



The effect of land use on taxonomical and functional diversity of lichens in an agricultural landscape

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ABSTRACT

The objective of this work was to determine the effect of land use on lichen richness and their functional groups in agricultural land. A significant overall effect of land use on the number of species was found. In general, there was a marked dissimilarity with regard to species richness of forest versus non-forest sites. An analysis of some ecological requirements of lichens showed the most apparent differences concerning tolerance to nutrients and acidity value of particular species. Richness of species capable of producing soredia, isidia and reproducing via fragmentation was higher in coniferous forest compared to non-forest habitats (including wooded patches). Functional traits were a more sensitive and more informative index of lichen response to land use intensity compared to species richness.

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1. Introduction

Lichenized fungi exist in many ecosystems, influence biodiversity, and are thus important for its conservation (e.g., Ellis and Coppins, 2006; McCune and Geiser, 2009). Lichens are commonly considered as bioindicators due to their sensitivity to anthropogenic pressure, even when the intensity of anthropogenic disturbances is low (Pinho et al., 2012). Even though lichens have a wider distribution range in comparison with, for example, plants, the lichen biota differs between biogeographic regions of Europe (Bergamini et al., 2005). Lichens exhibit sensitivity to different kinds of habitat disturbances, both of natural and human origin, e.g. land use. Landscape structure (Motiejūnaitė and Fałtynowicz, 2005), habitat parameters, e.g. size and degree of patch isolation, soil pH and depth, the cover of vascular plants, microtopography (Löbel et al., 2006), substrata and habitat diversity (Löbel et al., 2006; Wolseley et al., 2006) and the type of management regime (Ardelean et al., 2015; Boch et al., 2016) can all affect diversity and

richness of lichen species and their separate groups.

In general, land use changes usually cause lichen diversity loss (e.g., Stofer et al., 2006; Boch et al., 2016). Many studies have revealed a higher number of lichen species in less intensively managed habitats and their decline under intensive land use (e.g., Ruoss, 1999; Wolseley et al., 2006; Giordani et al., 2010; Boch et al., 2016). As early as the 1980s, Gilbert (1980) pointed to land use as the chief determinant of terricolous lichen abundance in the limestone plateau of Derbyshire, England. In Switzerland, the number of lichen species was found to decrease from mountains to agricultural areas (Ruoss, 1999). The response of lichen diversity to, among others, land use was intensively studied in W Sardinia, Italy (Giordani et al., 2010). The strongest differences were detected for the forested sites vs. managed agroforestry lands; a high within-group diversity (esp. for epilithic species), and a high homogeneity of species composition were observed in managed agroforestry lands (Giordani et al., 2010). An investigation in six different biogeographic regions in Europe showed a significant overall effect of land use on lichen species richness and significant differences in means were detected between natural forests and farmland (Bergamini et al., 2005). Even though the studied areas ranged from northern Europe to the Mediterranean, differences in lichen composition caused by land use were similar (Bergamini et al.,

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2005).

Recently, much attention has been focused on the diversity of functional groups, which helps to understand mechanisms leading to biodiversity impoverishment better than any analyses based on species richness alone. Functional groups consist of species exerting a comparable effect upon a particular process or responding in a similar manner to changes in their external constraints (Lévêque and Mounolou, 2003 in: Giordani and Brunialti, 2015). According to Pinho et al. (2008), the advantages of use of functional groups to detect environmental changes result from the fact that this approach provides a balance between the very robust indicators including all species (e.g. total number of species) and a more detailed analysis, based on individual species. Species traits could be indicative of lichen community adaptation to environmental conditions (Diaz and Cabido, 2001 in: Giordani et al., 2012), therefore providing relevant ecological information (Giordani et al., 2012). Trait patterns along environmental gradients have been interpreted as indicators of mechanisms behind lichen adaptation to the environment (Nelson et al., 2015). The recognition of how management affects environment based on lichen diversity can be a cost-effective and important source of information for environmental planning and biodiversity policies (Bergamini et al., 2005; Pinho et al., 2012). Such analyses also enable a better understanding of environmental conditions, their changes and the importance of agriculture landscape for lichen conservation (e.g., Boch et al., 2016).

