a farm and the region in which it is located. The indicator "aquatic landscape fragmentation" was excluded because the farm scale is too small to consider it. The indicator "spontaneous vegetation dynamics" originally weighted the proportion of area where vegetation was allowed to grow spontaneously by coefficients associated with three age classes (<50, 50–200, and > 200 years). For this indicator, we split the first age class into two (<15 and 15–50 years) to capture the rapid rate at which vegetation can become established on fallow land (Šebelíková et al., 2016) (Table 1). Then, because few farms can leave a large proportion of land fallow, we also added consideration of areas with low degrees of human management (i.e., semi-natural areas and permanent grasslands), but weighted their contribution to the indicator score at half that of fallow land (Table 1). The indicator "harmful invasive species" was modified to focus exclusively on invasive plant species, using the qualitative indicator "invasive plant species" from the IPE (Delzons et al., 2021).

Trophic processes are assessed using a single indicator of terrestrial animals heavier than 5 kg, including both wildlife and domestic species (e.g., Galloway cattle) (Torres et al., 2018). The indicator considers the proportion of total area on which each species is present (e.g., grazing area), the proportion of the year that it is present (e.g., grazing period), and the viability of the species' local population. The viability of domestic animals on the farms was assumed to equal 1. As in the initial method, the score of each principle was calculated as the arithmetic mean of its indicator(s), and the final score of ecological integrity (range: 0–1) was calculated as the geometric mean of the scores of three principles.

Table 1
Indicators of ecological integrity of the method of Torres et al. (2018) and their adaptations in the present study. The score of each principle equals the mean of the scores of its indicators, and the score of ecological integrity equals the mean of the scores of the principles.

Prin-ciple	Indicator	Initial method		Method adapted to agroecosystems	
		Definition	Calculation	Adaptation	Calculation
Disturbance regime	Natural avalanche and/or rock slide regimes	Degree of regulating risks of avalanches and rock slides	Qualitative from 0 (maximum regulation) to 1 (no regulation)	None	Qualitative from 0 (maximum regulation) to 1 (no regulation)
	Natural fire regimes	Degree of regulating risks of fire via prescribed burning and/or fire suppression	Qualitative from 0 (maximum regulation) to 1 (no regulation)	None	Qualitative from 0 (maximum regulation) to 1 (no regulation)
	Natural hydrological regimes	Degree of modification of the natural hydrological regime	Qualitative from 0 (maximum modification) to 1 (no modification)	Focused on the presence of irrigation and drainage	Qualitative classes: drainage and irrigation present = 0, drainage or irrigation present = 0.50, ditch management alone = 0.75, no drainage or irrigation = 1
	Natural pest regimes and mortality events	Degree of regulation of pests and management of mortality events	Qualitative from 0 (maximum regulation and management) to 1 (no regulation or management)	Focused on pesticide use	Proportion of farm area that receives no pesticide applications
Landscape composition and connectivity	Terrestrial landscape fragmentation	Degree of landscape fragmentation by human infrastructure	Qualitative from 0 (maximum fragmentation) to 1 (minimum fragmentation)	Use of an indicator of the density and connectivity of semi-natural areas (Manneville et al., 2014)	Density of human infrastructure in the landscape calculated from a simple cartographic study
	Aquatic landscape fragmentation	Degree of disturbance of migratory processes in river systems	Qualitative from 0 (maximum disturbance) to 1 (no disturbance)	Not considered	–
	Spontaneous vegetation dynamics	State of the natural regeneration of vegetation	$\sum SVD \times TSVD$: proportion of area where spontaneous vegetation dynamics are allowed; T: time since abandonment (years) (<50 = 0.1, 50–200 = 0.5, > 200 = 1)	State of the natural regeneration of vegetation	$\sum (SVD \times T + SN \times T \times 0.5) SVD$: proportion of farm area where spontaneous vegetation dynamics are allowed; SN: proportion of semi-natural and permanent grassland area on the farm; T: age of the area (years) (<15 = 0.10, 15–50 = 0.33, 51–200 = 0.50, > 200 = 1)
	Harmful invasive species	Impact of harmful invasive species	Qualitative from 0 (maximum impact) to 1 (no impact)	Focused on invasive plant species (Delzons et al., 2021)	Qualitative from 0 (maximum impact) to 1 (no impact)
Trophic processes	Terrestrial fauna heavier than 5 kg	Species composition of fauna heavier than 5 kg	$\sum S_i \times T_i \times V_i S$: proportion of maximum area occupied by species i; T: proportion of the year that species i is present in this area; V: viability of the population of species i (0–1)	None	$\sum S_i \times T_i \times V_i S$: proportion of maximum area occupied by species i; T: proportion of the year that species i is present in this area; V: viability of the population of species i (0–1)

2.1.2.2. Adapting the index of ecological potential to assess ecological integrity. The IPE (Delzons et al., 2021) has the advantage of being "ready-to-use", applicable at the farm scale, and based on easy-to-use qualitative indicators. The IPE assesses the ecological potential of a site based on nine indicators of its diversity (two), functionality (five), and natural heritage (two) (Table 2). It differs from the ecological integrity of the Torres method in not considering stochasticity, but in explicitly considering indicators of biodiversity and natural heritage (defined by UNESCO (1972) as "natural features, geological and physiographical formations and delineated areas that constitute the habitat of threatened species of animals and plants and natural sites of value from the point of view of science, conservation or natural beauty"). The IPE is based on cartographic analyses (to provide the ecological context), field studies (i.e., a survey route to record the habitats present and qualify indicators of ecological functionality), and bird surveys (Delzons et al., 2021). Habitats are described using level 4 habitat classifications of the European Nature Information System (EUNIS) (European Environment Agency, 2021). An indicator of undeveloped area considers the percentage of farm area not covered by crops, temporary grasslands, or infrastructure. A single score for the IPE can be calculated by multiplying the score of the eight positive indicators by its assigned weight (range: 5–20 %), summing the weighted scores, and then adding the negative indicator "invasive plant species" (range: −0.5 % to −3.5 %) (Delzons, 2015).

