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Potential of recording acoustic complexity index to monitor agri-environmental scheme effectiveness in grasslands☆

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ABSTRACT

To counteract the ongoing biodiversity loss due to intensified agricultural practices, agri-environmental schemes have been introduced in most European countries. A systematic monitoring is needed to optimise their effectiveness in promoting biodiversity. New and automated methods, such as passive acoustic monitoring (PAM), offer a promising alternative to traditional, labour-intensive, and costly methods. However, we know relatively little about whether differences in biodiversity due to land use can be related to differences in acoustic indices, and whether spatio-temporal factors are modifying the relationship. To assess the potential of PAM as an alternative to traditional methods for evaluating the effectiveness of agri-environmental measures, we explored whether bird and grasshopper species richness and insect abundance as grassland diversity indicators are related to the acoustic complexity index and to land use at local scales (e.g., fertilisation), and landscape scales (e.g., percentage of woody structure). We also examined whether these relationships depend on the peak vocalisation activity of the indicator group (morning vs. midday). Local vegetation structure increased insect abundance, which was related to an increased acoustic complexity index at midday. A higher percentage of agri-environmental schemes in the landscape increased bird species diversity, which was related to an increased acoustic complexity index in the morning. Passive acoustic methods to monitor biodiversity and evaluate the effectiveness of conservation measures are a promising tool. This study demonstrates that the acoustic complexity index is able to indirectly show changes in land-use via species richness or abundance. Furthermore we would like to point out that thesuitability depends on the indicator group and its activity patterns.

1. Introduction

Farmland represents the dominant land-use type in Europe (Eurostat, 2022). However, continued agricultural intensification and habitat loss have caused considerable declines in farmland biodiversity over the last 100 years (IPBES, 2019; Pilotto et al., 2020; Wagner et al., 2021). To counteract biodiversity loss, agri-environmental schemes (AES) have been introduced in most European countries (Batáry et al., 2015; EU, 2005; Kleijn et al., 2006). To determine and optimise their effectiveness, systematic farmland biodiversity monitoring is needed (Kleijn et al., 2001). Given that systematic monitoring using traditional methods, such as trapping or counting species in the field, is costly and labour intensive, most evaluations of the effectiveness of AES are limited to restricted spatial and temporal resolution (but see, for example, Meier et al. (2021). Those studies have shown marginal to moderately positive effects of AES on biodiversity at the field and farm scale (Aviron et al., 2009; Kampmann et al., 2012), and at the landscape scale (Meier et al., 2024; Zingg et al., 2019). Batáry and Tscharntke (2022) recommended considering different scales in evaluating agri-environmental schemes to understand their effects. The few farmland biodiversity monitoring schemes that currently exist in Europe have often focused on easily observable indicator species (e.g., plant, butterfly, and bird diversity) (Bühler, 2024; EEA, 2013; Meier et al., 2021). Compared to plants (7000–15,000 species in Switzerland) or birds (200 species), insects are the most species-rich group of all animals, and there are approximately 44,000–60,000 species in Switzerland (Widmer et al., 2021). Therefore, cost-efficient methods for monitoring insect and bird biodiversity are needed.

Passive acoustic monitoring (PAM) (Darras et al., 2024; Ross et al., 2023), eDNA-based detection of species (Sickel et al., 2023), the use of

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drones (Marchowski, 2021), or a combination of eDNA and acoustic monitoring using drones (Wang et al., 2023) are promising cost-efficient new methods for monitoring because they can be largely automated (Dufourq et al., 2021; Merchant et al., 2015). PAM is particularly promising because it allows the identification of acoustic morphospecies and their circadian rhythms (Aide et al., 2017; Eldridge et al., 2018). A further benefit of PAM is that it enables simultaneous data collection at multiple locations. It also helps to reduce the observer effect, when data from many different field workers may be biased. Animals are also less impacted by human disturbance from the acoustic loggers compared to by researchers conducting transect walks (Gibb et al., 2019).