Some studies on lichens in the agricultural landscape in Europe that involved a classification of species according to their tolerance to eutrophication gave similar results. Lichen communities in open agricultural landscape are mainly characterized by a high contribution of nitrophiles accompanied by low richness and abundance or even absence of oligotrophic species, compared to sites with less intensive land use (Wolseley et al., 2006; Pinho et al., 2008; Giordani et al., 2010; Munzi et al., 2014). Munzi et al. (2014) found positive relationships between nitrophilous, xerophilous and photophilous groups and agricultural areas, while sensitive functional groups (oligotrophic, hygrophilous and mesophilous) were associated with more forested sites. Land use also seems to affect reproductive traits and morphological forms. Growth form, reproductive strategy and photobiont type were among tested functional traits and their response to environmental factors related to climate, human disturbances and stand structure was investigated (Giordani et al., 2012). Studies on species richness of lichen functional groups across land use gradients in Europe showed that fertile species preferred open and intensively managed landscapes, while sterile species, as well as the substratum generalists, were found mainly in forests (Stofer et al., 2006). Other findings include increase of the importance of species with the principal photobiont *Trebouxia* s.l. and decrease of species with *Trentepohlia* with intensification of land use (Stofer et al., 2006). Despite the advantages of using functional groups, they were rarely used and functional response of lichens to land-use intensification has been poorly studied (Stofer et al., 2006; Pinho et al., 2008; Giordani et al., 2012; Ardelean et al., 2015; Matos et al., 2015; Spickerman, 2015).

To detect human impact in agricultural landscapes, it is necessary to conduct monitoring of organisms with high indicative value. Effect of land use on lichens in rural areas, also considering functional traits, remains still poorly understood in central Europe. Some authors (Boch et al., 2016) already pointed out a need for regional replication of studies that would facilitate our understanding of the degree to which land use decreases biodiversity, with regard to variation in abiotic conditions between the regions, to find general patterns ruling the process.

The objective of this work was to determine the effect of land

use on lichen richness and their functional groups in agricultural land. Based on the literature data presented above, we expected some differences in lichen species diversity and a shift in the lichen functional groups across land use spectrum, from forest communities (deciduous forest, coniferous forest) to rural areas under diverse intensity of human activities (wooded patches, grasslands, turf, ruderal, segetal and mixed). We studied the differences and shifts with the use of a possible indicator value of lichen diversity variables (i.e. total species richness and richness of functional groups of species classified according to their light, temperature, moisture, continentality, acidity and nutrient requirements, morphological forms and reproductive strategy) for measurement and monitoring of land use effect on lichens.

2. Materials and methods

2.1. Study area

The study area is situated on the Równina Nowotomska plain, a part of Wielkopolska province (Kondracki, 2009) in western Poland. According to the geobotanical division of the country, this area belongs to the western part of Kraina Wielkopolsko-Kujawska range (Szafer, 1972). The altitude ranges from 60 to 100 m a.s.l. The climate is transitional, marked by interlacement of oceanic and continental impacts, with average yearly precipitation of ca 550 mm and average yearly temperature of ca 8 °C (Graf, 2001; Kaniecki and Kostecki, 2001; Wrzesiński, 2001; Kozacki et al., 2006a,b). Agricultural landscape covers most of the investigated area. The persistence of scattered housing combined with a dense, quite regular network of wooded patches that separate fragments of fields, meadows and pastures is a typical spatial layout in the central part of the study region and is the remnant of an Oleder ('Dutch') settlement. The process of 'Dutch' settlement in Poland took place between the 16th and the 19th century (Chodyta, 2006). The colonists, first the Dutch and later principally Germans and Poles, created independent communities and their settlements were organized under a particular type of law. Conifer forests (mainly *Leucobryo-Pinetum*) growing on the dominant sandy soils, make up most of the forest communities.

