

## Article

# The Potential to Save Agrestal Plant Species in an Intensively Managed Agricultural Landscape through Organic Farming—A Case Study from Northern Germany

Eugen Görzen , Tim Diekötter, Maike Meyerink, Helen Kretzschmar and Tobias W. Donath \*

Department of Landscape Ecology, Institute for Natural Resource Conservation, Kiel University, Olshausenstr. 75, 24118 Kiel, Germany; egoerzen@ecology.uni-kiel.de (E.G.); tdiekoetter@ecology.uni-kiel.de (T.D.); maikemeyerink@gmx.de (M.M.); helen.kretzschmar@gmx.de (H.K.)

\* Correspondence: tdonath@ecology.uni-kiel.de; Tel.: +49-431-880-1198



**Citation:** Görzen, E.; Diekötter, T.; Meyerink, M.; Kretzschmar, H.; Donath, T.W. The Potential to Save Agrestal Plant Species in an Intensively Managed Agricultural Landscape through Organic Farming—A Case Study from Northern Germany. *Land* **2021**, *10*, 219. <https://doi.org/10.3390/land10020219>

Academic Editor:  
Andreu Bonet-Jornet

Received: 10 February 2021

Accepted: 15 February 2021

Published: 22 February 2021

**Publisher's Note:** MDPI stays neutral with regard to jurisdictional claims in published maps and institutional affiliations.



**Copyright:** © 2021 by the authors. Licensee MDPI, Basel, Switzerland. This article is an open access article distributed under the terms and conditions of the Creative Commons Attribution (CC BY) license (<https://creativecommons.org/licenses/by/4.0/>).

**Abstract:** Intensive agriculture is among the main drivers of diversity decline worldwide. In Central Europe, pressures related with agriculture include habitat loss due to the consolidation of farming units, pesticide and fertilizer use, and shortened crop rotations. In recent decades, this development has resulted in a severe decline of agrestal plant communities. Organic farming has been suggested as a biodiversity friendly way of farming, as it strongly restricts the use of synthetic pesticides and fertilizers and relies on longer crop rotations. It may thus help in saving agrestal plant communities in the future. In this study, we assessed the long-term effects of three types of arable field management (conventional farming, organic farming, and bio-dynamic farming) on three farms in the federal state of Schleswig-Holstein, Northern Germany. We collected data on above-ground plant communities and seed banks and analyzed them with regards to the impact of the farming system and their position in the field using nonmetric multi-dimensional scaling (NMDS) and linear mixed effects models (LME) combined with ANOVA and Tukey contrast tests. Plants in organically or bio-dynamically managed fields differed in their composition and traits from those occurring in conventionally managed fields, i.e., they showed a preference for higher temperatures and were dominated by insect-pollinated species. While conventional farming had negative effects on vegetation and the seed bank, organic and bio-dynamic farms had neutral or slightly positive effects on both. This highlights the potential of the latter two to conserve species even in an intensively managed landscape. In addition, this may halt or even reverse the decrease in arthropod, bird, and mammal species, since agrestal plants constitute an important component of food-webs in agricultural landscapes.

**Keywords:** land management; organic farming; conventional farming; agroecosystems; plant species diversity; segetal vegetation; arable weeds

## 1. Introduction

Agriculture is among the main drivers of biodiversity decline worldwide [1]. In Central Europe, pressures related with agriculture include habitat loss due to the consolidation of farming units, pesticide and fertilizer use, and shortened crop rotations [2]. Organic farming has been suggested as a biodiversity-friendly way of farming as it strongly restricts the use of synthetic pesticides and fertilizers and relies on longer crop rotations [3]. Here, we assessed the long-term effects of organic and conventional arable field management on the above-ground and seed bank compositions of segetal plant communities on three farms in the federal state of Schleswig-Holstein, Northern Germany.

In Central Europe, segetal plants have evolved under continuous agricultural practices over the last 7000 years. Thus, they are well adapted to agriculturally managed growth conditions. However, since the 1950s, due to the development of different crop varieties, sowing and harvest times, soil tillage, fertilizer application, and the control of pest species, these conditions have drastically changed [4]. For example, chemical herbicides have

largely replaced mechanical weed control. Consequently, between 1950 and 1977 pesticide use on German farmland increased from <5% to around 90% of the area [5]. Between 1974 and 1994, the area sprayed with herbicides increased 2.5-fold in England and Wales [6] and worldwide 490,000 tons of herbicides were used for agricultural purposes in 2015—16,806 tons in Germany alone [7]. Similarly, the amount of nitrogen (N) and phosphorus (P) fertilizer used per unit of cropland area increased approximately eight times (0.9 to 7.4 gNm<sup>-2</sup> cropland yr<sup>-1</sup> on average) and three times (0.4 to 1.2 gPm<sup>-2</sup> cropland yr<sup>-1</sup> on average), respectively, globally between 1961 and 2013 [8]. The consequences of these changes in arable management on agrestal plant species is well documented in numerous studies, which all show a dramatic decrease in agrestal plant diversity [9–12].

In Germany, the agrestal flora encompasses around 10% of the total flora (around 350 species; [13]). While about 200 of these species have a broad ecological range, 150 of them are limited to arable fields [14]. The increase of monotonous intensively managed crop cycles exerts a high selective pressure on these plant communities and favors a relatively small group of plant species well adapted to the dominating arable management practice, i.e., high pulses of nutrient availability, herbicide input, and changes in the competition structure [11]. In contrast, species that rely on specific site conditions and/or management schemes, e.g., *Agrostemma githago* or *Consolida regalis*, are red-listed [10]. A recent study from the state of Baden Württemberg estimates the loss of agrestal species between 1948/49 and 2011 at 64% [15]. Concurrently, the cover of these species in agricultural landscapes decreased from 40% in the 1950s and 1960s to 4% in 2009 [16].

In addition to its intrinsic value, the flora specific to arable fields plays an important role in the functioning of agroecosystems. Agrestal species promote habitat structures and food resources for insect species like pollinators or the natural antagonists of pest species [17]. A diverse flora may also promote nutrient cycling, regulate the microclimate, or support the decomposition of toxic substances [17–19]. For these reasons, beginning in the 1970s, measures were undertaken to conserve the specific flora in Central Europe [20]. These measures included the installation of buffer strips along crop fields and the establishment of reservoirs in botanical gardens or in the open countryside [21,22]. However, due to their limited spatial and temporal extent and their regional occurrence, these measures have had only marginal effects on the conservation of agrestal species in agricultural landscapes in Central Europe at large [23]. Consequently, one opportunity for the large-scale re-establishment of agrestal plant communities is seen in organic farming [24,25]. In their review, Hole et al. [26] summarized that the majority of studies investigating the flora of arable and mixed farming systems recorded higher agrestal plant abundance and species richness in organically managed fields, regardless of the arable crop being grown. This difference in biodiversity between organic and conventional fields was higher for broad-leaved forb species than for grasses. Fields under organic management also held considerably more rare and/or declining species such as *Galeopsis angustifolia* [27], *Spergula arvensis* [27], *Centaurea cyanus* [28], or *Ranunculus arvensis* [27,29]. Also, Gabriel et al. [30] showed that the species richness of plants was higher in organic than in conventional fields. Only for organically managed fields, the richness of rare species (present in 5% of total samples) differed substantially among fields and regions due to environmental heterogeneity [30]. In contrast to bumblebees, butterflies, solitary bees, and epigeal arthropods, plants were shown to benefit from organic management per se and not only via reduced yields [31]. Generally, a lower management intensity and a higher propagule pressure towards the edge of fields leads to higher plant diversity at field margins as compared to the centers of fields, independent of management type [26,30].

For organic agriculture to be able to promote the agrestal flora, propagule sources of the respective species still need to exist in the landscape. Agrestal species in agricultural landscapes can originate from propagules of the recent or the past vegetation that, in the latter case, may be stored in the seedbank [32]. Similar to the vegetation itself, the seedbank has also been shown to respond to agricultural management. Management type, i.e., organic, integrated, and conventional [33] or bio-dynamic, bioorganic, and conventional

appeared to exert a selection pressure on the species composition of the seedbank, building up different communities over time. At a within-field scale, species richness and agrestal plant abundance in the seedbank was greatest in organic farms [33,34]. At the regional and landscape scale, however, species richness increases with the number of crop types and cropping practices between fields, which was higher for integrated and conventional than organic farms [33]. Thus, both the above-ground vegetation and the seed bank should be studied when assessing the effects of agricultural management on the agrestal flora or the management-dependent potential for the re-establishment of agrestal flora in modern agricultural landscapes.

In the present study, we were interested to know whether (a) the diversity of the agrestal plant community differs among different farming systems, i.e., conventional, bioorganic, and biodynamic, (b) diversity patterns in the above-ground vegetation correspond to patterns in the seed bank, and (c) diversity patterns in agrestal plants vary with position within agricultural fields.

## 2. Materials and Methods

### 2.1. Study Area

Our study area is located in the administrative district Duchy of Lauenburg, in the southeastern part of Schleswig-Holstein, Germany, approximately 50 km East of Hamburg. The region is characteristic of the eastern hillside region of Schleswig-Holstein dominated by agriculture [35]. The three investigated farms are located in close proximity (maximum distance between two investigated fields: 5 km), allowing the comparison of different land management types under similar edaphic and climatic conditions. The climate in this part of Schleswig-Holstein is temperate, with a mean annual temperature of 8.9 °C and mean annual precipitation of 692 mm [35,36]. The continental climatic impact is more pronounced in this region compared to other parts of Schleswig-Holstein where the climate has a more Atlantic character [35]. Soil types in the young morainic landscape of the study area predominantly include cambisols and luvisols, while locally, colluvisols, gleysols, and lowland peat soils also exist [37,38]. The substrate is mainly composed of sand or loam covered by sand, primarily from glacial deposits [37–39].

Within this area, we selected a conventional farm and two organically managed farms. The conventional farm relies on intensive conventional arable management, e.g., input of synthetic fertilizer and pesticides, which both organic farms, i.e., farm Ritzerau and the farm Lämmerhof, are not allowed to make use of. Instead, they rely on organic fertilizer, e.g., manure, compost, and under- and inter-sown Fabaceae to increase soil fertility. In addition, they have to refrain from the use of pesticides but have to control weeds and pests mainly by wider crop rotations, tillage timing, and weed harrowing. These different management systems also imply differences in soil tillage and crop rotation. While the organic farms employ only ploughing (Ritzerau 25–30 cm deep; Lämmerhof around 20 deep), the conventional farm ploughs only half of the fields (around 30 cm deep). At the latter farm the other half of the fields are grubbed (30 cm deep). Also, the crops grown in the fields differ considerably. The conventional farm manages 166 ha of arable fields, which are split into 19 fields (average field size 8.7 ha). Its most important crop by far is winter wheat (90 ha), followed by rape (58 ha) and winter barley (18 ha). In contrast, the proportion of crops are more balanced at the organically managed Ritzerau farm, whose arable fields cover a total of 108 ha, divided into eight fields (average field size 13 ha; oats: 25 ha, winter rye: 35 ha, winter wheat: 25 ha, spelt: 22 ha). The bio-dynamically managed Lämmerhof farm is the largest farm (total arable area 253 ha) but its area is split into 50 fields and therefore fields cover on average only 5 ha. At these fields seven different crops are grown: winter rye (80 ha), oats (60 ha), spelt (50 ha), summer barley (25 ha), summer wheat (20 ha), winter wheat (10 ha), and rape (8 ha). Not unexpectedly, the farms differ with respect of their productivity, e.g., while the wheat harvest at the conventional farm ranges between 9 to 11 t/ha, both organic managed farms only yield 3 to 6 t/ha.