Identifying acoustic indices to predict species richness or abundance is important in PAM implementation. Acoustic indices are metrics for the quantification and comparison of different sounds of a landscape. These are summary metrics from sound recordings that capture patterns of acoustic activity. However, they do not directly predict species richness or abundance. Instead, they have been primarily tested as corollaries of biodiversity (Alcocer et al., 2022). Recent advances in artificial intelligence, particularly convolutional neural networks (CNNs), now enable the estimation of species richness directly from PAM data. Nevertheless, acoustic indices remain valuable tools, offering efficient and scalable ways to describe soundscape structure and complement species-level approaches (Mueller et al., 2023). There are correlations between acoustic indices and bird species richness (Aide et al., 2017; Bradfer-Lawrence et al., 2020; Buxton et al., 2018; Droege et al., 2021; Eldridge et al., 2018; Mammides et al., 2017; Shaw et al., 2021), bird abundance (Bradfer-Lawrence et al., 2020; Mueller et al., 2022) or bird composition (Mueller et al., 2022; Shaw et al., 2021), species richness of amphibians or insects (Aide et al., 2017), entire species communities (Sueur et al., 2008), vegetation structure (Bradfer-Lawrence et al., 2020; Droege et al., 2021; Retamosa Izaguirre et al., 2021), habitat type (Droege et al., 2021; Hayashi et al., 2020) and landscape homogenisation (Burivalova et al., 2019). Acoustic indices cannot be used as a proxy for species richness and abundance in every landscape context or habitat, and further studies are needed, specifically in the agricultural landscape (Buxton et al., 2018). Two meta-analyses showed that acoustic indices are partially able to quantify species richness or abundance. However, more research is needed to overcome the limitations of direct proxies (Alcocer et al., 2022; Bateman and Uzal, 2022). In the agricultural landscape, the relationship between bird vocal activity and landscape composition (Dixon et al., 2020), non-crop vegetation (Dixon et al., 2023), or pruning and pesticide application in a coffee landscape (Molina-Mora et al., 2024) was evaluated. Meanwhile, Biffi et al. (2024) studied the association of hedgerows of different ages and bird vocal activity. The responsiveness of acoustic indices to local-scale land-use was assessed (Quinn et al., 2024). However, these studies did not report land use, species richness, and vocal activity through PAM simultaneously, as was the case for Mueller et al. (2022). To fully understand the variable response, additional studies are necessary, and complementary species data need to be considered. To our knowledge, it is currently unknown whether the association between acoustic indices and species richness or abundance varies over the course of 24 h, depending on the time of acoustic activity in the indicator group (Farina et al., 2013; Metcalf et al., 2021). However, answering these open questions is important before PAMs can be used in a monitoring context to evaluate the effectiveness of AESs.

Here, we examined whether i) local land use variables (i.e., fertilisation, vegetation, and plant diversity) and landscape-scale land, measured as the percentage area managed under the Swiss agri-environmental schemes and the proportion of landscape woody structure, are directly linked to species indicators (e.g., bird richness, grasshopper richness, and total insect abundance) and indirectly related to the acoustic complexity index (ACI). We also explored whether ii) these relationship varied depending on the activity pattern of the indicator group. We expected that the characteristics of less intense land use at the local and landscape scales are positively related to the species indicators and, in turn, to a higher ACI. Further, we predicted that bird species richness is linked to the ACI recorded in the morning, whereas grasshoppers and total insects are linked to it at midday. The ACI measures the variability in acoustic energy across frequency bins (Bateman and Uzal, 2022). Since ACI (Pieretti et al., 2011) has been identified as a biodiversity proxy that outperformed others in a meta-analysis (Alcocer et al., 2022) and showed greater consistency across different frequency filters (Hyland et al., 2023), it holds potential for tracking bird species richness, given the high variability in bird vocalizations—even within a single frequency bin (Buxton et al., 2018; McGrann et al., 2022). Therefore, ACI was selected for testing in this study.

2. Materials and methods

2.1. Study design and study sites

The study was conducted in Switzerland (Fig. 1) on 15 squares of 1-km^2 that were part of the Swiss farmland monitoring programme ALL-EMA (www.allema.ch). ALL-EMA studies habitats, plants, breeding birds, and butterflies on a total of 170 defined 1-km^2 survey squares (Meier et al., 2022). To investigate the landscape-scale effects, the squares were subdivided into four 500 × 500 m subsquares (0.25 km^2) (Fig. 1). Of the 170 squares, we only included squares that comprised meadows that were being extensively managed following the management protocols of the Swiss agri-environmental scheme (AES) and a conventionally managed meadow in at least two subsquares per square. Extensively managed meadows under the Swiss AES refer to grassland areas cultivated with minimal management input, such as having no synthetic fertilisers applied or not having the first cut before 15 June. They must support a minimum number of plant taxa, for example, 3–6 typical meadow species per 28 m^2. To standardise the main noise factors of traffic and urban noise, only meadows that were at least 300 m from a two-lane road and at least 200 m from a settlement with more than two houses were selected. This selection was informed by studies demonstrating that anthropogenic noise can substantially alter animal behaviour and habitat use at these distances (Kleist et al., 2021; Ware et al., 2015). Only a small subsample of squares and subsquares met our strict criteria and our limited sampling effort. In total, 15 squares and 22 subsquares comprising 38% of all subquares out of the 15 squares) were considered for inclusion in the study. (See Table 1.)