2.2. Sampling design

Ninety six sample plots (20 × 50 m, N-S oriented) were located within a 45 × 12 km rectangle of the study area, considering physical-geographic conditions of the region (Kondracki, 2009), with the application of systematic sampling design (for example see Wohlgemuth et al., 2008). The distance between most plots amounted to ca 2.5 km. Some plots did not match the regular distribution grid for practical reasons (difficult access, presence of a highway). In such cases the sample plots were located in alternative, accessible sites with the same land-use category as the primary location. The sample plots represented eight land-use categories, which were distinguished based on vegetational features. They included a variety of habitats both from forest communities (deciduous forest, coniferous forest), and rural areas under diverse impact of human activities (wooded patches, grasslands, turf, ruderal, segetal and mixed). All lichenized fungi, except parasites, were considered. Species were recorded taking into account the type of substrate (especially wood, bark of living tree, and soil). Stumps, snags, fallen pieces of barks and branches were included in the 'wood' category. When species could not be determined in the field, specimens were collected for further identifications in the laboratory (with the use of microscope and thin-layer chromatography). Species nomenclature follows Diederich et al. (2016). We compiled data on ecological indicator

value for most of the recorded species according to Wirth (2010); groups of species were classified according to their light, temperature, moisture, continentality, acidity and nutrient requirements. Morphological forms and reproductive strategy were based on the data proposed by Nimis and Martellos (2008). However, some species were excluded from future analysis when this information was not available.

2.3. Statistical analysis

The total species richness was estimated with the use of four estimators: Chao 2, Jackknife 1, Jackknife 2 and Bootstrap. The One-way ANOVA and *post-hoc* Tukey test were applied for testing the effect of land-use on the total number of species (the variable with normal frequency distribution). The effect of land-use on species number in the given functional groups (the variables with non-normal frequency distribution) was analysed with the use of non-parametric Kruskal-Wallis ANOVA with *post-hoc* multiple comparison test. The sampling sites were grouped with the aid of cluster analysis (with Euclidean distance and Jaccard's similarity index applied). Total number of species per plot was normalized by logarithmic transformation. The analyses were performed with the use of Statistica 13.1 (Dell Inc. 2016) and PaST 3.11 (Hammer et al., 2001).

3. Results

3.1. Floristic data

The lichen biota in the study area consisted of 101 species (Table S1), corresponding to 80–93% of the total number of taxa estimated by Chao 2, Jackknife 1, Jackknife 2 and Bootstrap (Table 1). Some of them were included in the Polish Red List categories (Cieśliński et al., 2006), i.e. *Chaenotheca furfuracea* (NT), *Chaenotheca trichialis* (NT), *Cladonia* cfr. *bellidiflora* (EN), *Evernia prunastri* (NT), *Flavoparmelia caperata* (EN), *Hypogymnia tubulosa* (NT), *Lecanora persimilis* (DD), *Vulpicida pinastri* (NT). The study also revealed the presence of lichens protected by law, i.e. *Cladonia arbuscula*, *Cladonia portentosa*, *Cladonia rangiferina*, *F. caperata*, *Imshaugia aleurites*, *H. tubulosa*, *Melanelixia subaurifera*, and *V. pinastri* (Rozporządzenie, 2014).

3.2. Effect of land use on the number of species

We found a significant overall effect of land use (ANOVA: $F = 26.4$, $df = 7$ $p < 0.001$) on the number of species. Mean number of species was significantly lower in segetal habitats (Table 2, Fig. 1). Additionally, differences in means were significant between coniferous forest, wooded patches and grasslands (Table 2, Fig. 1).

3.3. The differences in species composition between habitats (cluster analysis)

Variability of the species composition between habitats is rather continuous (Fig. S1). A higher similarity of species composition was

Table 2

Mean number of species, and mean logarithmized (the natural logarithm) number of species in investigated land use. SD – standard deviation.

Habitat	N	Mean	SD	Logarithmized values	
				Mean	SD
deciduous	6	17.8	9.0	2.82	0.57
coniferous	29	19.7	7.7	2.94	0.48
wooded patches	19	9.4	4.5	2.18	0.70
grasslands	15	8.1	5.5	2.01	0.71
turf	7	9.1	8.4	2.14	0.57
ruderal	6	9.7	4.4	2.30	0.41
segetal	9	0.1	0.3	0.08	0.23
mixed	5	12.8	8.2	2.49	0.57
total	96	12.1	8.8	2.25	0.96