The Ritzerau farm was gradually transformed from 2001 to 2003 to organic farming according to EU Organic Regulation 2091/92 [40]. Since then, it has ended the use of chemical pesticides and synthetic fertilization. The arable fields of the Lämmerhof were organically managed according to the terms of the Bioland Association since 1989. In 2004 the farm management followed the terms of biodynamic organic farming according to the Demeter guidelines [41]. These put special emphasis not only on closed nutrient cycles, soil fertility, and biodiversity but also make use of various esoteric concepts. Demeter guidelines are even more restrictive for the input of substances used for plant protection and organic nitrogen fertilizers than regulations for organic farming in accordance with the EU Organic Regulations 2091/92. In general, animal husbandry is mandatory for bio-dynamic farms managed according to the Demeter guidelines [41]. It is important to note that the management of the Lämmerhof farm pays special attention to fostering biodiversity on the farmland [41].

## 2.2. Vegetation Sampling

In order to compare the composition and abundance structure of the above-ground agrestal flora between the different management types (conventional farming, organic farming, and bio-dynamic farming), a vegetation survey was carried out from 6 July 2015 until 18 August 2015. We sampled all vascular plant species in homogeneous 100 m<sup>2</sup> (1 × 100 m) plots using an ordinal scale that was optimized for the estimation of species abundance based on the number of individuals using long but narrow plots (Table 1). Since we expected species diversity to decrease along a gradient from the edge to the center of the field [42], paired samples (field center vs. field edge; three replicates per field) were obtained. The plots for the assessment of the agrestal flora at the field margin were placed 2 m away and parallel to the field margin. If feasible, vegetation assessment was finalized prior to the harvest. Only in a small number of cases (conventionally farmed rape fields and bio-dynamically farmed ray fields) we were forced to perform vegetation sampling after the harvest to prevent excessive damage to the crops. In total, 215 plots (78 in conventionally managed farming fields, 102 in bio-dynamic managed fields, and 35 in organically managed farming fields) were sampled within the framework of aboveground vegetation assessment.

**Table 1.** Abundance scale for vegetation assessment in the field (left) and transformed averaged scale for data analysis (right): Numbers refer to (averaged) species counts or estimates.

Field Assessment Scale	Transformed Analysis Scale
1	1
2–5	3
6–10	8
11–50	30
51–100	75
101–500	300
501–1000	750
1001–5000	3000
>5000	7500

Initially, during the vegetation survey, all vascular plant species were recorded. However, since we aimed to focus specifically on the agrestal flora, all crop species, under-sown species, and shrub saplings were excluded for all subsequent analyses.

For studying the seed bank, we selected six fields per farm and collected seed bank samples every 10 m within a plot of 100 m length and 1 m width using a soil auger ( $\varnothing = 3.5$  cm, depth = 20 cm). Equivalent to aboveground vegetation assessment, paired samples were collected from the field center and the margins (2 m distance to the edge) by the end of February 2017. Thus, in total we obtained 36 samples (12 for each land-use type) for seed bank analysis. The 10 soil cores from each plot (0–20 cm) were pooled in

sealed bags and mixed thoroughly to obtain homogeneous seed bank samples resulting in a sampled soil volume of 1924 cm<sup>3</sup> and a sampled surface area of 96.2 cm per unit. Species composition in the samples was determined by applying the seedling emergence method [43]. The samples were spread in Styrofoam trays on a 1 cm layer of sterilized potting soil and placed in an unheated greenhouse, watered regularly to field capacity, and exposed to artificial light (11 h/day) for five months. Seedling determination and counting was performed instantly after seed germination. In case immediate species identification after germination was not possible, individual seedlings were transferred to plant pots filled with potting soil and raised until determination was feasible. As soon as the first germination peak was passed after 14 weeks, we simulated a two-week period of drought to activate dormant seeds and to suppress the expansion of bryophytes by suspending irrigation and allowing the samples to dry. Following this, the soil was watered again and kept moist for an additional four weeks until the end of the experiment to record all subsequently germinating seeds.

### 2.3. Statistical Analyses

We used the information available from the German SL 1.2 dataset implemented in Turboveg 2.118 [44] to calculate cover-weighted Ellenberg indicator values [45] for the vegetation and the seed bank datasets separately. The Ellenberg indicator values for nitrogen (N-value), light availability (L-value), moisture (F-value), temperature (T-value), and soil reaction (R-value) were used as indicators of prevailing environmental conditions. Ellenberg indicator values, which are derived empirically from field studies, reflect the ecological requirements of a plant species towards a specific site factor under field conditions factoring in interspecific competition [45–47]. They usually range from 1 to 9 indicating a plant's preference for low and high levels of the site factor of interest, respectively. For diversity assessment between land use types, we calculated the number of species per plot, the Shannon diversity (H-index), and evenness; all calculations were based on the functions implemented in PC-ORD 7.03 [48].

Variations of agrestal species abundance between the three management types, conventional farming (CF), organic farming (OF), and bio-dynamic farming (BDF), as well as differences in species composition between the field edge and the center of the fields were assessed separately for aboveground vegetation and the seed bank using nonmetric multidimensional scaling (NMDS) in PC-ORD 7.03 [48]. For NMDS analysis, averaged occurrences of species in the vegetation and seed bank datasets were standardized to their percentage share within each vegetation record. For mathematical reasons, to enable calculation of the distance matrices within the framework of the NMDS, we excluded six plots with zero occurrences of plants from the vegetation dataset (final  $n = 209$ ) and one plot from the seed bank dataset (final  $n = 34$ ), but included the respective samples in all subsequent analyses to ensure accurate estimation of richness and diversity indices comparing the different levels of land use type and position. The axes for the final solutions of the NMDS for the vegetation and seed bank datasets (in both cases using the Bray–Curtis distance measure, 500 permutations, and three dimensions) were rescaled using the function “orthogonal principal axes” [49]. Cover-weighted Ellenberg indicator values for nutrient availability (N-value), light availability (L-value), temperature (T-value), moisture availability (F-value), and soil reaction (R-value) were included as categorical factors in both (aboveground vegetation and seed bank) analyses. Species were defined as “long-lived” if a considerable number of entries (>10% in the BIOFLOR vegetation database [50]) indicated at least a potential for species persistence. In both analyses, quantitative variables with a coefficient of determination of  $r^2 \geq 0.25$  were correlated with the ordination axes to explain variation in the distribution of the agrestal vegetation [48].

The evaluation of the diversity, Ellenberg, and ecosystem indicators of the vegetation and seed bank data sets (indicators specified in Table 2) started with the definition of an appropriate linear mixed effect model [51] using the software R, version 3.6.0 [52]. The included indicators red list (RL) species referring to categories V and 1–3 in the red list of

Schleswig-Holstein [12]. High nature value (HNV) species were derived from the recent biotope mapping manual for Schleswig-Holstein [53], which included all species from the categories high nature value (HNV) grassland, farmland, and fallows (see Table 2).

**Table 2.** ANOVA results—effects of land use type, position, and their interaction on agrestal vegetation and the seed bank of agrestal vegetation (diversity indicators, Ellenberg indicator values, and ecosystem indicators such as the abundance of species pollinated by insects, high nature value (HNV) species, and red list (RL) species).

Indicators	Transformation	pseudo R <sup>2</sup>		Land Use Type	Position	Land Use Type × Position
		Marginal	Conditional	p-Value	p-Value	p-Value
<b>Vegetation</b>						
<i>Diversity</i>						
Richness		0.41	0.83	<0.001	<0.001	<b>0.018</b>
Shannon		0.46	0.66	<0.001	<0.001	<b>0.005</b>
Evenness		0.47	0.66	<0.001	<0.001	<b>0.005</b>
No. of individuals	log (x + 1)	0.51	0.86	<0.001	<0.001	<0.001
<i>Ellenberg</i>						
L-value		0.02	0.56	0.634	0.896	0.159
T-value		0.12	0.29	<b>0.001</b>	0.712	0.113
N-value		0.10	0.40	<b>0.024</b>	0.687	0.190
F-value		0.07	0.42	0.908	<b>0.012</b>	<b>0.008</b>
R-value		0.10	0.42	0.051	0.256	0.824
<i>Ecosystem</i>						
Abundance insect pollination	log (x + 1)	0.50	0.85	<0.001	<0.001	<0.001
Abundance HNV species	log (x + 1)	0.64	0.83	<0.001	<0.001	<0.001
Abundance RL species	log (x + 1)	0.28	0.62	<b>0.001</b>	0.550	<b>0.024</b>
<b>Seed bank</b>						
<i>Diversity</i>						
Richness		0.71	0.82	<0.001	0.403	0.750
Shannon		0.53	0.53	<b>0.001</b>	0.737	0.596
Evenness		0.53	0.53	<b>0.001</b>	0.693	0.597
Seed density	sqrt	0.35	0.48	<b>0.002</b>	0.301	0.689
<i>Ellenberg</i>						
L-value		0.11	0.62	0.271	0.256	0.909
T-value		0.81	0.84	<0.001	0.369	0.141
N-value		0.09	0.25	0.098	0.331	0.917
F-value		0.19	0.22	<b>0.007</b>	0.215	0.906
R-value	box-cox	0.35	0.41	<b>0.011</b>	0.196	0.448
<i>Ecosystem</i>						
Abundance insect pollination	log (x + 1)	0.26	0.30	<b>0.006</b>	0.632	0.361
Abundance HNV species	log (x + 1)	0.19	0.23	<b>0.034</b>	<b>0.02</b>	0.330

Raw data were used unless transformation type is indicated. Bold font indicates significant differences ( $p \leq 0.05$ ): RL species abundance was not tested for the seed bank dataset since no red list species were present.

