Project no. 227161

BIOBIO

Indicators for biodiversity in organic and low-input farming systems

Thematic Priority: Food, Agriculture and Fisheries and Biotechnology

Funding scheme: KBBE-2008-1-2-01

Report on scientific analysis containing an assessment of performance of candidate farming and biodiversity indicators and an indication about the cost of indicator measurements

Deliverable D4.1

Due date of deliverable: Month 32

Actual submission date: 06/09/2012

Start date of project: 01.03.09

Duration: 3 1/2 years (42 months)

Organisation name of lead contractor for this deliverable: FDEA-ART (WP4)

Final Version

Project co-funded by the European Commission within the Seventh Framework Programme (2009-2012)		
Dissemination Level		
PU	Public	X
PP	Restricted to other programme participants (Including the Commission Services)	
RE	Restricted to a group specified by the consortium (Including the Commission Services)	
CO	Confidential, only for members of the consortium (Including the Commission Services)	

Report on scientific analysis containing an assessment of performance of candidate farming and biodiversity indicators and an indication about the cost of indicator measurements

Philippe Jeanneret[1], Gisela Lüscher[1], Manuel Schneider[1], Michaela Arndorfer[2], Luisa Last[1], Sebastian Wolfrum[9], Katalin Balázs[3], Debra Bailey[1], Marion M.I. Bogers[4], Peter Dennis[6], Sebastian Eiter[7], Wendy Fjellstad[7], Jürgen K. Friedel[2], Ilse R. Geijzendorffer[4], Tiziano Gomiero[8], Felix Herzog[1], Rob H.G. Jongman[4], Maximilian Kainz[9], Anikó Kovács[3], Roland Kölliker[1], Gerardo Moreno[10], Maurizio Paoletti[8], Jean-Pierre Sarthou[5], Siyka Stoyanova[11]

[1](FDEA-ART) Federal Department of Economic Affairs, Research Station ART, Zurich, Switzerland;

[2](BOKU) Division of Organic Farming, University of Natural Resources & Life Sciences, Vienna, Austria;

[3](SIU) Szent Istvan University, Institute of Environmental & Landscape Management, (SZIE-KTI), Gödöllö, Hungary;

[4](ALTERRA) Alterra, Wageningen UR, The Netherlands;

[5](INRA) UMR Dynafor, INRA-ENSAT AGRO, Toulouse, France;

[6](ABER) Institute of Biological, Environmental and Rural Sciences, Aberystwyth University, UK;

[7](NFLI) Norwegian Forest and Landscape Institute, Ås, Norway;

[8](UP) Department of Biology, Padova University, Italy;

[9](TUM) Chair for Organic Agriculture, Centre of Life and Food Science, Weihenstephan, Technical University of Munich, Germany;

[10](UEX) Forestry School, University of Extremadura, Plasencia, Spain;

[11] (IPGR) Institute of Plant Genetic Resources "K.Malkov", Bulgaria.

August 2012

Preface

This document (D4.1) reports on results of analysis of data collected on indicators during the BIOBIO WP3 activities, and calculation of them. The purpose was to scientifically evaluate the set of candidate indicators defined in WP2 for linkage between organic/low-input agricultural systems and biodiversity.

Based on templates prepared for data collection in parallel to WP2 and at the start of WP3, for an efficient computer-based data management, data were obtained by FDEA-ART from each case study. Although standardised data format and structure for analysis were foreseen and disseminated, a considerable workload was caused by the necessary check and standardization of data before analysis.

With the set of data, hypotheses were examined and tested with modern statistical procedures including correlation, regression, analysis of variance, and methods of numerical ecology like multivariate analyses, diversity analysis, i.e. diversity indices, rarefaction, diversity partitioning (alpha-, beta-, gamma-diversity). Main activities concentrated first on preparation and execution of a workshop held at FDEA-ART on 15th to 17th November 2011. Focus was set to correlation analysis to examine redundancy among and between indicators. The way of aggregating plot data at farm level (species diversity indicators) was investigated. Classification of habitats after in situ mapping with the Biohab method (EBONE) was partly adapted to the BIOBIO data. Data analysis and results obtained were then further complemented (factsheets) to be submitted to the stakeholder advisory board (SAB) during the SAB III meeting held on 25th and 26th January in Brussels.

Based on the result of that workshop and further additional analysis, the list of candidate indicators was discussed and agreed on (e.g. earthworms = number of earthworm species, species abundance, alpha-diversity, etc.). Data analysis allowed narrowing down the candidate indicator set proposed by WP2 to a set of biodiversity indicators for organic/low-input farming systems at the European level, possibly still with several options for partly redundant indicators. An analysis of the cost for the record of the indicators was performed. Results of the analysis of cost are integrally reported in Targetti et al. (2011) – Deliverable 3.3.

Content

Preface 3
1 Concept, design and methods of analysis in BioBio 7
1.1 Overview 7
1.2 Approach to farm selection 7
1.3 Mapping a farm (habitat map) 9
1.4 Selecting habitats for species sampling 9
1.5 Species recording 10
1.6 Genetic diversity assessment 11
1.7 Farm management interviews 11
1.8 Preparing the data 12
1.9 Data analysis and indicator selection approach 13
2 Characteristics of indicators assessed in the BioBio case studies across Europe 14
2.1 Genetic diversity indicators 14
2.1.1 Livestock breeds 14
2.1.2 Cultivars 15
2.1.3 Origin of cultivated accessions 16
2.1.4 Molecular Genetic Diversity of Grassland Species 17
2.2 Species diversity indicators 19
2.2.1 Overview 19
2.2.2 Correlations within indices of individual species groups 21
2.2.3 Gamma richness of species across farms and case studies 22
2.3 Habitat diversity indicators 25
2.3.1 Overview 25
2.3.2 Indicator value distribution 25
2.4 Farm management indicators 32
2.4.1 Indicator value distribution 32
3 Testing for correlations within and between indicator groups 39
3.1 Arable and horticultural case studies – Austria, France and The Netherlands 43
3.2 Grassland case studies – Bulgaria, Switzerland, Norway, Hungary and Wales 44
3.3 *Dehesa* case study – Spain 45
3.4 Mixed farming case study – Germany 46
3.5 Vineyard case study – Italy, and Olive plantation case study – Spain 47
3.6 Interpretation of analysis and consequences for monitoring 48
4 What do the indicators indicate? 50
4.1 Case studies, farm types and farming systems 50
4.2 Testing differences between organic and non-organic, and along intensity gradients in low-input farming systems for the direct biodiversity indicators 54

5 On the relationship between species diversity and semi-natural habitats 57
5.1 Relationship between species diversity and semi-natural habitats across the BIOBIO case study regions 57
5.2 Arable and horticultural case studies – Austria, France and The Netherlands 59
5.3 Grassland case studies – Bulgaria, Switzerland, Norway, Hungary and Wales 60
5.4 Case studies: Dehesa – Spain, Mixed farming – Germany, Vineyard – Italy and Olive plantation – Spain 61
6 On the relationship between species diversity and farm management indicators 62
6.1 Total nitrogen input 62
6.2 Total direct and indirect energy input 63
6.3 Intensification/Extensification – Expenditure on fertilizer, crop protection and concentrate feedstuffs 65
6.4 Field operations 67
7 Conclusions 68

APPENDIX 1 – SPECIES RICHNESS INDICES

APPENDIX 2 – CORRELATIONS BETWEEN SPECIES RICHNESS INDICES WITHIN CASE STUDY REGIONS

APPENDIX 3 – CORRELATIONS BETWEEN GENETIC, SPECIES, HABITAT AND FARM MANAGEMENT INDICATORS WITHIN CASE STUDY REGIONS

APPENDIX 4 – BIOBIO CATEGORIZATION OF THE FARM HABITATS: DETAILED CLASSIFICATION USING THE EBONE (HTTP://WWW.EBONE.WUR.NL) TERMINOLOGY

APPENDIX 5 – NAMES AND ADDRESSES OF TAXONOMISTS FOR SPECIES IDENTIFICATION

APPENDIX 6 - Wolfrum, S. et al. (2012). Cost-efficiency of measuring earthworm diversity in a German case study at farm-scale: lessons for application in monitoring and agricultural practice. 12th Congress of the European Society for Agronomy, Helsinki, Finland, 20-24th August 2012.

APPENDIX 7 – Lüscher, G. et al. (2012). Plant, earthworm, spider and bee diversity in agricultural fields of grazing and field crop farming systems in eight regions across Europe. 12th Congress of the European Society for Agronomy, Helsinki, Finland, 20-24th August 2012.

APPENDIX 8 – Amossé, A. (2012). Indicateurs de biodiversité dans les exploitations agricoles biologiques et conventionnelles du cas d'étude français « Vallées et Coteaux de Gascogne » du projet européen BIOBIO. Rapport de stage, INP/ENSAT, Dynafor, Toulouse. 33pp.

APPENDIX 9 – Gomiero, T. et al. (2012). Environmental Impact of Different Agricultural Management Practices: Conventional vs. Organic Agriculture. Critical Reviews in Plant Sciences, 30:95–124, 2011.

APPENDIX 10 – Friedel, J. et al. (2012). Biodiversität von Regenwürmern im Marchfeld, Österreich.

APPENDIX 11 – Nascimbene, J. et al. (2012). Organic Farming Benefits Local Plant Diversity in Vineyard Farms Located in Intensive Agricultural Landscapes. Environmental Management 49, 1054-1060.

APPENDIX 12 – Stoyanova, S. et al. (2011). Intensity in low-input farms and development of indicators for biodiversity. Agricultural Sciences, Vol. III, Issue 7, 55-62.

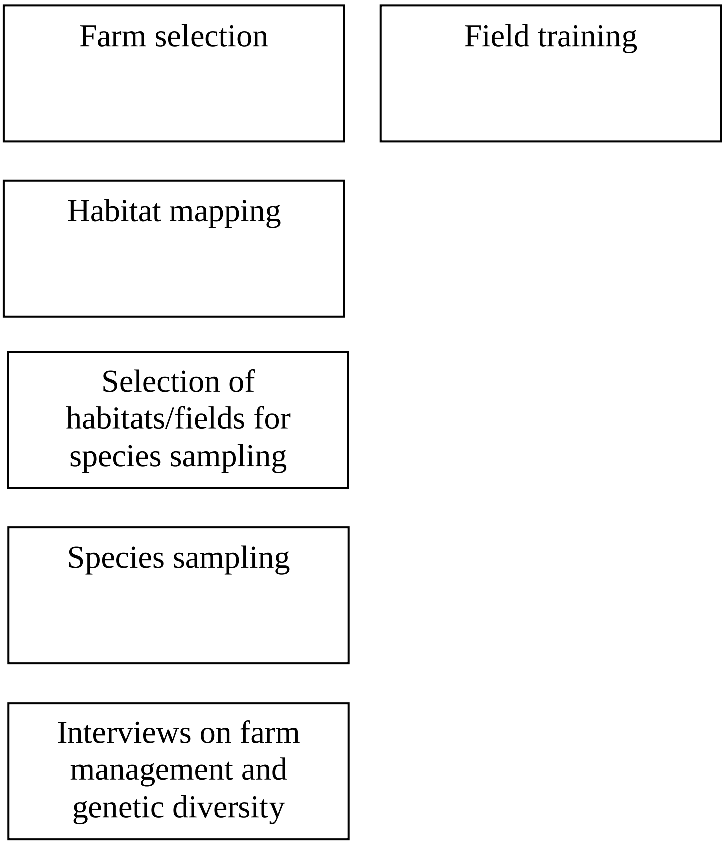
APPENDIX 13 – Kovács, A. et al. (2012). Earthworms, spiders and bees as indicators of habitat and management in a low-input farming region - a whole farm approach. Ecological Indicators (in revision, not included).

1 Concept, design and methods of analysis in BioBio

Concept, design and methods used in BioBio are described with detailed information in Dennis et al. (2009) and Dennis et al. (2012). Sections 1.1 to 1.7 are integrally included in Herzog et al. (2012a).

1.1 Overview

The activities for the farm scale monitoring of direct and indirect biodiversity indicators in BioBio were chronologically structured as follows:



First, farms were selected in regions (case study). In the meantime, field training were organised for the habitat mapping and the species sampling methods.

Second, habitats were mapped on farms.

Third, habitats/fields were selected from maps to be sampled for species indicators.

Fourth, species indicators were sampled in the selected habitats/fields.

Fifth, farmers were interviewed to enable assessment of the management and genetic diversity of crops and livestock.

1.2 Approach to farm selection

BioBio case study regions representing major organic and low-input farming systems were selected on the basis of the HNV Farmland method (Andersen et al., 2003) and statistical sources (EUROSTAT, Organic Farming in Europe – Country Reports http://www.organic-europe.net/) according to their relative importance and distribution across Europe. However, while case-study regions were homogeneous in terms of biogeographical conditions and farming types. They cover low to medium intensive organic and non-organic farming; very intensive conventional farming, industrial animal production, etc. were not covered.

Each of the 12 European case studies (as well as the additional case studies in Tunisia, Ukraine and Uganda) focused on a factor of interest, i.e. organic versus non-organic (baseline) systems or low-input systems along a gradient of farming intensity. To qualify as an “organic farm” in BioBio, the farm had to have been certified as organic since 2005, i.e. to have been continuously managed according to organic farming standards (EC 2007) for a minimum of five years. Farms that had been certified for less than five years or that were in the process of transition to organic farming were excluded from the project. Farms that were not certified organic are referred to here as “non-organic” or “conventional” (as commonly used in the literature). In the case of low-input systems, one or two significant variables were chosen to define the largest possible intensity gradient for farm selection.

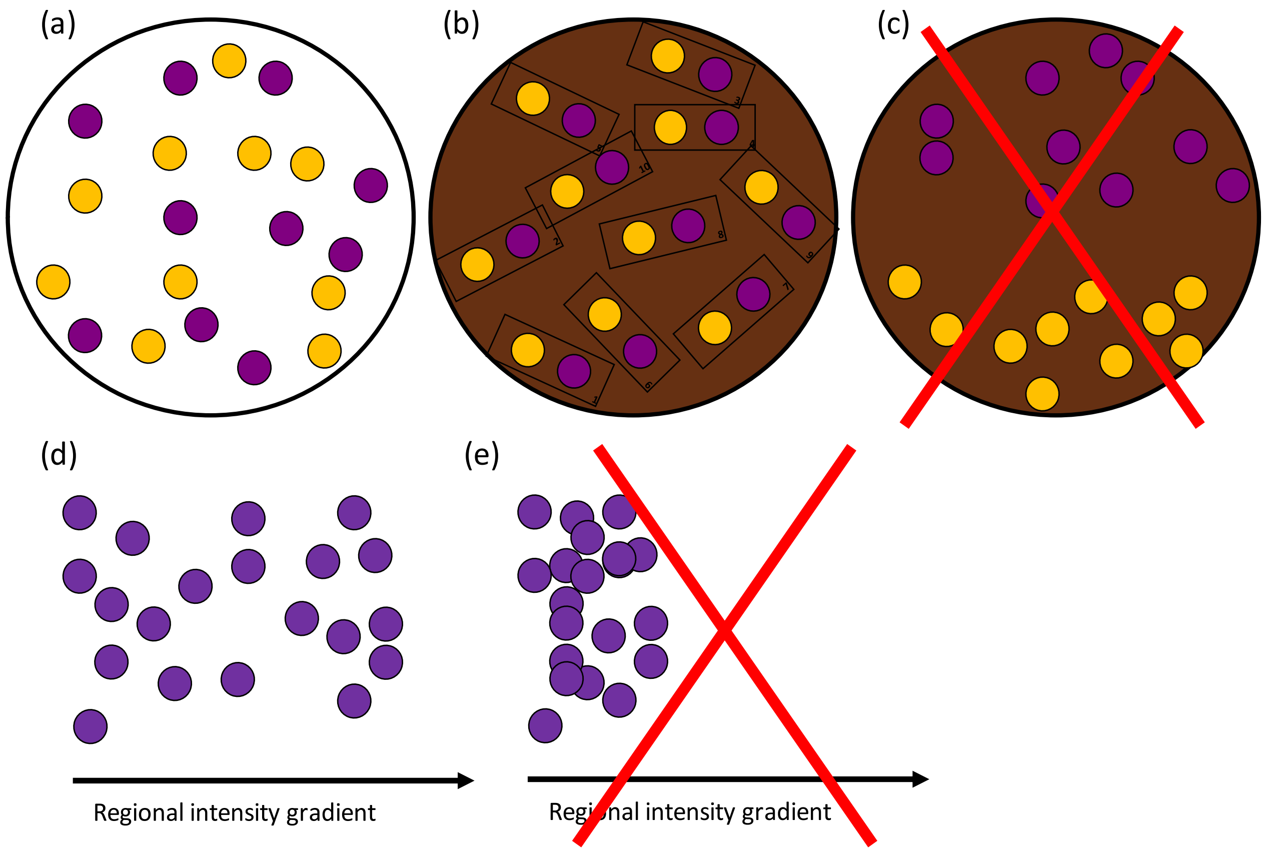


Figure 1: Acceptable patterns of farm selection for case studies with organic and non-organic farms (a) and (b), and for HNV regions with a gradient of intensity (d). Systematic bias in options (c) and (e) must be avoided.

When selecting farms for biodiversity monitoring confounding factors must be accounted for, especially if different farming systems are to be sampled (organic and non-organic in BIOBIO). Two sets of potentially confounding factors were recognized in BIOBIO:

1) Environmental conditions: biogeographical region, geomorphological and soil features, landscape situation, altitude;

2) Farm characteristics: type of farm (crops, forage, mixed farming, animal species), size, management intensity, uncultivated habitat types.

Examples of possible confounding effects and problems of interpretation caused by poor farm selection include the following:

a) All (or most) of the organic farms are selected at high altitude in a region, whilst all (or most) of the non-organic farms are selected at low altitude. An observed difference in biodiversity indicator values cannot be attributed unequivocally to the farming system because altitude is correlated with the latter. It is then difficult to determine whether an observed difference in measurements of biodiversity indicators is due to the farming system, or to the altitude (Figure 1).

b) All (or most) of the selected organic farms grow crops, whilst all (or most) of the selected non-organic farms have mixed farming (or vice versa). An observed difference in biodiversity indicators cannot be attributed unequivocally to the farming system because the type of farm is correlated with the latter. In this example, it is difficult to determine whether an observed difference in biodiversity indicator measurements is due to the farming system, or to the type of farm.

In each case study region, 16–20 farms were randomly selected out of the 30-40 which were available for the evaluation of candidate biodiversity indicators, and which had been preselected with the aim of avoiding potential confounding factors.

In the case of heterogeneous regions, farms were selected in pairs, i.e. one organic and one non-organic farm in the same environmental conditions in each case.

1.3 Mapping a farm (habitat map)

A farm is a business that usually consists of several components (economic activities such as crop production or animal husbandry) and several fields. Farms are often not consolidated, i.e. rather than being adjacent to one another, individual fields may be spatially distributed over relatively large areas. As a first step, the farm boundaries were obtained either from cadastral maps or from the farmer. The survey area was defined by the farm property boundary. Fields that were owned by the farmer, but managed by others were excluded from the survey, whilst rented land that was owned by others but managed exclusively by the farmer was included. Habitats of the farm were provisionally identified on the basis of aerial photographs or satellite images available on the internet. Detailed habitat mapping was conducted in the field. Habitats were defined on the prepared base maps (e.g. aerial photographs) and described on standard forms. Site, environmental and management conditions were described according to a predefined codes for geomorphology, geology, soil, etc.

Mapping the habitats of a farm is the first step of biodiversity indicator recording. BIOBIO has adopted a standard habitat mapping procedure for the European scale developed by Bunce et al. (2008). The habitat/land use classification method is based on a generic system of habitat definitions or so-called General Habitat Categories (GHCs); see Dennis et al. (2012) and www.ebone.wur.nl.

1.4 Selecting habitats for species sampling

Once a farm had been mapped, habitats were grouped into types (Figure 2). Generally, each GHC or combination of GHCs represents a different habitat type except for grasslands, for which a finer classification is used. Grassland GHCs are further subdivided according to their moisture and nutrient levels as indicated by the environmental conditions.

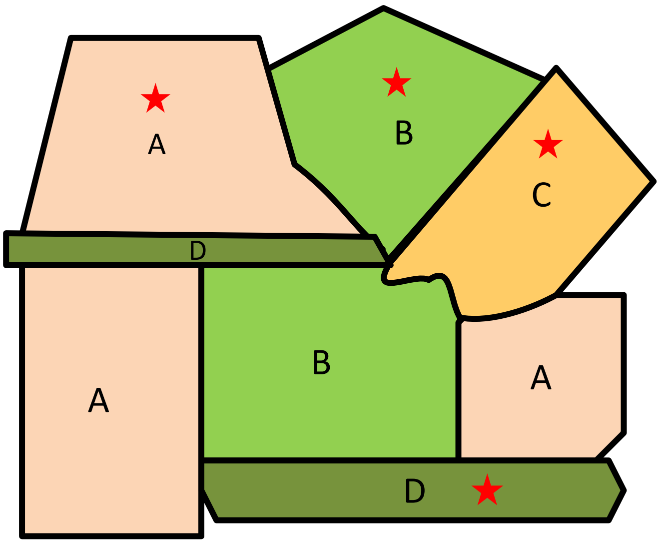


Figure 2: Schematic farm with six areal and two linear habitats belonging to four different habitat types (A, B, C, D). From each habitat type, one individual habitat (stars) is randomly selected for species diversity recording.

From each of the habitat types belonging to the farmed area, one habitat per type was randomly selected as a *survey habitat* for plant, bee, spider and earthworm species. This random selection of habitats, including all habitat types on a farm, one habitat per type, is the basic structure of the sampling design for assessing species diversity at the farm level.

1.5 Species recording

Species recording methods are described in detail by Dennis et al. (2012). Basically, in each selected habitat for flora and fauna surveys, all of the following species groups were sampled:

- Flowering (vascular) plants of farmland habitats;
- Wild and domestic bees and bumblebees of farmland habitats (hereinafter referred to simply as 'bees');
- Spiders of farmland habitats;
- Earthworms of farmland habitats.

The spatial location for sampling areal and linear habitats is illustrated in Figure 3.

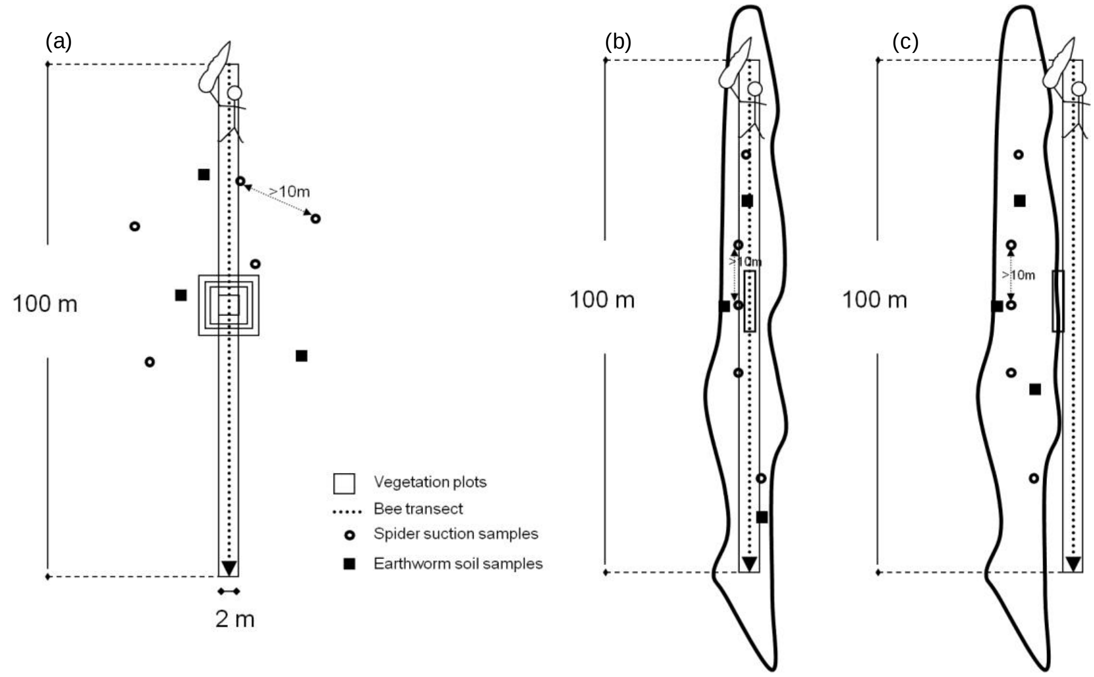


Figure 3: Flora and fauna sampling in areal (a) and linear habitats, without shrubs (b) and with shrubs (c). Source: Dennis et al. (2012).

Vegetation – The procedure for recording vegetation used two types of sampling plots, square and linear. Square plots were positioned in areal habitats (Figure 3) and linear plots were positioned in linear habitats. Vegetation was recorded in nested plots of $4m^2$, $25m^2$, $50m^2$ and $100m^2$ respectively in areal habitats, and of 10m x 1m in linear habitats. All vascular flowering plants were recorded (ferns, bryophytes and lichens were not recorded). Once the whole plot was recorded, the estimated cover percentage for the entire plot was listed against each species, using 5%-cover categories.

This procedure provides basic information on the species composition of vegetation within the habitats, as well as allowing a rating of quality for assessing future change.

Wild and domestic bees and bumblebees: Bees were captured with an aerial net. Each habitat was surveyed by a slow walk along a 100-metre-long, 2-metre-wide transect crossing the centre of the vegetation plot. Where habitat length was shorter than 100m, 2 x 50m transects were surveyed. The transect walk lasted 15 minutes. While walking, the collector caught all individual bees seen within the 2m-wide 'belt' with a standard entomological aerial net. Captured specimens were immediately transferred into a killing jar charged with ethyl acetate, and taken to a taxonomist for identification if they could not be immediately identified in the field by the collector. The transect walk was repeated three times throughout the season.

Spiders: Spiders were caught with a modified vacuum shredder powered by a two-stroke engine. A suction sample composed of five subsamples was taken in each habitat selected from the habitat map of each farm. Each of the five suction subsamples was taken within a sample ring (0.357 m internal diameter and 40 cm height) placed beforehand at random on the target vegetation within the habitat. Samples were stored in a cool-box. Plant material, soil and other arthropods were separated out in the laboratory, and the spiders were identified by a taxonomist. Sampling was repeated three times throughout the season.