To adapt the IPE to assess the ecological integrity of livestock farms, we first decided that if a farm contained non-contiguous fields, we would apply it only to the main group of fields around the farmstead, since the IPE was not designed to assess fragmented sites. In addition, we included consideration of EUNIS habitats described at lower detail (i.e., level 3, and one to level 2), since we were unable to describe all habitats to level 4. These adaptations were validated by the main developer of the IPE (O. Delzons, Muséum National d'Histoire Naturelle, Paris). We also excluded the indicators "bird diversity" and "heritage species" (weights of 15 % and 20 %, respectively) to ensure that the adapted IPE would not be based on the bird surveys used subsequently to evaluate the utility of the ARS that was based on it. Finally, we distributed their 35 % of total weight proportionally to the six remaining positive indicators (Table 2) and expressed the percentage scores as proportions before averaging them into a final score of ecological integrity (range: 0–1).

2.1.3. Calculating the adapted rewilding score

The human forcing and ecological integrity scores, which consider effects of human management in the short term and long term, respectively, were combined into the ARS (range: 0–1) as in the original method: ARS = *ecological integrity score* × (1 – *human forcing score*).

2.2. Application

We applied the adapted method to a sample of seven farms in western France and southeastern England with a wide variety of (i) use of external inputs and (ii) consideration of environmental issues, in particular biodiversity, in their farm strategy. We considered the farms that included biodiversity explicitly in their production strategies as "biodiversity-friendly". The sample farms ranged in intensity from extremely extensive (0.18 livestock units (LU, i.e., a 600 kg cow) ha^{-1}) to moderately intensive (ca. 1.3 LU/ha), without reaching the highest intensities observed in these regions (Foray et al., 2022) (Fig. 1, Table 3):

- Knepp (K): an agricultural rewilding project in England started in 2001 (Knepp Wildland), which combines restoration of ecological processes with low agricultural production (Corson et al., 2022). Specific actions to restore ecological processes and thus favor biodiversity include restoring habitats, introducing species, and extensive grazing. Only Knepp's "southern block", the most extensively documented part of the system, was analyzed. Its livestock production comes from managing herds of traditional breeds of cows, pigs, and ponies, as well as two deer species, on a contiguous fenced area of 450 ha composed of a mosaic of grassland, shrubland, and woodland.
- La Barge (B): an organic suckler beef farm in western France certified with a "Nature and Progress" certification of environmental quality and social equity, a member of the "Paysans de nature" network, and located in a coastal marshland landscape (i.e., ditches, pools, and little woody vegetation). Actions to favor biodiversity include flooding meadows, creating ponds, and extensive grazing.
- Saint Laurent de la Prée (S): an organic suckler beef farm operated as an experimental unit by INRAE in western France that researches agroecological production of beef in association with field cropping (Roche et al., 2022), located partially in a coastal marshland landscape equivalent to that of B's, and partially in a coastal marshland with hedgerows. Actions to favor biodiversity include creating ponds; planting grass or flower strips on arable land, refuge strips in

Table 2
Indicators of the index of ecological potential (Delzons et al., 2021) by dimension (Dim), their original weights (Delzons, 2015), and the modified weights when used to estimate ecological integrity in the present study.

Dim	Indicator	Definition	Calculation	Original weight	Modified weight
Diversity	Habitat diversity	Number of habitats (EUNIS level 4; levels 3 and 2 also considered in the present study)	Number of habitats divided by 25, expressed as a percentage and capped at 100 %	15 %	23.0 %
	Bird diversity	Number of bird species	Number of species divided by 50, expressed as a percentage and capped at 100 %	15 %	0 %
Functionality	Undeveloped area	Quantification of the undeveloped area	Percentage of farm area not covered by crops, temporary grasslands, or infrastructure	5 %	7.7 %
	Invasive plant species	Qualitative assessment of the presence of invasive plant species and their potential impact on the local ecosystem	Grade of A (87 %), B (62 %), C (37 %), or D (12 %)	–4%[a]	–4.0 %[a]
	Hosting potential	Qualitative assessment of the presence of microhabitats for hosting species	Grade of A (87 %), B (62 %), C (37 %), or D (12 %)	10 %	15.4 %
	Permeability	Qualitative assessment of the permeability to fauna	Grade of A (87 %), B (62 %), C (37 %), or D (12 %)	5 %	7.7 %
	Ecological networks	Qualitative assessment of ecological networks	Grade of A (87 %), B (62 %), C (37 %), or D (12 %)	20 %	30.8 %
Natural heritage	Heritage habitats	Quantification of the area covered by heritage habitats	Percentage of the area covered by heritage habitats	10 %	15.4 %
	Heritage species	Number of species[b] considered threatened on regional, national, or European Red Lists	Number of Red List species divided by 20, expressed as a percentage and capped at 100 %	20 %	0 %

[a] subtracted from the total after summing the other indicators.
[b] species considered: plants, birds, amphibians, reptiles, butterflies, and dragonflies.