2.2. Acoustic recording and data processing

To assess the association between the diversity of our species groups and the acoustic complexity index (ACI), acoustic recordings were performed with the acoustic logger AudioMoth 1.2.0 (Hill et al., 2018). Devices were programmed to pause for 10 min after each 10-min recording (Ross et al., 2021). This generated 30 min of sound recording per hour, with a sampling rate of 48 kHz (Hill et al., 2019; Mueller et al., 2022). The recording period was selected to ensure 24 consecutive hours of data for each site, allowing selection of the main vocalising periods: morning (5:00–9:00) for birds and midday (11:00–15:00) for insects.

The AudioMoth device was placed at the centre of the selected meadows and mounted to a wooden pole 1.2 m above ground. Recordings were conducted over four periods: (1) 20–30 July, (2) 4–9 August, (3) 17–24 August, and (4) 1–5 September. The less the targeted biophony is masked by geophony and anthrophony, the more sensitive the ACI is to the vocalising taxa (Metcalf et al., 2021). Therefore, for data processing, as wind and rain are disturbing geophony factors on recordings, first recordings with wind >10 km/h or/rain (>0 mm) were filtered out. Windspeed per hour (km/h) and precipitation (mm/h) were obtained from MeteoSwiss for the closest weather station. Anthrophony such as cowbells, children playing, and tractors could mask the recordings (Fairbrass et al., 2017). If anthrophony was the dominant noise, recordings were replaced in each meadow with recordings from

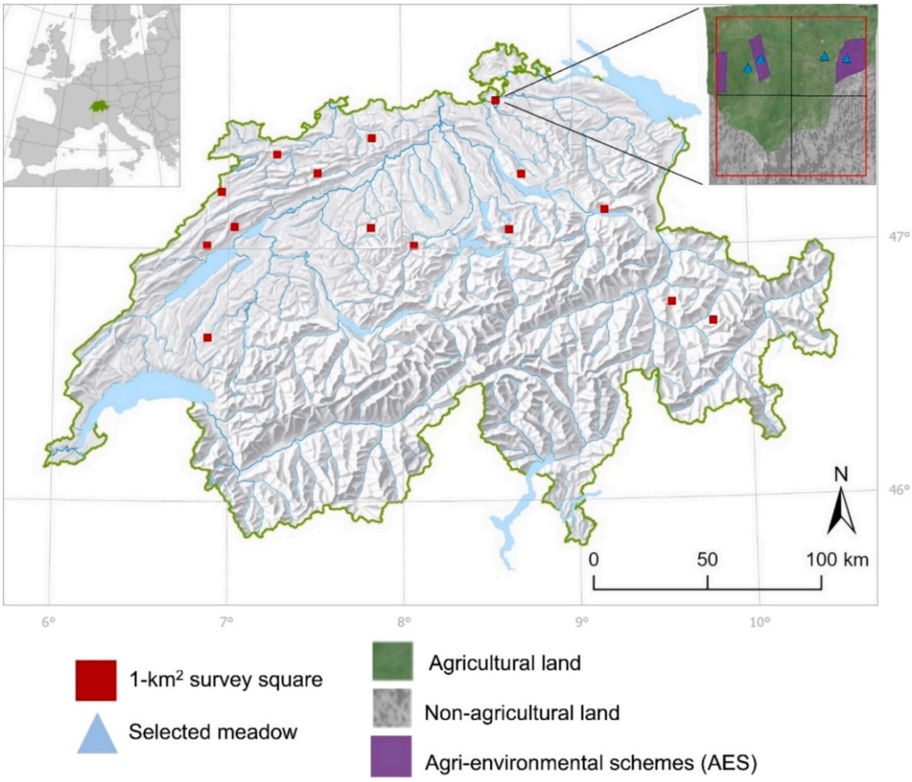

Fig. 1. Location of the study area in Europe and the 15 analysed survey squaresof each 1 km^2 in Switzerland. In the upper right corner, a single survey square is zoomed in. Each survey square (1 km^2) was divided into four subsquares (0.25 km^{2,} 22 subsquares), and a pair of meadows was selected.