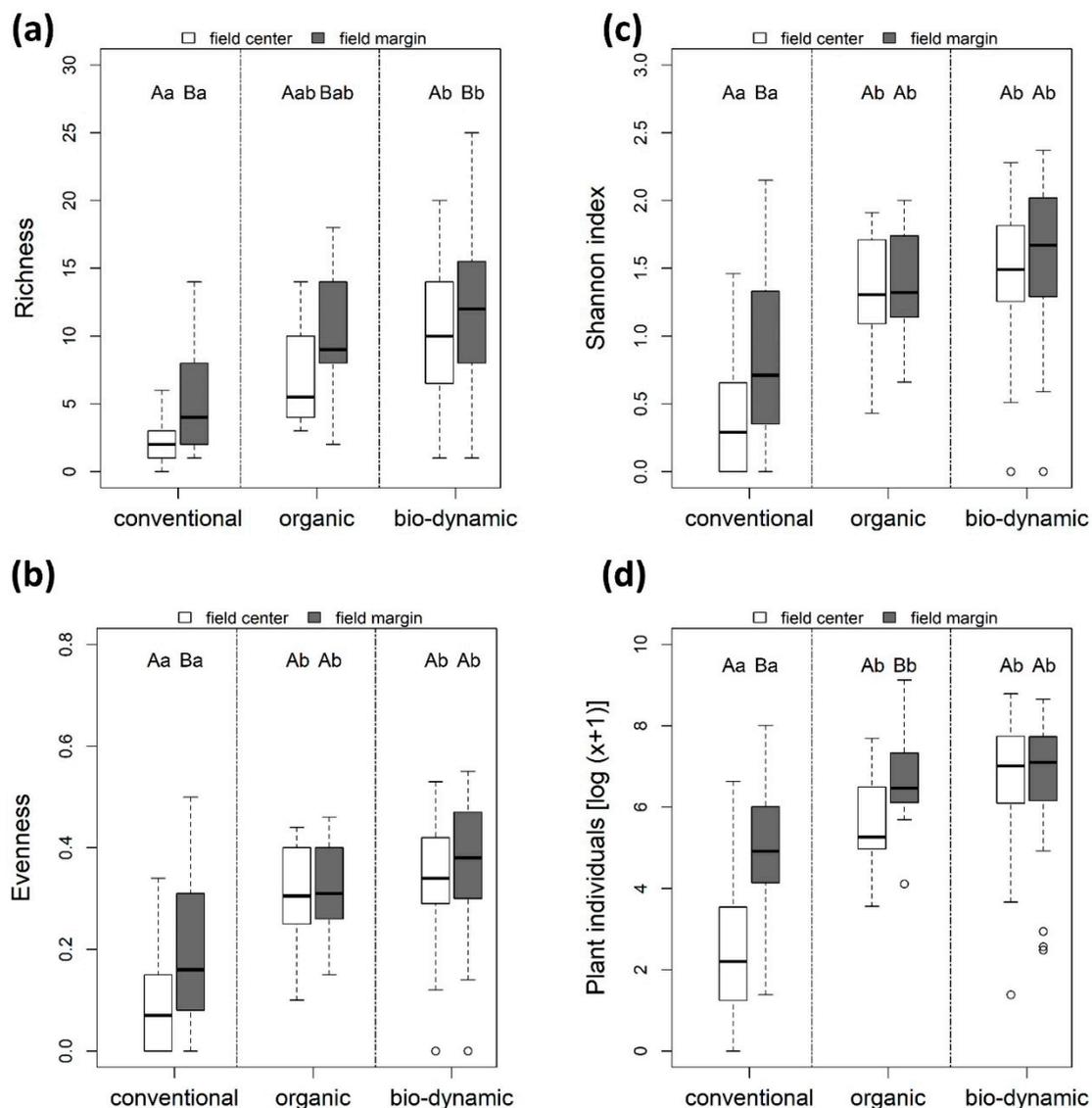
Based on visual residuals analysis, we concluded that the data met assumptions of normality for analysis of variance (ANOVA). In case the data were heteroscedastic (either due to the different levels of land use type (CF, OF, BDF) or due to the different levels of position in the field (margin, center), we used the weights function in R package nlme [54] to account for heteroscedasticity. The weights function calculates the variance for each level of position or land use instead of the variance for the entire model. If necessary, data transformations (specified in Table 2) were performed to meet the criteria of the model. The statistical model for the evaluation of the vegetation and seed bank data included the factors land use type (CF, OF, BDF), the level of position in the crop field (margin, center), and their interactions as fixed effects. Position (i.e., field margin vs. field center) and plot nested in field were defined as random factors. Based on our model, we used ANOVA followed by Tukey contrast post-hoc tests to compare the several levels of the influence factors [55]. Being primarily interested in the general effect of land use and position in the field, we focused on specific tests rather than testing all levels of variables of interest

against all others. Therefore, in the first step, we tested whether the indicators of interest varied in position between the edge and the center of the fields. In a second step, we tested for differences in the dependent variables between the three levels of land use type (CF, OF, BDF), distinguishing between the position at the margin or center of the field.

### 3. Results

#### 3.1. Agrestal Flora above Ground

The number of individuals of agrestal plants was significantly higher at the field margin compared to the field center in CF and OF but not in BDF (Table 2, Figure 1d). Averaged across field margins and centers, the number of plant individuals per plot in OF ( $1124 \pm 318$ ) or BDF ( $1742 \pm 173$ ) was more than five and eight times higher than in CF ( $202 \pm 47$ ), respectively.

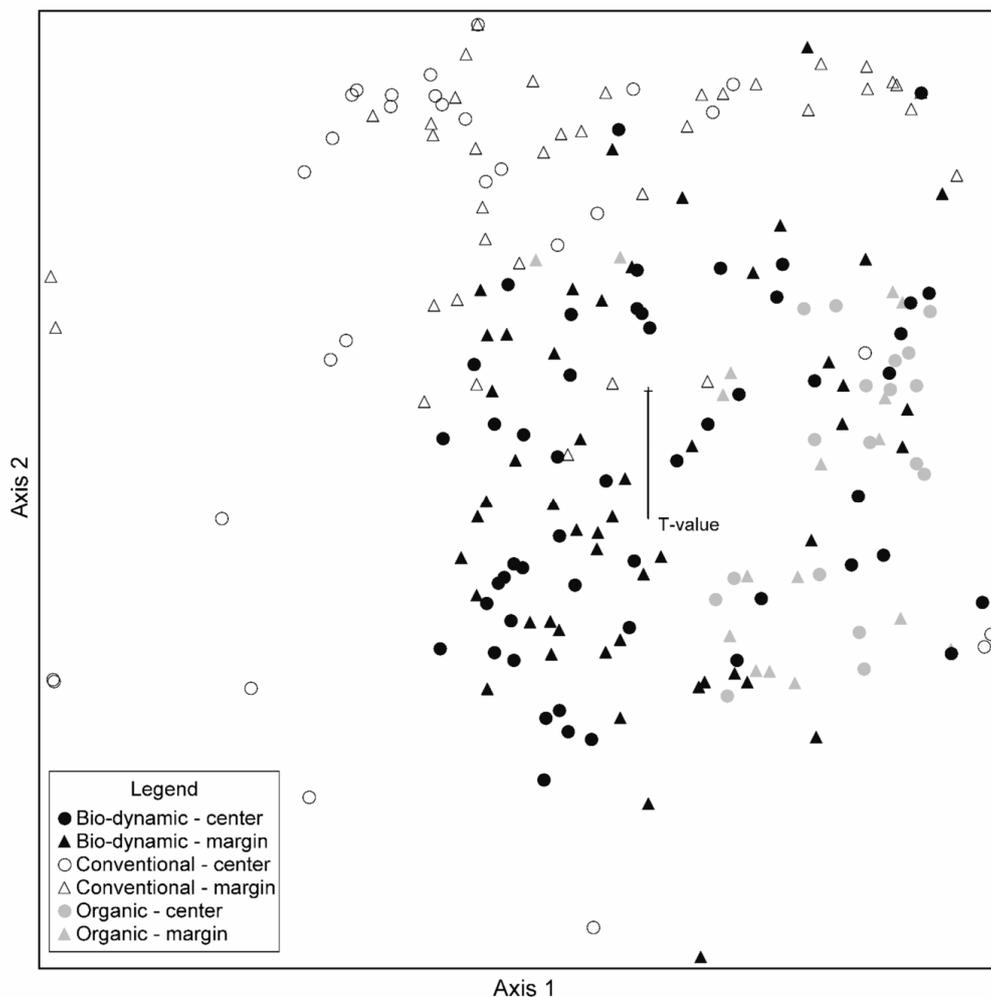


**Figure 1.** Richness (a), evenness (b), Shannon index (c), and the number of plant individuals (d) in the above-ground agrestal flora. For greater clarity, the number of plant individuals is presented as  $\log [x + 1]$ . Boxes indicate the range from the first quartile to the third quartile, whiskers cover 1.5 times the interquartile range (IQR) beyond the box and circles indicate outliers  $>1.5 \times$  IQR beyond the box. Capital letters (A, B) indicate significant differences ( $p \leq 0.05$ ) between the field center and the field margin within each of the three land use types. Lowercase letters (a, b) indicate significant differences ( $p \leq 0.05$ ) of the position field center between the three land use types and of the position field margin between the three land use types, respectively. Significant differences between the groups are based on ANOVA and post-hoc Tukey contrast tests.

Above ground, agrestal plant species richness was significantly influenced by an interaction of land use and position within the crop field ( $p = 0.018$ , Table 2). Richness at the field margins was on average 1.25 times higher compared to the field centers in all three land-use systems (Figure 1a). Pooled across field position, richness was more than twice as high in organically managed and almost three times as high in the bio-dynamically managed fields compared to the conventionally managed fields (mean  $\pm$  sd CF:  $3.8 \pm 0.4$ ; OF:  $8.3 \pm 0.39$ ; BDF:  $11.4 \pm 0.5$ ).

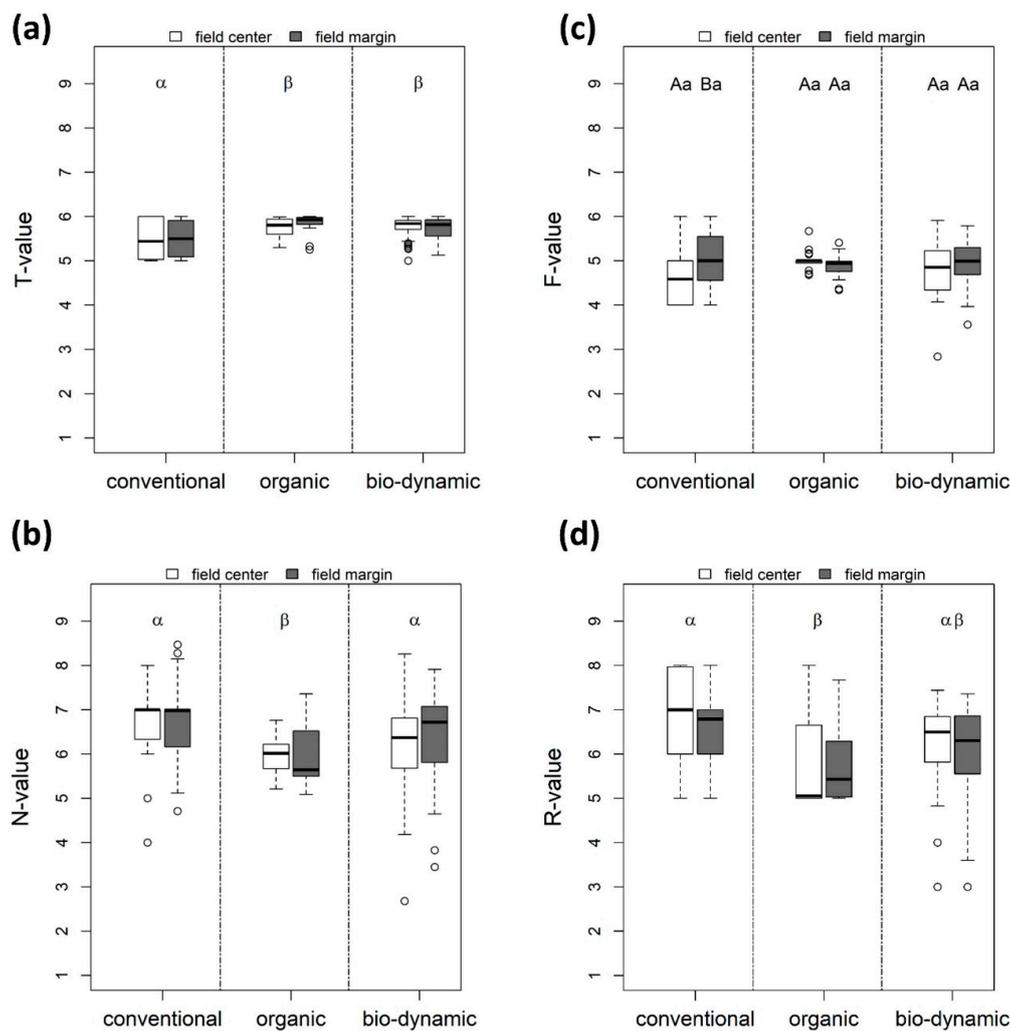
Also, Shannon and evenness indices were interactively affected by land-use system and position within the crop field ( $p = 0.005$  each; Table 2). Only in conventionally managed fields did both indices indicate significant differences between the field margins and the field centers (Figure 1b,c). The Shannon and evenness indices were more than twice as high in OF and BDF compared to CF (Figure 1b,c). No differences existed between OF and BDF.