Earthworms: Cool and wet weather were preferred for sampling earthworms. Extraction by applying an expellant solution (diluted allyl isothiocyanate (AITC)) causing the earthworms to come to the soil surface was first performed in three samples of 30 x 30 cm each. After this, three soil cores (each 30 x 30 x 20 cm deep) were taken with a spade. Earthworms were extracted from the soil core, stored in alcohol, and identified by a taxonomist.

A total of 1490 habitats were surveyed on 195 farms (Table 1).

Table 1: Number of farms and survey habitats for species per case study region. Data analysis shown here is based on the 12 European case study regions. ARA = Arable, HOR = Horticultural, GRA = Grassland, DEH = Dehesa, MIX = Mixed farming, VIN = Vineyards, OLI = Olives.

Number of:	ARA Austria	ARA France	HOR The Netherlands[1]	GRA Bulgaria	GRA Switzerland	GRA Hungary	GRA Norway	GRA Wales [2]	DEH Spain	MIX Germany	VIN Italy	OLI Spain	Total
Farms	16	16	14	16	19	18	12	20	10	16	18	20	195
Habitats	123	157	103	133	109	148	118	213	111	127	74	74	1490
Habitat types	15	36	19	51	19	58	23	45	31	14	11	14	n.a.
Habitat types per farm ∅	7.7	9.8	7.4	8.3	5.7	8.2	9.8	10.7	11.1	7.9	4.1	3.7	7.6

[1] No sampling of fauna species on 6 farms
[2] No sampling of fauna species on 4 farms

1.6 Genetic diversity assessment

A comprehensive set of indicators for detecting biodiversity in farming systems must include measures of genetic diversity within species. However, reliable detection of genetic diversity is generally labour-intensive, often technically demanding, and can be difficult owing to the lack of information on e.g. breeding pedigrees and seed sources. In BIOBIO, the assessment of on-farm genetic diversity is based on a questionnaire surveying data on the number and abundance of different breeds per farm animal species, the number and abundance of different varieties per crop species, the origin of crops, and pedigree-based genetic diversity. Data were collected together with the farm management data.

1.7 Farm management interviews

The farm management questionnaire is the basis for farm management data collection. Designed to cover the management practices of farms with and without livestock, the questionnaire takes into account different land-use types such as grassland, arable crops and permanent crops (olives and vineyards), as well as semi-natural habitats (field margins, hedges etc.). Data were recorded on different scales of measurement: farm level, crop level (standard operations for each crop), and field level (selected habitats of the species survey). All data collected in the farm management questionnaire derived from the interviews based on farmers' operational knowledge of their farm and on basic farm accounting.

The farm management questionnaire was divided into 4 main sections (A, B, C and D) and several subsections:

Form A surveyed general farm data collected at farm level, such as overall energy consumption, agri-environmental measures, organic matter fluxes, etc.;

Form B yielded parameters describing the farm's plant production system. Standard operation data such as fertilization practices, plant protection measures and mechanised field operations were collected for each crop or grassland type. Data were used to calculate nitrogen input and nitrogen balances as well as to assess farming intensity based on grazing management, plant protection measures and mechanised field operations. The synthesis of data from all completed 'B' forms reflects the complete plant production system of the farm.

Form C concerned the specific management of habitats where flora and fauna indicators are sampled (results not shown here).

Form D provided information on livestock management and livestock numbers on the farm, broken down by livestock category, enabling the calculation of livestock units. Additional parameters were meat production (indicator of productivity), use of pastures and common grazing land.

Data were processed in spreadsheets, where further indicator calculation was performed, as well as in the online tool Dialecte (http://dialecte.solagro.org/).

1.8 Preparing the data

Preparation of data for analysis is necessary but often difficult and tedious work. In BIOBIO, after collection in the 12 case study regions, raw data were prepared in electronic spreadsheets. However, before indicators can be calculated, e.g. the gamma species richness of bees, the total nitrogen input on farms etc., data were checked for overall consistency, synonyms were identified for species, GIS data and habitat records were adjusted to ensure comparability between case studies, etc. Although standardized methods were implemented in all the case study regions, and templates for electronic data format organized to help for correct preparation of the data, the work load for data check and correction was huge. This caused important delays for the data analysis. It must be emphasized that this step is necessary to ensure proper analysis of data across case studies and should not be underestimated.

After raw data were prepared in Excel® spreadsheets, indicators were calculated using the tool for statistical analysis R version 2.15.1 (R Development Core Team, 2011). The data and indicators were then checked for consistency during a second step including feedback loops between individual partners and FDEA-ART.

1.9 Data analysis and indicator selection approach

The following criteria for indicator selection were applied:

- Indicators must be reliably measurable across Europe;
- There must be no consistent correlations, or only minor ones with other indicators within the four categories of (i) habitat diversity indicators, (ii) species diversity indicators (iii) indicators for the genetic diversity of crops and livestock, (iv) farm management indicators (i.e. to avoid costly redundancy);
- Farm management indicators as well as habitat diversity indicators should correlate with species diversity indicators ("indicative performance", hypothesis driven);
- Indicators should detect differences between farms;
- Indicators must pass the stakeholder audit.

In the following sections, analysis concentrates on correlations within and among indicator groups as well as on the indication performance of indicators. The purpose is to help decision making about indicators. Most of the results were produced during the process of the stakeholders' consultation and of writing the document "Guidebook and Factsheets" (Herzog et al., 2012a).

While species indicators and specific farming practices were recorded on individual habitats, management and habitat indicators were surveyed for the entire farm (see also section 3). In the following sections, analysis concentrates on indicators derived at the farm scale. Further detailed analysis relying on species indicators and farming practices will be carried out and results published in scientific journals.

If two indicators correlate consistently both within and between case studies they convey the same information, and only one of them need be measured. Within the four indicator groups (habitat, species, genetic diversity and farm management), correlating indicators were identified. In a second step, correlations across the themes were investigated, in particular between management, habitat and species diversity indicators. These correlations may indicate relationships which help us to understand the biodiversity findings, and in the case of strong and consistent correlations across case study regions, surrogate indicators may be identified.

Correlation analysis used the Spearman rank correlation rho (Legendre and Legendre, 1998) and its test according to Hollander and Wolfe (1973), both implemented in the tool for statistical analysis R version 2.15.1 (R Development Core Team, 2011).

Principal Component Analysis (PCA) was used to investigate indicator performance to distinguish between farms of the 12 case study regions, farm types, i.e. "Field crops & horticulture", "Specialist grazing livestock", "Mixed crops & livestock" and the "Permanent crops", case study regions, and farming systems, i.e. organic, non-organic and low input. Based on PCA, ordination diagrams were prepared using the package 'vegan' (Oksanen et al., 2010) of R version 2.15.1 (R Development Core Team, 2011) and information from Borcard et al. (2011).

To test for relationships between management indicators and species indicators, fitted linear models were run using the 'lm' function from R version 2.15.1 (R Development Core Team, 2011) following Chambers (1992).

2 Characteristics of indicators assessed in the BIOBIO case studies across Europe

2.1 Genetic diversity indicators

Information on livestock breeds and cultivars used on each farm was tested as a surrogate for genetic diversity. This information can be collected as part of a farmer questionnaire for farm management.

2.1.1 Livestock breeds

Cross breeds were included since genetic diversity of an off farm sire breed is effectively introduced via artificial insemination or loan of a single bull or ram. Indicators are compared at the whole farm level since e.g. grazing animals use the full extent of the grazed or forage cropped land throughout each year. In the BIOBIO case studies, only cattle and sheep were significantly represented on the livestock specialist and mixed farms (Figure 4).

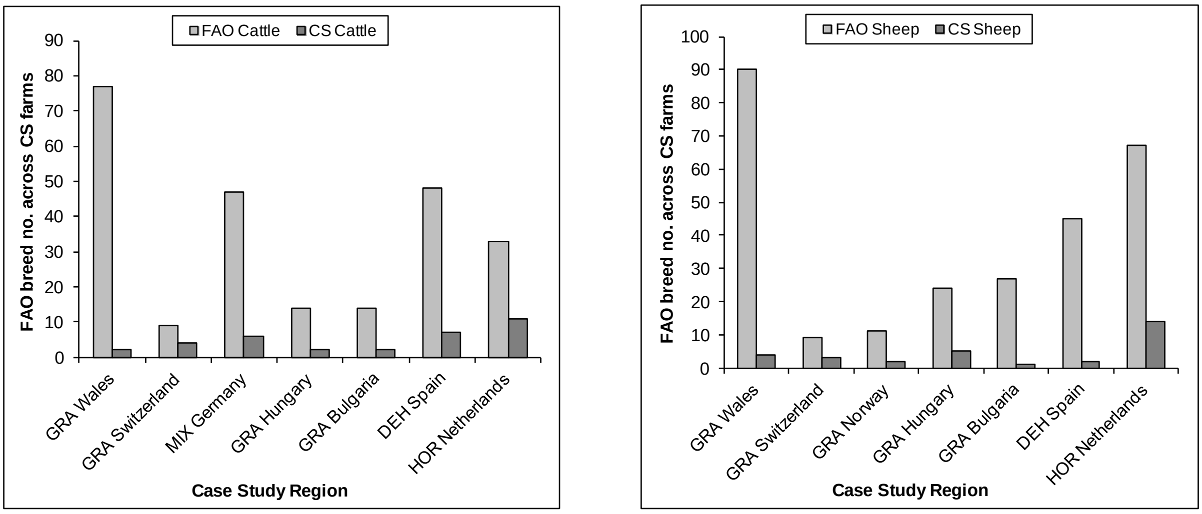


Figure 4. Number of breeds reported per country by FAO DAD-IS (Domestic Animal Diversity Information System) (DAD-IS online), and number of listed species of cattle / sheep represented in the BIOBIO Case Studies(CS) that had livestock.

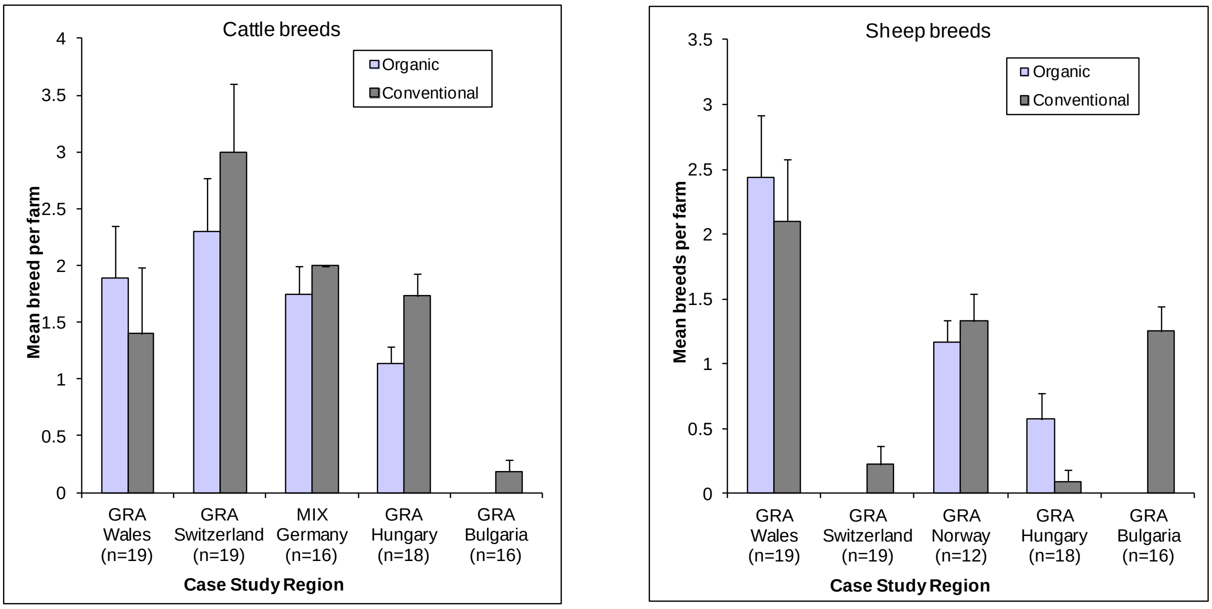


Figure 5. Average number of breeds (± SEM) per farm of cattle and sheep recorded in Case Study regions of BIOBIO project, 2010, for those CS that had rare or local breeds. Comparison of mean cattle breeds between organic and non-organic farms by t-test, GRA Hungary: t = -2.418; P <0.05; remaining CS regions t < 1.00; not significant. Comparison of mean sheep breeds between organic and non-organic farms by t-test, GRA Hungary: t = 2.169; P = 0.06; remaining CS regions t < 1.00; not significant.

In the seven case study regions with animal farming, 26 cattle and 30 sheep breeds were recorded. Rare and local breeds were represented in 5 case studies (Figure 5). The average number of breeds per farm was 1-2, irrespective of production system and was characteristic of the specialization of individual farms to particular enterprises, regardless of management system. The mean breeds per species are calculated separately for cattle or sheep from the individual farm values. These farm values can be for different categories of livestock production system to allow analysis of differences, for instance organic versus non-organic systems. In the BIOBIO case studies, there was no significant difference in number of breeds between organic and non-organic farms, except in the Hungarian case study where non-organic farms had more cattle breeds per farm on average.

2.1.2 Cultivars

Application of various cultivars on a farm is supposed to increase resistance and also resilience after abiotic (temperature, drought) and biotic (pests, diseases) disturbances (see literature review in Dennis et al., 2009). In particular, agricultural systems dominated by only one cultivar might be more susceptible to any kind of disturbance. Also, an increased amount of various cultivars might attract more pollinators enhancing ecosystem functioning in agricultural landscapes. Furthermore, different cultivars of one species might have different requirements for soil composition and properties. An increase of cultivars would, therefore, lead to a more balanced use of below-ground resources by any kind of crops in agriculture. An increase of cultivar diversity on a farm may be due to the farmers' personal preferences and experiences, but may also be due to political issues supporting the use of multiple cultivars per species either to maintain genetic resources of crops, or cultural heritage or to enhance variability and productivity of agricultural products. A decline in indicator value (change in state) may indicate pressures on biodiversity, e.g. increased use of one variety as monoculture due to intensification. Besides annual changes of crop rotations and changes in cultivar use, an increased cultivation of multiple cultivars of one species per farm can also be a response to farmers' preference, taking into account unpredictable climatic events or higher chances of pest resistance by using various cultivars of one species. Planting multiple cultivars on a farm may indicate a change from intensive and high yielding farming to more locally adaptive farming by using more or less diverse crop genetic resources.

Cultivar diversity has been calculated by dividing the total number of cultivars by the number of crop species on the farm. Cultivar diversity has been calculated for all farms in all case studies. The cultivar diversity was highly variable across farms in case studies but was around 1.5 cultivars per species over the case studies on average except in the Italian case study (vineyard, Figure 6).

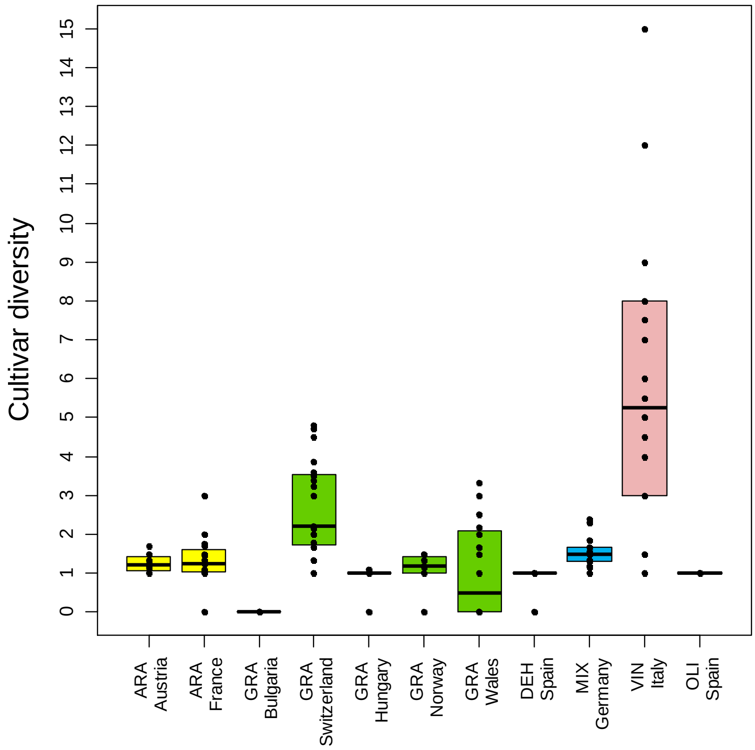
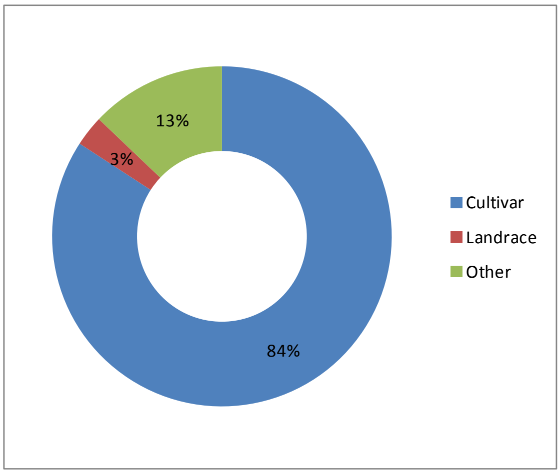


Figure 6. Average number of cultivars per crop species on farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and farms (black dots).

2.1.3 Origin of cultivated accessions

Origin of cultivated accessions is an indicator based on landraces cultivated on a farm. A landrace is a local variety of a domesticated plant species highly adapted to local conditions due to natural selection and evolutionary processes. Compared to cultivars, landraces are heterogeneous but less yielding. However, the great amount of landraces worldwide provides a major basis for plant breeding. The unit is percentage of landraces grown on the farm, measured across all crop species and varieties. Calculation of the indicator "Origin of cultivated accessions" uses the percentage of landraces on a farm.



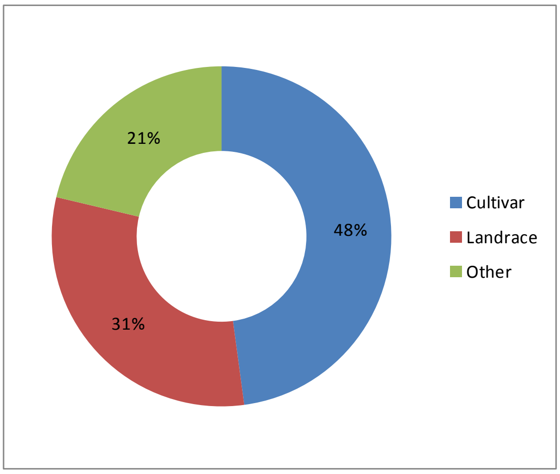


Figure 7. Origin of accessions compared between (a) European case study regions (all varieties = 171), and (b) the case study region in Uganda (all varieties = 119).

The analysis of data showed that there were very few landraces in general in the BIOBIO case studies, i.e. 5 landraces in all 195 farms of the case study regions (which occurred in Germany, Dehesas and Olive plantations in Spain), compared to the case study in Uganda where 37 landraces were found on only 16 farms. In Uganda, the use of landraces is very common and provides a important part of agricultural production. Whereas in agrarian countries, represented by selected case study regions, landraces are of minor importance. Here, landraces are a curiosity in greater agricultural production systems, i. e. 1 single wheat landrace in the German case study, or belong to the category of tree crops, which have been used over many decades in natural production systems as in the Spanish Dehesas or olive groves.

2.1.4 Molecular Genetic Diversity of Grassland Species

The use of molecular tools, e.g., molecular marker or genotype sequencing, allows a much more detailed insight into diversity of individuals or population structures by addressing genetic variation directly at the DNA-level and excluding environmental factors. These tools enable the measurement and evaluation of genetic diversity of ex situ / in vitro and in situ / in vivo genetic resources. Thus, the variability of genetic material can be investigated. Potential breeding material can be analysed in detail and the genetic structure of populations can be investigated and valuable alleles, allele sequences, genotypes, individuals of wild relatives and populations can be detected and conserved. The continued development of molecular tools, e.g., genotyping methods such as next generation sequencing, enables rapid, precise and efficient investigation.

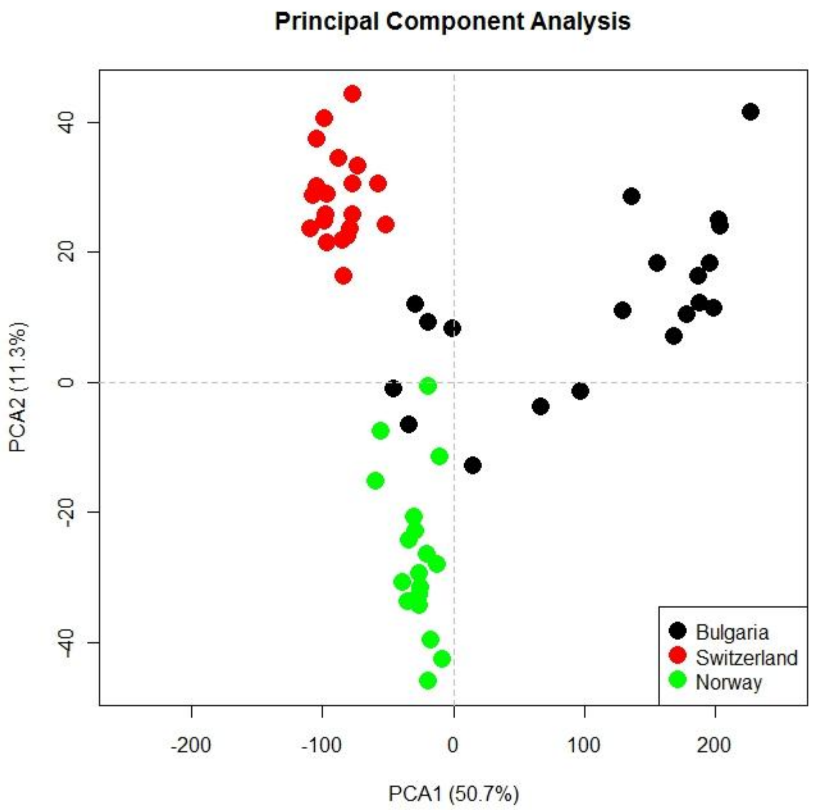


Figure 8. Principal Component Analysis of *Dactylis glomerata* populations based on the total number of alleles per loci and population in the case study regions of Bulgaria, Switzerland and Norway. 20 Swiss, 20 Bulgaria and 18 Norwegian populations are presented.

Dactylis glomerata populations in case study regions of Bulgaria, Switzerland and Norway were investigated. Figure 8 shows that the three populations are well separated in the multidimensional space of alleles of plant populations (represented on axis 2 of the PCA). However, some Bulgarian and Norwegian populations were close to each other showing similarity in the allele composition.

Gene diversity or expected heterozygosity, introduced by Nei (1973), is a common diversity index in studies on genetic diversity of populations based on allele frequencies occurring within and among populations.

$$H_E = \left(1 - \sum_{i=1}^{m} x_i^2\right)$$

In which n is sample size, x_i is the frequency of allele i, m is the number of alleles at one locus; $H_E = 0$ (two populations have no alleles in common), $H_E = 1$ (two populations have same allele frequencies).

In Switzerland and Norway, populations were sampled in intensively managed and low-input meadows (categories based on the habitat mapping and vegetation plots). Gene diversity of *Dactylis glomerata* populations in farms depended on the country (df = 2, F = 4.02, p = 0.02)(Figure 9). Tukey post-hoc test revealed that the populations of Norway and Bulgaria were significantly different (p = 0.01). In Switzerland and Norway, the interaction between the management intensity and the country was not significant, neither was the management intensity (df = 1, F = 0.04, p = 0.84).

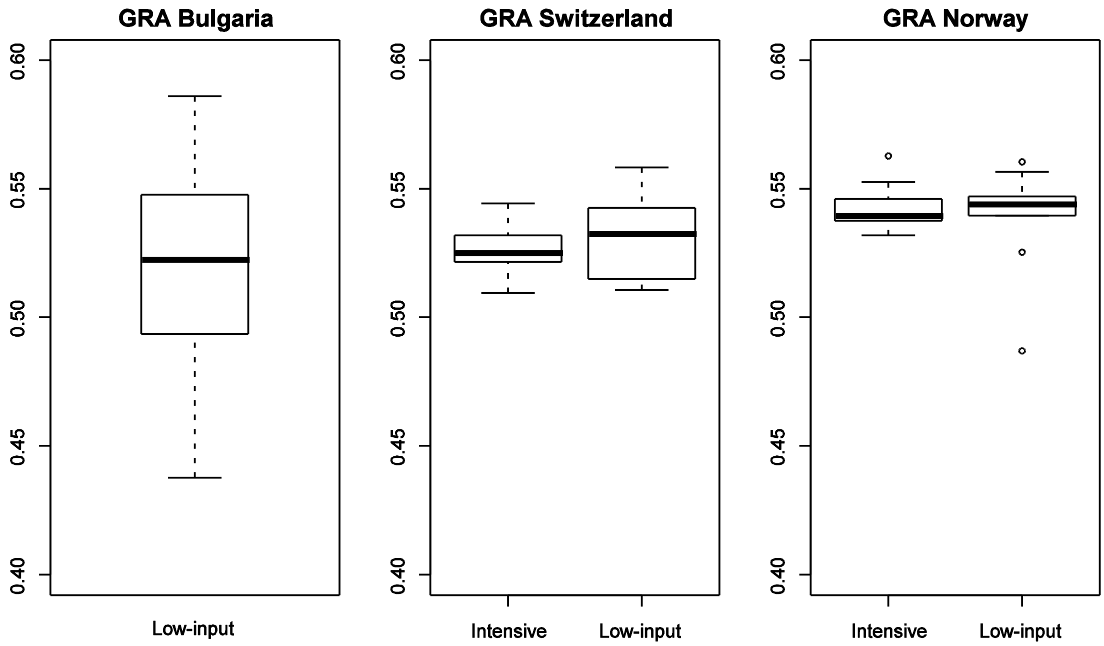


Figure 9. Boxplot (lower quartile, median, upper quartile) of the gene diversity in 3 BIOBIO case study regions. Each country is represented by 10 farms. In Bulgaria, low-input faming is the only farming system. Whereas, the Swiss and Norwegian case studies are represented by 5 organic and 5 non-organic farms. On each farm, one intensively and one extensively managed plot was sampled leading to 59 populations of n = 20 (CH), 19 (NO) and 20 (BG).

The allele diversity in populations of the Bulgarian and Swiss case studies were compared to the plant species richness found in habitats to investigate whether the genetic diversity of *D. glomerata* could represent the plant species richness. Neither the Bulgarian nor the Swiss populations were significantly related to the plant species richness found there (df = 18, t value = 1.06, p = 0.3 and df = 18, t value = 0.4, p = 0.7).