Table 1

Factors directly or indirectly related to the acoustic complexity index (ACI) and used as explanatory and/or response variables in the SEM (see Table 2).

Category	Variable	Description	Unit	Mean ± SD
Soundscape	ACI morning	Mean acoustic complexity index between 5 am and 9 am per site	–	18,318.77 ± 454.41
	ACI midday	Mean ACI between 11 am and 3 pm per site	–	18,706.83 ± 553.87
Species groups	Bird species richness	Species richness of birds per subsquare	–	31.91 ± 6.73
	Grasshopper species richness	Species richness of grasshoppers per site	–	3.20 ± 1.44
	Insect abundance	An abundance of flying insects over 4 mm in size per site	–	105.16 ± 122.30
Land use (landscape)	% woody structure	Proportion of woody structure area per square	%	0.15 ± 0.08
	% AES	Area managed according to the prescriptions of the Swiss AES	%	0.39 ± 0.33
Land use (local)	Fertilisation	Number of fertiliser applications per year per site	–	0.98 ± 1.30
	Vegetation height	Mean vegetation height per site	cm	12.41 ± 5.91
	Vegetation structure	Variance of vegetation density per site	-	2.91 ± 2.33
	Plant species richness	Species richness of vascular plants in the meadow	–	26.77 ± 11.46

the same time of the day but from another date. Approximately 3–6 h of recordings had to be replaced. In the end, due to the strict selection of sound recordings that were not masked by geophony and anthrophony, the analysis was based on 15 squares and 22 subsquares comprising 44 meadows.

The ACI was calculated as the average distance between every adjacent value of intensities and then divided by the sum of all the intensity values (Pieretti et al., 2011) (eq. 1). In doing so, the ACI quantifies the biophony by the variability of intensities (Pieretti et al., 2011). The ACI was calculated for each selected sound file using the package *soundecology* (Villanueva-Rivera et al., 2011) in R Version 4.4.1 (R Core Team, 2023). Due to the different activity times for birds and insects, and to minimise acoustic interference from overlapping vocal signals (Metcalf et al., 2021), two different time slots were used for data analysis. It was assumed that birds were the most active in the morning, between 5 am and 9 am. Vocalising insects such as grasshoppers on the other side were more active during midday, between 11 am and 3 pm. The mean of ACI for those time frames was calculated for each plot for analysis using structural equation modelling.

$$\mathrm{ACI} = \sum_{f=1}^{F} \frac{\sum_{t=1}^{T-1} \left| A_{f,t+1} - A_{f,t} \right|}{\sum_{t=1}^{T} A_{f,t}} \tag{1}$$

$A_{f,t}$: Amplitude in frequency bin f at time step t, F: total number of frequency bins, T: total number of time steps.

2.3. Bird and grasshopper species richness and insect abundance

Data on breeding bird occurrence in the agricultural landscape were obtained from the common breeding bird surveys conducted by the Swiss Ornithological Institute in 2022. Ornithologists mapped breeding farmland birds, excluding forest birds, by identifying territories along a transect measuring 3 to 5 km in length. Each survey square was visited three times per breeding season, with observations conducted in the morning for 2 to 3 h before 11 am. To assess bird species richness at the subsquare level, all territories with centroids located within the agricultural land and within the 0.25 km^2 subsquare were included. The number of observed bird species per subsquare was then summed to calculate the species richness.