NMDS-ordination of the agrestal flora above ground (Figure 2) indicated a strong separation between conventional farming (CF) and both organic farming (OF) and bio-dynamic farming (BDF) along Axis 2, and a weaker separation between BDF and OF along Axis 1. No clear distinction between the field center and the field margin was observed. Ellenberg T-values ( $r^2 = 0.28$ ) were correlated with NMDS Axis 2.



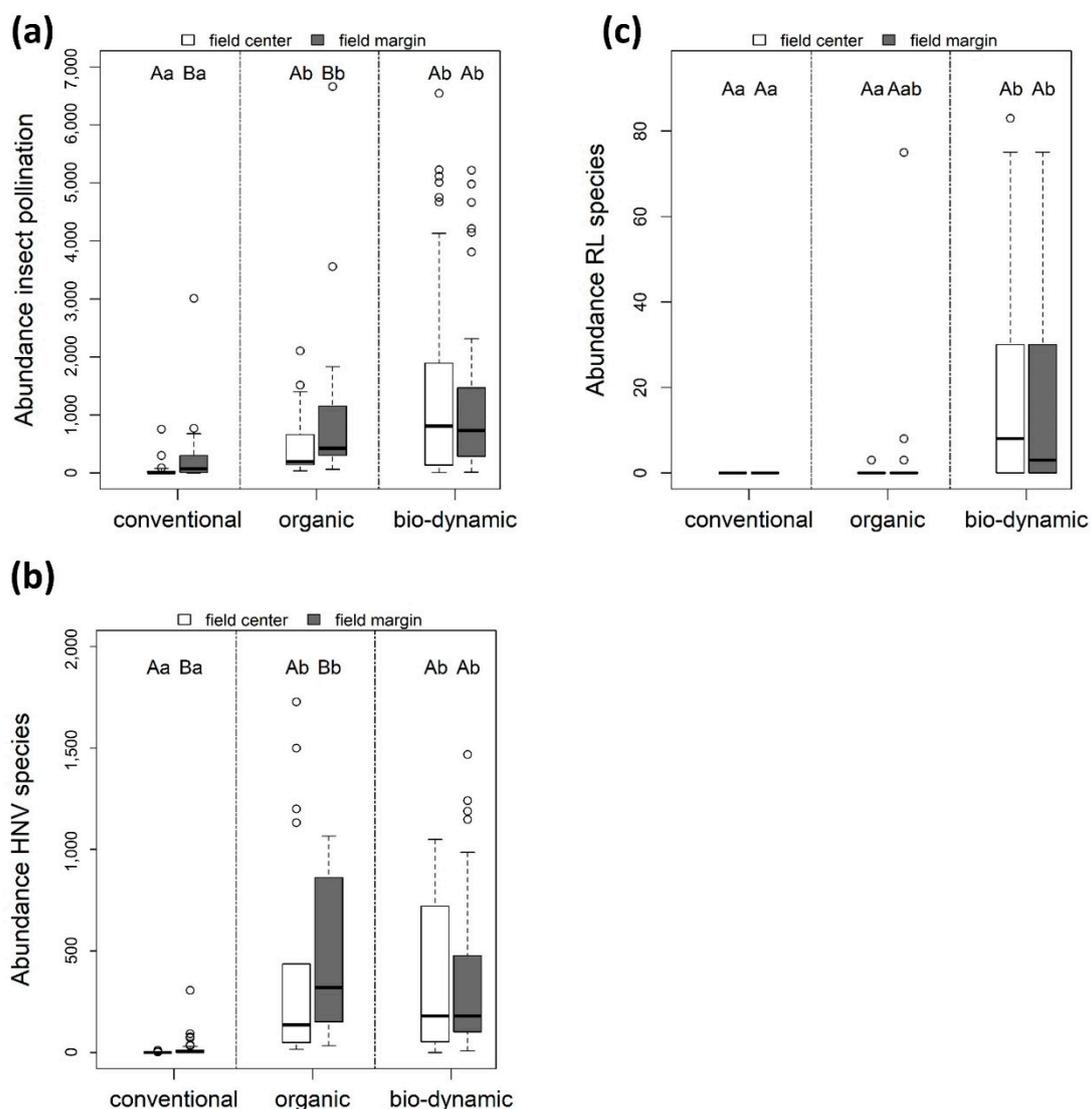
**Figure 2.** Non-metric multidimensional scaling (NMDS, Bray–Curtis dissimilarity, three dimensions, 500 iterations, final stress = 16.9) of the agrestal flora (above-ground vegetation): Black signatures represent bio-dynamic land use, grey signatures represent organic land use, and white signatures represent conventional land use, circles represent the field center and triangles represent the field margin. Arrows indicate correlation of Ellenberg indicator values with the NMDS axes ( $r^2 \geq 0.25$ ), arrow length denotes the strength of the respective correlation. Only the correlation of the Ellenberg temperature indicator value (T-value) was higher than the threshold value  $r^2 \geq 0.25$ .

Higher Ellenberg T-values (Figure 3a) were observed for BDF ( $5.7 \pm 0.02$ ,  $p = 0.003$ ) and OF ( $5.8 \pm 0.04$ ;  $p = 0.004$ ) as compared to CF ( $5.48 \pm 0.05$ ;  $p = 0.001$ ; Table 2), whereas the position within fields or the interaction of land use and this position had no effect. Ellenberg N-values (Figure 3b) were significantly influenced by land use type ( $p = 0.024$ ; Table 2). Pooled across position, they were significantly lower in OF ( $5.96 \pm 0.09$ ;  $p = 0.019$ ) as compared to CF ( $6.76 \pm 0.10$ ), but not BDF ( $6.34 \pm 0.10$ ;  $p = 0.324$ ). Ellenberg F-values were influenced by the interaction of land-use and position ( $p = 0.008$ ; Table 2). Post-hoc Tukey contrasts indicated a significant difference between the field centers and the margins only in CF ( $p = 0.002$ ; Figure 3c). Land use showed a tendency to influence Ellenberg R-values ( $p = 0.051$ , Table 2), and post-hoc Tukey contrasts confirmed significantly higher mean R-values in OF than in CF fields ( $p = 0.04$ ; Figure 3d). Neither land-use type nor position affected the Ellenberg L-values.



**Figure 3.** Ellenberg indicator values for temperature (T-value, a), nutrient availability (N-value, b), moisture availability (F-value, c), and soil reaction (R-value, d) in above-ground vegetation. Only those Ellenberg indicator values are presented where significant effects of land use type, position, or their interaction were detected. Boxes indicate the range from the first quartile to the third quartile, whiskers cover 1.5 times the interquartile range (IQR) beyond the box and circles indicate outliers  $>1.5 \times$  IQR beyond the box. In the case of the F-value, capital letters (A, B) indicate significant differences ( $p \leq 0.05$ ) between the field center and the field margin within each of the three land use types. Lowercase letters (a, b) indicate significant differences ( $p \leq 0.05$ ) of the position field center between the three land use types and the position field margin between the three land use types, respectively. In the case of the T-value, N-value, and R-value Greek letters ( $\alpha$ ,  $\beta$ ) indicate significant differences ( $p \leq 0.05$ ) between the land use types pooled across position. In these cases, no significant interaction of land use type and position or effect of position was detected. Significant differences between the groups are based on ANOVA and post-hoc Tukey contrast tests.