2.2 Species diversity indicators

2.2.1 Overview

Across the 12 case study regions altogether, 1'581 plant species were observed with on average 83 species per farm (Table 2). 24'183 earthworm individuals, 33'129 spiders and 6'230 bee individuals were collected, belonging to 49, 604 and 382 species respectively, with on average 6, 31 and 11 species per farm. The number of species and individuals per case study region is detailed in Table 2 and shows high variability among regions.

In BIOBIO, the farm level of analysis allows the investigation of relationships among biodiversity indicators, i.e. gene, species, habitat and management indicators, at a common spatial unit, i.e. the farm. While habitat diversity, genetic diversity and farm management surveys directly yield indicators at the farm level, species surveys are undertaken in individual habitats and need to be upscaled to the farm level, which is the common denominator for BIOBIO indicators. Species indicator measurements in individual habitats must be aggregated to a farm level value to be related to indicator measurements that only occur at the farm level.

In BIOBIO, species data consisted of lists of species and abundance of individuals (percentage cover for plants), collected or observed in habitats. Once the species lists are available, there are at least five different methods to aggregate species data to a farm level from species data obtained in habitats.

Gamma richness:	Total number of species of the farm pooled over the sampled habitats
Alpha richness:	Average number of species per habitat, averaged over the sampled habitats
Area weighted richness:	Total number of species over the sampled habitats weighted by the area of the habitat types to which each habitat belongs
Rarefied richness:	Average number of species calculated for the smallest number of habitats found in a farm (rarefaction) of the respective case study
Chao estimated richness:	Estimated number of species in a farm based on the accumulated number of species found in habitats (after Chao, 1987)

Given the approach of sampling one habitat per habitat type over the farm, the five indices should similarly approximate the species richness of that farm. The main critical points of this approach for the species richness at farm level are (1) all habitats of the same habitat type are implicitly attributed to the same number of species, i.e. the possible and probable variability of species diversity among habitats of the same habitat type (so called beta-diversity within the habitat type) is not considered, and (2) the definition of the habitat types used for the habitat mapping is fundamentally based on life forms, vegetation types and some environmental conditions which perhaps neglects important habitat features for e.g. bees that might be better represented by another habitat type definition; the consequence is that two habitats may be attributed to one habitat type with respect to the vegetation type found there, but could correspond to two distinct habitat types if bee habitat characteristics would be considered.

Table 2: Summary of species and individual numbers found in the 12 BIOBIO case study regions. ARA = Arable, HOR = Horticultural, GRA = Grassland, DEH = Dehesa, MIX = Mixed farming, VIN = Vineyards, OLI = Olives.

		ARA Austria	ARA France	HOR The Nether-lands	GRA Bulgaria	GRA Switzer-land	GRA Hungary	GRA Norway	GRA Wales	DEH Spain	MIX Germany	VIN Italy	OLI Spain	Total
	Farms	16	16	14	16	19	18	12	20	10	16	18	20	195
	Habitat types	15	36	19	51	19	58	23	45	31	14	11	14	336
	Habitat types per farm	7.7	9.8	7.4	8.3	5.7	8.2	9.8	10.7	11.1	7.9	4.1	3.7	7.9
Plants	Species	247	360	207	364	269	388	200	321	403	211	246	288	1‘581
	Species per farm	50.4	101.2	49.6	78.0	84.5	90.9	88.0	84.0	164.1	70.1	60.4	71.9	82.8
Earthworms	Individuals[1]	1'164	7'962	671	293	2'321	474	928	4'226	2'337	2'664	219	924	24'183
	Species	10	16	16	8	17	8	10	18	17	11	14	19	49
	Individuals per farm	72.8	497.6	47.9	18.3	122.2	26.3	77.3	211.3	233.7	166.5	12.2	46.2	1'532
	Species per farm	4.7	10.4	4.4	3.4	10.4	2.3	5.8	8.6	6.4	7.8	3.4	4.5	6.0
Spiders	Individuals[1]	1'470	4'879	500	770	2'200	1'816	3'175	9'214	2'921	4'272	466	1'446	33'129
	Species	128	215	76	106	125	163	104	159	116	110	86	123	604
	Individuals per farm	91.9	304.9	35.7	48.1	115.8	100.9	264.6	460.7	292.1	267.0	25.9	72.3	2'079.9
	Species per farm	30.2	64.5	11.6	19.8	28.9	29.3	36.8	45.8	38.0	35.9	12.2	22.5	31.3
Bees (*Apis mellifera* excluded)	Individuals	101	2'127	73	356	570	298	812	588	485	115	453	252	6'230
	Species	49	153	22	91	64	101	23	13	51	34	64	44	382
	Individuals per farm	6.3	132.9	5.2	22.3	30.0	16.6	67.7	29.4	48.5	7.2	25.2	12.6	403.8
	Species per farm	5.2	33.6	2.6	11.4	14.0	10.4	10.6	5.7	12.2	5.1	9.4	6.6	10.6

[1] "Individuals" here means adults that could be identified to species. Juveniles are excluded.

2.2.2 Correlations within indices of individual species groups

Once the species lists are available from field surveys, all five richness parameters (and others) can be derived. However, not all five indices need to be calculated. Indeed, in most BIOBIO case study regions and for the four species groups, the gamma richness, i.e. the total number of species pooled over the sampled habitats, was significantly correlated to the other four richness indices, i.e. alpha richness (B8_1.2), area weighted richness (B8_1.3), rarefied richness (B8_1.4), and Chao estimated richness (B8_1.5), as exemplified in Figure 10 for spider data from the case study in Wales. Therefore, investigations of relationships between indicator species groups and habitat and farm management indicators were further carried out with the gamma richness of the indicator species groups (value distribution of indices across the 12 case study regions and correlograms for each indicator species group and each case study region are available in Appendix 1 and Appendix 2, respectively).

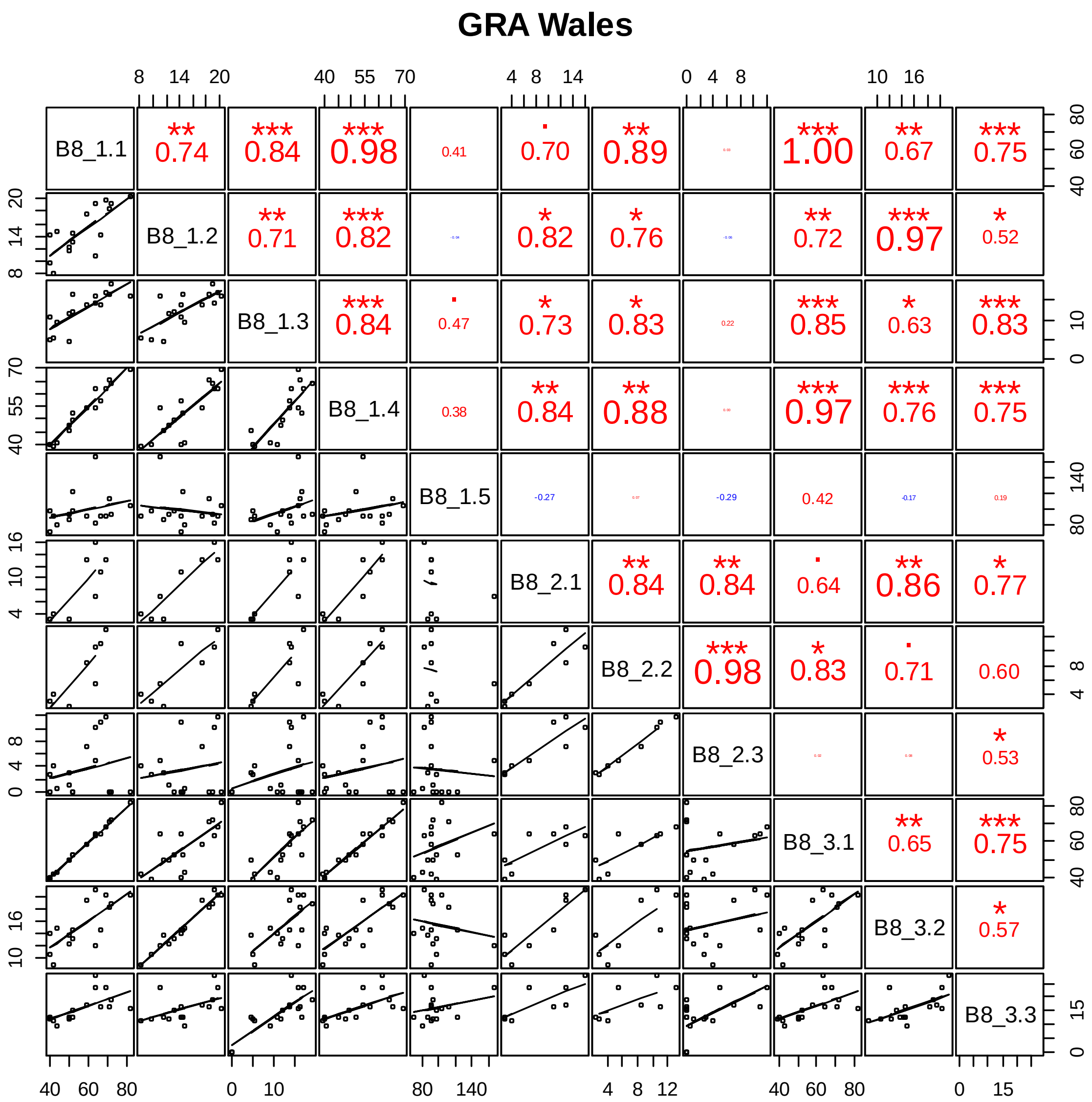


Figure 10. Correlogram of species diversity indices of spiders calculated from the species list with abundance. B8_1.1 = Gamma richness of spiders on farms, B8_1.2 = Alpha richness, B8_1.3 = Area weighted richness, B8_1.4 = Rarefied richness, B8_1.5 = Chao estimated richness, B8_2.1 = Gamma richness of spiders in cultivated forage and food crops, B8_2.2 = Alpha richness of spiders in cultivated forage and food crops, B8_2.3 = Area weighted richness of spiders in cultivated forage and food crops, B8_3.1 = Gamma richness of spiders in semi-natural habitats, B8_3.2 = Alpha richness of spiders in semi-natural habitats, B8_3.3 = Area weighted richness of spiders in semi-natural habitats.

Furthermore, indicator values were calculated for cultivated forage and food crops, and semi-natural habitats separately, i.e. gamma richness of spiders in cultivated forage and food crops (B8_2.1), alpha richness of spiders in cultivated forage and food crops (B8_2.2), area weighted richness of spiders in cultivated forage and food crops (B8_2.3), gamma richness of spiders in semi-natural habitats (B8_3.1), alpha richness of spiders in semi-natural habitats (B8_3.2), area weighted richness of spiders in semi-natural habitats (B8_3.3). This is especially important to analyse the relationship between species diversity and the management indicators which are mostly related to the cultivated habitats and less to the semi-natural habitats.

2.2.3 Gamma richness of species across farms and case studies

The following 4 figures show the gamma richness distribution over the 12 case study regions for the 4 species groups investigated. In the following graphs, the "Field crops & horticulture" case study regions are yellow, the "Specialist grazing livestock" case study regions are green, the "Mixed crops & livestock" case study region is blue and the “Permanent crops” case study regions are pink.

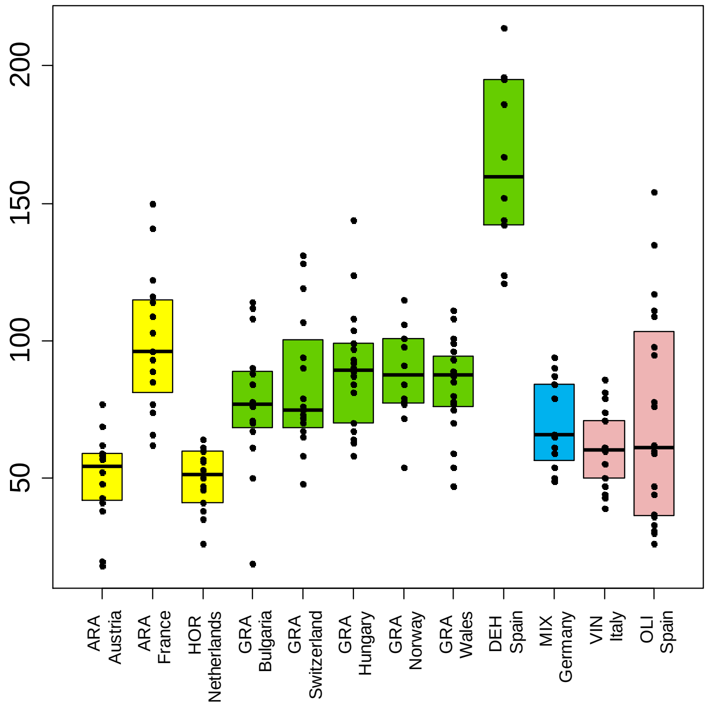


Figure 11. **Plant** gamma richness in farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

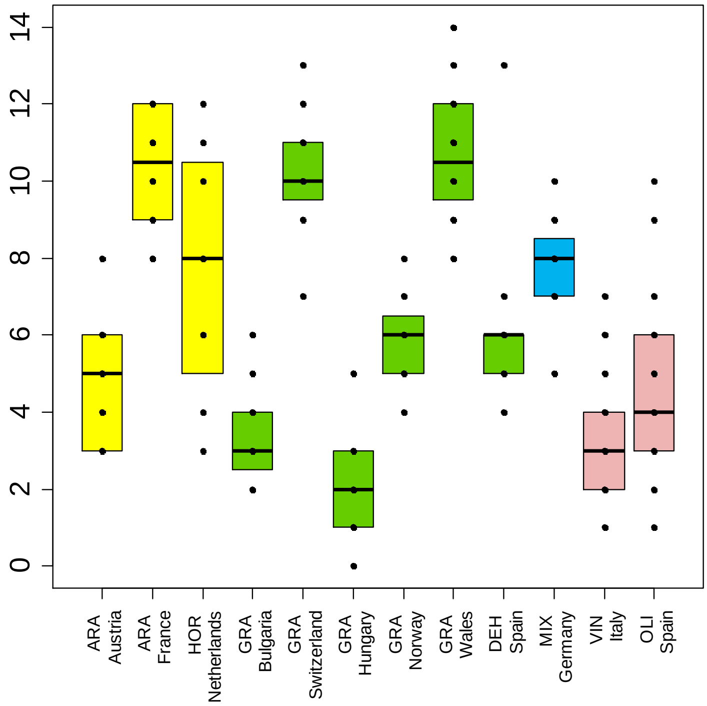


Figure 12. **Earthworm** gamma richness in farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

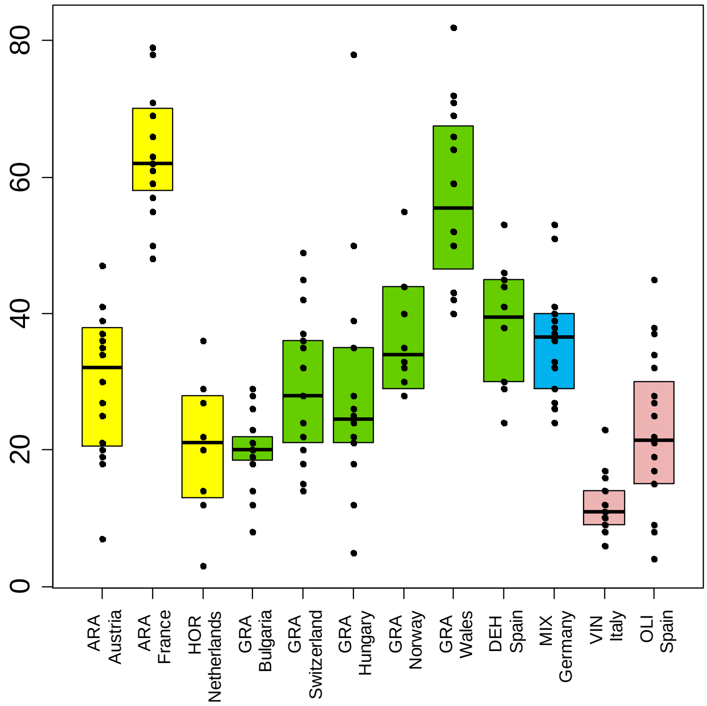


Figure 13. **Spider** gamma richness in farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

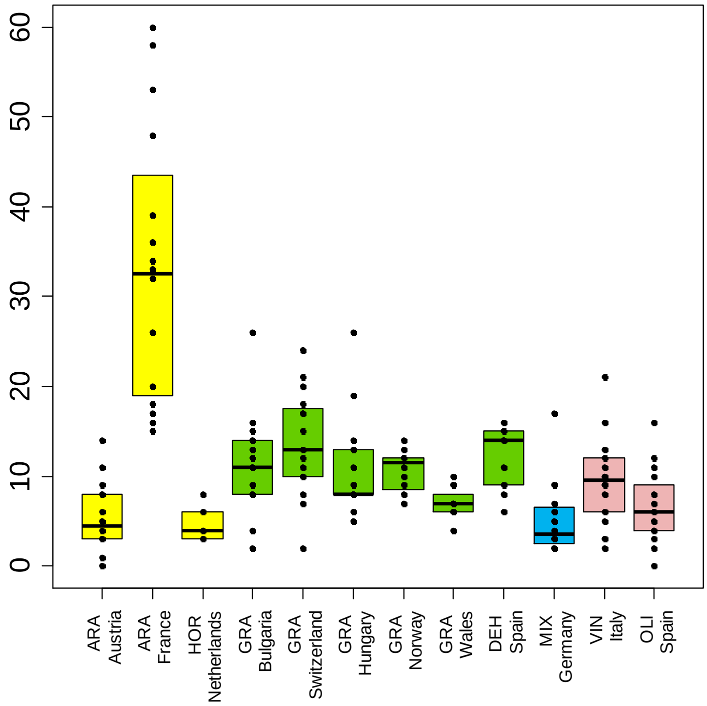


Figure 14. **Bee** gamma richness in farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

There were an exceptionally high number of plant species in the Spanish Dehesas (Figure 11), a high number of spider species in France and Wales (Figure 13), and a high number of bee species in France (Figure 14). However, comparison of species richness level across case study regions is certainly not the most relevant information that can be derived from Figures 11 to 14 but rather the variability among farms within case study regions. Indeed, for plants, richness ranged from 26 to 154 species in olive plantation farms in Spain, for instance. Spider species richness varied from 5 to 78 in the Hungarian case study and bee species showed high variation from 15 to 60 species in farms of the French case study.

2.3 Habitat diversity indicators

2.3.1 Overview

A habitat is an area with relatively homogeneous environmental conditions, occupied by plants and animals that are adapted to those conditions. On farmland, the term "habitat" is sometimes associated with "semi-natural habitats" or elements of "ecological infrastructure". However, wild species also occur in fields of crops and some wild species such as ruderal plant species or seed feeding birds are even specialized to these environmental conditions. Farm habitat indicators should therefore also relate to arable crop fields, sown and permanent grasslands, intensively managed vineyards and orchards, etc. At the other end of the spectrum, they must also account for less intensively managed parts of the farm such as marginal grasslands, hedgerows, grazed forest, etc. The BIOBIO method for measuring biodiversity indicators therefore starts with establishing a habitat map of the entire farm, i.e. the Utilized Agricultural Area (UAA) that is managed by a specific farmer. (In BIOBIO the definition of UAA is rather broad, including semi-natural habitats on the farm that may be excluded in e.g. national agricultural census data). The habitat mapping method follows the EBONE approach (http://www.ebone.wur.nl/), which has been adapted to farm scale mapping. The method of habitat/land use classification is based on a generic system of habitat definitions, so called General Habitat Categories. See Dennis et al. (2012) for a description of the method. Furthermore, in order to calculate the BIOBIO habitat indicators (see Bailey et al., 2012), the individual habitats were categorized. Detailed explanation on the habitat categorization is given in Appendix 4.

2.3.2 Indicator value distribution

18 habitat indicators have been tested in the BIOBIO case studies, most of them based on the habitat mapping. Eight of them have passed this test and the audit of the BIOBIO Stakeholder Advisory Board, and are presented in this section. Six indicators are not considered sufficiently developed for standard application and merit further research and testing. The remaining four indicators have been discarded either because they cannot be reliably measured or because they yield information that is redundant when other habitat indicators are being employed. See Herzog et al. (2012b) for an overview on all indicators which were tested in the case study regions.

Several of the habitat indicators are normalized over the farm size, i.e. calculated on a per hectare basis, in order to correct for the tendency that the number of habitats will tend to increase with farm size (Figure 15). The relationship was usually positive, and significant ($p < 0.05$) in 4 case study regions, i.e. Bulgaria (t value = 2.3), Switzerland (t value = 4.1), Germany (t value = 3), olive plantations in Spain (t value = 6.8). These indicators cannot be calculated for farms which are smaller than 1 ha as this would lead to an increase of the indicator values due to simple mathematical reasons.

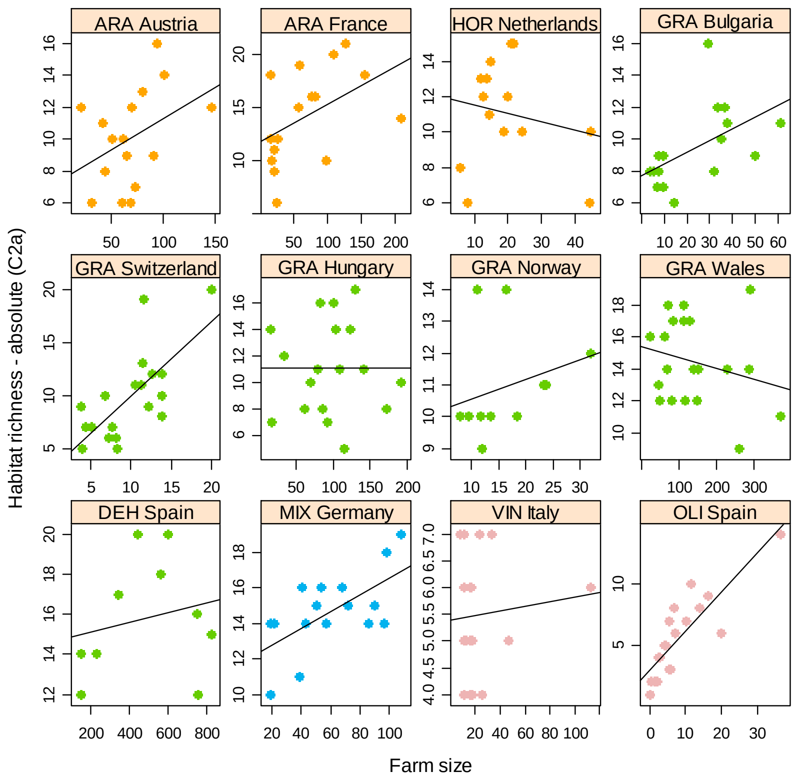


Figure 15: Relationship between the habitat richness (number of habitats per farm) and the farm size (ha) in 12 BIOBIO case study regions.

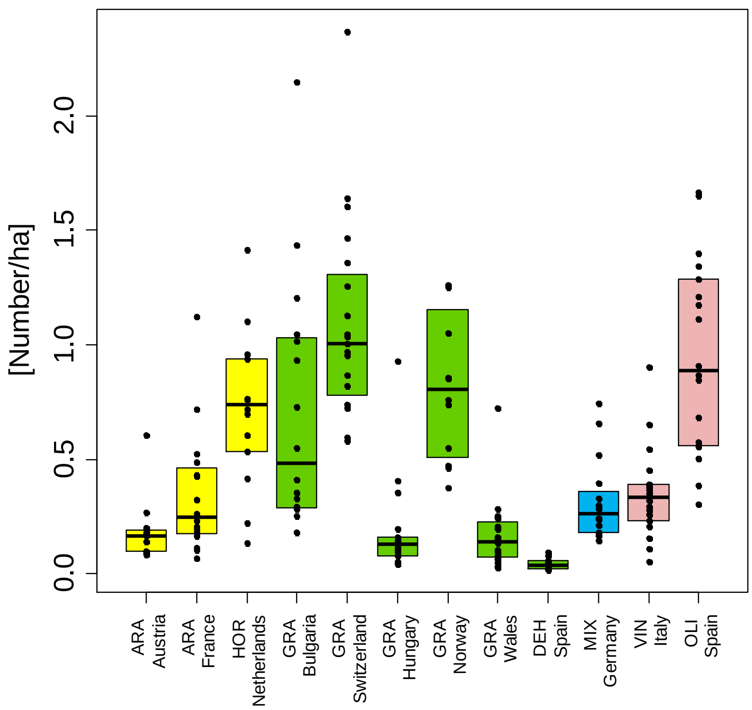


Figure 16: Habitat richness (number of habitat types per ha) in farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

The variability in habitat richness among farms is usually high within case study regions, e.g. in Bulgaria where the number of habitat types per ha varied from 0.2 to 2.1 (Figure 16). The Dehesas in Spain are quite habitat poor with a maximum value at 0.09 habitat types per ha in the richest farm. However, it must be emphasized that this indicator relies on values calculated per hectare. This shows the density of habitat types in farms but not the absolute habitat richness. To complement the information of Figure 16, the absolute values of habitat richness are also provided (Figure 17).