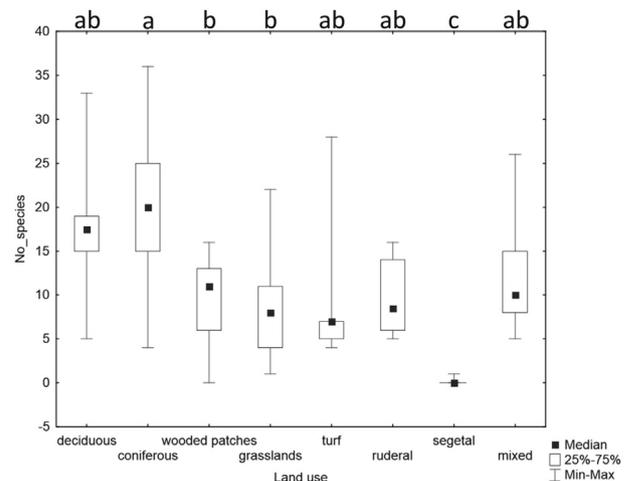


Fig. 1. Effects of habitats on mean number of species (Anova test on log-transformed value, $p < 0.05$). Data not sharing a common letter differ significantly (Tukey test, $p < 0.05$).

found between coniferous and deciduous forests. The lowest similarity value was shown between segetal and the other land use types. There was also little similarity in the cluster including mainly wooded patches and grasslands. The highest overall similarity value among all habitat types (Jaccard index's value > 0.6) was found for coniferous stands, however turfs from two adjacent plots (40 and 41) exhibited highest similarity values out of all studied plots (ca 0.8).

3.4. Functional diversity of lichens in response to land use

In general, species exhibited significantly lower acidity and eutrophication values in coniferous forest than in wooded patches, grasslands and ruderal habitats (Fig. 2). Lichens in turf showed significantly higher indicator value for light compared to coniferous forest and grasslands. Significantly lower indicator value for temperature was found in coniferous forest compared to grasslands,

Table 1

Estimated species richness in all land use types in the Równina Nowotomyska plain, Poland (\pm SD).

Number of species observed	Estimators	Number of species estimated	SD	Completeness (%)
101	Chao2	113.7	8.6	89
	Jackknife 1	117.8	4.9	86
	Jackknife 2	125.7	NA	80
	Bootstrap	108.3	NA	93