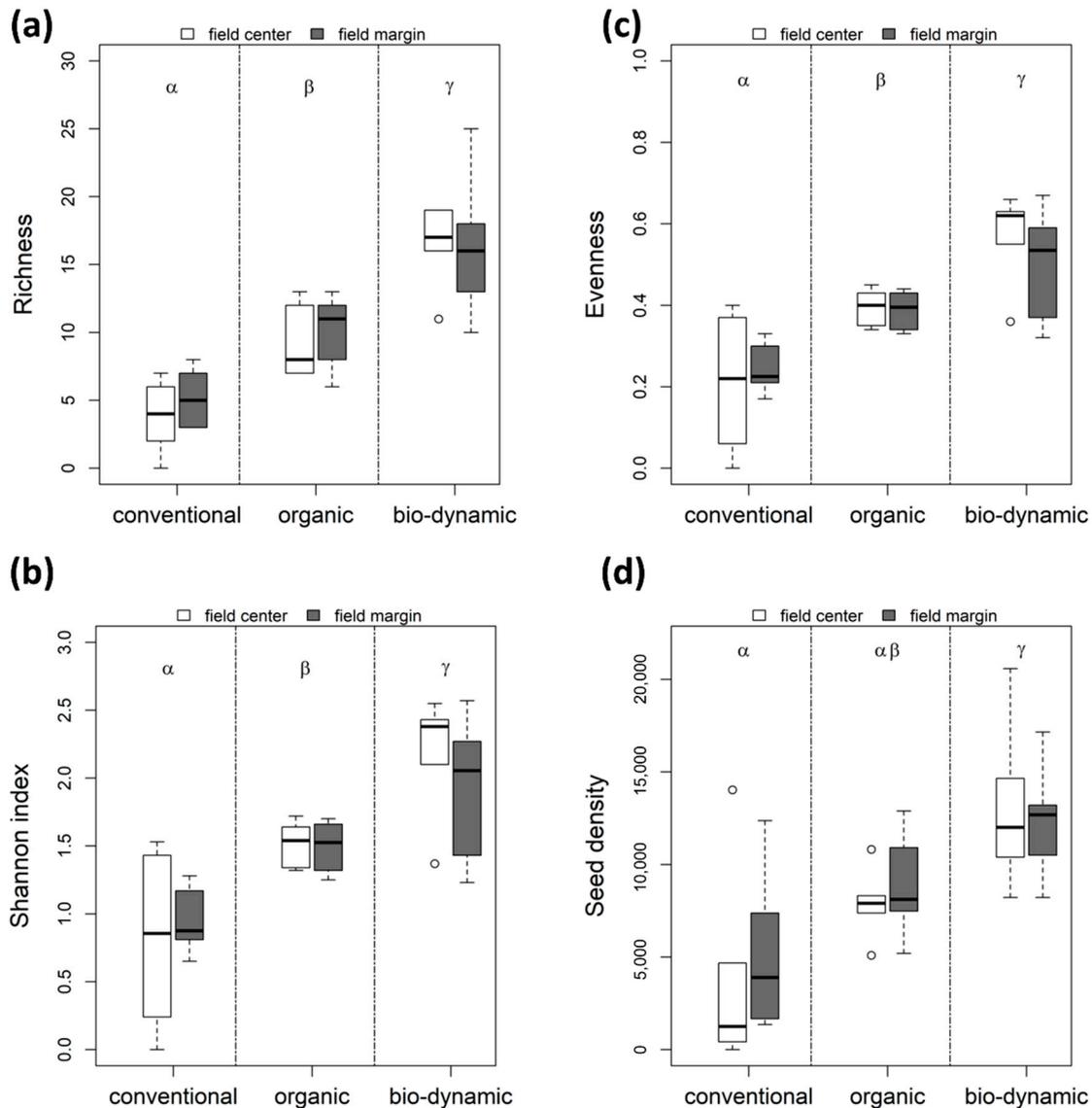
The abundance of individuals of insect pollinated agrestal plants (Figure 4a) was higher in field margins compared to the field center in CF ( $p \leq 0.001$ ) and OF ( $p = 0.006$ ), but not in BDF ( $p = 0.984$ ). In both field centers and margins, the abundance of insect pollinated agrestal plants was significantly lower in CF ( $137.09 \pm 42.18$ ) compared to OF ( $1066.31 \pm 318.97$ ) and BDF ( $1353.31 \pm 156.57$ ). Similarly, the abundance of HNV species (Figure 4b) was higher in the field margins compared to the field centers in CF ( $p < 0.001$ ) and OF ( $p = 0.0213$ ), but not in BDF ( $p = 1.0$ ). In both field centers and margins, the abundance of HNV species was significantly lower in CF ( $11.10 \pm 4.36$ ) compared to OF ( $794.57 \pm 273.47$ ) and BDF ( $629.60 \pm 102.36$ ). The interaction of land-use type and position significantly influenced the abundance of red list species ( $p = 0.024$ ; Table 2). Within the field centers, RL species abundance (Figure 4c) in BDF was significantly higher compared to CF ( $p \leq 0.001$ ) and OF ( $p = 0.004$ ). At the field margin BDF differed significantly only from CF ( $p = 0.003$ ).



**Figure 4.** Abundance of insect pollinated agrestal plants (a), abundance of high nature value (HNV) species (b), and abundance of red-list (RL) species (c) in above-ground vegetation. Boxes indicate the range from the first quartile to the third quartile, whiskers cover 1.5 times the interquartile range (IQR) beyond the box and circles indicate outliers  $>1.5 \times$  IQR beyond the box. Capital letters (A, B) indicate significant differences ( $p \leq 0.05$ ) between the field center and the field margin within each of the three land use types. Lowercase letters (a, b) indicate significant differences ( $p \leq 0.05$ ) of the position field center between the three habitat types and the position field margin between the three land use types, respectively. Significant differences between the groups are based on ANOVA and post-hoc Tukey contrast tests.

### 3.2. Agrestal Flora Seed Bank

Seed density in the seed bank varied with land-use type ( $p = 0.002$ ; Table 2; Figure 5d). Seed density in BDF ( $12,689 \pm 1032$ ) but not in OF ( $8341 \pm 652$ ) was significantly higher than in CF ( $4308 \pm 1365$ ). No effect of position or interaction of land use type and position was found.

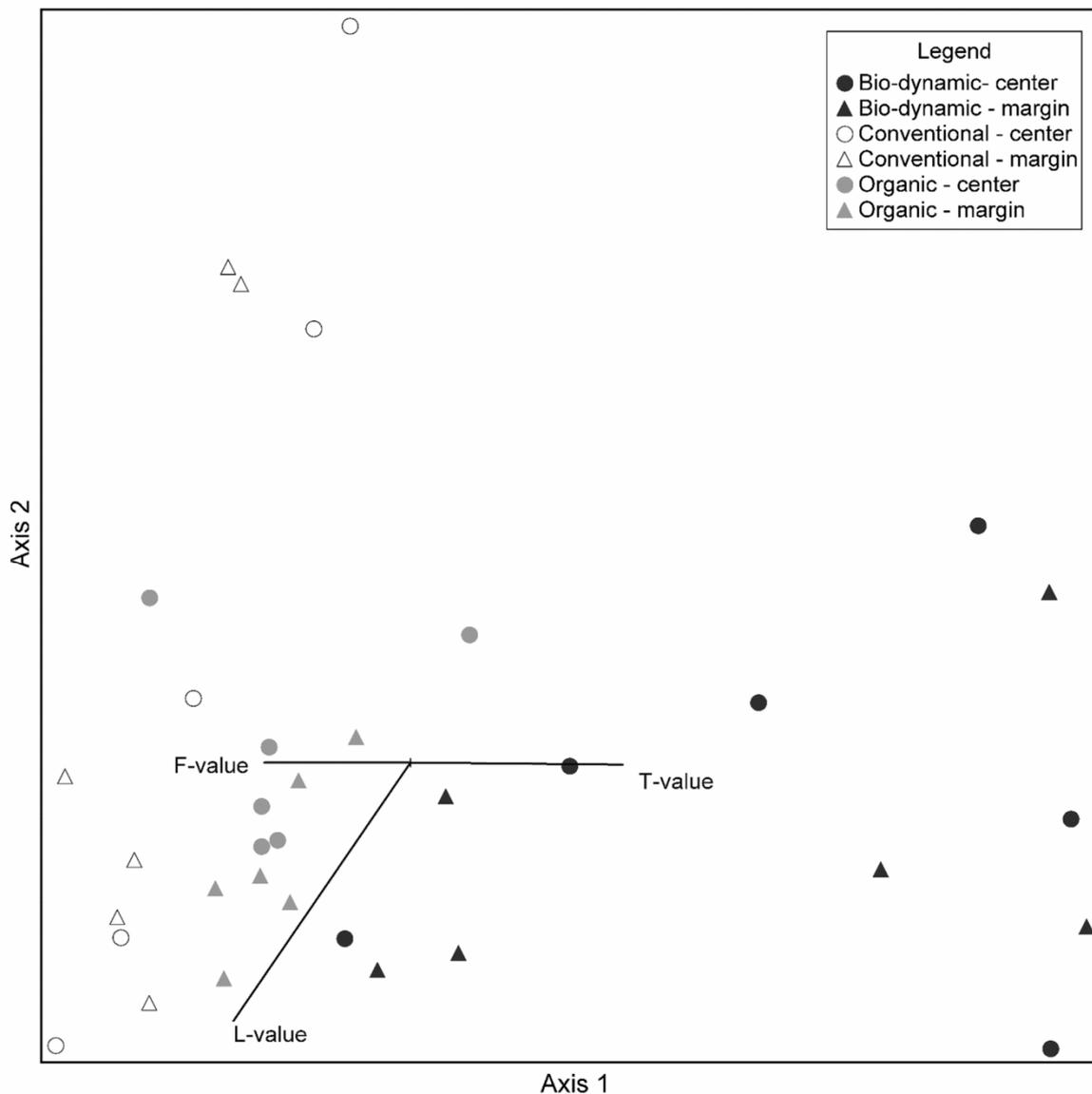


**Figure 5.** Richness (a), Shannon index (b), evenness (c), and seed density (d, raw data presented) in the seed bank of the agrestal flora. Boxes indicate the range from the first quartile to the third quartile, whiskers cover 1.5 times the interquartile range (IQR) beyond the box and circles indicate outliers  $>1.5 \times$  IQR beyond the box. Greek letters ( $\alpha$ ,  $\beta$ ,  $\gamma$ ) indicate significant differences ( $p \leq 0.05$ ) between the land use types pooled across position. In these cases, no significant interaction of land use type and position or effect of position was detected.

Seed bank richness varied significantly with land-use type ( $p \leq 0.001$ ; Table 2; Figure 5a), and it was around three times higher in BDF ( $16.4 \pm 1.2$ ) than in CF ( $4.5 \pm 0.7$ ), and still more than one and a half times higher than in OF ( $9.7 \pm 0.8$ ). No effect of position or interaction of land-use type and position was found. This general pattern, i.e., higher values for the diversity measures in OF as well as BDF fields than under CF, corresponded to the Shannon and evenness indices as well ( $p \leq 0.001$  each; Table 2 and Figure 5b,c).

In the NMDS the seed bank was strongly separated in its composition among CF, OF, and BDF along Axis 1, whereas field centers and field margins were not clearly separated