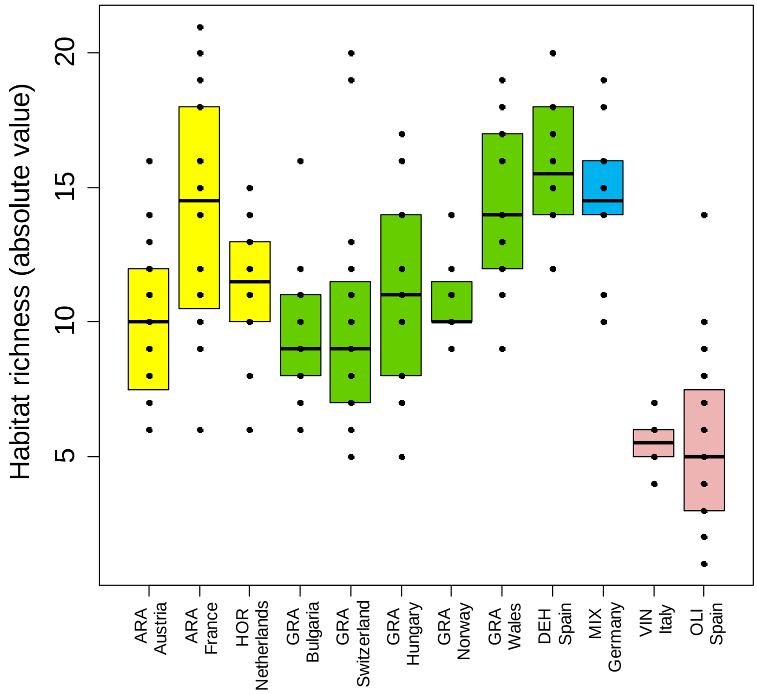


Figure 17: Habitat richness in absolute value in farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

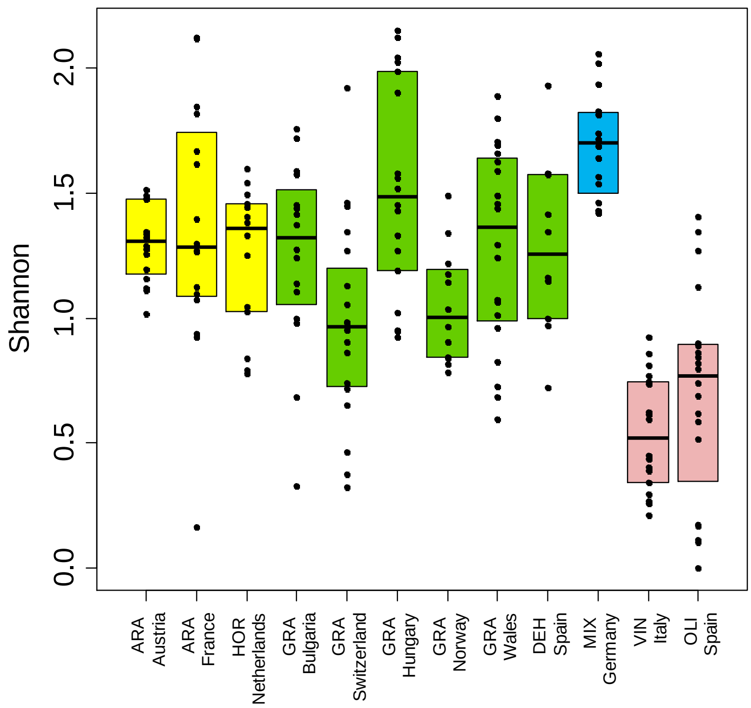


Figure 18: Habitat diversity (Shannon index) in farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

Comparison of Figure 16 and Figure 17 shows that the Dehesas in Spain are among the richest farms in terms of absolute number of habitat types, but due to the large size of the farms (ranging from 150 to 827 ha), the habitat density per hectare was very low. In contrast, the farms of the Swiss case study were among the poorest in terms of absolute number of habitat types, but the habitat density per hectare is high because farms were relatively small (ranging from 4 to 20 ha).

Taking into account the area of habitats belonging to each habitat type in farms by calculating the Shannon value (Figure 18), the case study in France showed the highest variability among farms, with a farm among the lowest Shannon values, i.e. highly dominated by very few habitat types (Shannon value = 0.2), and a farm among the highest values, i.e. having a balanced area of habitats in each type (Shannon value = 2.1).

The indicator Crop Richness (Figure 19) shows the number of crops cultivated per hectare of farm area. This is similar to the Habitat Richness indicator, but is restricted to frequently disturbed farm habitats, including arable, fruit and vegetable crops. The indicator provides a measure of the degree of specialisation on the farm. Generally, the farms of the European case studies had very low numbers of crops per hectare. One olive farm in Spain stands out as having a much higher crop richness than all other farms, with almost three crops per hectare.

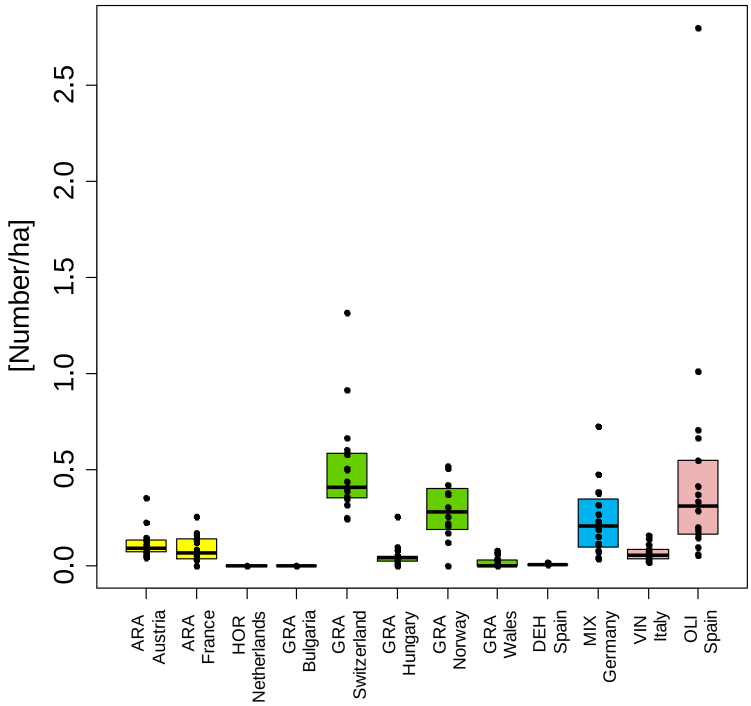


Figure 19: Crop richness in farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

The indicator Tree Cover shows, not surprisingly, that the case study regions with farming systems vineyard (Italy), olive plantations and Dehesas (Spain) are most dominated by trees (Figure 20). The case studies in Wales and The Netherlands showed very high variability in tree cover amongst farms.

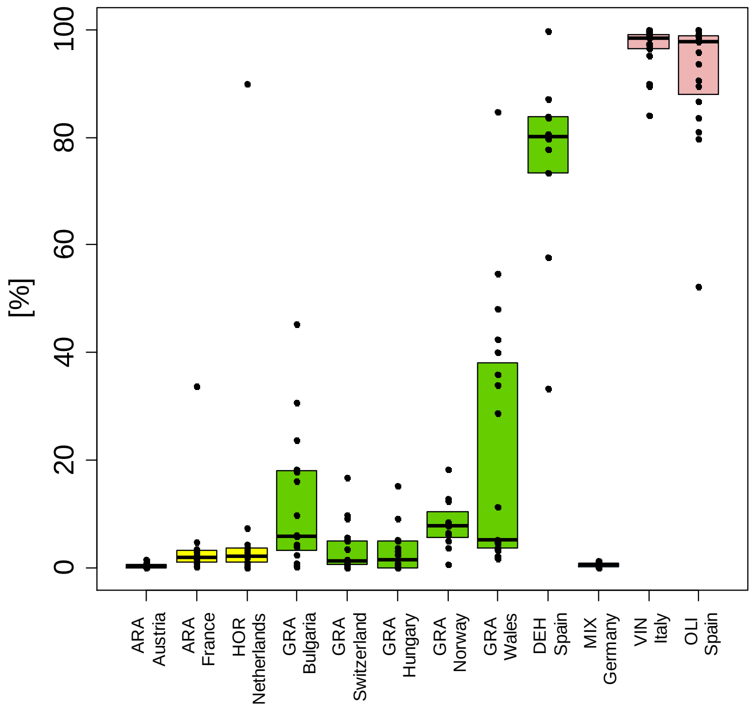


Figure 20: Tree cover in farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

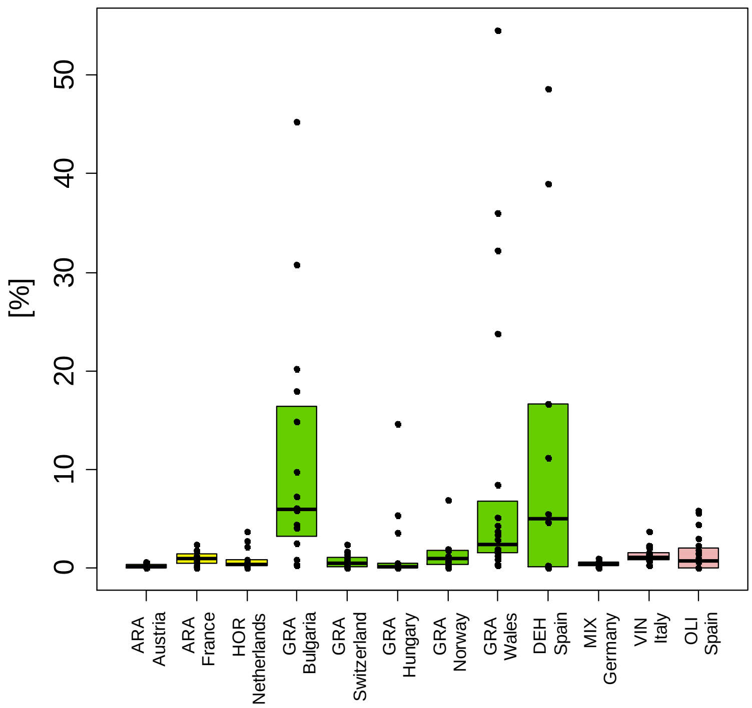


Figure 21: Shrub cover in farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

Most of the BIOBIO farms do not have a high cover of shrubs (Figure 21) with the exception of the case studies in Bulgaria, Wales and the Dehesas where shrubs covered 30% to 50% of some farms.

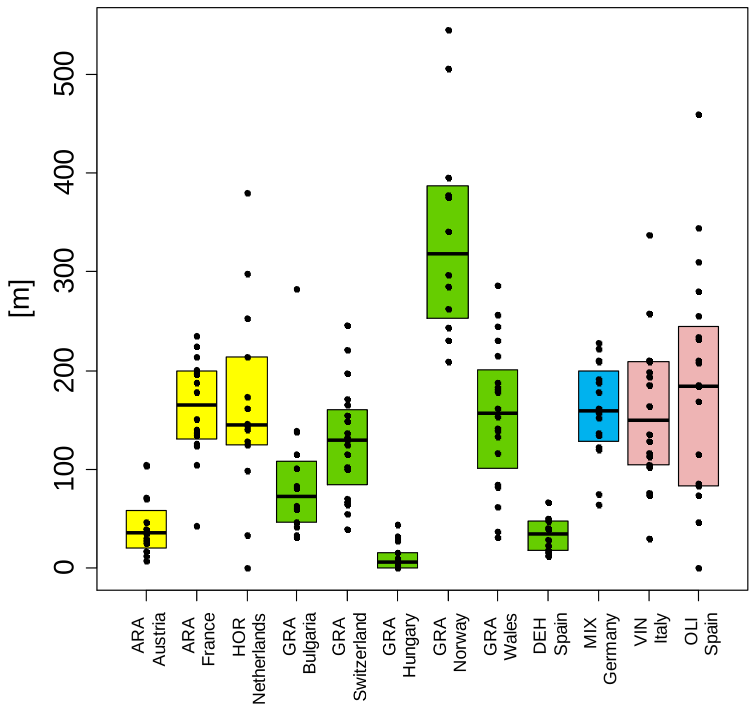


Figure 22: Length (metres) per hectare of linear elements in farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

The champion case study for the linear elements in farms is without doubt Norway with all farms with more than 200 m per hectare (Figure 22). Most of the linear features there were herbaceous strips.

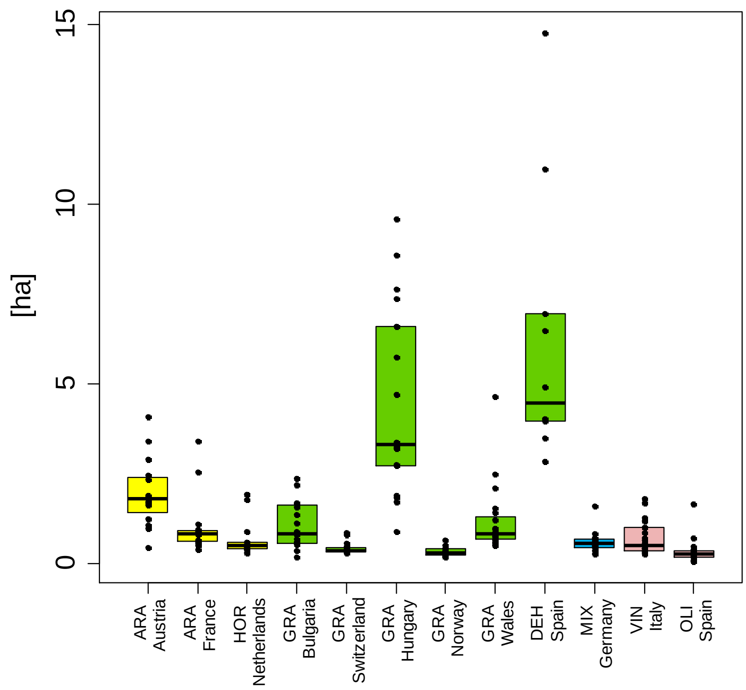


Figure 23: Average size of habitat patches (hectares) on the farm in farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

Habitat patch size on average showed low variability among and between the BIOBIO case studies (Figure 23), except in Hungary and the Dehesas where some farms have patches of approaching 10 or even more than 10 hectares on average.

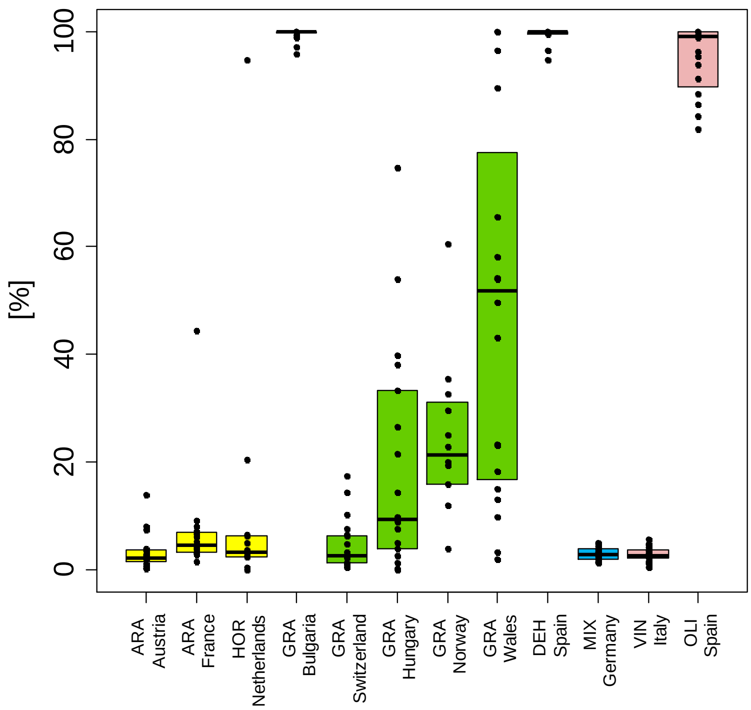


Figure 24: Cover of semi-natural habitats in farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

Perhaps one of the most interesting habitat diversity indicators is the cover of semi-natural habitats in farms, since semi-natural habitats generally support a much greater biodiversity than cultivated habitats. Figure 24 shows that the case study regions in Bulgaria and the Dehesas were nearly entirely considered semi-natural, followed by the olive plantations in Spain which had very high proportion of semi-natural habitats. The case study in Wales showed a very highly variable cover of semi-natural habitats in farms, ranging from 2% to 100%.

2.4 Farm management indicators

2.4.1 Indicator value distribution

BIOBIO farm management indicators are conceptualized as indirect indicators for biodiversity. They reflect the level of farming intensity and, thus, the major impact exerted on habitats and species in agricultural landscapes. By shaping the habitat structure, farming methods determine the conditions for the diversity of species that is assessed by direct indicators of biodiversity, mainly on the managed area of the farms. Therefore, farming practices are key drivers to maintain and restore biodiversity. Eight management indicators relating to energy and nutrient input, pesticide applications and disturbance by mechanical operations have been identified and represent different categories of pressure indicators. The management indicators are extensively described in Herzog et al. (2012a), see Herzog et al (2012b) for an overview on discarded indicators. Generally, the pressure on biodiversity rises with increasing values of farm management indicators. They signify increased nutrient input to the farm land, progressive mechanization of farm operations, more frequent pesticide applications or higher livestock densities on the farm.

With respect to the proportion of the farm area on which mineral fertilizer was used, farms ranged from 0 to 85% at least (Bulgaria) in all case study regions except in the grassland farms in Hungary and in the Dehesas in Spain where all farms were under 20% (Figure 25). However, in these case study regions as well as in the Netherlands, Bulgaria and Switzerland, farms that used mineral fertilisers were exceptions. With a median value of 50%, the mixed farming system in Germany had the highest proportion of area with use of mineral fertilizers.

The total nitrogen input is an important indicator of the management intensity of farms. In BIOBIO, the highest level was observed in farms of the mixed farming and grassland system of Germany and Switzerland, respectively, while the lowest occurred in the vineyards of Italy, and the olive plantations of Spain (Figure 26). The highest variability was observed among farms of the horticulture case study in the Netherlands, with the lowest value of nearly 0 and the highest value of nearly 400 kg N/ha UAA. Farms in the Welsh case study were also very variable, with values ranging from 0 to 327 kg N/ha UAA. Farms in Hungary (organic and non-organic) had lower values of total nitrogen input than the “low input farms” in Bulgaria.

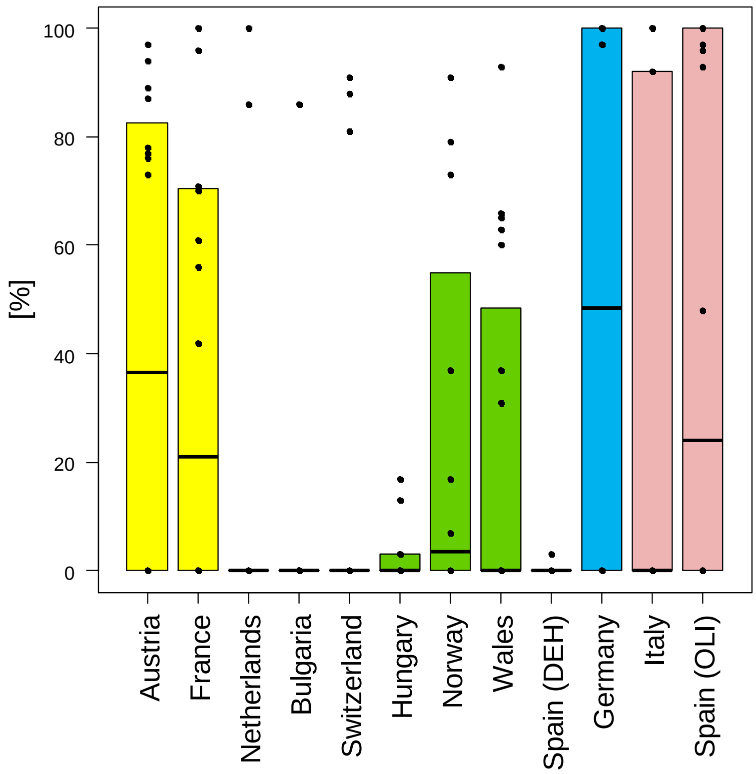


Figure 25: Proportion of the farm area (%) with use of mineral N-fertiliser in the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

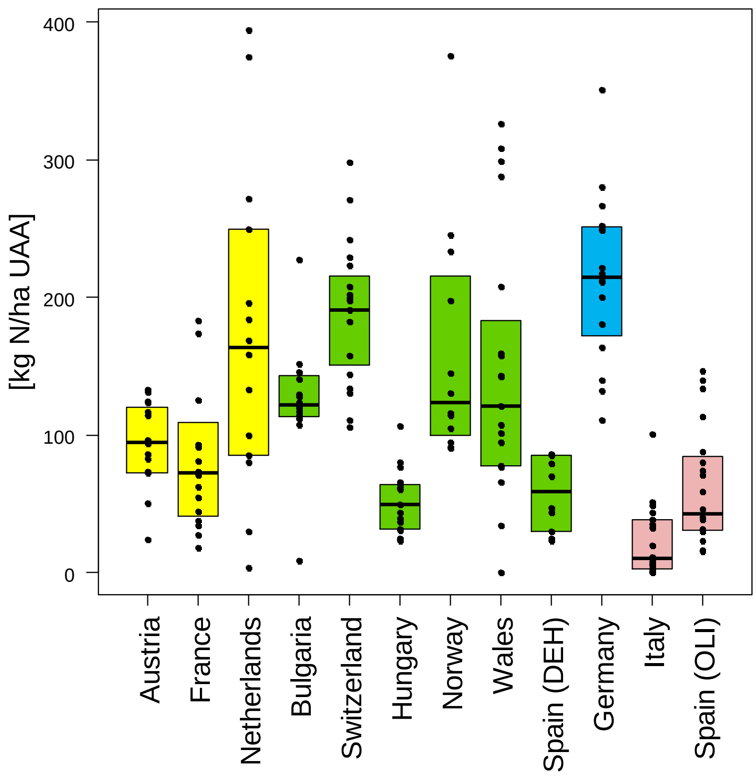


Figure 26: Total nitrogen input per hectare on the farms of the 12 BioBio case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

With respect to the energy input, the highest level of consumption was reached in the horticultural farms of the Netherlands and the vineyards of Italy (Figure 27). On vineyard farms in Italy, energy input needed for wine production was included. The lowest energy farms were found in the Dehesas in Spain, and in Hungary and Wales.

For the indicator of intensification/extensification (expenditure on fertiliser, crop protection and concentrate feedstuffs) there were outlier farms in the Netherlands and in Bulgaria with exceptionally high expenditure compared to the average farm of the BIOBIO case study regions (Figure 28). The high expenditure correlated with the high number of pesticide applications on three of the farms of the Netherlands case study and the nitrogen input on farms of Bulgaria. The lowest cost farms were found in the Dehesas in Spain as well as in Hungary and Wales, illustrating the significant positive correlation between costs and energy consumption occurring in 9 of the 12 case study regions.

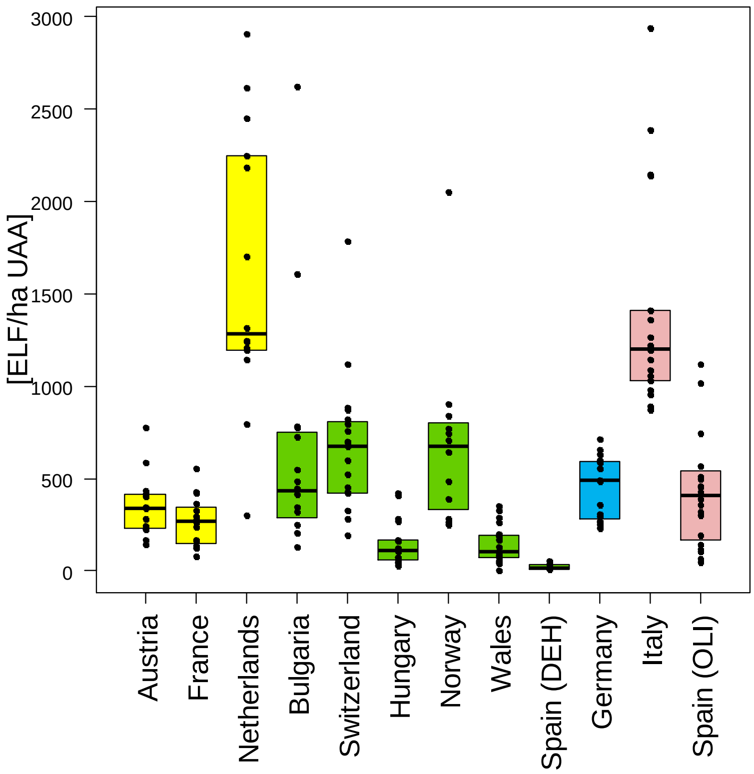


Figure 27: Total direct and indirect energy input (Equivalent Litre of Fuel on the farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

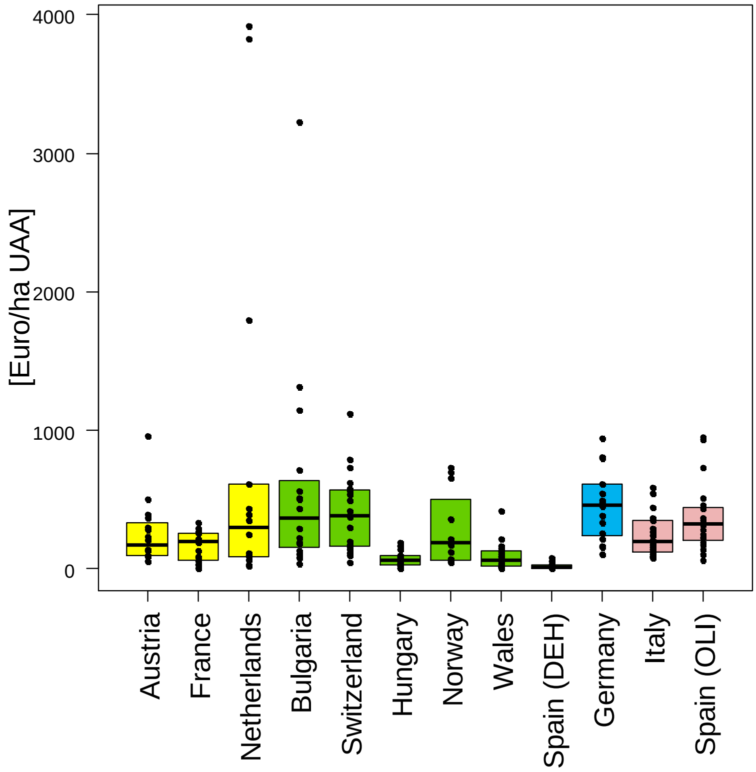


Figure 28: Intensification/extensification: expenditure on fertiliser, crop protection and concentrate feedstuffs on farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

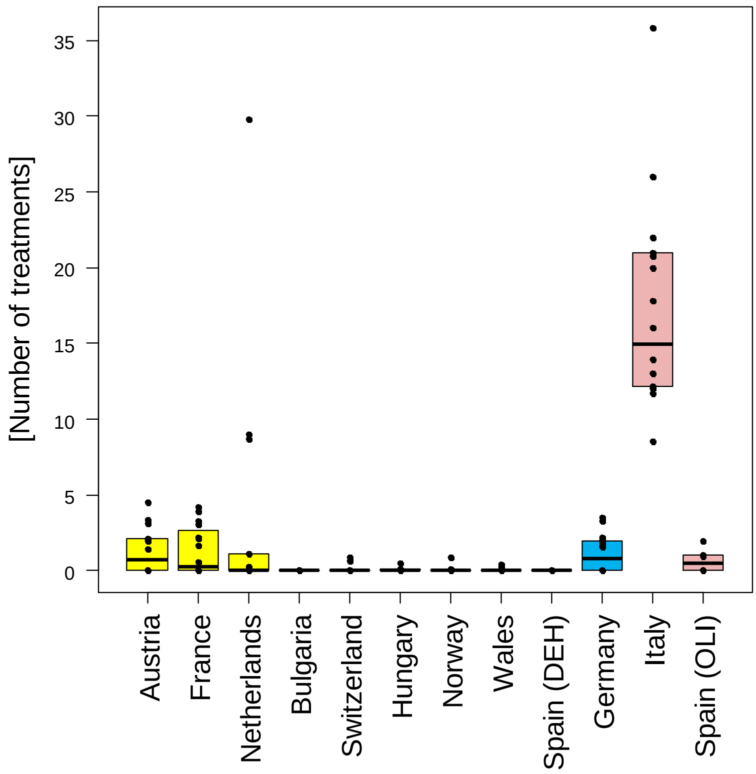


Figure 29: Pesticide use on farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

The pesticide use indicator (number of pesticide treatments) showed clearly the high to very high level of intensity in vineyards of farms in Italy and on some farms of the horticulture case study in the Netherlands (Figure 29). Pesticides were also commonly applied in the case study regions with mixed farming (Germany), arable crops (France, Austria) and olive plantations (Spain). Pesticides were only marginally used in grassland case study regions, i.e. in Bulgaria, Switzerland, Norway, Wales and the Dehesas in Spain.