Fig. 1. Locations of the seven farms in the sample in France and the United Kingdom.

meadows, and hedges; decreasing field sizes; and increasing crop diversity.

- Thorigné d'Anjou (Th): an organic suckler beef farm operated as an experimental unit by the Chamber of Agriculture of the Pays de la Loire region in west-central France that researches feed self-sufficiency in organic crop-livestock systems (Farm-XP, 2021).
- Trévarn (Tr): an organic dairy farm in extreme northwestern France that applies agroecological practices and is located in a hedgerow landscape with scattered woodlands (Glinec, 2019). Actions to favor biodiversity include transforming arable land to permanent grassland, maintaining hedges and embankments, and creating brush piles.
- Oasys (O): a dairy farm operated as an experimental unit by INRAE in western France that researches adaptation of dairy production to climate change (Novak et al., 2020) and is located in a plain landscape with scattered hedgerows.
- Derval (D): a dairy farm operated as an experimental unit by the Chamber of Agriculture of the Pays de la Loire region in west-central France that represents a typical dairy farm in western France and is located in a landscape with partially restored hedgerows (Farm-XP, 2022).

One person (L.V.) surveyed all six French farms in the sample in spring 2021 using the IPE protocol (Delzons et al., 2021) to collect data on the presence of EUNIS habitats, bird diversity and, according to regional lists (Conservatoire Botanique National), plant species considered invasive (Table S1). For farm K, EUNIS habitats and invasive plant species were assessed by another person (A.M.) during a visit in Oct 2021. We calculated the indicator of terrestrial animals heavier than 5 kg using only data on managed animals (i.e., Longhorn cattle, Tamworth pigs, Exmoor ponies, fallow deer, and red deer on K; domestic cattle on all other farms), since we had only anecdotal data on the presence of wildlife on these farms. The energy demand of the farms, which ranged from 90 to 20 496 MJ ha^{-1} (Mondière et al., in review), was normalized into the indicator of human forcing (range: 0–1) by dividing it by 37 000 MJ ha^{-1}, which corresponded to the energy demand estimated for intensive dairy farms in western France (Salou et al., 2017). Most of the energy demand of the farms in the sample came from direct (i.e., on-field) energy use (Fig. S1).

2.3. Assessment of avian biodiversity

To provide an initial assessment of the utility of using the ARS to assess the biodiversity potential of the farms, we compared the ARS to simple measurements of biodiversity obtained from bird surveys. We did not include other taxonomic groups because birds are good indicators of a site's overall biodiversity, which was sufficient for this initial assessment (Chiatante et al., 2021). Birds occur high in food webs and are sensitive to environmental changes, and it is relatively inexpensive to survey them (Gregory and van Strien, 2010). During the bird survey of the IPE protocol, the bird species present on the French farms were detected in the morning by sight, sound, or physical traces (e.g., feathers, nests) based on two 5-min observation periods at ca. 10 points along a survey route on each farm. Each point covered an area 300 m in diameter, and the set of points covered all habitats in the main group of fields of each farm (Fig. S2). For farm K, logistical constraints required that we use data from 10 bird surveys performed from late Mar to mid-June 2018 (James, 2018), each consisting of a slow walk along a permanent transect in the "southern block" to detect birds by sight or sound according to the method of the Common Birds Census of the British Trust for Ornithology (Marchant, 1983). These data were then analyzed to calculate two indicators of avian biodiversity: species richness (i.e., number of species) and an index based on the Red List of Threatened Species of the International Union for Conservation of Nature. The Red List index equaled the mean, for all species, of the score assigned to each species to reflect its conservation status at the national scale (France or the United Kingdom) (Butchart et al., 2004): least concern = 0, near threatened = 1, vulnerable = 2, endangered = 3, and critically endangered = 4.

3. Results

3.1. The adapted rewilding score and its components

3.1.1. Human forcing

Human forcing scores ranged from 0.002 to 0.55 among the farms in the sample, with higher scores for dairy farms (0.22, 0.38 and 0.55 for Tr, O, and D, respectively), lower scores for suckler beef farms (0.02, 0.10, and 0.11 for B, S, and Th, respectively), and the lowest score for K (0.002) (Fig. 2, Table 4).

3.1.2. Ecological integrity

Except for Th, ecological integrity scores of the farms based on the adapted Torres method were lower than those based on the adapted IPE, and all farms except Th followed the same ranking according to the two methods: highest for K (0.81 and 0.90, respectively), second highest for B (0.64 and 0.69, respectively), lowest for D (0.36 and 0.41, respectively), and intermediate for S, Th, Tr, and O (Fig. 2, Table 4). The two sets of ecological integrity scores were strongly correlated (r = 0.90) (Table 5).

Using the adapted Torres method, the indicators that differed most among farms were natural hydrological regimes (from 0 for O to 1 for K), natural pest regimes and mortality events (from 0.51 for D to 1.00 for K, B, S, Th, and Tr), spontaneous vegetation dynamics (from 0.004 for D to 0.32 for K), harmful invasive species (from 0.37 for B and S to 0.87 for

Table 3
Three-year mean production and livestock-feeding management data of the seven (C)ommercial or (E)xperimental farms in the sample (2018–2020, except for Derval (2016–2018)) (K: Knepp, B: La Barge, S: Saint Laurent de la Prée, Th: Thorigné d'Anjou, Tr: Trévarn, O: Oasys, D: Derval). Data represent mean annual production per ha of the total area used for the animal production system (off- and on-farm). AR: Agricultural rewilding.