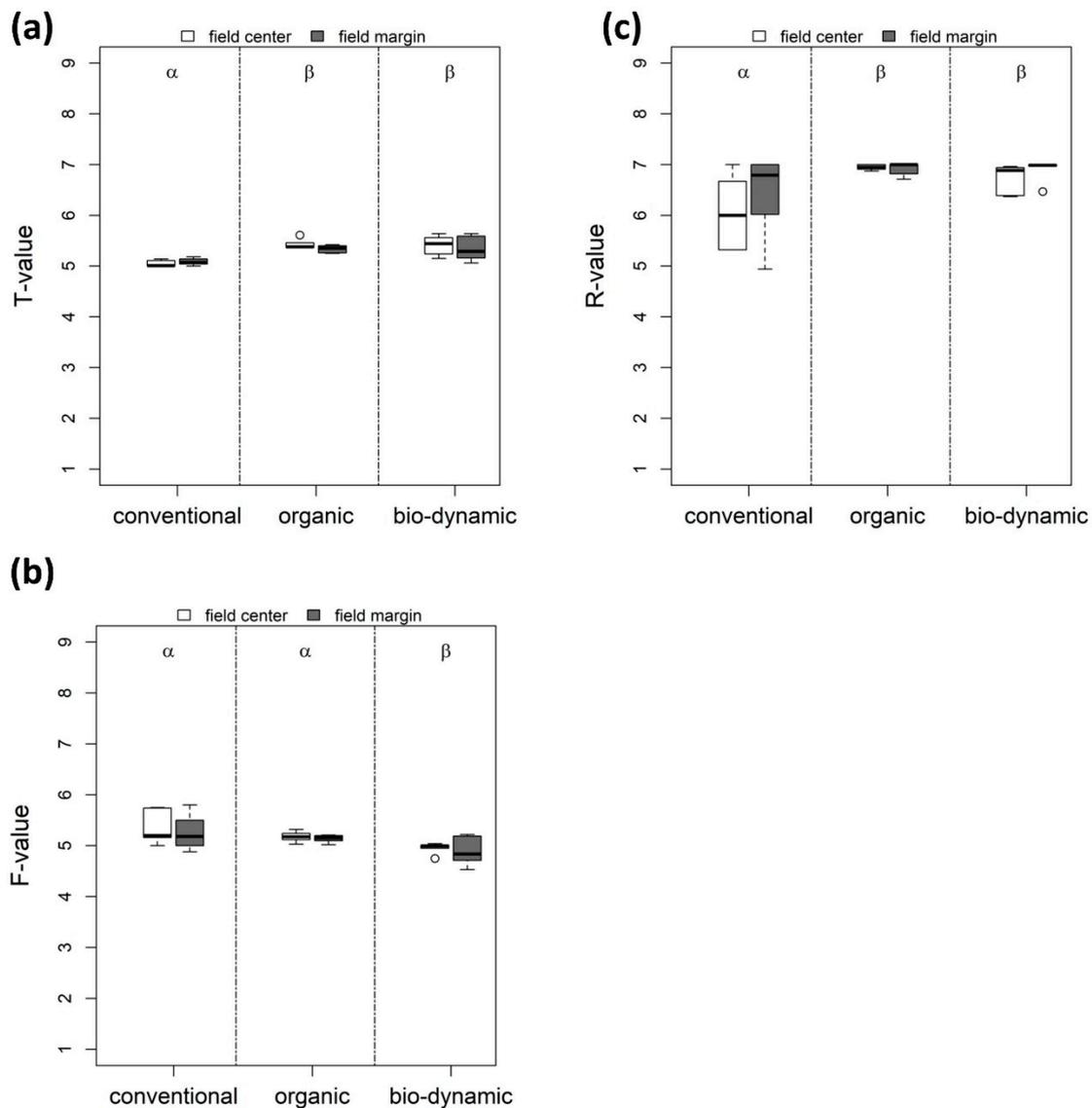
(Figure 6). Ellenberg T-values ( $r^2 = 0.41$ ) were positively correlated, and L-values ( $r^2 = 0.35$ ) and F-values ( $r^2 = 0.28$ ) were negatively correlated with Axis 1. Ellenberg L-values were also negatively correlated with Axis 2 ( $r^2 = 0.50$ ).



**Figure 6.** Non-metric multidimensional scaling (NMDS, Bray–Curtis dissimilarity, two dimensions, 500 iterations, final stress = 13.5) of the agrestal seed bank data. Black signatures represent bio-dynamic land use, grey signatures represent organic land use, and white signatures represent conventional land use, circles represent the field center and triangles represent the field margin. Arrows indicate the correlation of Ellenberg indicator values with the NMDS axes ( $r^2 \geq 0.25$ ), arrow length denotes the strength of the respective correlation. Only the correlation of the Ellenberg T-values (temperature), L-values (light availability), and F-values (moisture availability) with the NMDS axes were higher than the threshold value  $r^2 \geq 0.25$ .

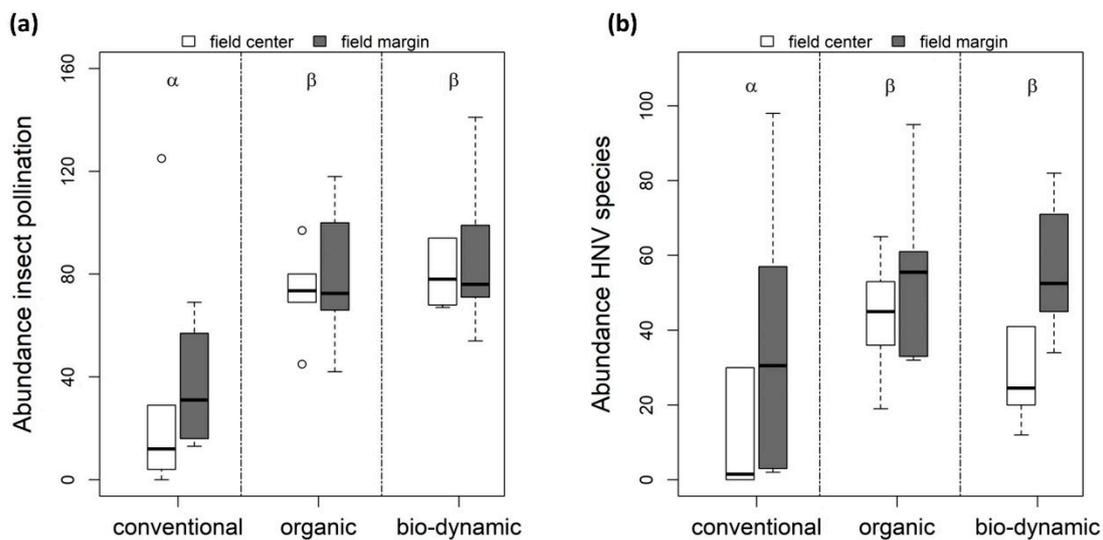
There was significant variation in the seed bank with regard to Ellenberg T-values ( $p < 0.001$ ), F-values ( $p = 0.007$ ), and R-values ( $p = 0.011$ ) with land-use type but not with position or the interaction of position and land-use type (Table 2). T-values (Figure 7a) were significantly higher in BDF ( $5.38 \pm 0.06$ ) and OF ( $5.38 \pm 0.03$ ) compared to CF ( $5.07 \pm 0.02$ ), while F-values (Figure 7b) were significantly lower in BDF ( $4.92 \pm 0.06$ ) and OF ( $5.16 \pm 0.03$ ) compared to CF ( $5.31 \pm 0.10$ ). R-values (Figure 7c) were significantly higher in OF

( $6.93 \pm 0.03$ ) and BDF ( $6.82 \pm 0.07$ ) compared to CF ( $5.78 \pm 0.62$ ). No effects were found for Ellenberg L-values and N-values (data not shown).



**Figure 7.** Ellenberg indicator values for temperature (T-value, **a**), moisture availability (F-value, **b**), and soil reaction (R-value, **c**) in the seed bank of the agrestal flora. Only those Ellenberg indicator values are presented where significant effects of land use type were detected. Boxes indicate the range from the first quartile to the third quartile, whiskers cover 1.5 times the interquartile range (IQR) beyond the box and circles indicate outliers  $>1.5 \times$  IQR beyond the box. Greek letters ( $\alpha$ ,  $\beta$ ) indicate significant differences ( $p \leq 0.05$ ) between the land use types pooled across position. In these cases, no significant interactions of land use type and position or effect of position was detected. Significant differences between the groups are based on ANOVA and post-hoc Tukey contrast tests.

The abundance of plants pollinated by insects in the seed bank (Figure 8a) varied with land-use type ( $p = 0.006$ ; Table 2). It was more than twice as high in OF ( $75.75 \pm 6.23$ ) and BDF ( $89.50 \pm 9.90$ ) compared to CF ( $33.25 \pm 10.38$ ). The abundance of individuals of HNV species in the seed bank (Table 2; Figure 8b) varied with land use type ( $p = 0.034$ ) and position ( $p = 0.020$ ). However, the impact of position was not confirmed by post-hoc Tukey contrast tests. The abundance of individuals of HNV species was significantly lower in CF ( $31.42 \pm 12.16$ ) compared to BDF ( $48.52 \pm 9.01$ ) and CF ( $49.58 \pm 5.70$ ).



**Figure 8.** Abundance of insect pollinated agrestal plants (a) and abundance of HNV species (b) in the seed bank of the agrestal flora. Boxes indicate the range from the first quartile to the third quartile, whiskers cover 1.5 times the interquartile range (IQR) beyond the box and circles indicate outliers  $>1.5 \times$  IQR beyond the box. Greek letters ( $\alpha$ ,  $\beta$ ) indicate significant differences ( $p \leq 0.05$ ) between the land use types pooled across position. In these cases, no significant interactions of land use type and position were detected. Significant differences between the groups are based on ANOVA and post-hoc Tukey contrast tests.

#### 4. Discussion

In the present study, we showed at a small spatial scale that both organic and bio-dynamic fields harbored two- to three-times more agrestal plant species above ground and in the seed bank compared to conventionally managed fields. In addition, organically managed fields harbored five times more and bio-dynamically managed fields 11 times more arable weed individuals above ground than conventionally managed fields. Plants in organically or bio-dynamically managed fields differed in their composition and traits from those occurring in conventionally managed fields in that they showed a preference for higher temperatures and were dominated by insect-pollinated species.

Preservation of biodiversity in agricultural landscapes is, besides food production, a major aim of organic farming. Yet, the intensity of management may differ among different approaches even within the organic sector. Consequently, the diversity of plant species may also differ, a phenomenon that was also apparent in our study. Nonetheless, the abundance and richness of species in the above-ground vegetation in arable fields were very variable but on average systematically higher in organic fields cultivated in accordance with agri-environmental contracts than in conventionally managed fields. This accords with other studies that also found elevated plant diversity in fields managed according to agri-environmental schemes [56]. According to the Ellenberg indicator values of both the seed bank and the aboveground vegetation, differences in site conditions between the three management systems were small. Concurrent to studies assessing long-term changes in agrestal plant communities in the course of agricultural intensification [16,57], however, we found on average lower T-values in the above-ground vegetation and seed bank and higher N-values in the above-ground vegetation of intensively conventionally managed fields compared to the organic fields.

Light is arguably among the most important resources affecting the flora in agricultural systems [56,58] and plants in favor of high levels of light are therefore characteristic of these open land systems. As such, high and undifferentiated L-values for the aboveground vegetation among the three management types in our study followed this expectation. The obvious structural difference among the three management types, however, manifested itself in higher T-values in the vegetation of organically managed fields as compared to conventionally managed fields. Higher T-values in organically managed fields could

indicate a more open structure of the crop vegetation in these fields. Besides the effects of stand structure on the agrestal vegetation, such an open structure allows flower-visiting insects to enter the crop. Consequently, insect-pollinated plants, which are in steep decline in modern agricultural landscapes [59–61], were more abundant in both types of more extensively managed fields than in conventional ones. The density of crops, in turn, is not only a result of sowing densities but also of the availability of nutrients in the system.

Agri-environmental schemes promoting the organic or bio-dynamical management of arable fields prohibit the application of synthetic fertilizers. Interestingly, though, the N-values differed only between conventional and organic fields but not between conventional and bio-dynamic fields. This may result from the biodynamic farm striving towards closed nutrient loops and, in this course, an intensive use of Fabaceae-species as undersown crops and intertillage for taking advantage of these groups' ability to fix atmospheric nitrogen [62]. In contrast, the organic farm does not follow this approach and the crop's condition actually reflects a lower nutrient availability and may indicate a suboptimal management of the organic fields. That biodynamic fields harbored not only the highest number of agrestal plant species in general but also of red list species, i.e., typically plant species with narrower tolerances, indicates that not higher nutrient levels in general but rather the level of competition with the crop constitutes the main obstacle for their conservation in the absence of herbicides. In organic farming, the application of organic fertilizer with a delayed nutrient release as compared to chemical fertilizers and lower sowing densities could lead to reduced competition levels.