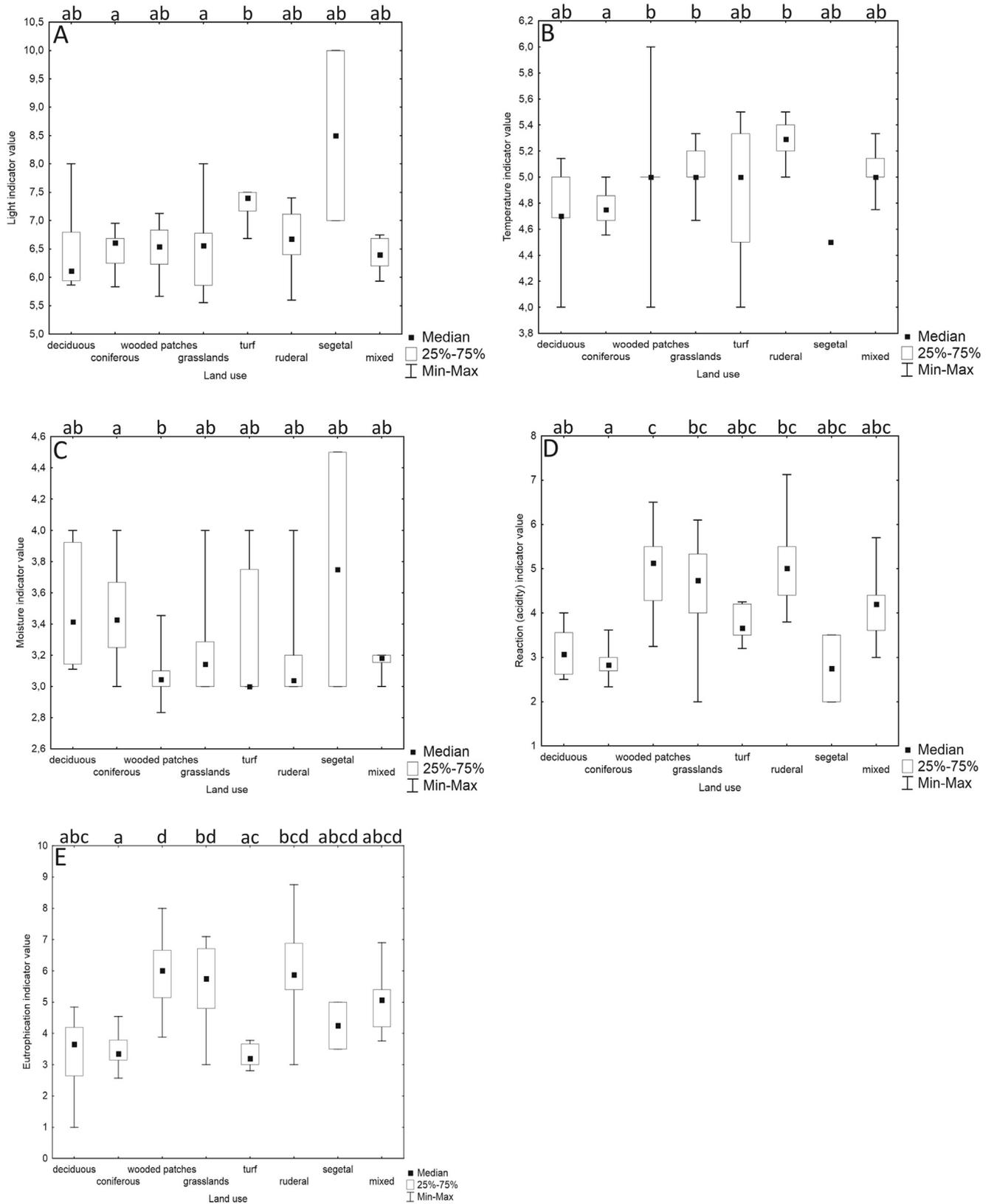


Fig. 2. Effects of land use on ecological requirements of lichens, defined by: A) light indicator value, B) temperature indicator value, C) moisture indicator value, D) reaction indicator value, E) eutrophication indicator value, according to *post-hoc* multiple comparison test in Kruskal-Wallis ANOVA test. Data not sharing a common letter differ significantly ($p < 0.05$).

wooded patches and ruderal habitats. Lichens in deciduous forest exhibited significantly lower acidity and eutrophication values than in wooded patches. Also significantly higher eutrophication values were observed for lichens growing in grasslands and wooded patches compared to turfs. Significantly lower humidity value was recorded for lichens growing in wooded patches compared to coniferous forests. No differences were detected between habitats regarding the continentality value.

Mean number of species with isidia (vegetative symbiotic propagules) was significantly higher in coniferous forests than in wooded patches, grasslands and segetal habitats (Fig. 3). Similarly, a significantly higher mean number of species capable of producing soredia was recorded in coniferous forests in comparison with wooded patches, grasslands, segetal habitats, and additionally with ruderal habitats. Significantly lower species richness of lichens producing this type of propagule, as well as isidia and reproducing via fragmentation was observed in segetal habitats in comparison with deciduous forests. Mean number of species that reproduce mainly through fragmentation was also significantly higher in coniferous forests than in wooded patches, grasslands, turf and segetal habitats. Mean number of fertile species was significantly

lower in segetal habitats than in coniferous forests, wooded patches, grasslands, ruderal and mixed habitats.

We found a weak effect of land use on growth forms of lichens (Fig. S2). Most differences were detected between segetal habitats and other land-use types. A significantly higher number of crustose lichens was observed in coniferous forest than in wooded patches, grasslands and segetal habitats. The latter also had a significantly lower mean number of species than deciduous forests and turf. Differences were also found with species with foliose-narrow thalli; a significantly higher number of species was recorded in coniferous forest than in wooded patches and segetal habitats, as well as in deciduous forest than in segetal habitats. Mean number of species with foliose-broad thalli was significantly higher in coniferous forests than in grasslands, wooded patches and segetal habitats. The number of fruticose species was significantly lower in segetal habitats than in coniferous forest, wooded patches, grasslands, ruderal and mixed habitats.