Type of system		AR	Suckler beef			Dairy		
Name		K	B	S	Th	Tr	O	D
Farm type		C	C	E	E	C	E	E
Biodiversity considered?		Main objective	In the overall strategy	In the overall strategy	Not specifically	In the overall strategy	Not specifically	Not specifically
Milk produced (t FPCM[a] ha^{-1})		-	-	-	-	3.10	5.12	6.71
Animal live weight produced (kg ha^{-1})		54.2	114.2	218.2	272.5	97.3	163.1	197.3
Human-edible protein produced (kg ha^{-1})		5.0	11.0	21.0	26.0	110.6	183.1	238.6
Mean herd composition (number)[b]	Cows	24.1	50.0	48.0	70.0	70.0	69.0	86.2
	Heifers	40.8	60.0	42.0	71.5	33.0	49.5	56.6
	Other species[c]	300.9	–	–	–	–	–	–
	Cattle breed	Longhorn	Maraîchine	Maraîchine	Limousin	Crossbreed[d]	Crossbreed[e]	Prim'Holstein
Annual feed ration for productive cows[b] (kg dry matter day^{-1})	Annual crop silage[g]	–	–	–	–	–	2.9	10.9
	Grass silage	–	–	–	1.3		0.1	3.0
	Wrapped bales	–	–	1.7	–	6.8	4.7	–
	Hay	–	4.4	6.2	4.5	0.9	1.1	–
	Grazed grass	–	9.7	8.0	6.3	7.4	6.0	2.6
	Cereals	–	–	–	–	–	–	1.1
	Concentrate feed made on-farm	–	–	0.6	0.1	–	–	–
	Concentrate feed made off-farm	–	–	–	–	–	0.9	2.7
Grazing management	% of year outside	100	75	59	67	67	50	41
	Stocking rate (LU/ha)[h]	0.18	0.53	0.63	0.89	0.79	1.11	1.30
Total area used for animal production (ha)	Crops/ temporary grassland on-farm	–	8.0	18.0	118.0	12.0	85.0	90.4
	Permanent grassland on-farm	–	154.0	105.0	29.0	91.0	2.0	1.8
	Off-farm	–	–	–	–	0.8	5.2	21.5
	Total	450.0	162.0	123.0	147.0	103.8	92.2	113.7
Agricultural context		Low Weald clay soils in West Sussex	Coastal marsh in Vendée	Coastal marsh in Charente-Maritime	Hedge landscape in Maine et Loire	Peneplain in Finistère	Clayey plateau in Poitou-Charentes	Wooded plateau in Loire-Atlantique

[a]FPCM: fat- and protein-corrected milk.
[b]for commercial farms, the herd composition and feed ration were estimated by the farmer.
[c]pigs, ponies, fallow deer, and red deer.
[d]Prim'Holstein, Jersey, and Montbéliarde crossbreeds.
[e]Prim'Holstein, Jersey, and Scandinavian Red crossbreeds.
[f]steers fattened with concentrate feed produced on-farm.
[g]maize, sorghum and maslin in Oasys; maize in Derval.
[h]livestock units per ha of on-farm area used to feed the livestock.

the other farms), and terrestrial animals heavier than 5 kg (from 0.17 for D to 1.00 for K) (Table S3). Using the adapted IPE, the indicators that differed most among farms were habitat diversity (from 0.24 for S and Th to 1.00 for K), undeveloped area (from 0.032 for O to 1.00 for K), hosting potential (from 0.37 for S to 0.87 for K and Tr), invasive plant species (from 0.37 for B and S to 0.87 for the other farms), and heritage habitats (from 0 for D, O, Th, and Tr to 1 for B) (Table S4).

3.1.3. *Adapted rewilding score*

The ARS depended on the method used to estimate ecological integrity, being lower when based on the adapted Torres method for all farms except Th. Based on the adapted Torres method or the adapted IPE, the ARS was highest for K (0.81 and 0.90, respectively), second highest for B (0.63 and 0.67, respectively), lowest for O (0.26 and 0.27, respectively) and D (0.16 and 0.18, respectively), and intermediate for S, Th, and Tr (Table 4). The two sets of ARS were strongly correlated ($r = 0.96$) (Table 5).

As with the ecological integrity score, all farms except Th followed the same ranking by ARS, regardless of the method (Fig. 2, Table 4), and the ARS had lower standard deviation when based on the adapted Torres method (0.22) than when based on the adapted IPE (0.25). Visually analyzing the human forcing and ecological integrity scores at the same time identified three groups of farms (Fig. 2, Table 4):

- Farms with low human forcing (0.002–0.02) and high ecological integrity (0.64–0.90) (K and B)
- Farms with intermediate human forcing (0.10–0.22) and ecological integrity (0.41–0.60) (S, Th, and Tr)
- Farms with high human forcing (0.38–0.55) and intermediate ecological integrity (0.36–0.43) (O and D)

3.2. *Assessment of avian biodiversity*

Bird species richness was highest for S (64 species), followed by B (54), K (49), Th (39), Tr (33), O (30), and D (29). The Red List index was highest for K (0.45), followed by S (0.40), B (0.39), O (0.35), D (0.21), Tr (0.13), and Th (0.12). The two indicators were moderately correlated ($r = 0.64$) (Table 5).