Not only did the differently managed fields vary in the abundances and richness of agrestal plants above ground, but they also differed below ground. Interestingly, while the species composition of the seed bank differed between the two organic-management systems, the composition of the aboveground vegetation only differed between organic/bio-dynamic and conventional management. This hints at a general phenomenon in the interplay between the aboveground vegetation and the seed bank. The aboveground vegetation responds much faster to changes in management or site conditions than the seed bank. The latter typically shows a time lag of several years until it reflects the aboveground composition of the vegetation [63,64]. Although Albrecht [65] identified a threefold increase in seed density after the transformation from conventional to organic farming, this development was neither unidirectional nor linear, i.e., these authors found a steep increase during the first years but a slight decrease in the following years.

Since seed banks buffer against stochasticity of the environment and management, the development of a seed bank corresponding to the above ground vegetation is an important aim in ecological restoration, especially in dynamic habitats such as agricultural ones [63]. A study by Simmering et al. [66] taking place in a very intensively managed agricultural landscape in Hesse, however, showed that seed banks were not only very strongly reduced in species and seed number but often even completely depleted. Thus, in line with many studies from a wide variety of habitats [67,68], our results point at a low potential of seed banks to restore typical species-rich plant communities after sites experienced a phase of intensive agricultural management. If at all, such a restoration may naturally start from the field margin, where agrestal plant species richness is usually higher than in the center.

Our results showed a decline of species numbers from the field margin to the field center, similar to previous studies [11,57,66]. Yet, this diversity gradient was only apparent for the aboveground vegetation and did not translate into an equipollent pattern in the seed bank [66,69]. The higher number and abundance of agrestal species regularly observed at the margin as compared to the field center are caused by the—mostly short term—establishment of species from neighboring non-arable vegetation or regular seed input from adjacent unploughed strips which often still harbor some agrestal species [11,57]. Although some species may persist in margins of conventionally managed fields [57], due to frequent herbicide use, these margins may function as sinks rather than sources for agrestal plants. While a differentiation between field margin and center does also occur in the agrestal vegetation of both bio-dynamic and organic fields, only in the latter

these differences are as pronounced as under conventional management. The significant differences in Shannon diversity and evenness between margins and centers of the conventional fields further emphasized the high negative impact of this farming type on agrestal plant species. In contrast, for the bio-dynamic farm, where higher diversity is not only a side effect but a declared management aim, the quantified diversity indices not only indicated a higher diversity but also homogeneity of the agrestal plant communities across the management units.

## 5. Conclusions

In the current study, we showed that the organic farms had presumably neutral or slightly positive effects on the vegetation as well as the seed bank, while conventional farming had negative effects on both vegetation compartments. While this result was not surprising, it highlights the potential to save species even in an intensively managed landscape. Under less intensive agricultural management schemes, which might get applied in future through changes in the agricultural regulations within the European Union, these remnant populations can serve as sources for the re-establishment of more species-rich agrestal plant communities at the landscape level. Since coherent agrestal plant communities constitute an important component in food-webs of agricultural landscapes [17,70], such an integrated approach might translate into a reversal of population decrease in arthropod, bird, and mammal species.

**Author Contributions:** Conceptualization, T.W.D., T.D., M.M. and H.K.; methodology, T.W.D., M.M. and H.K.; software, E.G., M.M. and H.K.; validation, T.W.D. and E.G.; formal analysis, E.G. and T.W.D.; investigation, M.M. and H.K.; resources, T.D. and T.W.D.; data curation, E.G., H.K. and M.M.; writing—original draft preparation, E.G.; writing—review and editing, E.G., T.W.D. and T.D.; visualization, E.G.; supervision, T.W.D. and T.D.; project administration, T.D. and T.W.D.; funding acquisition, T.D. All authors have read and agreed to the published version of the manuscript.

**Funding:** This research was conducted in the long-term project “Organic Farming on Farm Ritzerau”, which is funded by Prof.h.c. Dr.h.c. Günther Fielmann.

**Institutional Review Board Statement:** Not applicable.

**Informed Consent Statement:** Not applicable.

**Data Availability Statement:** The data presented in this study are openly available in OSF at DOI 10.17605/OSF.IO/YC4XG.

**Acknowledgments:** The author thank the owners of the farms Koop, Ritzerau and Lämmerhof for allowing data collection on their fields. We are also grateful to David Bennett for language editing. This study was conducted in the long-term project “Organic Farming on Farm Ritzerau”. We gratefully acknowledge this support.

**Conflicts of Interest:** The authors declare no conflict of interest.

## References

1. Brondizio, E.S.; Settele, J.; Díaz, S.; Ngo, H.T. *Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*; IPBES Secretariat: Bonn, Germany, 2019.
2. Storkey, J.; Meyer, S.; Still, K.S.; Leuschner, C. The impact of agricultural intensification and land-use change on the European arable flora. *Proc. R. Soc. B Boil. Sci.* **2011**, *279*, 1421–1429. [[CrossRef](#)] [[PubMed](#)]
3. Friebe, B.; Prolingheuer, U.; Wildung, M.; Meyerhoff, E. Aufwertung der Agrarlandschaft durch ökologischen Landbau—Eine Möglichkeit der produktionsintegrierten Kompensation? *Naturschutz und Landschaftsplanung* **2012**, *44*, 108–114.
4. Robinson, R.A.; Sutherland, W.J. Post-war changes in arable farming and biodiversity in Great Britain. *J. Appl. Ecol.* **2002**, *39*, 157–176. [[CrossRef](#)]
5. Köhler, E. Moderne Landwirtschaft und Herbizide/Modern Agriculture and Herbicides. *Zeitschrift für Naturforschung C* **1979**, *34*, 895–899. [[CrossRef](#)]
6. Chamberlain, D.E.; Fuller, R.J.; Bunce, R.G.H.; Duckworth, J.C.; Shrubbs, M. Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. *J. Appl. Ecol.* **2000**, *37*, 771–788. [[CrossRef](#)]
7. FAO. FAOSTAT: Food and Agriculture Data. 2017. Available online: <http://www.fao.org/faostat/en/#data/RP> (accessed on 28 October 2017).

8. Lu, C.C.; Tian, H. Global nitrogen and phosphorus fertilizer use for agriculture production in the past half century: Shifted hot spots and nutrient imbalance. *Earth Syst. Sci. Data* **2017**, *9*, 181. [[CrossRef](#)]
9. Marshall, J.; Brown, V.; Boatman, N.; Lutman, P.; Squire, G. *The Impact of Herbicides on Weed Abundance and Biodiversity*; DEFRA review PN0940; Rothamsted Research: Hertfordshire, UK, 2001; p. 147.
10. Metzger, D.; Hofbauer, N.; Ludwig, G.; Matzke-hajek, G. Rote Liste gefährdeter Tiere, Pflanzen und Pilze Deutschlands. Band 7: Pflanzen. *Naturschutz und Biologische Vielfalt* **2018**, *70*, 1–784.
11. Meyer, S.; Bergmeier, E.; Becker, T.; Wesche, K.; Krause, B.; Leuschner, C. Detecting long-term losses at the plant community level—Arable fields in Germany revisited. *Appl. Veg. Sci.* **2015**, *18*, 432–442. [[CrossRef](#)]
12. Mierwald, U.; Romahn, K.S. Die Farn- und Blütenpflanzen Schleswig-Holsteins: Rote Liste. *Schriftenreihe LANU SH* **2006**, *18*, 1–1212006.
13. Ellenberg, H.; Leuschner, C. Vegetation Mitteleuropas mit den Alpen. In *Ökologischer, Dynamischer und Historischer Sicht*; UTB: Stuttgart, Germany, 2010; p. 1357.
14. Meyer, S.; Hilbig, W.; Steffen, K.; Schuch, S. Ackerwildkrautschutz—Eine Bibliographie. Bundesamt für Naturschutz. *BfN-Skripten* **2013**, *351*, 1–224.
15. Gerhards, R.; Dieterich, M.; Schumacher, M. Rückgang von Ackerunkräutern in Baden-Württemberg—ein Vergleich von vegetationskundlichen Erhebungen in den Jahren 1948/49, 1975–1978 und 2011 im Raum Mehrstetten—Empfehlungen für Landwirtschaft und Naturschutz. *Gesunde Pflanzen* **2013**, *65*, 151–160. [[CrossRef](#)]
16. Meyer, S.; Wesche, K.; Krause, B.; Leuschner, C. Dramatic losses of specialist arable plants in Central Germany since the 1950s/60s—A cross-regional analysis. *Divers. Distrib.* **2013**, *19*, 1175–1187. [[CrossRef](#)]
17. Altieri, M.A. The ecological role of biodiversity in agroecosystems. *Agric. Ecosyst. Environ.* **1999**, *74*, 19–31. [[CrossRef](#)]
18. Storkey, J. A functional group approach to the management of UK arable weeds to support biological diversity. *Weed Res.* **2006**, *46*, 513–522. [[CrossRef](#)]
19. Tschardtke, T.; Klein, A.M.; Krüess, A.; Steffan-dewenter, I.; Thies, C. Landscape perspectives on agricultural intensification and biodiversity—Ecosystem service management. *Ecol. Lett.* **2005**, *8*, 857–874. [[CrossRef](#)]
20. Sukopp, H.; Schneide, C.; Sukopp, U. Biologisch-ökologische Grundlagen für den Schutz gefährdeter Segetalpflanzen. *Naturschutz und Landschaftspflege in Brandenburg Sonderheft* **1994**, *1*, 14–16.
21. Meyer, S.; Wesche, K.; Leuschner, C.; Van Elsen, T.; Metzner, J. Schutzbemühungen für die Segetalflora in Deutschland—Das Projekt “100 Äcker für die Vielfalt”. *Treffpunkt Biologische Vielfalt* **2010**, *9*, 58–64.
22. Schumacher, W. Schutz und Erhaltung gefährdeter Ackerwildkräuter durch Integration von landwirtschaftlicher Nutzung und Naturschutz. *Natur und Landschaft* **1980**, *55*, 447–453.
23. Sommer, M. Schutz der Ackerwildkrautflora in Bayern—Geschichte und Empfehlungen zum nachhaltigen Schutz auf Grundlage aktueller Erfassungen. *ANLiegen Nat.* **2014**, *36*, 19–28.
24. Andreassen, C.; Andresen, L.C. Managing farmland flora to promote biodiversity in Europe. *Plant Sci. Rev.* **2012**, 199–205. [[CrossRef](#)]
25. Lang, M.; Albrecht, H.; Kollmann, J.; Van Elsen, T.; Gärtner, A.; Hotze, C.; Wiesinger, K. *Naturschutzleistungen des Ökologischen Landbaus: Wiederansiedlung Seltener und Gefährdeter Ackerwildpflanzen Naturräumlicher Herkünfte auf Ökobetrieben*; University of Kassel: Kassel, Germany, 2016; p. 160.
26. Hole, D.G.; Perkins, A.J.; Wilson, J.D.; Alexander, I.H.; Grice, P.V.; Evans, A.D. Does organic farming benefit biodiversity? *Biol. Conserv.* **2005**, *122*, 113–130. [[CrossRef](#)]
27. Kay, S.; Gregory, S. *Rare Arable Flora Survey 1999*; Northmoor Trust: Abingdon Oxfordshire, UK, 1999.
28. Rydberg, N.T.; Milberg, P. A Survey of Weeds in Organic Farming in Sweden. *Biol. Agric. Hortic.* **2000**, *18*, 175–185. [[CrossRef](#)]
29. Frieben, B.; Köpke, U. Effects of farming systems on biodiversity. In *Organic Farming in Land Use Systems. Proceedings of the First ENOF (The European Network for Scientific Research Co-Ordination in Organic Farming) Workshop*; ENOF: Bonn, Germany, 1995; pp. 11–21.
30. Gabriel, D.; Roschewitz, I.; Tschardtke, T.; Thies, C. Beta diversity at different spatial scales: Plant communities in organic and conventional agriculture. *Ecol. Appl.* **2006**, *16*, 2011–2021. [[CrossRef](#)]
31. Gabriel, D.; Sait, S.M.; Kunin, W.E.; Benton, T.G. Food production vs. biodiversity: Comparing organic and conventional agriculture. *J. Appl. Ecol.* **2013**, *50*, 355–364. [[CrossRef](#)]
32. Baskin, C.C.; Baskin, J.M. *Seeds: Ecology, Biogeography, and Evolution of Dormancy and Germination*; Academic Press: Cambridge, MA, USA, 2014; p. 1586.
33. Hawes, C.; Squire, G.; Hallett, P.; Watson, C.; Young, M. Arable plant communities as indicators of farming practice. *Agric. Ecosyst. Environ.* **2010**, *138*, 17–26. [[CrossRef](#)]
34. Rotchés-Ribalta, R.; Armengot, L.; Mäder, P.; Mayer, J.; Sans, F.X. Long-Term Management Affects the Community Composition of Arable Soil Seedbanks. *Weed Sci.* **2016**, *65*, 73–82. [[CrossRef](#)]
35. Schröter, L.; Irmeler, U. Lauf- und Kurzflügelkäfer der Ackerflächen. Faunistisch-Ökologische Mitteilungen. *Supplement* **2008**, *35*, 145–158.
36. Deutscher Wetterdienst. Climate Data Center CDC. 2018. Available online: <https://www.dwd.de> (accessed on 16 December 2018).
37. Reiss, S.; Bork, H.R.; Hoernes, U.; Rinker, A.; Mitusov, A. Die Verbreitung der Böden auf den Ackerflächen von Hof Ritzerau. *Faunistisch-Ökologische Mitteilungen* **2008**, *35*, 59–73.