4. Discussion

We found a significant overall effect of land use on the number

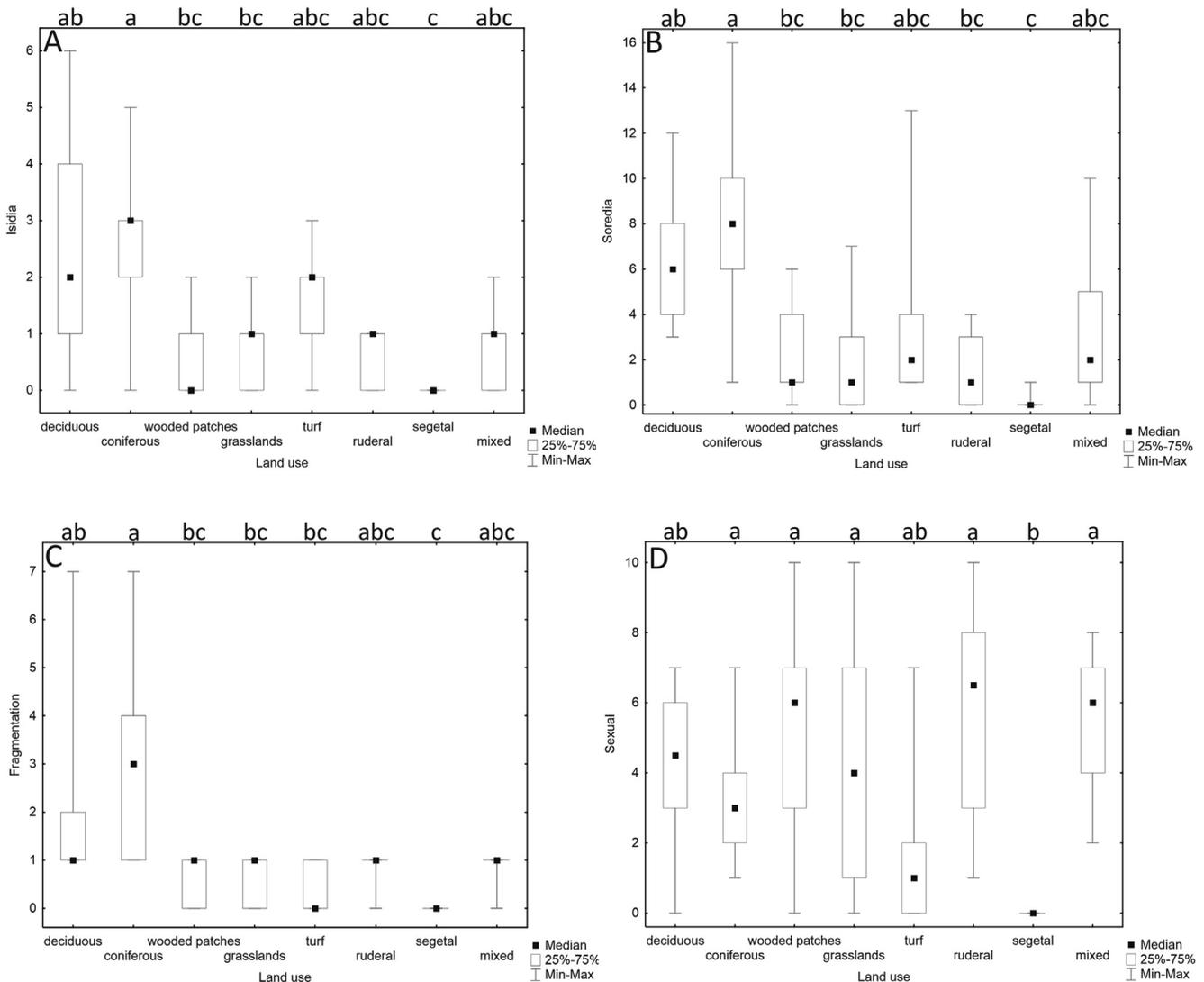


Fig. 3. Effects of habitats on species richness of classes defined by propagules: A) asexual reproduction – isidia; B) asexual reproduction – soredia; C) asexual reproduction – fragmentation; D) sexual reproduction – spores, according to *post-hoc* multiple comparison test in Kruskal-Wallis ANOVA test. Data not sharing a common letter differ significantly ($p < 0.05$).