Two measures for insect diversity were recorded, namely, the species richness of grasshoppers and insect abundance. Data for both measures were sampled next to each recording device at the local meadow scale in the same week that the recordings took place. The grasshopper species were acoustically identified and quantified in a cross-like 2 × 40 m long transect per site based on our experience from monitoring programmes. We walked twenty steps in all four cardinal directions, starting from the pole with the recording device. Every ten steps, we stopped walking for two minutes and listened to the grasshoppers for two minutes. To obtain comparable values, all the sampling was conducted between 11 am and 3 pm, under conditions of more than 15 °C, less than 50% cloud cover, no rain, and not more than light to moderate wind. Grasshopper species richness was calculated as the total number of grasshopper species identified per site. To quantify flying insect abundance, a cross-like transect was sampled with a sweep net to catch insects. This was performed on the transect on which the grasshoppers were also recorded, but always after the grasshoppers were recorded. We swept the net 180° once per step and changed direction after each step. After catching the insects, they were placed in freezer bags and immediately placed in a cooling box. Frozen insects larger than 4 mm, hereafter referred to as large insects, were counted in the laboratory at a later date to obtain a proxy for insect abundance. Due to the high level of effort required, we were unable to identify the insects to species level.

2.4. *Land use variables at the local and landscape scale*

Land-use variables were assessed once during the acoustic recording to link them to the diversity of species groups and to the ACI. Land-use variables at the local scale included fertilisation, vegetation height, vegetation structure, and plant species richness of the individual meadows. Fertilisation was recorded by interviewing the farmers, namely, by asking how often the meadow was fertilised per year. The average vegetation height was measured with a piece of cardboard (70 × 70 cm, 416 g), placed horizontally above the vegetation at the location where the recording device was placed. Then the soil-to-carton distance was measured at a metre distance from this site in all four cardinal directions (Russel and Detling, 2003). As a proxy for vegetation structure, the variance in vegetation height was estimated. Therefore, we placed a stick vertically on the ground at five positions of the cardboard (centre, at all four corners) and then estimated how many grass blades touched the stick vertically. The more similar the vegetation height was, the more grass blades touched the stick. The greater the variance of all the counted touching grasses per site, the greater the vegetation structural diversity. Vascular plant species richness was surveyed by professional botanists on a circular plot of 10 m^2 in the selected meadows between April 1st and August 25th, 2022.

At the landscape scale, defined as 0.25 km^2, land use was assessed by the proportion of areas with woody structure (percentage woody structure) and managed according to the protocols of the Swiss AES (% AES) of the agricultural area used. The percentage woody structure was assessed in 2017 based on stereoscopic interpretation of aerial images. The percentage AES was determined using data delivered by the Swiss cantons. 15 AES types were considered. The percentage AES included various types of meadows and pastures, such as extensively managed meadows and pastures, arable land including flower strips, and permanent crops such as hedges. All these land-use types have specific management prescriptions, including no synthetic fertilisers, limited manure applications, the first cut not being before a specific date, pesticides being prohibited, and a minimum number of plant taxa present. More details have been provided by Meier et al. (2022).

2.5. *Statistical analysis*

Structural equation modelling (SEM) is a multivariate statistical process that allows testing multiple hypotheses in a complex network, and examining direct and indirect relationships (Fan et al., 2016). To determine if extensive local management and a more diverse landscape were associated with higher species richness or abundance, which is potentially linked to a higher ACI, a SEM with different mixed effect submodels was built based on our structural concept (Fig. 2). ACI, species richness or abundance, vegetation structure, and plant species richness were selected as dependent variables (arrowhead) and modelled with different explanatory variables at the local and landscape scale (arrow base). Each square contained at least two subsquares and each subsquare included at least four meadows.

To determine whether the impacts of variables on species groups varied depending on the scale of the explanatory variable, land-use variables at the local scale and at the landscape scale were used as explanatory variables in the mixed-effect models. To assess whether the relationship between insect or bird diversity and the ACI varied during the time of the day, two SEM were built. One related morning ACI to our species indicators and local or landscape variables, and one related midday ACI to our species indicators and local or landscape variables. The SEM submodels were based on different generalized linear models (GLM) to meet the assumptions of homoscedasticity and no overdispersion (Bates et al., 2015). The residual plots and tests for heteroscedasticity and overdispersion have been provided in the supplementary materials. For ACI morning and ACI midday, a gamma GLM was used. Meanwhile, for grasshopper abundance and bird species richness, a Poisson GLM was used. For insect abundance and plant species richness, a negative binomial GLM was built. For vegetation structure, a linear model (LM) with log transformation was used.

As a measure of model fit, Fisher's exact test, with a good fit indicated by Fisher's C p-value $>.05$, was used, and the proportion of variance explained (R^2) was calculated for each SEM submodel. The correlations of all the explanatory and response variables were checked, and the strength of the correlation between the variables was measured using the variance inflation factor (VIF). Only variables with a VIF less than 5 were selected to reduce multicollinearity. To build the SEM, the *piecewiseSEM* package version 2.3.0 (Lefcheck, 2016) was used. Statistical analyses were performed in R version 4.4.1. (R Core Team, 2023).