The number of field operations was the highest in the vineyards of the Italian case study, followed by the mixed farms of Germany, and grassland farms of Switzerland. The latter was an exception among the grassland case study regions, which generally had a lower number of field operations (Figure 30). The low input case study regions of Bulgaria and the Dehesas in Spain demonstrated the lowest values for this indicator. More specific information on pesticide use and field operations can be derived from the sub-indicators "Herbicide use", "Fungicide use", "Insecticide Use" and "Mowing frequency", "Mowing time" and "Ploughing", respectively.

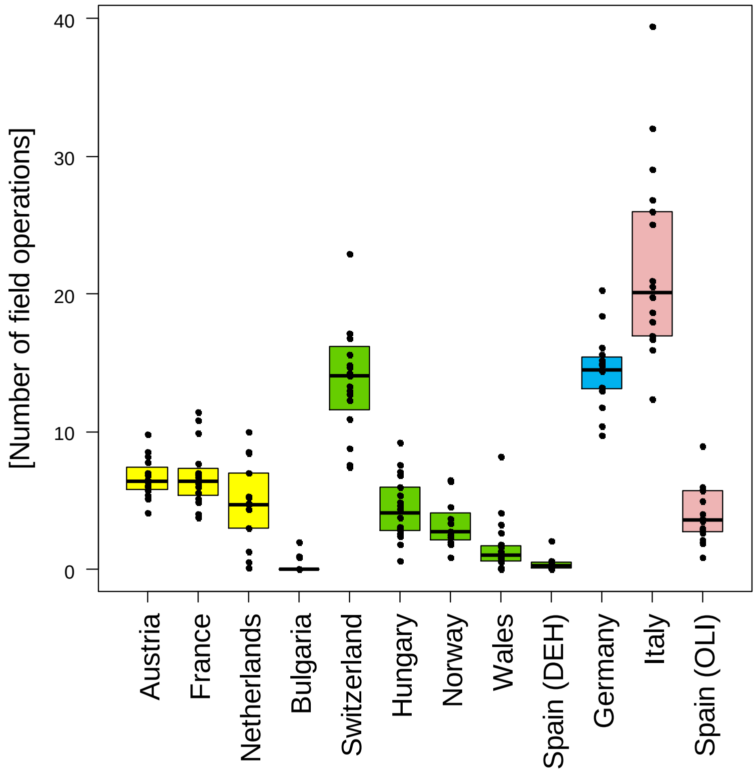


Figure 30: Field operations on the farms of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

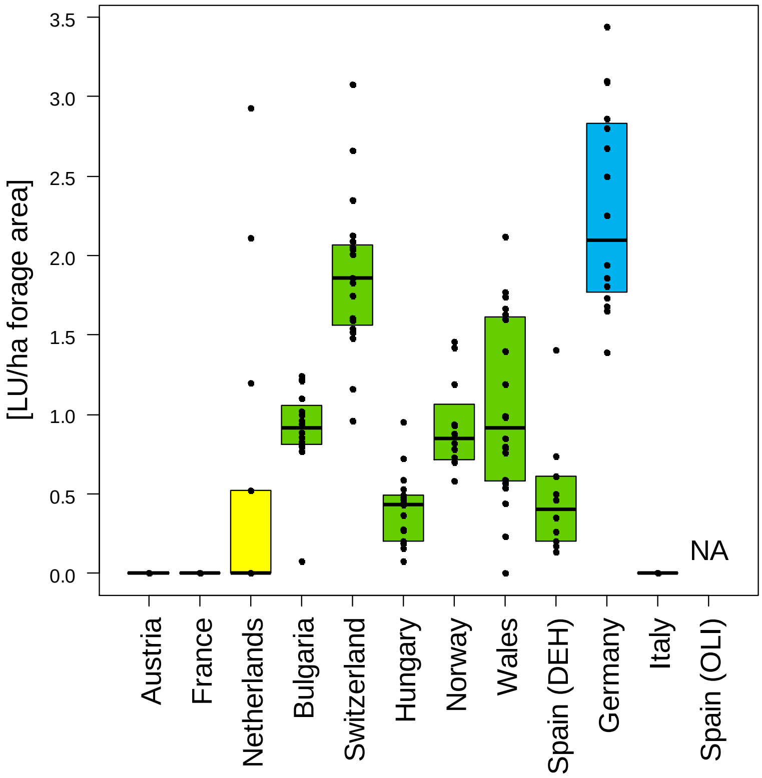


Figure 31: Average stocking rate (Lifestock Unit) per hectare forage area of the 12 BIOBIO case study regions. Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

In case studies with livestock, the mixed farms in Germany showed the highest level of intensity, followed by the grassland farms in Switzerland (Figure 31). Interestingly, farms in Bulgaria did not demonstrate the lowest average stocking rate despite considered low input. The highest variability occurred in farms of the horticulture case study in the Netherlands.

The grazing intensity is an important indicator in the grassland case studies because it reflects part of the overall management intensity. However, it is difficult to estimate in some cases as described below. The Dehesa farms were more intensively grazed than the other grassland case study regions (Figure 32). However, the range of grazing intensity was rather high in all case study regions. The situation for Norway was complicated by the fact that large areas of common grazing land are used in the summer. These areas were not included in BIOBIO, so the stocking rate on the farm applies only for the spring and autumn, when the sheep are on the inbye land. In the case study of Wales, this indicator could not be measured because only the overall average stocking rate per hectare of forage area could be obtained from farmers.

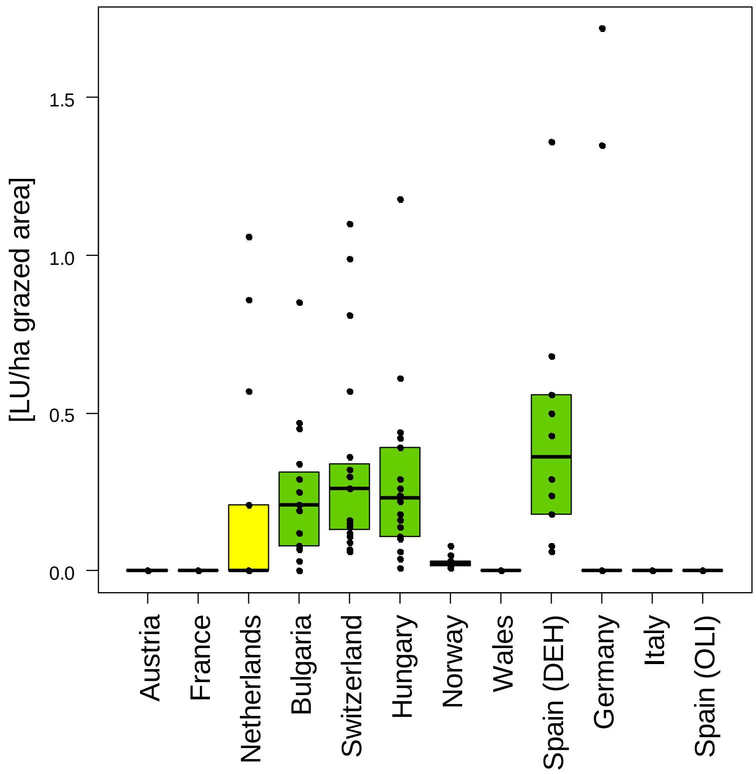


Figure 32: Grazing intensity, expressed as Livestock Units per hectare of grazed land, on the farms of 11 BIOBIO case study regions (this indicator could not be calculated for Wales). Boxplot with lower quartile (25% of the farms), median (50% of the farms), upper quartile (75% of the farms), and individual farms (black dots).

3 Testing for correlations within and between indicator groups

Several candidate indicators examined in BIOBIO were discarded at an early stage because they could not be measured and calculated in all case study regions (Herzog et al. 2012b). Examples of these are indicators based on Ellenberg values, which are not available for all European plants, and the 'permanent grassland' indicator, because in many instances neither the farmers during the interviews nor the field staff during habitat mapping were able to distinguish reliably between permanent and sown grassland. This chapter deals with analyses of the remaining candidate indicators – those that could be accurately measured - to test whether they provide useful information. For indicators of farmland biodiversity to be useful, they should be able to detect real differences between farms, they must reflect some relevant aspect of biodiversity and, to be cost effective, there should not be too much overlap in the information provided by different indicators.

In searching for indicators, a relevant question concerns the representativeness of one or the other indicator for the whole set. In the case of species groups, surrogates may be detected that indicate the diversity of other groups. For example, plants are primary producers and also structure the physical environment, thus providing both food and various ecological niches for faunistic groups. It is therefore often expected that plant diversity may be representative for (i.e. be a good indicator of) wider biodiversity.

The consequence for monitoring programs is that if two indicators correlate, they provide the same information and measuring only one of them is sufficient. We therefore tested for correlations within and between indicator groups. Within the four indicator groups (habitat, species, genetic diversity and farm management), correlating indicators were identified first. In a second step, correlations across the indicator groups were investigated, in particular between management, habitat and species diversity indicators. These correlations may indicate relationships which help us to understand the biodiversity findings, and in the case of strong and consistent correlations across case study regions, surrogate indicators may be identified.

The correlations among the four species groups in the BIOBIO case studies were very diverse with no general pattern. Correlations among all four species groups ranged from most of them being highly significant (Austria, Switzerland, olive plantations in Spain) to a few (France, Bulgaria, Germany), to one or no significant correlations (The Netherlands, Hungary, Norway, Wales, Dehesa in Spain, Italy). One example of each case is illustrated in Figure 33.

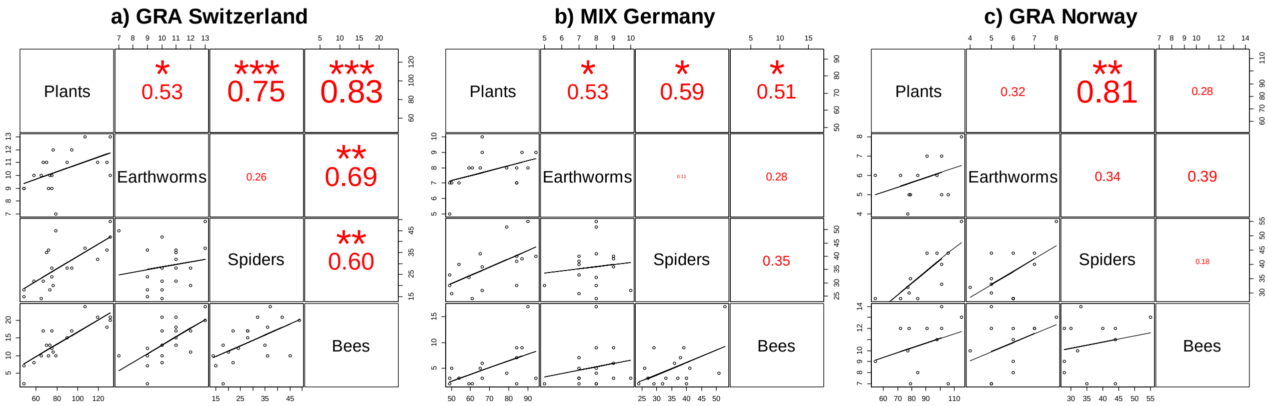


Figure 33: Spearman correlations of the gamma richness in farms of the four species indicator groups in (a) the grassland case study region in Switzerland, (b) the mixed farming region of Germany and (c) the grassland case study in Norway. Relationships between indicators are graphically shown below the diagonal. Correlation coefficients with significance (stars) are given above the diagonal. The font size is proportional to the coefficient value.

Farm-scale indicators

Because species diversity indicators were measured in plots (from the different habitat types found on the farm), and other indicators – genetic, habitat diversity and farm management indicators – were recorded for the entire farm, the species diversity measurements had to be scaled up to the farm level in order to yield a comprehensive and consistent indicator set.

	Level of measurement	**Indicator level**
Habitat diversity	Farm	Farm
Species diversity	Plot	Farm
Genetic diversity	Farm	Farm
Farm management	Farm[1]	Farm

[1] farming practices were also recorded at plot level in habitats surveyed for species.

There are various possible ways to upscale species diversity to the farm scale. 'Gamma richness', the species richness index selected, yields the total number of species on the farm aggregated over the sampled habitats. The unit of measurement is then the number of species per farm. By contrast, habitat diversity and farm management indicators, e.g. the number of habitat types (HabRich) and the number of livestock units (AvStock), are usually expressed on a per-hectare basis. Calculating values per hectare has the advantage of allowing comparisons between farms of various sizes, in particular when indicator values are dependent on farm size. We may, for example, expect habitat richness to depend on farm size, i.e. larger farms may tend to consist of a greater number of different habitat types (and thus also possess greater species richness) but see section 2.3.2.

Great care must be taken when selecting which data to use for analysis of correlations between species diversity, habitat diversity and farm management. Expressing gamma richness per hectare is an unsuitable approach, since it would require an extrapolation (due to the standard methods used to assess the species indicators on selected plots). Moreover, when calculating correlations between two indicators which are adjusted for farm size, i.e. where values are divided by the size of the farm to obtain values per hectare, correlation coefficients are artificially increased. For this reason, absolute 'per farm' values were used to analyse correlations across indicator groups (species, habitat, management). Some farming practices, e.g. pesticide use, were also measured at plot level and could therefore be directly related to the species diversity at plot level. In parallel, they could be scaled up to farm level by calculating area-weighted means.

A question which arises by investigating species richness at farm level is whether larger farms in area are species richer than smaller ones. This relationship can eventually occur when larger farms show higher habitat richness (but see section 2.3.2), and in particular a higher richness in semi-natural habitat types (see also section 5).

The plant gamma richness correlated only significantly with the farm size in the olive plantations in Spain ($t = 4.01$, $p < 0.05$, Figure 34).

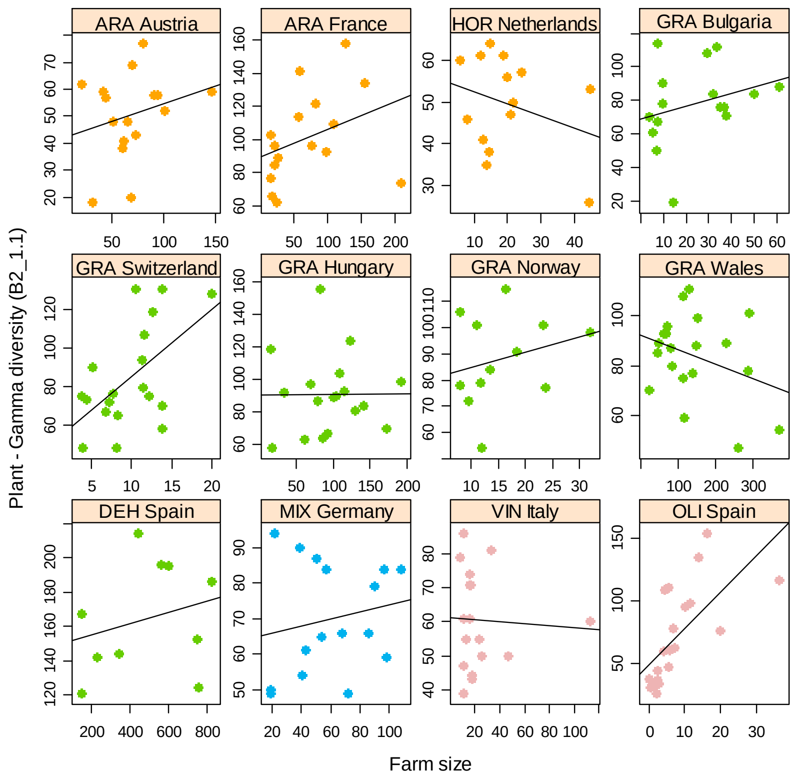


Figure 34: Relationship between plant gamma richness and farm size (ha) in farms of 12 BIOBIO case study regions.

Earthworm gamma richness correlated significantly but negatively with farm size in the Dehesas in Spain (t = 45.2, $p < 0.05$, Figure 35).

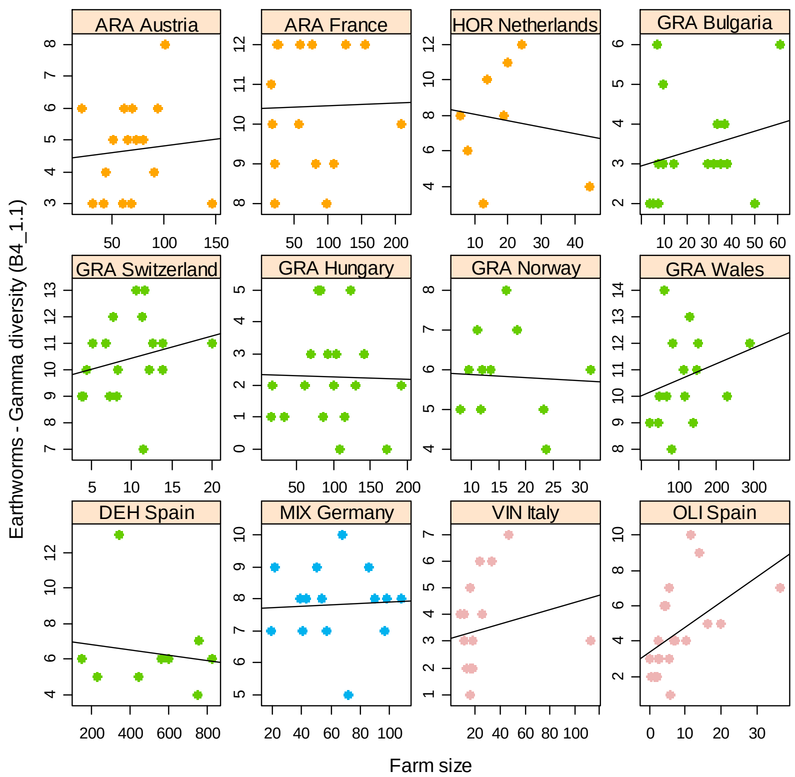


Figure 35: Relationship between earthworm gamma richness and farm size (ha) in farms of 12 BIOBIO case study regions.

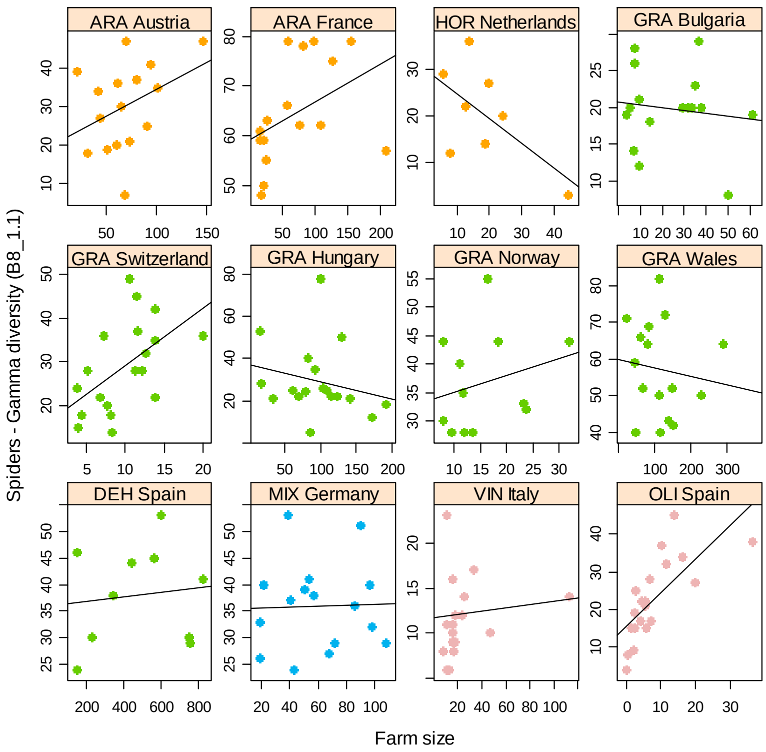


Figure 36: Relationship between spider gamma richness and farm size (ha) in farms of 12 BIOBIO case study regions.

The spider gamma richness correlated in none of the case study regions significantly with the farm size (Figure 36). The bee gamma richness correlated with the farm size only in the olive plantations in Spain ($t = 4.7$, $p < 0.05$, Figure 37).

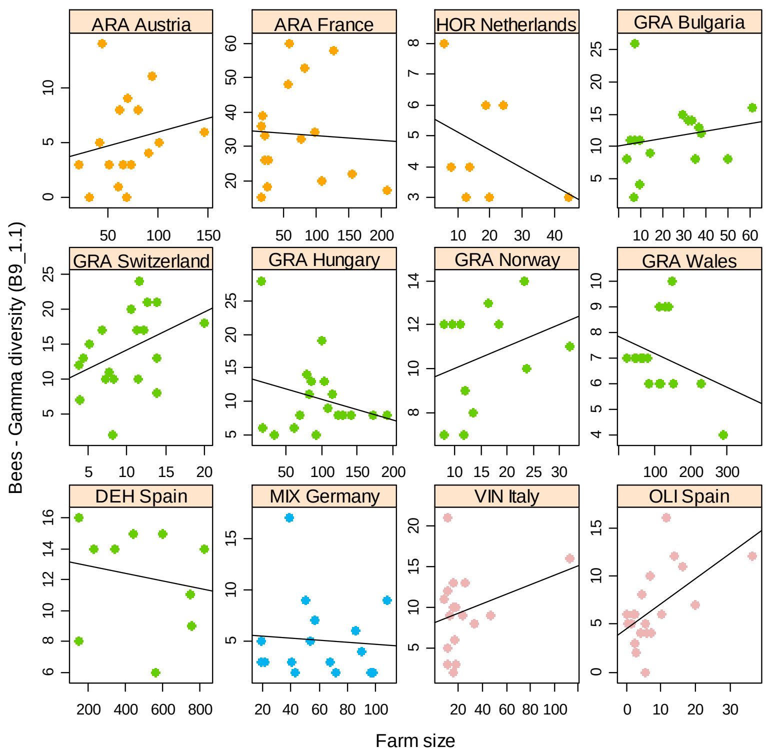


Figure 37: Relationship between bee gamma richness and farm size (ha) in farms of 12 BIOBIO case study regions.

3.1 Arable and horticultural case studies – Austria, France and The Netherlands

No general pattern could be derived for the arable case studies on the basis of the correlation analysis between the selected indicators. Nevertheless, there was a degree of similarity, e.g. the number and abundance of different cultivars per crop species (genetic diversity indicator A4_1) correlated positively with the number of pesticide applications (D9). However, while more varieties on average in farms could not be linked to less pesticide use in France as expected (see also section 2.1.2), there was no significant relationship in the Austria case study when considering the non-organic farms only. The management indicators correlated significantly with each other (e.g. arable case study in Austria, Figure 38). In Austria, all species indicators correlated positively with one another, while in France only plant species richness correlated positively with bee and spider species richness.

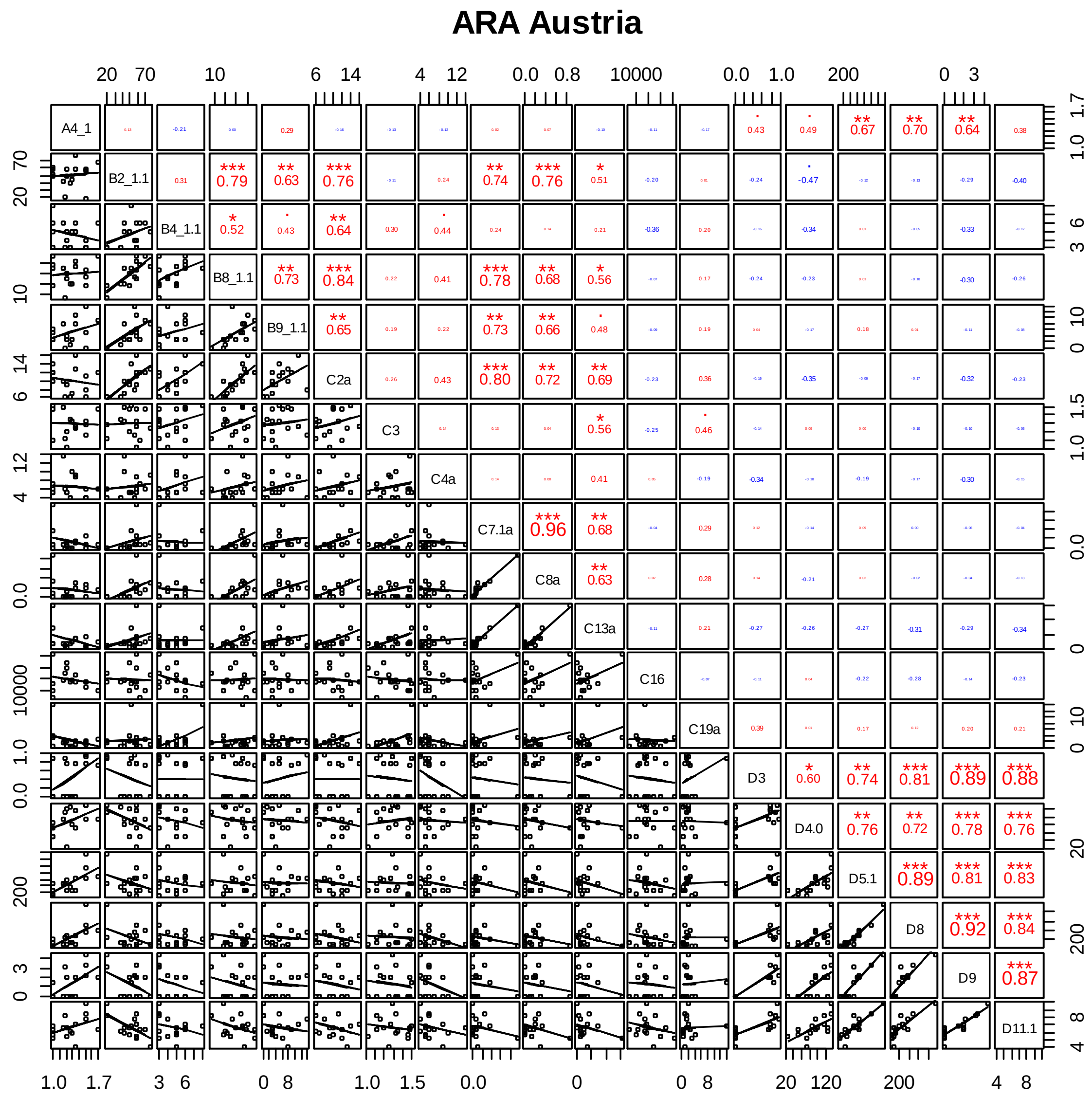


Figure 38: Example of Spearman correlations of BIOBIO indicators in the Austrian arable case study. The relationship between indicators is shown graphically below the diagonal. Positive (red) and negative (blue) correlation coefficients with significance (stars) are given above the diagonal. Font size is proportional to coefficient value.