of species. Similar results were obtained by Bergamini et al. (2005) in lichenological investigations along land use gradients ranging from old-growth forest to farmland across Europe. Although many studies have shown a decrease in species number along land-use gradients from less to more disturbed sites (e.g., Ruoss, 1999; Wolseley et al., 2006; Giordani et al., 2010; Boch et al., 2016), we did not observe such a clear pattern. Also Loppi and De Dominicis (1996) detected only a slight influence of agriculture on lichens based on analyses of the lichen biota growing on oak trees. More subtle and harder to detect differences in lichen diversity in our study area might be also related to a low gradient of land use intensity. It is also worth noting that the statistical assessment of differences between the habitats could be hindered to some extent by the significant variation in sample size between the habitat types (see Table 2).

In general, low species richness in Wielkopolska compared to other regions of Poland seems to be associated with its climatic conditions. The shifting of climatic conditions in Wielkopolska lowland towards steppe-like ones (or towards more continental ones with warm summer), which is caused in some part by human activity (deforestation and drying of agricultural land), reflected by the decrease in precipitation and change in its distribution during the growing season, has been pointed out by some authors (e.g. Jackowiak et al., 2007; lit. cited there). Even in the case of substrate availability, climatic circumstances (low air humidity) do not encourage a species-rich lichen biota in this part of Poland. Thus, differences in species richness, if there are any, between various land use types, especially under less intensive management, can be more subtle and their detection more difficult. Boch et al. (2016 and lit. cited there) linked dissimilarity of lichen species richness in grasslands and forest between the Schwäbische Alb (SW Germany) and Hainich-Dün (central Germany) to regional differences in biotic and abiotic factors, including annual precipitation.

Weak differences in lichen species richness and mainly weak responses of selected functional traits in our study area might be also related to a relatively high anthropogenic pressure present in the area, in all of our study plots (including forests, that were managed). Thus, exclusion of natural habitats from our analyses could also lead to detection of a lower overall lichen species richness. In Scotland, epiphytic lichen communities in ancient pine forest significantly differed from those found in managed sites, both in forest and agricultural areas (Wolseley et al., 2006). Data received from many inventories in Europe suggest that forest management negatively affects lichen species richness in comparison with unmanaged forest (Paillet et al., 2010). The lower lichen species richness found in our area may result from the mono-specific character of locally dominant conifer stands. For example, Motiejūnaitė and Fajtnowicz (2005) pointed out that silviculture combined with relatively intensive agriculture did not encourage a high lichen diversity. Wolseley et al. (2006) also detected low species richness and an increase in tolerant species in conifer plantations. It can be expected that since pine, due to its bark properties, i.e. low pH, low moisture, a high rate of evaporation and unstable surface does not create favourable conditions for lichens (e.g., Kapusta et al., 2004).

In general, dissimilarity of forest versus non-forest sites is marked in our area by species richness and composition, as well as to frequency of species based on their records within plots and the number of occupied substrates (authors' own data not presented in this paper). These findings are in accordance with Giordani et al. (2010) who identified the strongest differences for the forested sites vs. managed agroforestry lands. Also Bergamini et al. (2005) identified differences in mean lichen species richness only between extreme land use sites – old-growth, more or less unmanaged forest and farmland. There are many studies pointing to the

negative influence of agricultural activity on lichenized fungi (e.g., Ruoss, 1999; Motiejūnaitė and Fajtnowicz, 2005; Wolseley et al., 2006; Giordani et al., 2010; Boch et al., 2016). In our area, segetal habitats hosted the lowest number of species. A successful establishment of lichens is prevented here by the application of mineral fertilizers and agrochemicals, as well as some agricultural activities (e.g. ploughing) and a lack of suitable substrates or their removal (e.g., Fajtnowicz, 1991; Lipnicki and Tobolewski, 1991–1992; Gilbert, 2000; Zarabska-Bożejewicz, 2016).