3. Results

Bird species richness was positively related to the ACI in the morning (Table 2, Fig. 2). The species richness of grasshoppers and the abundance of insects did not explain the ACI in the morning (Table 2). Insect abundance was positively related to the ACI at midday (11 am–3 pm) (Table 2, Fig. 2). In contrast, the species richness of birds and grasshoppers showed no significant relationship with the ACI at midday (Table 2, Fig. 2). R^2 was higher for ACI morning (0.24) compared to ACI midday (0.14), indicating model fit was equally good (Table 2). Fisher's C p-value as a global fit measure for the entire SEM was 0.0001 (ACI morning) and 0.0001 (ACI midday).

Relevant indirect land-use variables at the local scale were the vegetation height, which had the strongest positive association with the vegetation structure, which was positively linked with the insect abundance and the ACI (Table 2, Fig. 2). Fertilisation and plant species richness were not associated with insect abundance (Table 2, Fig. 2). Plant species richness was negatively related to fertiliser intensity. However, this was not associated with the ACI (Table 2, Fig. 2). Local land-use variables were not significantly associated with grasshopper species richness or bird species richness (Table 2, Fig. 2). Indirect land-use variables at the landscape scale were mainly the percentage AES, which was positively related to bird species richness (p-value: 0.027; Table 2, Fig. 2). There was no significant relationship between the percentage AES or percentage woody structure and insect abundance