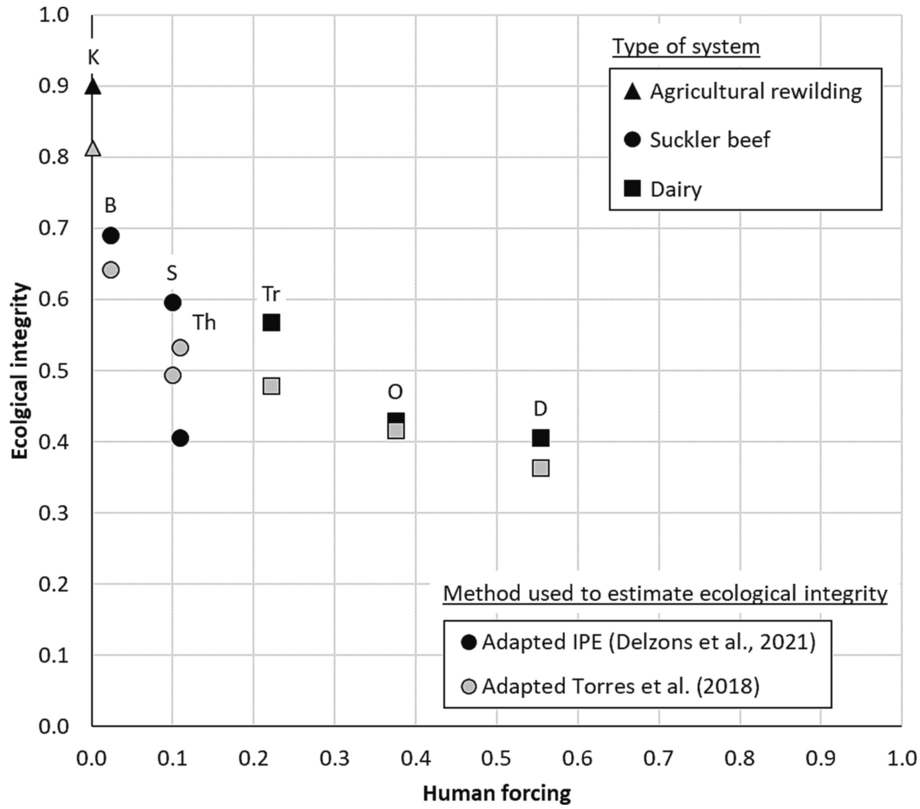

Fig. 2. Human forcing and ecological integrity of the seven farms in the sample (K: Knepp, B: La Barge, S: Saint Laurent de la Prée, Th: Thorigné d'Anjou, Tr: Trévarn, O: Oasys, D: Derval), the latter estimated according to two methods. IPE: index of ecological potential. Maximum human forcing (1.0) was assumed to equal that of intensive dairy farms in western France (Salou et al., 2017).

Table 4
Scores of human forcing, ecological integrity, and rewilding based on the adapted method of Torres et al. (2018) and the adapted indicator of ecological potential (IPE) (Delzons et al., 2021) (K: Knepp, B: La Barge, S: Saint Laurent de la Prée, Th: Thorigné d'Anjou, Tr: Trévarn, O: Oasys, D: Derval).

Item	Method used	Farms							Standard deviation
		K	B	S	Th	Tr	O	D	
Human forcing	Energy demand (MJ/ha)	90	862	3679	4030	8203	13 902	20 496	7481
	Normalized energy demand[a]	0.002	0.023	0.099	0.109	0.222	0.376	0.554	0.202
Ecological integrity	Adapted Torres	0.81	0.64	0.49	0.53	0.48	0.42	0.36	0.15
	Adapted IPE	0.90	0.69	0.60	0.41	0.57	0.43	0.41	0.18
Adapted rewilding score	Adapted Torres	0.81	0.63	0.45	0.48	0.37	0.26	0.16	0.22
	Adapted IPE	0.90	0.67	0.54	0.36	0.44	0.27	0.18	0.25

[a] divided by the energy demand estimated for intensive dairy farms in western France (37 000 MJ ha^{-1}) (Salou et al., 2017).

Table 5
Linear correlation matrix between human forcing (i.e., normalized energy demand), scores of ecological integrity, and adapted rewilding scores (according to the adapted method of Torres et al. (2018) or the adapted index of ecological potential (IPE) (Delzons et al., 2021)) and indicators of avian biodiversity calculated for the farms.

		Human forcing	Ecological integrity		Adapted rewilding score		Avian biodiversity	
			Adapted Torres	Adapted IPE	Adapted Torres	Adapted IPE	Species richness	Red List index
Human forcing		1.00						
Ecological integrity	Adapted Torres	−0.83	1.00					
	Adapted IPE	−0.71	0.90	1.00				
Adapted rewilding score	Adapted Torres	−0.91	0.99	0.88	1.00			
	Adapted IPE	−0.85	0.95	0.97	0.96	1.00		
Avian biodiversity	Species richness	−0.76	0.54	0.59	0.64	0.69	1.00	
	Red List index	−0.40	0.55	0.69	0.53	0.65	0.64	1.00

3.3. Correlations between human forcing, ecological integrity, the adapted rewilding score, and avian biodiversity

Human forcing and ecological integrity were negatively correlated, and more strongly so when ecological integrity was based on the adapted Torres method (r = –0.83) than on the adapted IPE (r = –0.71) (Table 5). Whether based on the adapted Torres method or the adapted IPE, the ARS was correlated more strongly with the same method's ecological integrity (r = 0.99 and 0.95, respectively) than with human forcing (−0.91 and −0.85, respectively).