However, some misinterpretations can occur when only total richness or abundance are analysed, because they do not involve species composition and thus a possible turnover of species due to, for example, eutrophication may remain undetected (Van Dobben and Ter Braak, 1998; Kapusta et al., 2004). Recently, some functional traits of species have been used to analyse effects of human activities on biota, including lichenized fungi (e.g., Stofer et al., 2006; Pinho et al., 2008; Giordani et al., 2012; Munzi et al., 2014). We analysed some ecological requirements of lichens recorded across investigated habitats, and the most apparent differences, which concerned tolerance to nutrients and acidity value of particular species, were found. Functional traits constituted a more sensitive and more informative index of lichen response to land use intensity compared to species richness. In general, lichens growing in coniferous forest exhibited, among others, significantly lower acidity and eutrophication values (for the definitions see Wirth, 2010), compared with wooded patches, grasslands and ruderal habitats. Also lichens in deciduous forest showed significantly lower acidity and eutrophication values than those in wooded patches. Significantly higher eutrophication values were observed for lichens growing in grasslands and wooded patches in comparison with turfs. These findings are in accordance with Pinho et al. (2012), who observed that both richness and abundance of nitrophilous species increased in areas under agricultural activities (Pinho et al., 2012). Wolseley et al. (2006) found positive correlations between agricultural intensification, loss of the acidophilous, and increases in the nitrophilous lichens. Also in Central Italy a higher frequency of nitrophiles growing on oaks characterized agricultural sites compared with study plots located within non-agricultural areas (Loppi and De Dominicis, 1996). Investigations in the SW coast of continental Portugal showed that presence of artificial areas, barren land and agricultural areas in the neighbourhood of sampling sites was negatively related to oligotrophic species, and positively to nitrophilic ones (Pinho et al., 2008). Also in W Sardinia (Italy) the gradient of land use intensity was highly correlated with the abundance of nitrophiles and/or broadly distributed species (e.g., *Physcia adscendens*, *Xanthoria parietina*) (Giordani et al., 2010). Some other studies in north-western Italy revealed that nitrophiles were generally more abundant than oligotrophs under high emissions of eutrophication substances (Giordani and Malaspina, 2016). Commonness of eutrophic lichens in agricultural sites may be related, among others, to drier microclimate, dust impregnation of bark (Loppi and De Dominicis, 1996) or a high input of nitrogen compounds into the environment through farming (Ruoss, 1999; Szymczyk and Zalewska, 2008; lit. cited there).

Among functional traits reproductive strategy also seems to be related to land use types. In general, richness of species capable of producing soredia, isidia as well as reproducing via fragmentation was higher in coniferous forest in comparison with non-forest habitats (including wooded patches). Also Stofer et al. (2006) observed that more species with vegetative symbiotic propagules occurred in forest than in open and more intensively managed sites. Symbiotic diaspores are usually larger than fungal propagules and species with larger propagules seem to be adapted to short-distance dispersal (Stofer et al., 2006; lit. cited there). Larger

amounts of suitable habitat available at shorter distance within forest may encourage occurrence of mostly sterile lichens. It should also be stressed, that in particular due to favourable conditions, many *Cladonia* species were recorded in our coniferous forest; most of them are capable of producing soredia or reproducing via fragmentation.

5. Conclusions

In this study, conducted in agricultural land in western Poland, we found some significant differences in lichen species richness and in selected functional traits among studied habitats, in accord with the pattern observed in Europe, i.e., species richness decline in a gradient from woodland to farmland and from less to more intensive land-use. However, these differences were smaller than expected, which might be related to: (1) relatively low overall lichen species richness probably associated with unfavourable climatic conditions in Wielkopolska province; (2) forest communities included in the analyses, i.e. coniferous forest, that is known not to be inhabited by species-rich lichen biota; and (3) low gradient of land use intensity, making differences in lichen diversity more subtle and harder to detect. However, we recognized that species' functional traits, especially based on nutrients/acidity tolerance, gave a better response to land use intensity than the lichen species richness in sites under less intensive management (including agricultural activities). Thus, they may play a role as ecological indicators of anthropogenic impact in an agricultural landscape. It should be also underlined that even in the regions with less favourable conditions for lichens, the agroecosystems can host some interesting species, at least at a local scale, and maintain a moderately rich lichen biota. Our results support those of some earlier studies that highlighted the importance of low-intensity land use for lichen conservation (Szymczyk and Zalewska, 2008; Boch et al., 2016).

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Supplementary data

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