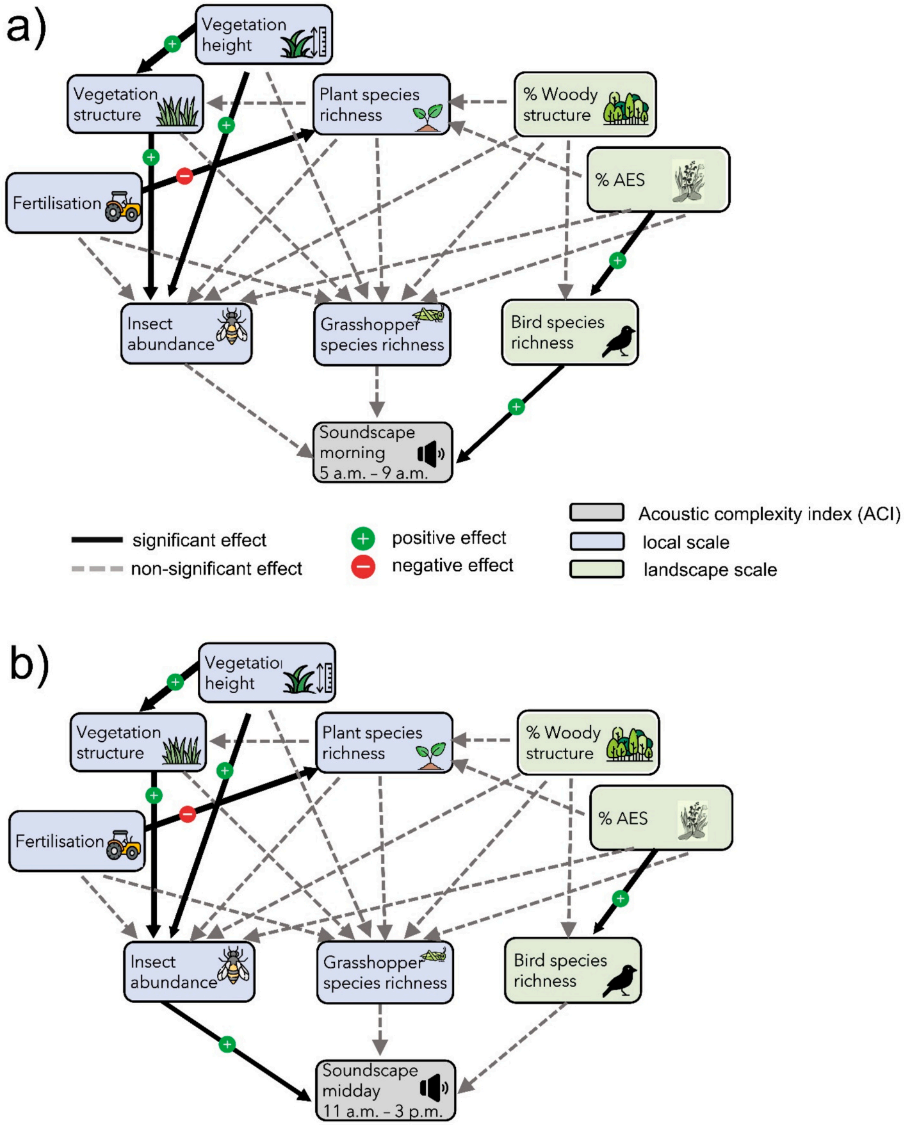

Fig. 2. Direct and indirect drivers of structural equation models related to a) morning acoustic complexity index (ACI) and b) midday ACI. Solid lines indicate significant paths. Thicker arrows represent a higher effect size. The grey dashed arrows indicate non-significant paths. Green-filled rectangles indicate drivers from the landscape scale, and blue-filled rectangles indicate drivers from the local scale. AES: Area managed according to the protocols of the Swiss agri-environmental scheme. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

and grasshopper species richness (Table 2, Fig. 2).

4. Discussion

4.1. *Association between the ACI and bird species richness, insect abundance, or grasshopper species richness*

Our study shows that the Acoustic Complexity Index (ACI) can serve as a proxy for species richness or abundance and is sensitive enough to detect management-related changes in agricultural landscapes. Bird species richness was positively related to the ACI in the morning, whereas insect abundance showed a positive relationship with the ACI at midday. These time-specific patterns suggest that the ACI can be used to monitor different species, provided that recordings are collected during periods of highest acoustic activity and under conditions with low levels of anthrophony and geophony. Despite these promising results, the lack of a relationship between grasshopper species richness and the ACI at midday was unexpected, given that Orthoptera typically show peak vocalisation activity around this time (Fischer et al., 1996). Possible explanations include low variability in species richness within this taxonomic group or dominance effects of a single vocally prominent species.

Our finding that bird species richness contributes substantially to acoustic complexity aligns with previous research in meadow ecosystems (Mueller et al., 2022). In contrast, the positive relationship observed between ACI and insect abundance has not previously been reported. Pieretti et al. (2011) did not detect such a relationship, which may be due to differences in data processing: we excluded recordings with high levels of anthrophony and geophony, reducing acoustic masking and increasing the sensitivity of ACI to biological signals (Metcalf et al., 2021). Regarding Orthoptera, the absence of a relationship between grasshopper richness and ACI is consistent with Mueller et al. (2023), who also found no link between grasshopper diversity and acoustic diversity indices.

Although ACI appears useful for tracking bird richness and insect abundance, further evidence is required to ensure its applicability in agricultural monitoring. In particular, it remains unclear whether passive acoustic monitoring (PAM) can reliably detect management effects

Table 2

Results of the two SEM, ACI birds in the morning and ACI insects at midday. Std. Estimate and *P*-value for each SEM sub-model, including the explanatory and response variables, are provided. For each explanatory variable and driver, the standard estimation, p-value, and R^2 are shown. Significant *p* values are shown in boldface type (*** $p < .001$, ** $p < .01$, * $p < .10.05$).

Explanatory variable	Response variable	Std. Estimate	*P*-value
ACI morning *R^2: 0.24*[1]	Bird species richness	**2.336**	**0.025****
	Grasshopper species richness	1.507	0.139
	Insect abundance	0.989	0.328
ACI midday *R^2: 0.14*	Bird species richness	0.996	0.336
	Grasshopper species richness	−0.466	0.644
	Insect abundance	**2.287**	**0.028****
Bird richness *R^2: 0.14*	Percentage woody structure	−1.138	0.255
	Percentage AES	**2.209**	**0.027****
Insect abundance *R^2: 0.84*	Fertilisation	−1.411	0.158
	Vegetation height	**0.047**	**0.0001******
	Vegetation structure	**3.527**	**0.0004******
	Plant species richness	−0.1012	0.312
	Percentage woody structure	−0.499	0.618
	Percentage AES	−0.313	0.754
Grasshopper species Richness *R^2: 0.32*	Fertilisation	0.592	0.555
	Vegetation height	1.629	0.103
	Vegetation structure	0.874	0.382
	Plant species richness	0.978	0.328
	Percentage woody structure	0.425	0.671
	Percentage AES	1.388	0.165
Plant species richness *R^2: 0.45*	Fertilisation	**−3.622**	**0.0003******
	Percentage woody structure	0.350	0.726
	Percentage AES	0.470	0.638
Vegetation structure *R^2: 0.12*	Vegetation height	**2.354**	**0.024****
	Plant species richness	0.835	0.409