Human forcing was negatively correlated with bird species richness and the Red List index (r = −0.76 and −0.40, respectively), while ecological integrity was positively correlated with them, whether based on the adapted Torres method (r = 0.54 and 0.55, respectively) or the adapted IPE (r = 0.59 and 0.69, respectively) (Table 5, Fig. S3). The ARS was correlated more strongly with bird species richness than with the Red List index, and more so when based on the adapted IPE (r = 0.69 and 0.65, respectively) than when based on the adapted Torres method (r = 0.64 and 0.53, respectively) (Table 5, Figs. 3 and S3).

3.4. Relation between productivity and the adapted rewilding score

Productivity and environmental impacts of the sample farms were assessed by Mondière et al. (in review). The quantity of human-edible animal protein produced was negatively correlated with the ARS based on the adapted IPE (Fig. 4). Compared to suckler beef farms (Th, S, B), dairy farms (D, O, Tr) produced more protein, which decreased more as ARS increased. Tr (the least productive dairy farm) had an ARS (0.44) that lay between those of the most productive suckler beef farms Th (0.36) and S (0.54), while its protein production (111 kg ha^{-1}) was ca. 4–5 times that of Th (26 kg ha^{-1}) and S (21 kg ha^{-1}). K combined the highest ARS and lowest productivity.

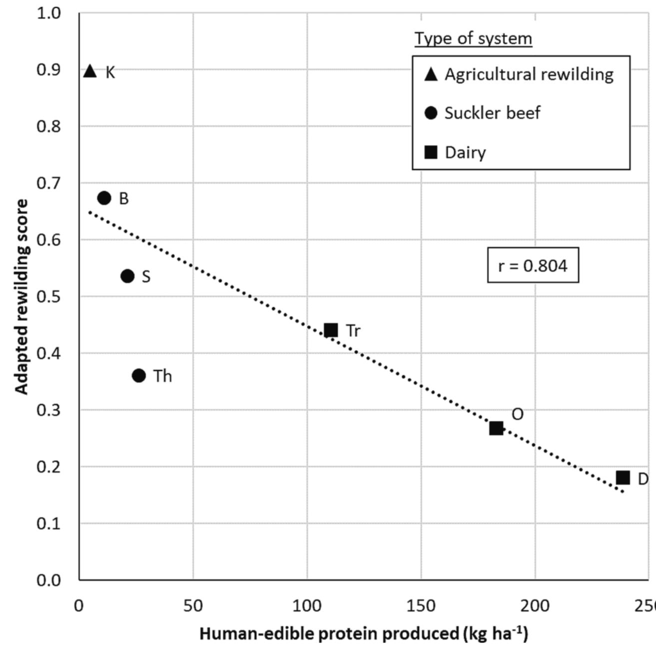

Fig. 4. The adapted rewilding score (based on using the adapted index of ecological potential to estimate ecological integrity) as a function of the quantity of human-edible animal protein produced of the seven farms in the sample (K: Knepp, B: La Barge, S: Saint Laurent de la Prée, Th: Thorigné d'Anjou, Tr: Trévarn, O: Oasys, D: Derval).

4. Discussion

4.1. The adapted rewilding score and its components

4.1.1. Human forcing

For assessment of livestock systems instead of rewilding projects, summarizing human forcing as only the quantity of energy input into the system seemed an appropriate indicator of short-term effects, since livestock systems require more management to maintain their current functioning. However, estimating cumulative energy demand using LCA requires much more data, technical expertise, and time than estimating human forcing using the quantitative and qualitative indicators in the Torres method. In the absence of the LCA expertise, software, and databases needed to estimate cumulative energy demand, our adapted method could rely on simpler methods to assess energy demand, such as the open-source Farm Energy Analysis Tool (Camargo et al., 2013).

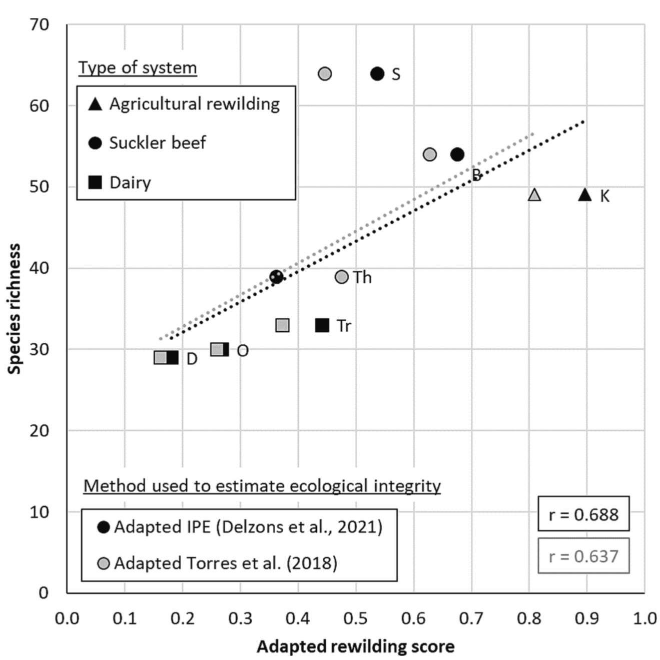

Fig. 3. Correlation between observed bird species richness and the adapted rewilding score of the seven farms in the sample (K: Knepp, B: La Barge, S: Saint Laurent de la Prée, Th: Thorigné d'Anjou, Tr: Trévarn, O: Oasys, D: Derval) as a function of the method used to estimate ecological integrity. IPE: index of ecological potential.