38. Burbaum, B.; Filipinski, M. *Die Böden Schleswig-Holsteins: Entstehung, Verbreitung, Nutzung, Eigenschaften und Gefährdung*; Landesamt für Landwirtschaft, Umwelt und Ländliche Räume Schleswig-Holstein: Flintbek, Germany, 2012; p. 108.
39. Richter, F.; Fleige, H.; Horn, R. Verteilung von Bodentypen, Subtypen und Varietäten in einer Jungmoränenlandschaft Schleswig-Holsteins. *Mitteilungen Deutschen Bodenkundlichen Gesellschaft* **2004**, *103*, 153–154.
40. Neumann, H.; Loges, R.; Roweck, H.; Taube, F. Naturschutz und ökologischer Landbau—Rahmenbedingungen, Stand der Forschung und Konzeption des Projektes, Hof Ritzerau. *Faunistisch-Ökologische Mitteilungen* **2008**, *35*, 7–19.
41. Brüggemann, C.; Hack, D.; Thode, U. Der Lämmerhof. Historie. 2017. Available online: <https://www.laemmerhof.de/> (accessed on 11 November 2017).
42. Van Elsen, T. Ackerwildkraut-Bestände im Randbereich und im Bestandesinnern unterschiedlich bewirtschafteter Halm- und Hackfruchtäcker. *Veröff. Bundesanst. Agrarbiol.* **1990**, *20*, 21–39.
43. Thompson, K.; Bakker, J.P.; Bekker, R.M. *The soil seed Banks of North West Europe: Methodology, Density and Longevity*; Cambridge University Press: Cambridge, UK, 1997; p. 276.
44. Hennekens, S.M.; Schaminée, J.H.J. TURBOVEG, a comprehensive data base management system for vegetation data. *J. Veg. Sci.* **2001**, *12*, 589–591. [[CrossRef](#)]
45. Ellenberg, H.; Weber, H.E.; Dull, R.; Wirth, V.; Werner, W.; Paulißen, D. Zeigerwerte von Pflanzen in Mitteleuropa. *Scr. Geobot.* **1991**, *18*, 1–248.
46. Diekmann, M. Species indicator values as an important tool in applied plant ecology—a review. *Basic Appl. Ecol.* **2003**, *4*, 493–506. [[CrossRef](#)]
47. Silvertown, J.; Dodd, M.; Gowing, D.; Lawson, C.; McConway, K. Phylogeny and the Hierarchical organization of plant diversity. *Ecology* **2006**, *87*, S39–S49. [[CrossRef](#)]
48. McCune, B.; Mefford, M.J. *PC-ORD. Multivariate Analysis of Ecological Data*; Version 7.03; MjM Software Design: Gleneden Beach, OR, USA, 2016; p. 273.
49. Kenkel, N.C.; Burchill, C.E. Rigid rotation of nonmetric multidimensional scaling axes to environmental congruence. *Abstr. Bot.* **1990**, 109–119.
50. Klotz, S.; Kühn, I.; Durka, W. BIOFLOR—a database on biological and ecological traits of vascular plants in Germany. *Schriftenreihe für Vegetationskunde* **2002**, *38*, 1–334.
51. Verbeke, G.; Molenberghs, G. Linear mixed models for longitudinal data. *Linear Mixed Models in Practice. Lect. Notes Stat.* **2009**, *126*, 63–153.
52. R Core Team. *R: A Language and Environment for Statistical Computing (Version 3.4.2)*; R Foundation for Statistical Computing: Vienna, Austria, 2016.
53. LLUR (Landesamt für Landwirtschaft, Umwelt und ländliche Räume des Landes Schleswig-Holstein). *Kartieranleitung und Biotoptypenschlüssel für die Biotopkartierung Schleswig-Holstein mit Hinweisen zu den gesetzlich geschützten Biotopen sowie den Lebensraumtypen gemäß Anhang I der FFH-Richtlinie-Kartieranleitung, Biotopenschlüssel und Standardliste*, 5th ed.; LLUR: Flintbek, Germany, 2019; p. 386.
54. Pinheiro, J.; Bates, D.; DebRoy, S.; Sarkar, D.R. *nlme: Linear and Nonlinear Mixed Effects Models*; R Package Version 3.1-141; 2019. Available online: <http://cran.r-project.org/web/packages/nlme/index.html> (accessed on 21 February 2021).
55. Schaarschmidt, F.; Vaas, L. Analysis of Trials with Complex Treatment Structure Using Multiple Contrast Tests. *HortScience* **2009**, *44*, 188–195. [[CrossRef](#)]
56. Seifert, C.; Leuschner, C.; Meyer, S.; Culmsee, H. Inter-relationships between crop type, management intensity and light transmissivity in annual crop systems and their effect on farmland plant diversity. *Agric. Ecosyst. Environ.* **2014**, *195*, 173–182. [[CrossRef](#)]
57. Fried, G.; Petit, S.; Dessaint, F.; Reboud, X. Arable weed decline in Northern France: Crop edges as refugia for weed conservation? *Biol. Conserv.* **2009**, *142*, 238–243. [[CrossRef](#)]
58. Hautier, Y.; Niklaus, P.A.; Hector, A. Competition for light causes plant biodiversity loss after eutrophication. *Science* **2009**, *324*, 636–638. [[CrossRef](#)]
59. Biesmeijer, J.C.; Roberts, S.P.M.; Reemer, M.; Ohlemüller, R.; Edwards, M.; Peeters, T.; Schaffers, A.P.; Potts, S.G.; Kleukers, R.; Thomas, C.D.; et al. Parallel Declines in Pollinators and Insect-Pollinated Plants in Britain and the Netherlands. *Science* **2006**, *313*, 351–354. [[CrossRef](#)]
60. Hallmann, C.A.; Sorg, M.; Jongejans, E.; Siepel, H.; Hofland, N.; Schwan, H.; Stenmans, W.; Müller, A.; Sumser, H.; Hörren, T.; et al. More than 75 percent decline over 27 years in total flying insect biomass in protected areas. *PLoS ONE* **2017**, *12*, e0185809. [[CrossRef](#)] [[PubMed](#)]
61. Wilson, J.D.; Morris, A.J.; Arroyo, B.E.; Clark, S.C.; Bradbury, R.B. A review of the abundance and diversity of invertebrate and plant foods of granivorous birds in northern Europe in relation to agricultural change. *Agric. Ecosyst. Environ.* **1999**, *75*, 13–30. [[CrossRef](#)]
62. Stewart, W.D.P. *Nitrogen Fixation in Plants*; Athlone Press: London, UK, 1966; p. 168.
63. Burmeier, S.; Eckstein, R.L.; Otte, A.; Donath, T.W. Spatially-restricted plant material application creates colonization initials for flood-meadow species. *Biol. Conserv.* **2011**, *144*, 212–219. [[CrossRef](#)]
64. Schmiede, R.; Donath, T.W.; Otte, A. Seed bank development after the restoration of alluvial grassland via transfer of seed-containing plant material. *Biol. Conserv.* **2009**, *142*, 404–413. [[CrossRef](#)]

65. Albrecht, H. Development of arable weed seedbanks during the 6 years after the change from conventional to organic farming. *Weed Res.* **2005**, *45*, 339–350. [[CrossRef](#)]
66. Simmering, D.; Waldhardt, R.; Otte, A. Erfassung und Analyse der Pflanzenartenvielfalt in der “Normallandschaft”—Ein Beispiel aus Mittelhessen. *Berichte der Reinhard-Tüxen-Gesellschaft* **2013**, *25*, 73–94.
67. Bakker, J.P.; Poschlod, P.; Strykstra, R.J.; Bekker, R.M.; Thompson, K. Seed banks and seed dispersal: Important topics in restoration ecology. *Acta Bot. Neerlandica* **1996**, *45*, 461–490. [[CrossRef](#)]
68. Bossuyt, B.; Honnay, O. Can the seed bank be used for ecological restoration? An overview of seed bank characteristics in European communities. *J. Veg. Sci.* **2008**, *19*, 875–884. [[CrossRef](#)]
69. Wilson, P.J.; Aebischer, N.J. The Distribution of Dicotyledonous Arable Weeds in Relation to Distance from the Field Edge. *J. Appl. Ecol.* **1995**, *32*, 295. [[CrossRef](#)]
70. Marshall, E.J.P.; Brown, V.K.; Boatman, N.D.; Lutman, P.J.W.; Squire, G.R.; Ward, L.K. The role of weeds in supporting biological diversity within crop fields. *Weed Res.* **2003**, *43*, 77–89. [[CrossRef](#)]