[1] An R-squared of 0.24 indicates that the parameters explain 24% of the SEM submodel.

across different seasons or habitat types. The development and implementation of standardised protocols will be essential for comparing results across landscapes and taxa. For Orthoptera, species-poor assemblages or dominance by a single species may limit the detectability of richness patterns through acoustic indices. Future work should therefore assess how community composition, calling behaviour, and acoustic interference influence the performance of ACI and related metrics.

4.2. Association between bird species richness, insect abundance, or grasshopper species richness and land-use variables at local or landscape scale

With respect to land-use variables at the local scale, vegetation height was positively related to vegetation structure. This was positively related to insect abundance and acoustic complexity at midday. This corresponds to our expectations, as a complex vegetation structure is known to be positive for insect diversity (Lengyel et al., 2016; Minor et al., 2021; Schirmel et al., 2019). Fertilisation was not associated with insect abundance or grasshopper richness, but was only negatively related to plant species richness, which is a well-known relationship (Boob et al., 2021; Dickson and Gross, 2013; Francksen et al., 2022; Gross and Mittelbach, 2017; Mueller et al., 2022; Yamauchi et al., 2023). The lack of a negative relationship between fertilisation and insect diversity might be explained by vegetation height, which was the consequence of a mowing event, being the more important factor influencing insect abundance or grasshopper diversity. Passive acoustic monitoring likely has the potential to capture the relationship between local land use and insect abundance. However, our study shows that its ability to do so depends on the local land use variables.

The percentage of areas managed according to the protocols of the Swiss AES was positively associated with bird species richness, which was related to the ACI. Management prescriptions include for example no synthetic fertilisers, limited manure applications, the first cut not being before a specific date, pesticides being prohibited, and a minimum number of plant taxa present. This is in line with the finding that landscape improvement has a relevant impact on farmland bird species (Jenny et al., 2003; Meichtry-Stier et al., 2014; Meier et al., 2024; Ritschard et al., 2019; Zingg et al., 2019). Our study revealed that the percentage woody structure was not significantly related to bird species richness. One reason for this unexpected finding might be that we focused on farmland birds, which depend less on woody structures compared to forest birds, and we did not consider forest birds. The ACI likely has the potential to evaluate the positive effect of the percentage AES within the landscape on bird species richness.

There was no significant positive relationship between percentage AES or percentage woody structures and insects at the local scale. These results are unexpected because habitat structures and heterogeneity affect distribution patterns of insects and specifically grasshopper species (Guido and Gianelle, 2001; Reid and Hochuli, 2007). Furthermore, the proportion and quality of AES promote butterflies as well as vocalising and non-vocalising birds (Zingg et al., 2019), and an increasing percentage of grassland AES were directly positively related to the abundance of food-specialised butterflies (Meier et al., 2022). This could be because our results were dominated by the strong effect of local habitat conditions on insect abundance. Given that we found no effect of the surrounding landscape on insect abundance or grasshopper richness, we cannot conclude whether the ACI could potentially be used for quantifying the effect of landscape-scale biodiversity measures on species diversity at the local scale.

5. Conclusion

Overall, we concluded that ACI can be used for tracking management-related changes in grassland diversity in an automated and relatively cheap way. Our findings help to prioritise biodiversity-promoting measures in agricultural landscapes and to evaluate the effectiveness of agri-environmental measures. However, the sensitivity of acoustic indicators calculated from PAM depends on the species group and the spatiotemporal scale at which the species group is most active and audible. As the next step, we recommend assessing the costs of PAM and comparing them with traditional monitoring costs.

CRediT authorship contribution statement

Sibylle Stoeckli: Writing – review & editing, Writing – original draft, Formal analysis, Data curation. **Sina Helfenberger:** Writing – original draft, Methodology, Investigation, Formal analysis, Data curation. **Eliane Meier:** Methodology, Formal analysis, Conceptualization. **Eva Knop:** Writing – review & editing, Methodology, Conceptualization.

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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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2026.111723.

Data availability

Data will be made available on request.

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