4.1.2. Ecological integrity

Ecological integrity was influenced by indicators of landscape characteristics, regardless of the method used to estimate it. Many studies highlight the high diversity and complexity of the landscapes in which livestock farms occur, due to the many semi-natural areas (Lemauviel-Lavenant and Sabatier, 2017; Vollet et al., 2017) and/or the diversity of crops on these farms (Rodriguez-Ortega et al., 2014). Since the farms studied were located in these types of landscapes, the indicators of landscape complexity (i.e., terrestrial landscape fragmentation for the adapted Torres method, ecological connectivity and site permeability for the adapted IPE) differed little among the farms, regardless of the method used. Spontaneous vegetation dynamics (for the adapted Torres method) depended on the degree to which farm management allowed them, with areas of completely spontaneous vegetation (K) or vegetation that was managed more extensively than cropland (e.g., permanent grassland, especially for B and Tr). The undeveloped area (for the adapted IPE) was also related to the management of cropland (considered to be developed), which sometimes covered nearly the entire farm (O and D), little of it (B) or none of it (K). This management influenced trophic processes (for the adapted Torres method), because most animals on the farms were domestic and grazed on variable percentages of the farms for variable durations.

Management also influenced the ecological integrity of the farms by influencing disturbance regimes (for the adapted Torres method), with farms having the same management of natural fires but differing management of hydrological and pest regimes as a function of their practices (i.e., drainage, irrigation, and pesticide use). In comparison, the hosting potential (for the adapted IPE) was related directly to the presence of microhabitats on the farms and, like habitat diversity (for the adapted IPE), caused the ecological integrity of farms to vary. Finally, the ecological context of the farms (i.e., presence of heritage habitats for the adapted IPE and presence of invasive species for both adapted methods) also caused the ecological integrity to vary.

4.1.3. Adapted rewilding score

The diversity of farms, which varied from an agricultural rewilding project (K) to a conventional dairy farm (D), resulted in a wide range of ARS, with the former having the highest ARS and the dairy farms having the lowest ARS, but with little difference between the most extensive dairy farm (Tr) and the suckler beef farms (especially Th). Despite having low human forcing and high ecological integrity, K could increase its ARS even more, in part because its rewilding began relatively recently (20 years ago). In contrast, although D had the highest human forcing and lowest ecological integrity, its ARS was not close to zero, which likely reflects that the farm sample, despite its wide range, did not include highly intensive farms within ecosystems that had extremely low ecological integrity. Thus, the ARS was able to assess and compare production systems, and it seems relevant to apply it to farms that have a wide range of intensity, even though it was not applied to extremely intensive farms.

4.2. Relations between avian biodiversity and the adapted rewilding score

4.2.1. Assessment of avian biodiversity

The sample farms that hosted more Red List bird species also tended to host more birds of all species (higher species richness). This result needs to be interpreted cautiously, however, since the farms' ecological contexts differed, and differences in species richness and Red List species are related not only to farm functioning but to the context's biodiversity potential. For example, Tr is located on the edge of the Breton peninsula, which hosts fewer bird species than other regions in France (Witté and Touroult, 2014), while B and S are located in coastal marshes, which host many more species, especially heritage species (Roche et al., 2022). In addition, surveying birds along a transect on 10 days at K but at points along a survey route on 1 day each on the French farms likely influenced the number of birds counted at K, since estimates of species richness increase as sampling effort increases (Walther and Martin, 2001). The differences in the surveyor and survey year at Knepp may also have biased its count. Ultimately, the low sampling effort of the bird surveys on the French farms weakens conclusions drawn about relations between avian biodiversity and the ARS.

4.2.2. Relations between avian biodiversity or human-edible protein production and the adapted rewilding score

The two indices of avian biodiversity were positively correlated with the ARS, and farms generally followed the same ranking for the ARS and the biodiversity indices. Farms that had high human forcing and low ecological integrity (O and D) had low species richness and a variable Red List index. Among farms that had intermediate human forcing and ecological integrity (S, Th, and Tr), S had the highest species richness, perhaps because it is located in a coastal marsh. Farms with low human forcing and high ecological integrity (K and B) had a relatively high and similar species richness and Red List index. However, these results need to be interpreted with caution since avian biodiversity represents only part of total biodiversity. Nonetheless, the positive correlations observed suggest that the ARS may be useful for assessing the biodiversity potential (at least for birds) of livestock farms, including those that are biodiversity-friendly. Unsurprisingly, the ARS decreased as the production of human-edible protein increased; however, it decreased less rapidly for dairy farms than for suckler beef farms, which suggests that dairy farms may provide a better tradeoff between the goals of higher production of animal protein and higher potential biodiversity.

4.3. Methodological assessment

4.3.1. Relative advantages and limits of the adapted rewilding score

Regardless of the adapted method used to estimate ecological integrity, estimating the ARS for the farms in the sample was simple, which is a key criterion for disseminating and using a method (Dardonville et al., 2022; González-Chang et al., 2020). One of the main strengths of the ARS is its consideration of two dimensions, which distinguish effects of short-term management (human forcing) from those of long-term management (ecological integrity) (Torres et al., 2018).