A4_1 *Number of different cultivars per crop species (cultivar diversity)*

B2_1.1 *Plants: Flowering plants of farmland habitats (gamma diversity)*

B4_1.1 *Earthworms: Earthworms of farmland habitats (gamma diversity)*
B8_1.1 *Spiders: Spiders of farmland habitats (gamma diversity)*
B9_1.1 *Bees: Wild bees, domestic bees and bumblebees of farmland habitats (gamma diversity)*
C2a *Habitat richness (number of habitat types on farms)*
C3 *Habitat diversity*
C4a *Crop richness*
C7.1a *Tree cover*
C8a *Area of farmland with shrubs*
C13a *Length of linear elements*
C16 *Average size of habitat patches on the farm*
C19a *Area of semi-natural habitats*
D2.2 *Average stocking rate per ha forage area*
D3 *Area with use of mineral N-fertiliser*
D4.0 *Total nitrogen input*
D5.1 *Total direct and indirect energy input*
D8 *Intensification/extensificatio:n expenditure on fertiliser, crop protection and concentrate feedstuffs*
D9 *Pesticide use*
D11.1 *Field operations*
D12.1 *Grazing intensity*

Furthermore, most of the habitat diversity indicators correlated significantly with each other in France, but less so in Austria and The Netherlands. In addition, the number of habitat types (C2a) on farms in Austria had strong positive correlations with species indicator richness, i.e. plants (B2_1.1), earthworms (B4_1.1), spiders (B8_1.1) and bees (B9_1.1), whilst this pattern did not occur in France. In both case studies, management indicators did not correlate with species indicators. In the horticultural case study, patterns of correlation were similar to those of the arable case studies, but usually weaker except in the case of bees, for which diversity correlated with management indicators. The number of pesticide applications (D9) could be taken as a surrogate for the other management indicators and for the number of cultivars per crop species (A4_1) in arable case studies. Because contrasted results do not allow surrogates to be proposed either for the species richness or for the habitat diversity indicators, however, no surrogate can be proposed between the indicator groups.

3.2 Grassland case studies – Bulgaria, Switzerland, Norway, Hungary and Wales

The general pattern in grassland case studies showed a few significant and weak correlations among indicator groups. Where significant correlations occurred, they were mainly within indicator groups, i.e. within species, habitat and management indicators. For example, the species richness of vascular plants (B2_1.1) in Switzerland was significantly positively correlated with the three other species indicator groups (Figure 39), but this was not a general pattern in Bulgaria, Hungary and Norway. In Wales, the correlations within indicator groups were sometimes significant, and some habitat diversity indicators were correlated with species indicators as for Switzerland, where the number of habitat types (C2a) could be taken as a surrogate for the species richness (except for earthworms). Based on these results, however, no surrogate or additional elimination of indicators can in general be proposed for grassland case studies, as each indicator represents specific conditions.

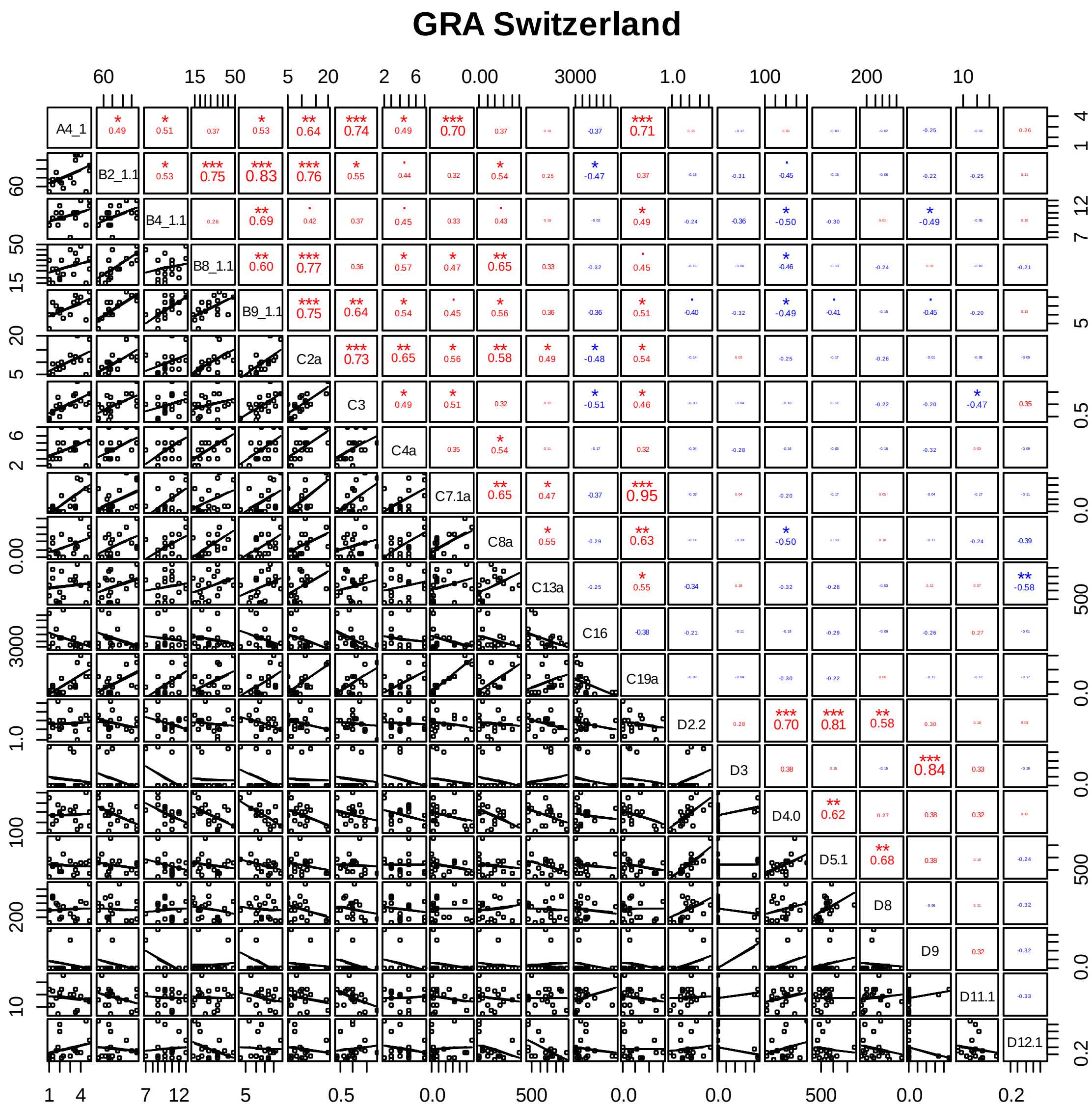


Figure 39: Example of Spearman correlations of BIOBIO indicators in Switzerland's grassland case study. The relationship between indicators is shown graphically below the diagonal. Positive (red) and negative (blue) correlation coefficients with significance (stars) are given above the diagonal. Font size is proportional to coefficient value. See Figure 38's legend for abbreviations.

3.3 *Dehesa* case study – Spain

In the Dehesa case study, significant correlations occurred between the management and habitat diversity indicators (Figure 40). In particular, the proportion of farmland with shrubs (C8a) significantly and negatively correlated with the average stocking rate (D2.2) and the grazing intensity (D12.1), whilst the share of semi-natural habitats (C19a) was negatively correlated with total nitrogen input (D4.0). With increasing numbers of livestock, there were fewer linear elements per hectare. Species indicators weakly correlated to each other and did not significantly correlate to any other indicators. Thus, a number of habitat diversity indicators could be represented by management indicators in Dehesas with the advantage that they are easier to collect.

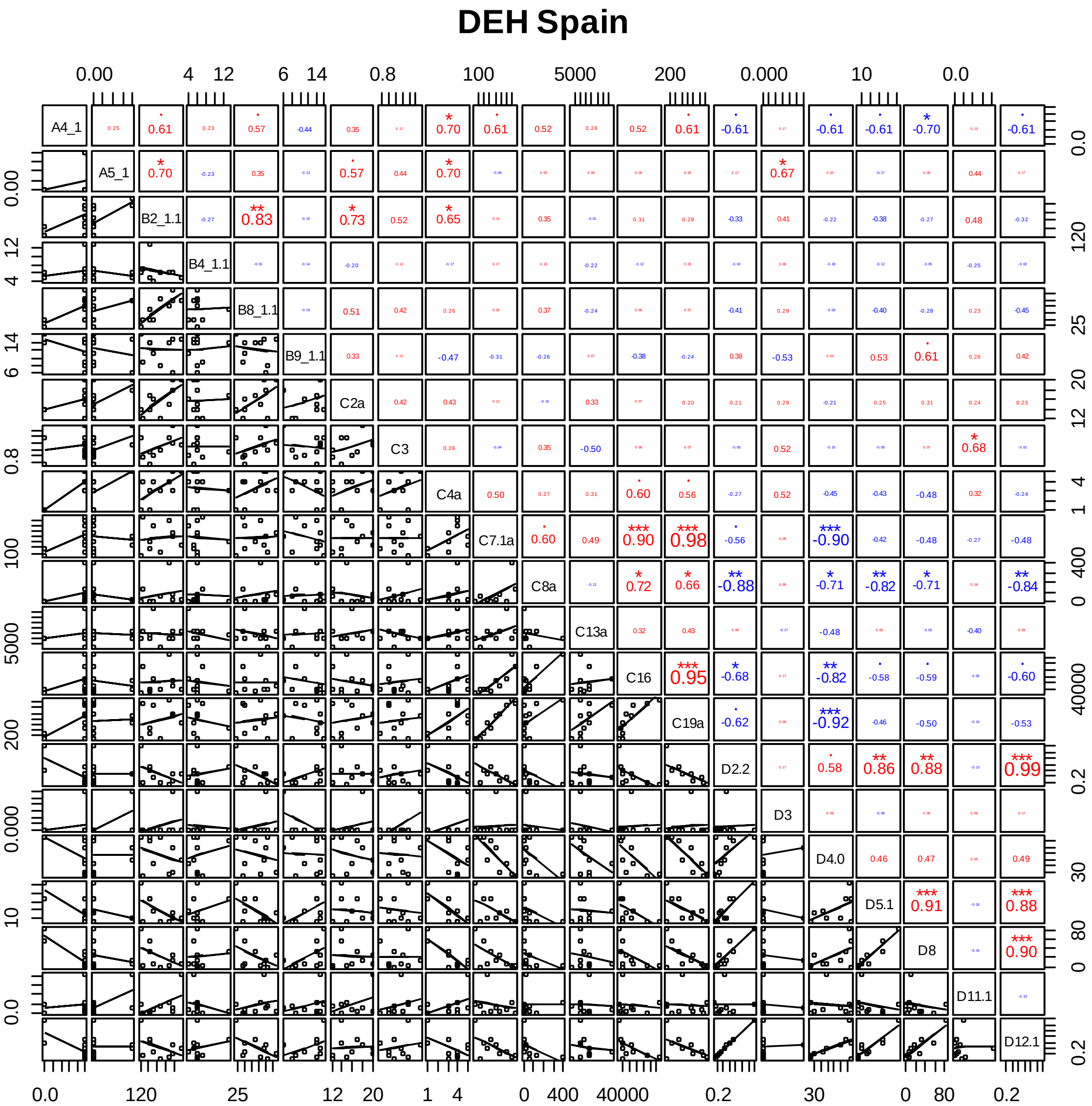


Figure 40: Example of Spearman correlations of BIOBIO indicators in the Spanish Dehesa case study. The relationship between indicators is shown graphically below the diagonal. Positive (red) and negative (blue) correlation coefficients with significance (stars) are given above the diagonal. Font size is proportional to coefficient value. See Figure 38's legend for abbreviations.

3.4 Mixed farming case study – Germany

In the mixed farming case study (Figure 41), total nitrogen input (D4.0) was the management indicator with the highest correlation with all other management indicators (e.g. positive correlation with the area with use of mineral N-fertiliser , D3, and with the total direct and indirect energy input, D5.1), excluding the number of field operations (D11.1). Total nitrogen input was also negatively correlated with plant (B2_1.1) and bee (B9_1.1) species richness and crop richness (C4a). Furthermore, the species richness of fauna indicators could be represented by plant species richness because of positive correlations. However, no indicators could be proposed as proxies for the habitat indicators.

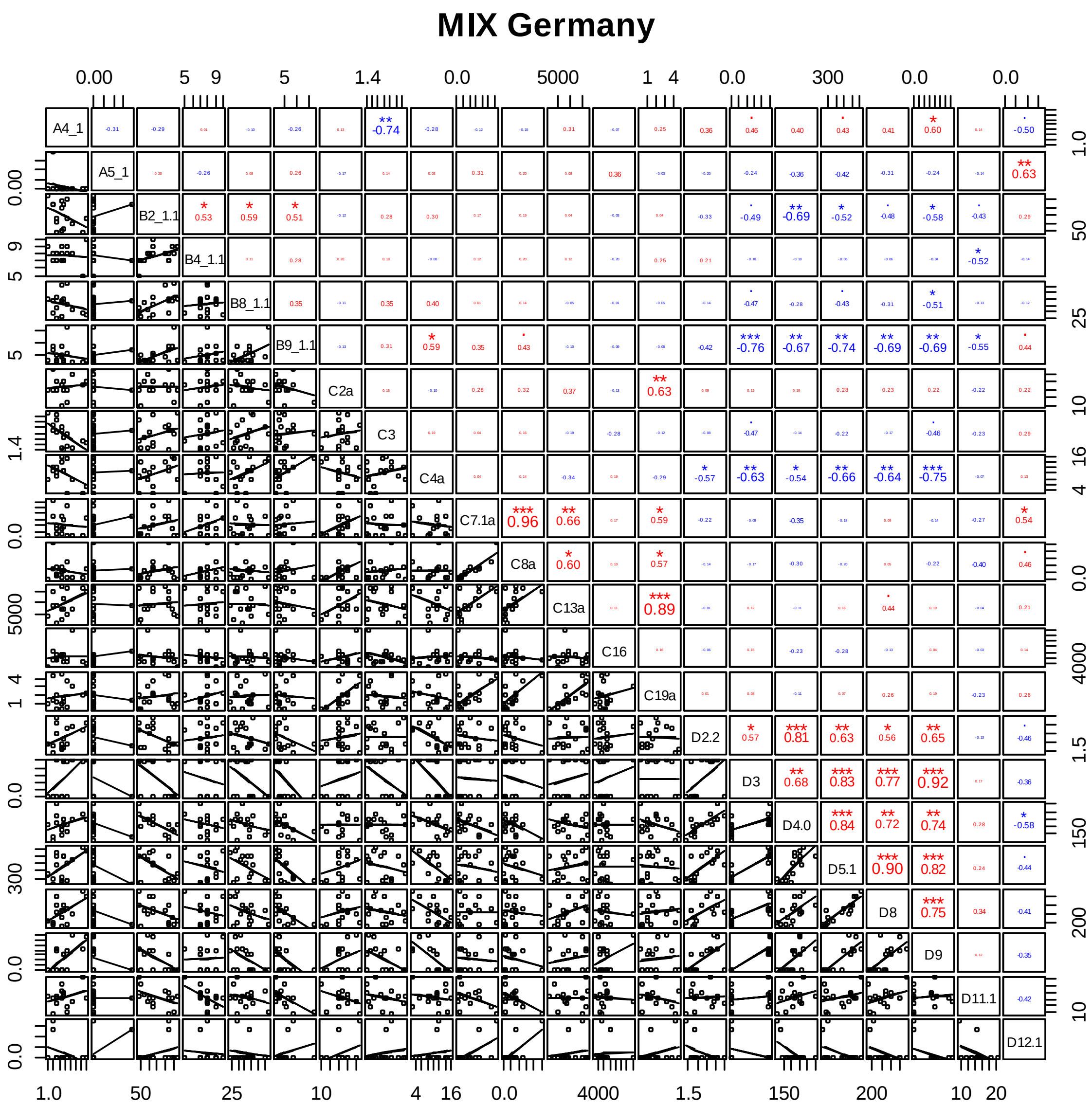


Figure 41: Example of Spearman correlations of BIOBIO indicators in the German mixed farming case study. The relationship between indicators is shown graphically below the diagonal. Positive (red) and negative (blue) correlation coefficients with significance (stars) are given above the diagonal. Font size is proportional to coefficient value. See Figure 38's legend for abbreviations.

3.5 Vineyard case study – Italy, and Olive plantation case study – Spain

In the vineyard case study, habitat indicators of farms such as tree cover (C7.1a), length of linear elements per hectare (C13a), average patch size (C16), and the share of semi-natural habitats (C19a) significantly and positively correlated to each other, so that the most appropriate indicator could be suggested as a surrogate for the others (Figure 42). By contrast, management and species indicators only marginally correlated to each other or to other indicators, except for plant species richness, for which habitat richness (C2a) or diversity (C3) could be proposed as a proxy as well as the management indicators area with use of mineral N-fertiliser (D3) and total nitrogen input (D4.0).

In the olive plantation case study, most of the indicators correlated with each other within groups – i.e. species, habitats and management – as well as between groups. In fact, species indicators were highly and positively correlated with habitat indicators such as habitat richness (C2a) or diversity (C3), as well as with management indicators such as total direct and indirect energy input (D5.1) and expenditures on fertiliser, crop protection and concentrate feedstuff (D8). Biodiversity assessment in this type of farming system could therefore be approached with a relatively short list of indicators, including the habitat diversity of farms, total nitrogen input, and number of field operations.

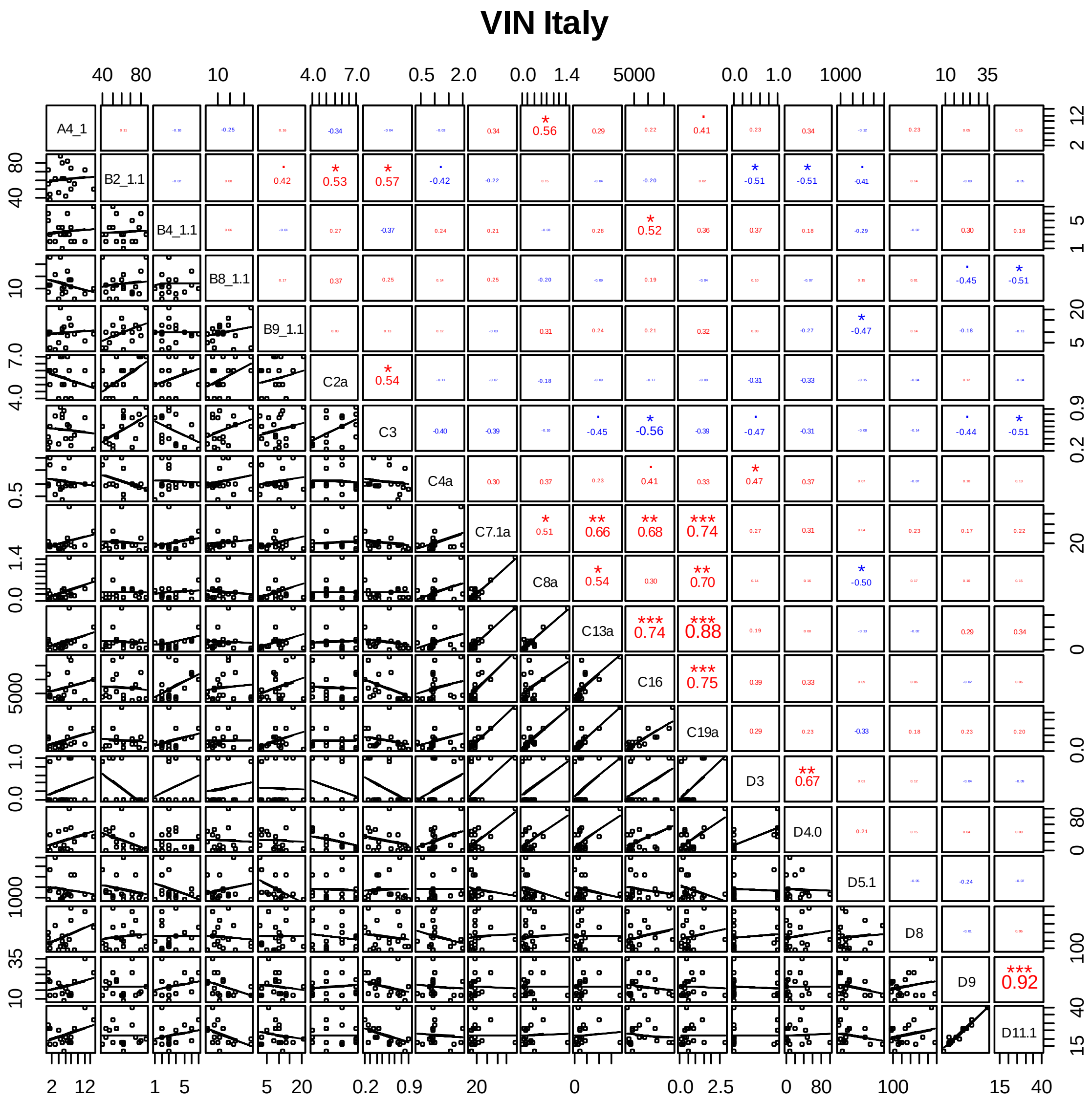


Figure 42: Example of Spearman correlations of BIOBIO indicators in the Italian vineyard case study. The relationship between indicators is shown graphically below the diagonal. Positive (red) and negative (blue) correlation coefficients with significance (stars) are given above the diagonal. Font size is proportional to coefficient value. See Figure 38's legend for abbreviations.

Correlograms for all the other case studies are provided in Appendix 3.

3.6 Interpretation of analysis and consequences for monitoring

Although correlations between farm management indicators and actual genetic, species and habitat indicators were observed in some case study regions, no general pattern emerged. The same applies to correlations within habitat, species and genetic diversity indicators, which were observed in some case study regions, but which were not consistent for types of farming, let alone for the whole range of case study regions. This shows that the remaining indicator set cannot be further reduced without losing information. Even in comparable farm types, biodiversity patterns and relationships between indicators are case specific. Consequently, all of the resulting biodiversity indicators should be monitored when using a European approach. Despite this, not all indicators are applicable for all farm types, for example the crop richness in grassland regions. However, an additional argument for not reducing the indicator set further – even for specific farm types where indicators did correlate – is that these observed correlations may disappear over time. This is because disturbances occurring in the agricultural landscape may support some indicators (e.g. species) whilst adversely affecting others. Thus, the removal of each indicator from the indicator set must be approached with caution.

4 What do the indicators indicate?

4.1 Case studies, farm types and farming systems

In monitoring and survey programmes, it is important to know what the indicators measured indicate. In the BIOBIO project, the main focus of the questioning was geared towards the capacity of indicators to distinguish between case studies, farming systems (organic vs. non organic), farm types (e.g. grassland vs. arable land), and to investigate the role of indirect indicators, i.e. the farm management indicators.