One criticism of certain methods that assess the overall status of ecosystems or biodiversity is that they focus on a pre-defined ecosystem or soil and climate region (Jeanneret et al., 2014), or that they are too general, which means that, if applied to agroecosystems, they would not be able to consider their specific characteristics (Manneville et al., 2014). The approach in the present study remained general, and, depending upon the method used to estimate ecological integrity, the ecological context was considered via the presence of heritage habitats or invasive species, although it could have been considered more completely, such as by assessing soil, climate, or geographic characteristics.

4.3.2. Comparing the methods adapted to estimate ecological integrity

The two methods adapted to estimate ecological integrity include indicators that consider landscape effects, the variety of land uses on farms, their management, and their ecological context. Although ecological integrity scores differed between the methods, the ranking of farms remained nearly the same. When estimating ecological integrity, the adapted Torres method was more similar than the adapted IPE to the initial method, but the relevance of the adapted Torres method for agroecosystems remains questionable. For example, trophic processes were assessed using the presence of animals heavier than 5 kg. The adapted method focused mainly on domestic animals because they were more common than large wildlife on the farms, but domestic animals influence food webs less because they use primary production only through grazing. This indicator may need to be rendered more precise, such as by specifying the position of each species heavier than 5 kg in the food web; however, doing so would require collecting additional data in order to calculate the ecological integrity score.

In contrast, the natural heritage, functionality, and diversity of ecosystems of the IPE can be assessed at the farm scale, although they can be adapted better to this scale. The resolution of the "undeveloped area" indicator remains much too low for farms because it considers cropland to be as developed as an asphalt surface. Further differentiating the degree of development may better capture effects of different land uses on the overall functionality of an ecosystem. It may also be useful to consider readjusting the weights of the IPE indicators that remained after excluding the two biodiversity indicators, since they initially represented 35 % of the IPE. In addition, the indicator of heritage habitats could be improved by comparing the habitats and/or species present to regional inventories.

The Torres method was designed to be able to assess sites with contrasting ecological contexts, while the IPE was designed to consider a site in relation to its ecological context. Thus, ecological integrity based on the adapted Torres method can be compared more readily among sites than that based on the adapted IPE. In contrast, regarding the scale of assessment, the Torres method, even adapted, seems less suitable to estimate ecological integrity at the farm scale than the adapted IPE, which considers local characteristics by considering heritage habitats. However, for assessment at the scale of a livestock-producing region, the Torres method could be more relevant.

4.3.3. Relevance of using the adapted rewilding score to assess overall biodiversity potential

The two methods yielded similar moderate correlations between the

ARS and the avian biodiversity indicators, which suggests that neither method was more relevant than the other for assessing biodiversity potential. These correlations suggest that the ARS may be able to indicate a certain biodiversity potential (here, bird species richness and Red List index) of livestock farms by being able to distinguish the influence of production strategies on biodiversity. Nonetheless, relations between ecosystem structure and avian biodiversity can vary among landscapes (Syrbe et al., 2013). Based on these preliminary results, bird diversity alone seemed oversimplified as a measure of overall biodiversity. Thus, relations between the ARS and the diversity of other taxonomic groups need to be assessed. In addition, it would be useful to compare the ARS to biodiversity indicators that consider the relative abundance of each species (e.g., Shannon diversity indicator), since relative abundance appears to be key to improve estimates of effects of human activities on biodiversity (Burns et al., 2021). However, Kok et al. (2020), citing Jost (2006), think that measures of evenness, such as the Shannon diversity index, are more important to ecologists than to conservationists, and perhaps poorer indicators of diversity than species richness or a ratio scale (e.g., $e^{\text{Shannon index}}$).

4.4. Perspectives

Assessment of human forcing could be improved by considering all human activities on a farm, such as cash-crop production (S, Th, O, and D) and tourism (K). It would also be useful to apply the method to additional farms in a larger geographic area, such as farms throughout France in the “Paysans de nature” network, which already perform regular field surveys of biodiversity, as well as more intensive farms, to assess the potential range of validity of relations between the ARS and biodiversity indicators. Because the ARS can distinguish effects of long-term vs. short-term practices, it could help farmers consider biodiversity more explicitly in their production strategies over both temporal horizons or help governments decide how to distribute subsidies to farmers who implement recommended practices for preserving biodiversity (European Commission, 2021; Gargano et al., 2021).

5. Conclusion

We adapted the rewilding score, which was developed to estimate and monitor rewilding of ecosystems over large areas with relatively few human activities, to agroecosystems, specifically livestock farms. Applying the ARS to seven cattle-oriented farms and comparing it to indicators of avian biodiversity revealed moderate correlations between them, which suggests that the ARS can assess the biodiversity potential of agroecosystems for birds to some degree. It can distinguish effects of long-term vs. short-term production strategies on biodiversity. Nonetheless, correlations between the ARS and biodiversity need to be assessed for additional taxonomic groups, indicators of biodiversity, and farms.

CRediT authorship contribution statement

Aymeric Mondière: Methodology, Formal analysis, Investigation, Writing – original draft. **Michael S. Corson:** Methodology, Formal analysis, Writing – review & editing. **Lou Valence:** Methodology, Formal analysis, Investigation, Writing – original draft. **Loïs Morel:** Methodology, Writing – review & editing. **Hayo M.G. van der Werf:** Methodology, Formal analysis, Writing – review & editing.

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.ecolind.2023.111165.

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