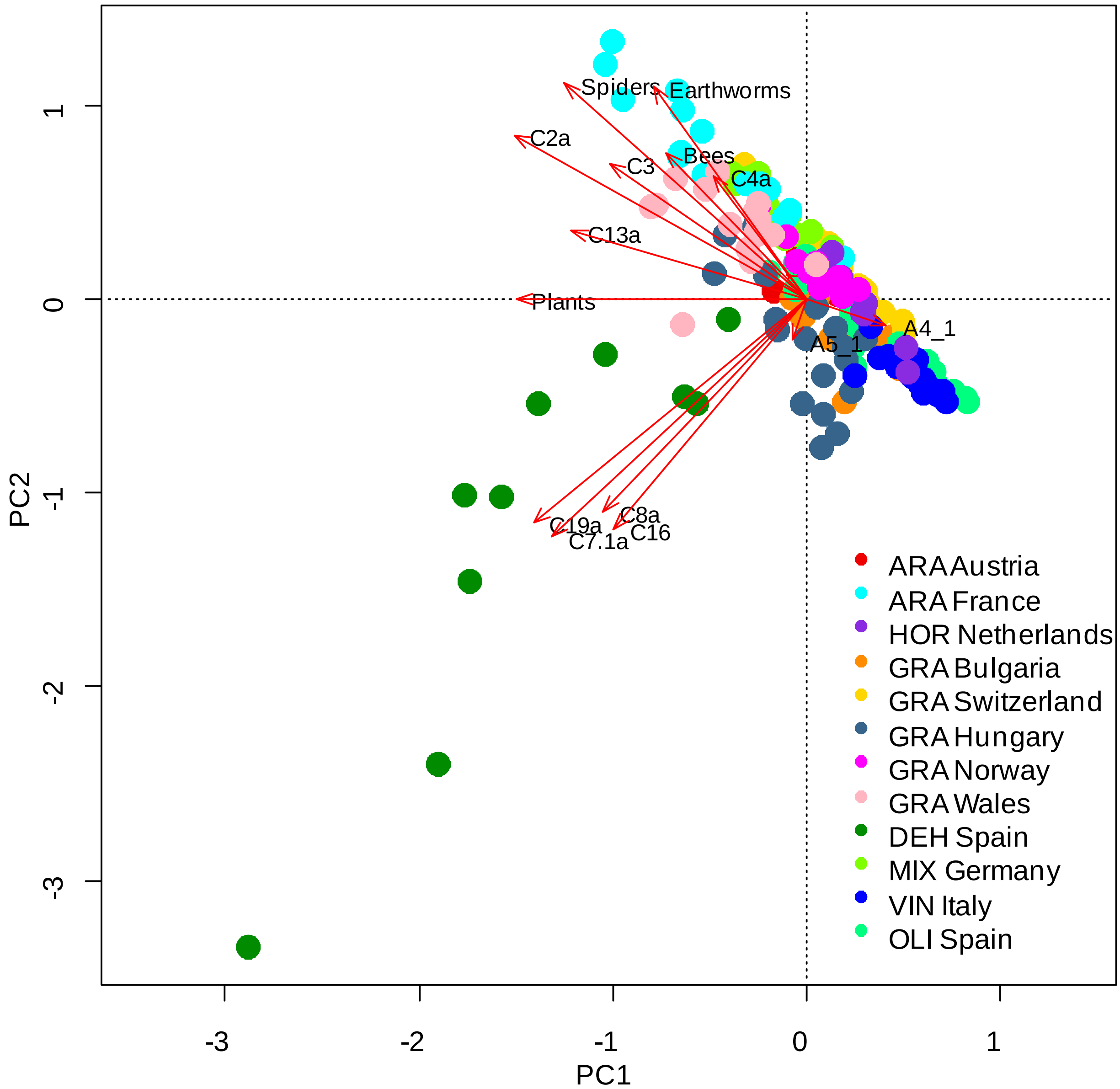


Figure 43: Principal component analysis of 185 BIOBIO farms (points) (10 farms were discarded because faunistic indicators could not be sampled) and 14 (direct) biodiversity indicators (arrows). Genetic diversity indicators are: A4_1 = Cultivar diversity, A5_1 = Origin of crops; Species diversity indicators are the gamma richness of plants, earthworms, spiders and bees; Habitat diversity indicators are C2a = habitat richness (number of habitat types on farms), C3 = habitat diversity (Shannon), C4a = Crop richness, C7.1a = Tree cover, C8a = Area of farmland with shrubs, C13a = Length of linear elements, C16 = Average size of habitat patches on the farm, C19a = Area of semi-natural habitats. The proportion of variance accounted for by the two axes represented is 51.6%.

In the BIOBIO project, genetic, species, habitat and management indicators represented a set of variables that were recorded in farms, i.e. a multivariate data set. A multivariate data set can be viewed as a collection of sites, the farms in BIOBIO, positioned in a space where each variable defines one dimension. There are thus as many dimensions as variables. To examine the structure of the data, it is interesting to represent the main trends in the form of scatterplots of the farms. Since the indicator set contained more than two variables, i.e. the 14 direct biodiversity indicators and the 8 indirect management indicators, it is tedious and not very informative to draw the farms in a series of planes defined by all possible pairs of indicators. The use of ordination methods (e.g. a principal component analysis) allows the data structure to be summarized. The aim of ordination method is to represent the data along a reduced number of orthogonal axes, constructed in such a way that they represent, in decreasing order, the main trends of the data. These trends can then be interpreted visually or in association with other methods such as clustering or regression.

A principal component analysis (PCA) was undertaken with the genetic, species and habitat diversity indicators to visualize the position of the farms in the multidimensional space (Figure 43). This PCA plot showed firstly that the Dehesa farms are different from the other farms, mainly because they had the highest values of tree cover (C7.1a), the area of farmland with shrubs (C8a), the average size of habitat patches on the farm (C16) and the area of semi-natural habitats (C19a). Secondly, there was a gradient ranging from the case studies in France, Germany and Wales to the olive plantation in Spain and the vineyard in Italy. This gradient was explained by higher values on average of gamma richness for three faunistic species groups and for the habitat indicators (C2a, C3, C4a) in France, Germany and Wales compared to Italy and olive plantations in Spain.

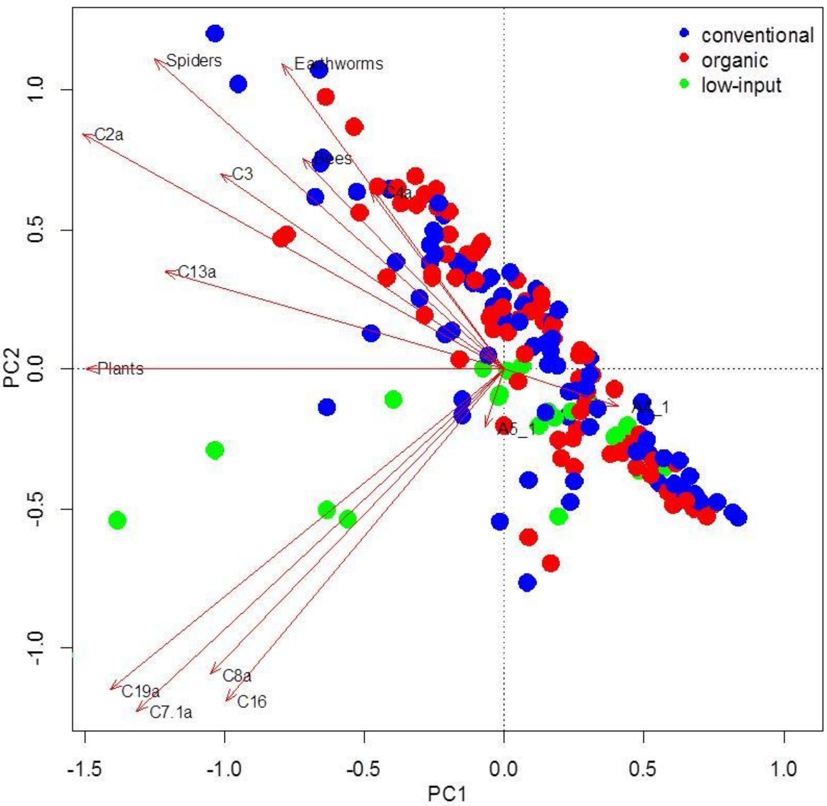


Figure 44: Principal component analysis of 185 BIOBIO farms (points) (10 farms were discarded because faunistic indicators could not be sampled) and 14 (direct) biodiversity indicators (arrows). For clarity of presentation, the zoom level in this plot excludes five of the Dehesa farms. Genetic diversity indicators are: A4_1 = Cultivar diversity, A5_1 = Origin of crops; Species diversity indicators are the gamma richness of plants, earthworms, spiders and bees; Habitat diversity indicators are C2a = habitat richness, C3, habitat diversity (Shannon), C4a = Crop richness, C7.1a = Tree cover, C8a = Area of farmland with shrubs, C13a = Length of linear elements, C16 = Average size of habitat patches on the farm, C19a = Area of semi-natural habitats. The proportion of variance accounted for by the two axes represented is 51.6%.

The same ordination plot with farms coloured according to the farming system, i.e. non-organic, organic and low-input, did not show a clear pattern to distinguish between the systems except for the low-input Dehesa farms (Figure 44, and compare to Figure 43). By contrast, indicators distinguished the farm type with permanent crops, i.e. farms with olive plantation in Spain and farms with vineyard in Italy, from the other farm types (Figure 45, and compare with Figure 43).

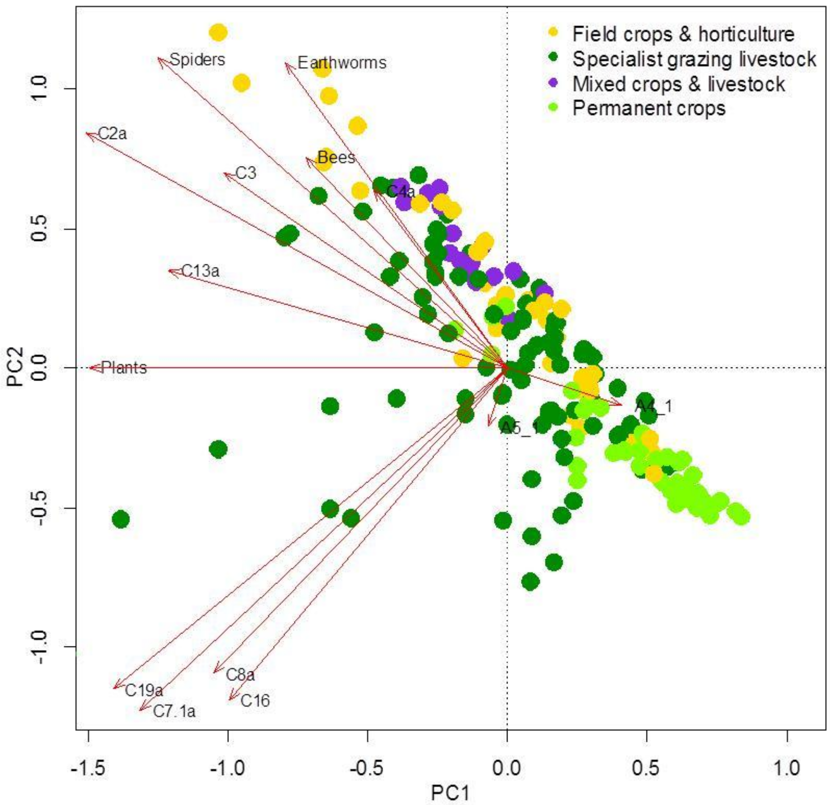


Figure 45: Principal component analysis of 185 BIOBIO farms (points) (10 farms were discarded because faunistic indicators could not be sampled) and 14 (direct) biodiversity indicators (arrows). For clarity of presentation, the zoom level in this plot excludes five of the Dehesa farms. Genetic diversity indicators are: A4_1 = Cultivar diversity, A5_1 = Origin of crops; Species diversity indicators are the gamma richness of plants, earthworms, spiders and bees; Habitat diversity indicators are C2a = habitat richness, C3, habitat diversity (Shannon), C4a = Crop richness, C7.1a = Tree cover, C8a = Area of farmland with shrubs, C13a = Length of linear elements, C16 = Average size of habitat patches on the farm, C19a = Area of semi-natural habitats. The proportion of variance accounted for by the two axes represented is 51.6%.

Therefore, the genetic, species and habitat indicators mainly indicate differences between the case study regions.

The 8 management indicators retained for further analysis clearly separate, on the one hand, farms with vineyard in Italy by higher values of pesticide use (D9) and the number of field operations (D11.1), and, on the other hand, the German mixed farming and the Swiss grassland systems by the average stocking rate (D2.2) and the total nitrogen input (D4.0) (Figure 46). Average stocking rate (D2.2), total nitrogen input (D4.0), area with use of mineral N-fertiliser (D3), expenditures on fertilizers, etc. (D8) have higher values in non-organic farms compared to organic farms (Figure 47).

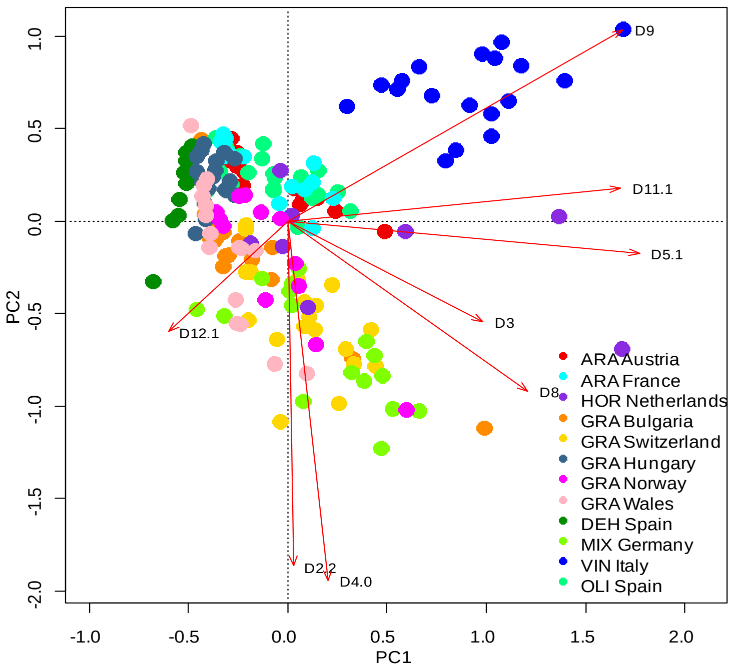


Figure 46: Principal component analysis of 195 BIOBIO farms (points) and 8 (indirect) management indicators (arrows). D2.2 = Average stocking rate per ha forage area, D3 = Area with use of mineral N-fertiliser, D4.0 = Total nitrogen input, D5.1 = Total direct and indirect energy input, D8 = Intensification/Extensification Expenditures on fertiliser, crop protection and concentrate feed stuff, D9 = Pesticide use, D11.1 = Field operations, D12.1 = Grazing Intensity. The proportion of variance accounted for by the two axes represented is 55.3%.

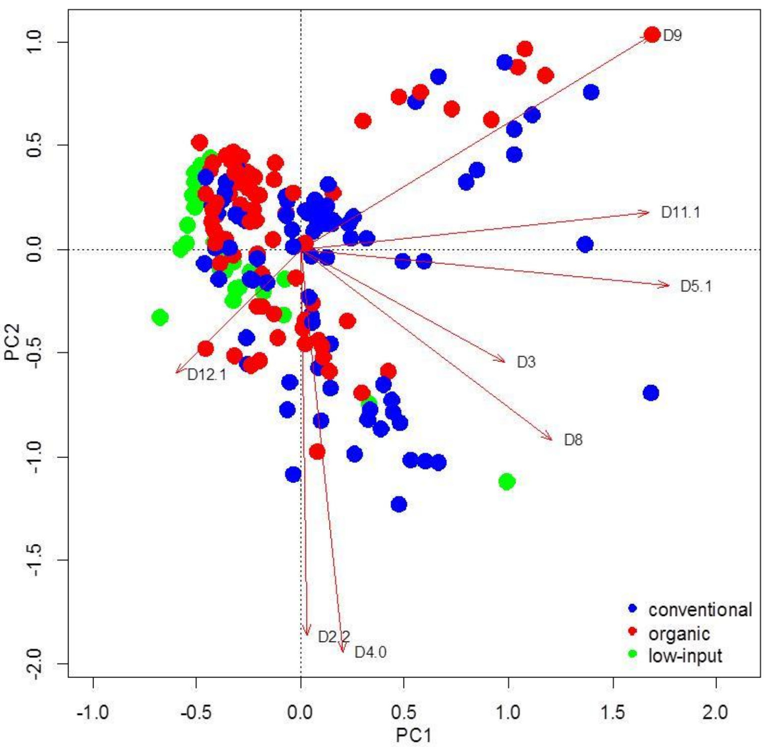


Figure 47: Principal component analysis of 195 BIOBIO farms (points) and 8 (indirect) management indicators (arrows). D2.2 = Average stocking rate per ha forage area, D3 = Area with use of mineral N-fertiliser, D4.0 = Total nitrogen input, D5.1 = Total direct and indirect energy input, D8 = Intensification/Extensification Expenditures on fertiliser, crop protection and concentrate feed stuff, D9 = Pesticide use, D11.1 = Field operations, D12.1 = Grazing Intensity. The proportion of variance accounted for by the two axes represented is 55.3%.

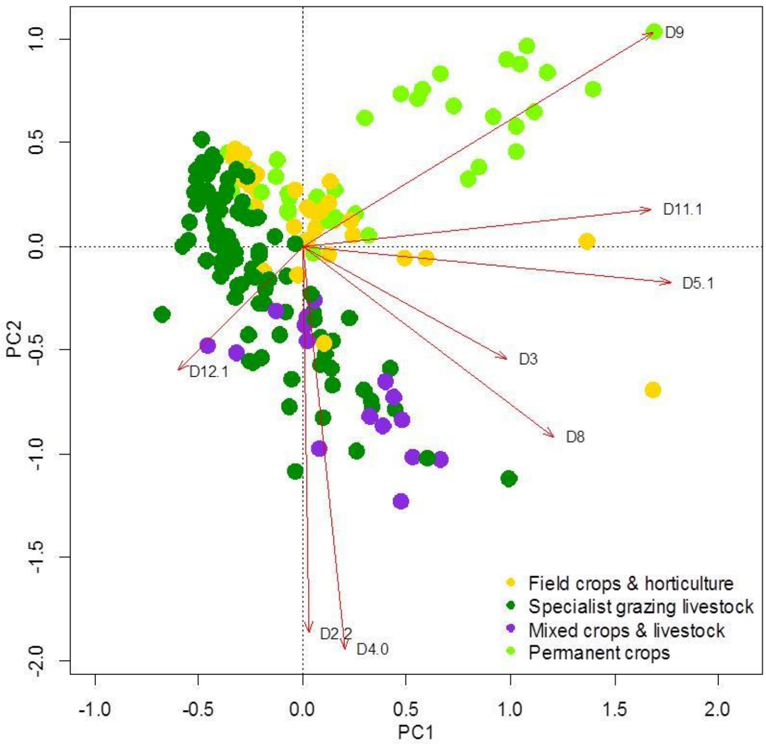


Figure 48: Principal component analysis of 195 BIOBIO farms (points) and 8 (indirect) management indicators (arrows). D2.2 = Average stocking rate per ha forage area, D3 = Area with use of mineral N-fertiliser, D4.0 = Total nitrogen input, D5.1 = Total direct and indirect energy input, D8 = Intensification/Extensification Expenditures on fertiliser, crop protection and concentrate feed stuff, D9 = Pesticide use, D11.1 = Field operations, D12.1 = Grazing Intensity. The proportion of variance accounted for by the two axes represented is 55.3%.

Average stocking rate (D2.2), total nitrogen input (D4.0), area with use of mineral N-fertiliser (D3), expenditures on fertilisers (D8), etc. display a gradient within specialist grazing livestock farms.

4.2 Testing differences between organic and non-organic, and along intensity gradients in low-input farming systems for the direct biodiversity indicators

We also investigated the use of ratios of species found in both organic and non-organic farming systems as a possible means to detect differences between the two groups of farms. Species numbers found on organic farms were predominantly higher than in the counterpart non-organic farms. This can be seen from the ratios between the average number of species on organic and non-organic farms (Figure 49) which tend to be above 1 in most cases. Values below 1 are more frequent in arable systems, vegetable farms and vineyards. To date only one of these ratios has been published; the positive effect of organic farming on plants species in Italian vineyards (Nascimbene 2012). Here, it can be seen that picking individual groups out of a dataset can lead to misleading interpretations and that mixed-effects modelling enables overall effects in the dataset to be analysed.

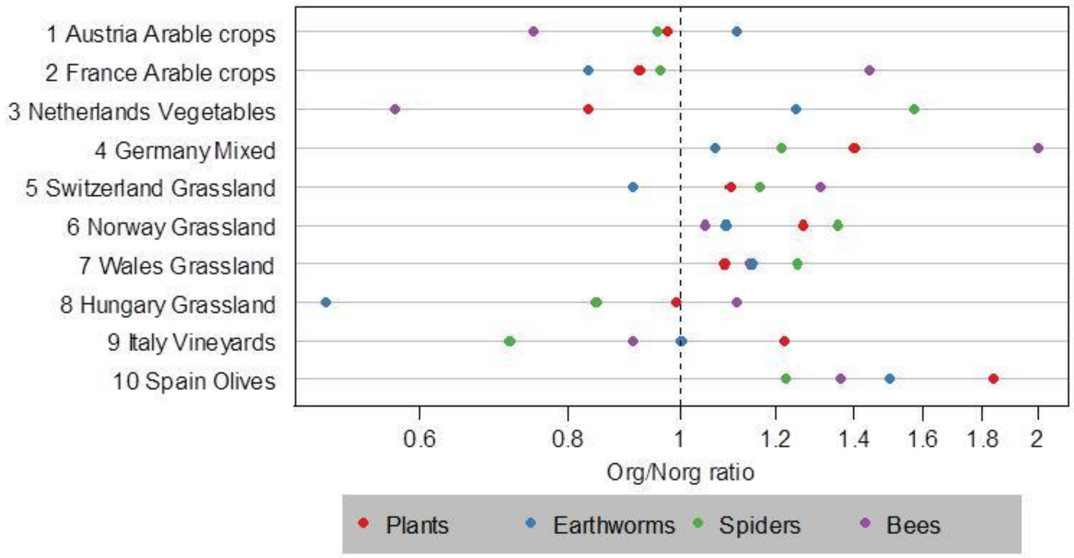


Figure 49: Ratios of total species numbers per farm in organic farms versus the non-organic baseline for the four taxonomic groups in ten investigated case study regions of the BIOBIO project.

In each of the four taxonomic groups, organic farming had a positive effect on total species numbers per farm (+10.8% for plants, +6.6% for earthworm, +4.6% for spiders and +17.8% for bees), but only the effect for plants was significant. In a joint model over all four groups, organic farming has a significantly positive effect (+10.4%). Comparable overall effects were also found if species numbers were aggregated after weighting with habitat area (+15.8%) as well as if only semi-natural habitats were considered (+12.9%).

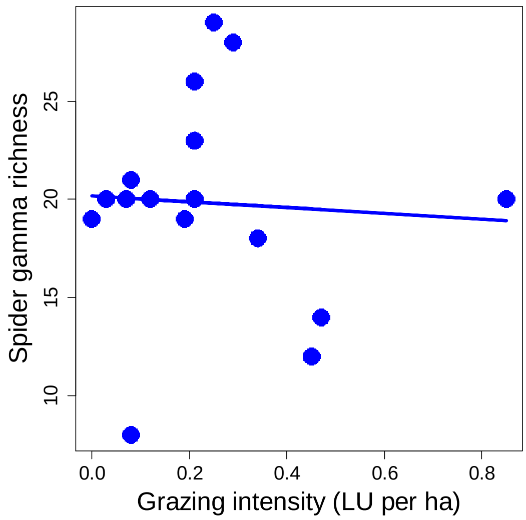


Figure 50: Trend in spider gamma species richness in farms of the Bulgarian case study explained by the grazing intensity (not a statistically significant relationship).

In case a grouping of farms could not be done *a priori*, and no group could be defined as a baseline, biodiversity data were compared to a range of farming characteristics, i.e. the management indicators that are supposed to reflect impacts and pressures on biodiversity. In the specific case of low-input farming, farming indicators have to be found that correlate well with direct biodiversity indicators. In the BIOBIO case study regions where low-input farming occurred, i.e. in the grassland systems of Bulgaria and the Dehesas in Spain, there were only a few management indicators that were significantly related to the species diversity of any group. However, the relationships could not be easily interpreted with ecological background information and should therefore be regarded with caution. This suggests that at low level of management intensity, differences with respect to species richness in farms may be due to other factors. It must be emphasized that in this analysis, the relationship between species diversity indicators and farming indicators has been investigated at the farm level, possibly obscuring more accurate relationships at the habitat and field level. As examples, in the Bulgarian case study and the Dehesa system in Spain, the spider gamma richness was negatively correlated to the grazing intensity (Figure 50), and the total nitrogen input was slightly positively correlated to the earthworm gamma richness although none of these correlations were statistically significant ($p > 0.05$).

5 On the relationship between species diversity and semi-natural habitats

In farmland, semi-natural habitats are supposed to play an important role for species diversity. In the BIOBIO farms, beside typical intensive agricultural fields, semi-natural habitats were part of the investigated habitats for species. See Appendix 4 for an explanation for the categorisation of habitats.

5.1 Relationship between species diversity and semi-natural habitats across the BIOBIO case study regions

One of the important questions arising with respect to biodiversity in farmland is how much of biodiversity effectively occurs in semi-natural habitats compared to intensive agricultural fields. This can provide useful information about the most effective sampling strategy for monitoring biodiversity. The comparative contribution is summarized in Figure 51 which shows that the percentage of species exclusively found in semi-natural habitats (Herzog et al., 2012a) is usually higher than in intensive agricultural habitats in the BIOBIO case study regions.

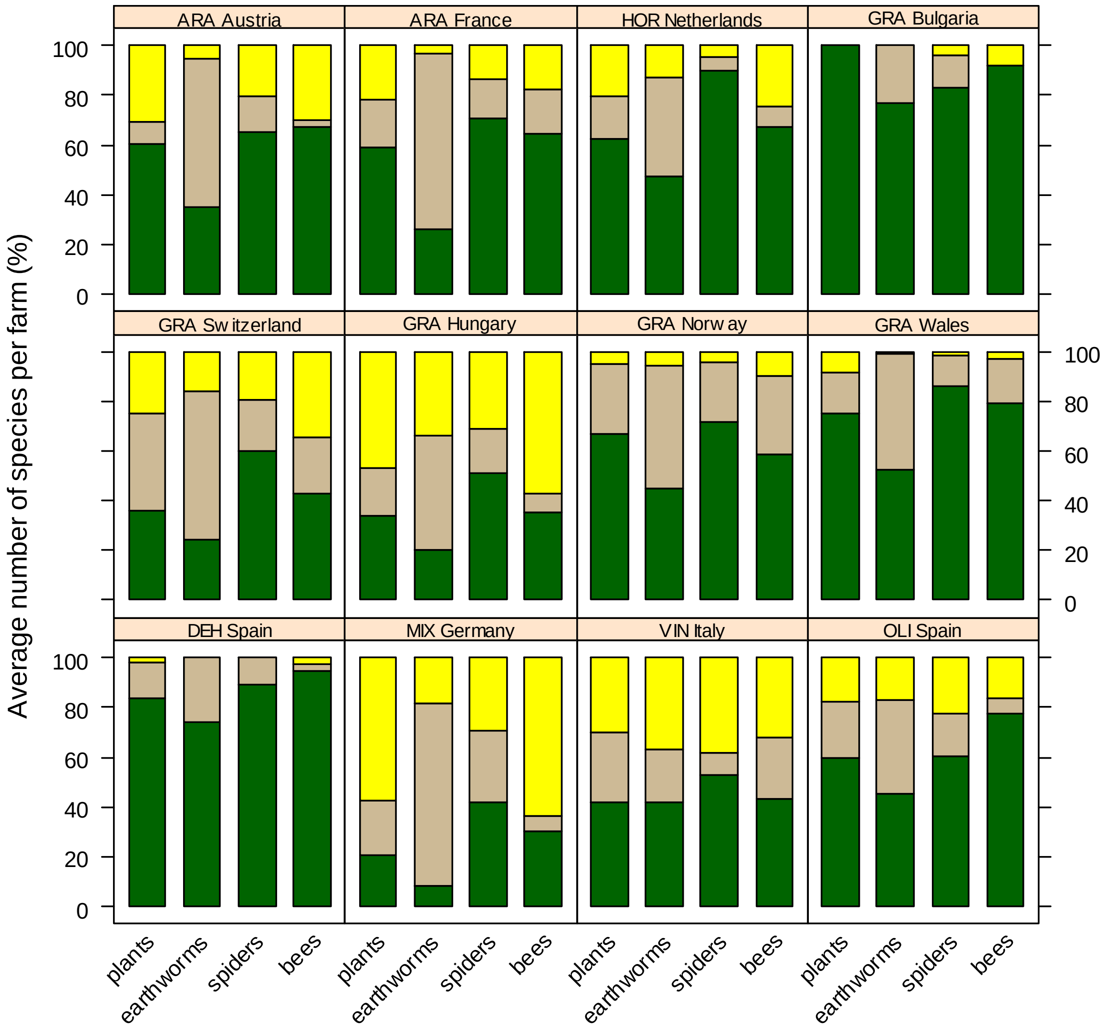


Figure 51: Percentage of plant, earthworms, spider and bee species found exclusively in semi-natural habitats (green stack), exclusively in intensive agricultural fields (yellow stack) and in both types of habitat (beige stack) based on the average number of species across farms in each case study (CS). ARA = arable CS, HOR = horticulture CS, GRA = grassland CS, DEH = Dehesa CS, MIX = mixed farming CS, VIN = vineyard CS and OLI = olive plantation CS.

Table 3: Number of semi-natural and intensive agricultural habitat types, and cover (% area) of semi-natural habitats in the BIOBIO case studies. ARA = arable CS, HOR = horticulture CS, GRA = grassland CS, DEH = Dehesa CS, MIX = mixed farming CS, VIN = vineyard CS and OLI = olive plantation CS.

Case study	Average No. of semi-natural / intensive agricultural habitat types found and sampled per farm	Average Cover (area percentage) of Semi-Natural Habitats (C19) per farm
ARA Austria	4.1 / 3.6	3.5
ARA France	6.6 / 3.2	7.5
HOR Netherlands	4.7 / 2.6	11.1
GRA Bulgaria	8.2 / 0.06	99.4
GRA Switzerland	3.4 / 2.3	4.7
GRA Hungary	3.7 / 4.5	19.5
GRA Norway	8 / 1.8	24.5
GRA Wales	9.7 / 0.9	48.6
DEH Spain	10.8 / 0.3	99.0
MIX Germany	2.5 / 5.4	3.0
VIN Italy	2.5 / 1.5	2.9
OLI Spain	3 / 0.6	95.0

The decisive factor with respect to the species richness at farm level (gamma diversity) was the overall number of habitat types, i.e. more habitat types meant more species. The results showed that usually more semi-natural than intensive agricultural habitat types were found in the BIOBIO farms across regions (TABLE 3). Therefore, more semi-natural habitats were sampled for species in farms as a result of the sampling method, i.e. one habitat sampled per habitat type found in the farm. Thus, semi-natural habitats have logically had more impact on species richness at farm level than the intensive agricultural ones in absolute value. However, the share of semi-natural habitat types on the farms (e.g. if 10 out of 15 habitat types recorded in a farm are semi-natural, the share of semi-natural habitat types is 15/10 = 66%) clearly impacted the number of plant, earthworm and spider species richness, for which the association was positive in 8 of 12 case studies (Figure 52). This trend was less obvious for the bee species richness.

For the most part, the species richness of the farms was not related to the cover (percentage area) of semi-natural habitats on farms (Figure 53). The reason is that a few semi-natural habitat types may dominate on the farm, comprising a large proportion of the area, without necessarily increasing the species richness at farm level. Case studies can be divided into three groups with respect to the cover of semi-natural habitats, i.e. case studies showing high cover (% area) of semi-natural habitats on farms (> 80%, e.g. olive plantations in Spain), case studies with low cover (% area) of semi-natural habitats on farms (< 20%, e.g. arable land in Austria) and case studies covering a large range of covers (% area) of semi-natural habitats on farms (0% < cover < 100%, e.g. grassland systems in Norway) (Figure 53). Farms with a high cover (% area) of semi-natural habitats on farms did not exhibit higher species richness in general except plants in olive plantations in Spain (Figure 53). Most positive associations were observed in grassland systems which have a higher variability in the cover (% area) of semi-natural habitats, e.g. in Norway and Wales for plants and spiders. In case studies with low cover (% area) of semi-natural habitats, significant positive trends were observed for earthworms in the vineyards of Italy and the mixed systems of Germany.

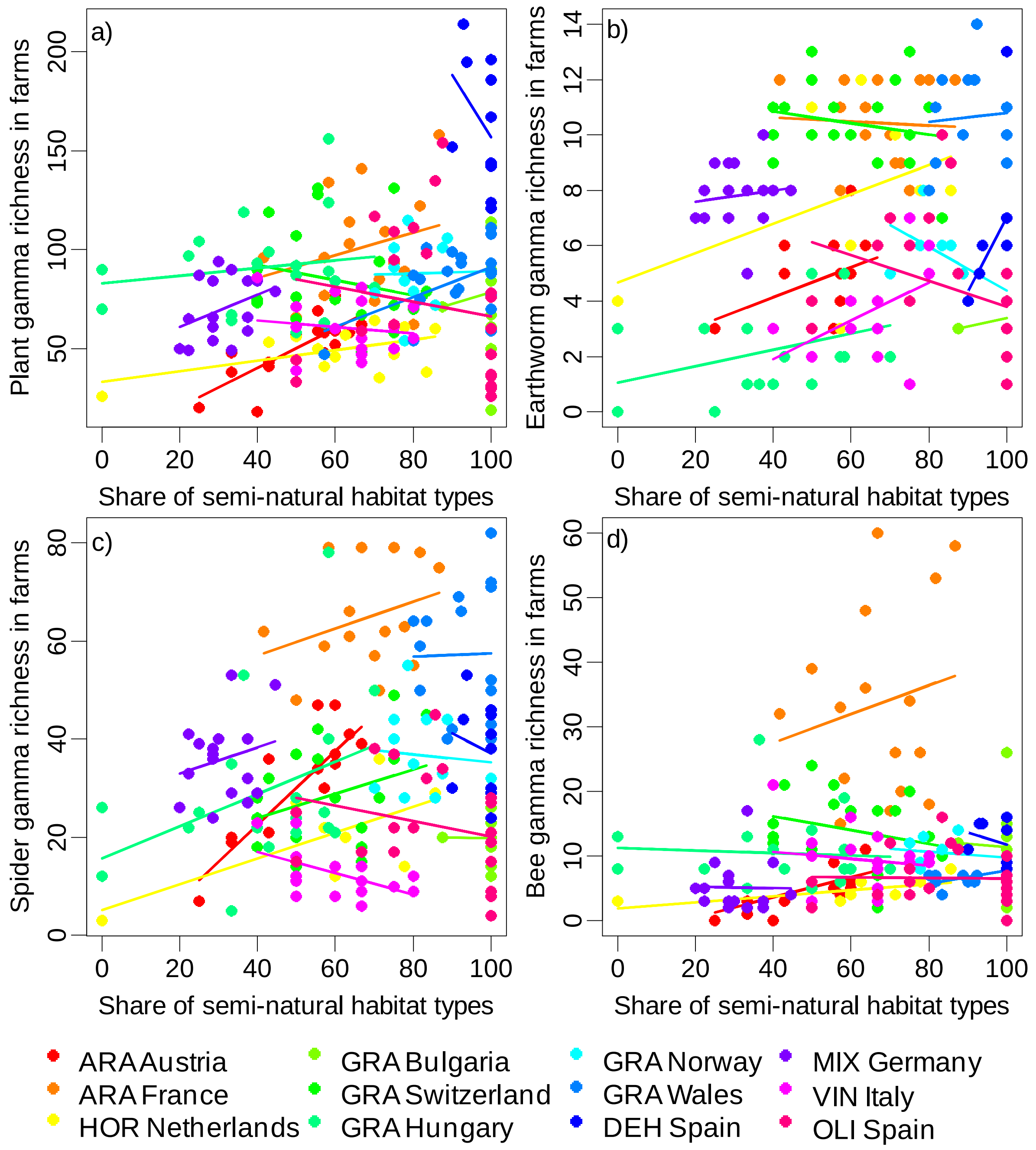


Figure 52: Gamma diversity of plants, earthworms, spiders and bees (overall species richness) related to the share of semi-natural habitat types (in terms of number of habitat types) on farms in 12 BIOBIO case study regions. ARA = arable CS, HOR = horticulture CS, GRA = grassland CS, DEH = Dehesa CS, MIX = mixed farming CS, VIN = vineyard CS and OLI = olive plantation CS.

5.2 Arable and horticultural case studies – Austria, France and The Netherlands

In these farming systems, the relationship between species indicators and semi-natural habitats was not the same for the four species groups, although the relationship for particular species groups was similar across case studies. Indeed, while about 60% of the plant species were exclusively found in semi-natural habitats, a large proportion of earthworm species were captured in both semi-natural and in intensive agricultural habitats (Figure 51). More than 65% of the spider and bee species occurred exclusively in semi-natural habitats. With the exception of earthworms in France, the number of species on farms was positively correlated to the share of semi-natural habitat types (Figure 52). The arable case study in France had the highest numbers of spider and bee species.

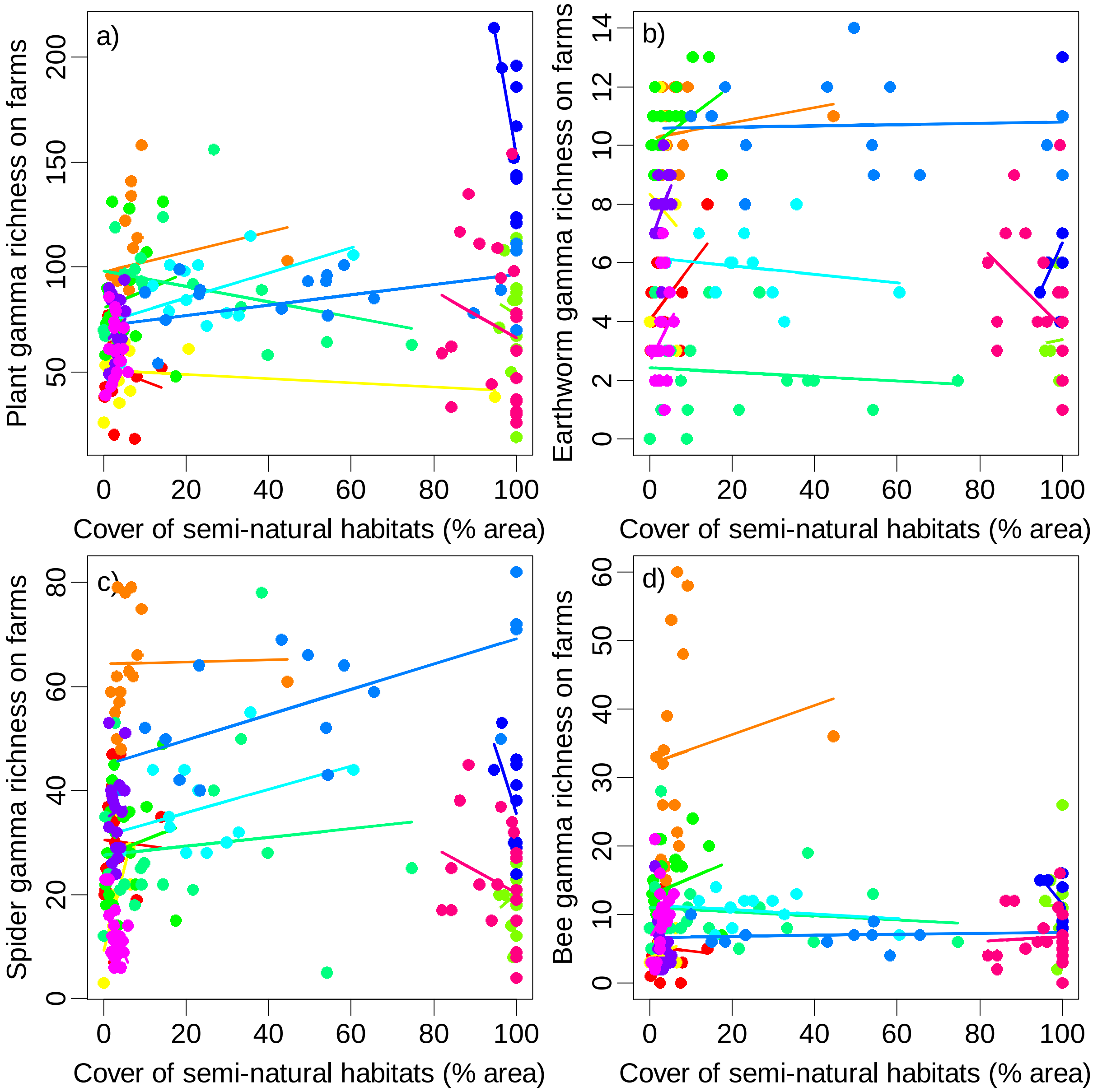


Figure 53: Gamma diversity of plants, earthworms, spiders and bees (overall species richness) related to the cover of semi-natural habitats (% area) on farms in 12 BIOBIO case study regions. Colours of points and lines refer to the case studies as for Figure 52.

5.3 Grassland case studies – Bulgaria, Switzerland, Norway, Hungary and Wales

In grassland case studies, the role of semi-natural habitats varied depending upon the amount of semi-natural habitat types. For example in Bulgaria, 132 habitats were classified as semi-natural whilst only one intensive agricultural field was reported. Both Hungary and Switzerland had fewer semi-natural habitats and the number of species found exclusively in semi-natural habitats or exclusively in intensive agricultural habitats was similar. Wales and Norway had a higher proportion of semi-natural habitat types and the proportion of species exclusively found in semi-natural habitats was also higher (Figure 51). On Bulgarian farms with more than 80% semi-natural habitat types, there was little variation in the number of species and the number of species was not the highest there (Figure 52). The case study in Wales had among the highest number of earthworm and spider species.

5.4 Case studies: Dehesa – Spain, Mixed farming – Germany, Vineyard – Italy and Olive plantation – Spain

With very high levels of semi-natural habitat types, the Dehesas and olive plantations in Spain also had a high level of species exclusively found in these habitats, in particular bees (Figure 51). The Dehesas were also exceptionally plant species-rich but there was no effect on the faunistic groups (Figure 52). The mixed farming system of Germany showed the highest effect of intensive agricultural habitats on species exclusively found there, due to the higher number of these habitat types in this region. However, the number of species generally increased with the share of semi-natural habitat types in German farms. In vineyards in Italy, both semi-natural and intensive agricultural habitats exhibited a similar number of exclusive species, and only the number of earthworm species increased with increasing share of semi-natural habitat types. In olive plantations in Spain, the number of species in farms did not increase with the share of semi-natural habitat types or even decreased even though high numbers of such habitats were found there.

6 On the relationship between species diversity and farm management indicators

6.1 Total nitrogen input

Nutrients are a limiting factor in plant production. They define the growth conditions for plant species and, thus, the differentiation of habitats. Nitrogen is a key element facilitating biomass production. Environments that are limited in nitrogen generally favour plant species diversity, whereas high nitrogen availability promotes a rather restricted number of highly competitive plants. To achieve satisfactory yields, farming strives to raise the nutrient level of the soil, changing the trophic conditions of the habitats for wild species. Nitrogen input is a pressure indicator that has proven useful for the assessment of land-use intensity in a series of studies in Europe and beyond. Nitrogen input largely determines the production intensity, e.g. the number of possible cuts in grasslands or the plant density in arable crops like cereals. The unit of measurement is the average input of nitrogen on the farm (kg N per ha UAA).

As shown in Figure 26, the total nitrogen input (D4.0) varied largely among and within case studies. With increasing inputs, there was a trend of decreasing gamma species richness of all four indicator groups recorded in cultivated forage and food crops (Figure 54). However, the negative linear relationship was only statistically significant ($p < 0.05$) for plant richness in the case study regions of Austria, Germany and Italy, for earthworm richness in Hungary, and for bee richness in Switzerland. Further individual analysis within organic and non-organic farms showed that the plant richness on non-organic farms of Austria and of Switzerland was significantly and negatively related to the total nitrogen input but not on the organic ones. Earthworm richness was further significantly and negatively related to the total nitrogen input on the organic farms of Austria. Organic and non-organic farms did not show significant relationships between the total nitrogen input and the spider richness in any case study region. In Switzerland, bee richness dependence on total nitrogen input was significant on non-organic farms but not on organic ones.

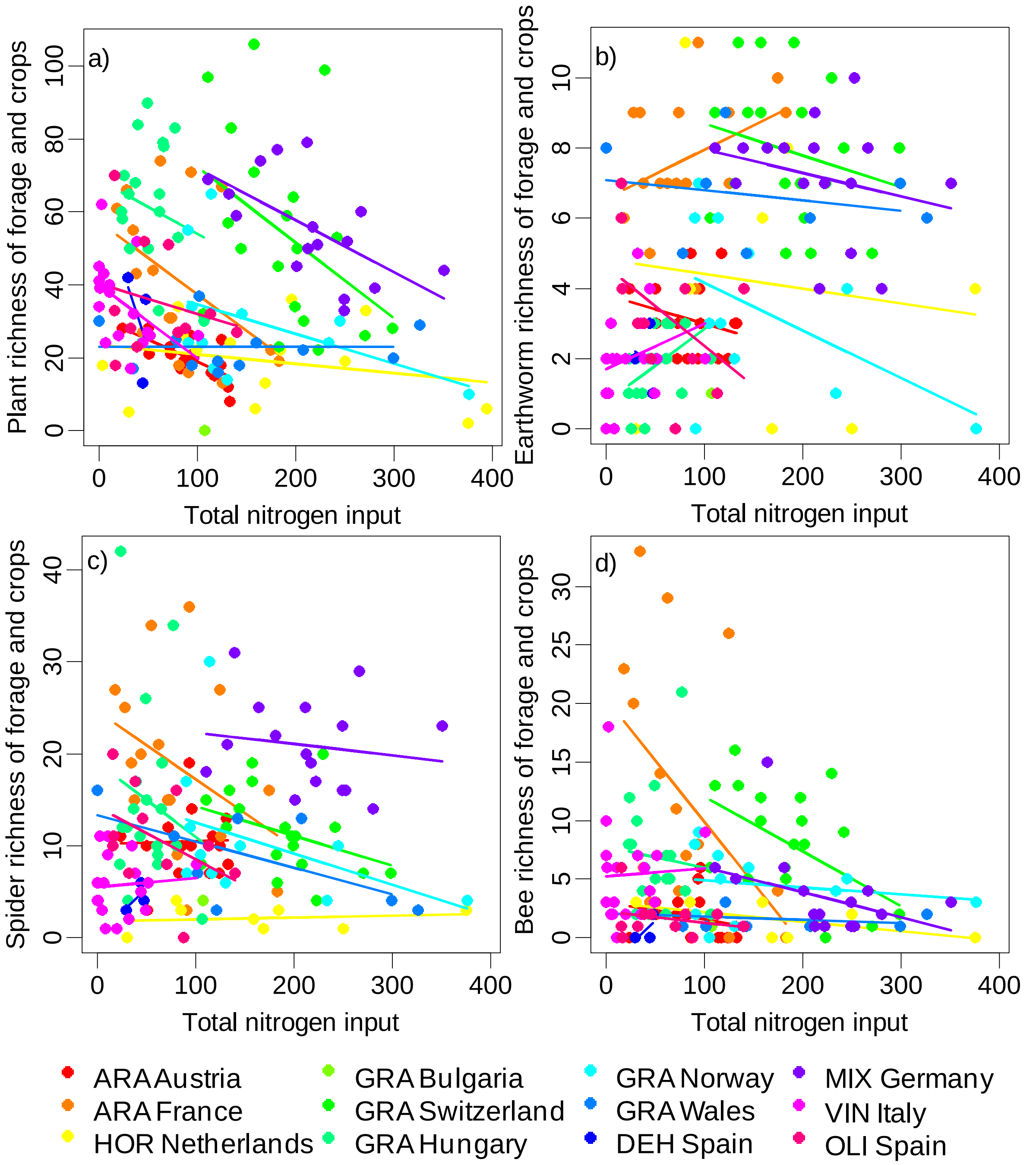


Figure 54: Gamma diversity of plants, earthworms, spiders and bees (overall species richness) collected or observed in cultivated forage and food crops, related to the total nitrogen input (kg N / ha UAA)(D4.0) on farms in 12 BIOBIO case study regions. ARA = arable CS, HOR = horticulture CS, GRA = grassland CS, DEH = Dehesa CS, MIX = mixed farming CS, VIN = vineyard CS and OLI = olive plantation CS.

6.2 Total direct and indirect energy input

Consumption of direct (fuel, electricity) and indirect energy (synthetic fertilisers, pesticides, production of feedstuff and machinery) for production of crops and livestock is a measure of the energy intensity of the farms. The measurement unit of this indicator is the equivalent litre of fuel per hectare of utilised agricultural area (UAA). It is known that the intensification of agriculture that has taken place during recent decades raises the energy use. Almost every activity on a farm is connected with an input of energy. Especially the production of modern crops is characterized by high inputs of fossil energy. The indirect energy consumption for farm inputs like fertilisers or pesticides is often higher than the direct energy consumption. The energy intensity is mainly dependent on the individual farming system. There is a strong correlation between the energy intensity, the farming structure (cropping system, livestock grazing regime), the resources used, matter fluxes (imported fertilizer, forage and feedstuff), the yields and the operation practices (frequency of operations, machines used). High input systems are e.g., characterized by the heavy use of fertilizers, pesticides, and labour-saving, high-power machines. The introduction of these techniques has led to a dramatic increase in the input of fossil energy. The higher the energy intensity, the higher is the control or regulation intensity in the agro-ecosystem. With this the potential for environmental effects increases and the farming system has a greater impact on biodiversity.

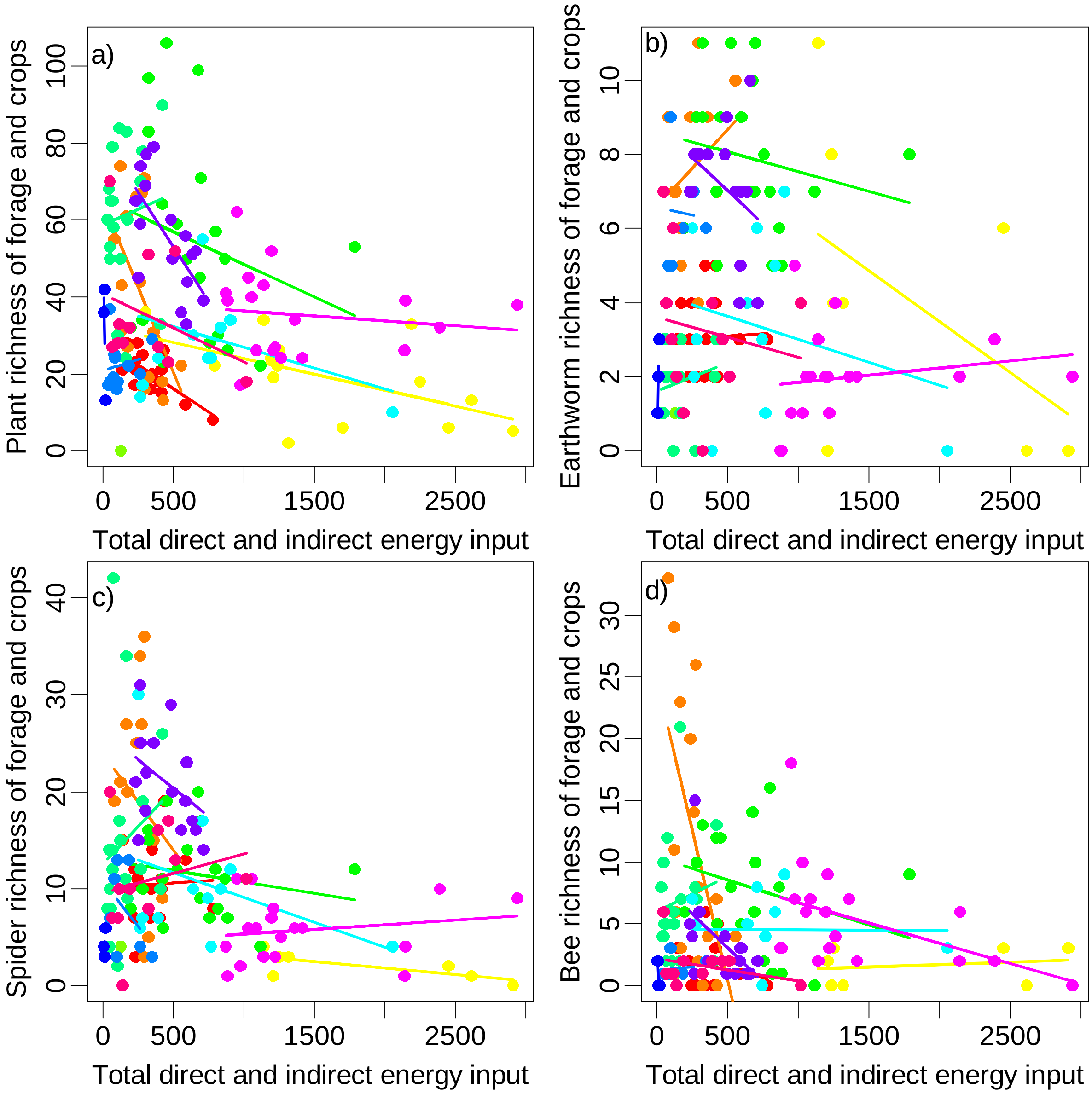


Figure 55: Gamma diversity of plants, earthworms, spiders and bees (overall species richness) collected or observed in cultivated forage and food crops, related to the total direct and indirect energy input (ELF)(D5.1) on farms in 12 BIOBIO case study regions. Colours of points and lines refer to the case studies as for Figure 54.

Regarding the total direct and indirect energy input (D5.1), negative linear relationships were significant ($p < 0.05$) for the plant richness surveyed in cultivated forage and food crops on farms of Austria, France, The Netherlands and Germany as well as for spider richness on farms of The Netherlands, and bee richness on farms of France and Germany (Figure 55). No correlations were found for earthworm richness. Interestingly, further analysis within management systems showed that plant richness was not particularly dependent on the energy input for organic or non-organic farms of the case study regions mentioned above apart from organic farms of Wales. The earthworm richness on organic farms of Norway was significantly and negatively related to the energy input as well as the spider richness on organic farms of France. The bee richness was significantly and negatively related to total energy input in the organic farms in Norway and the non-organic farms in Italy.

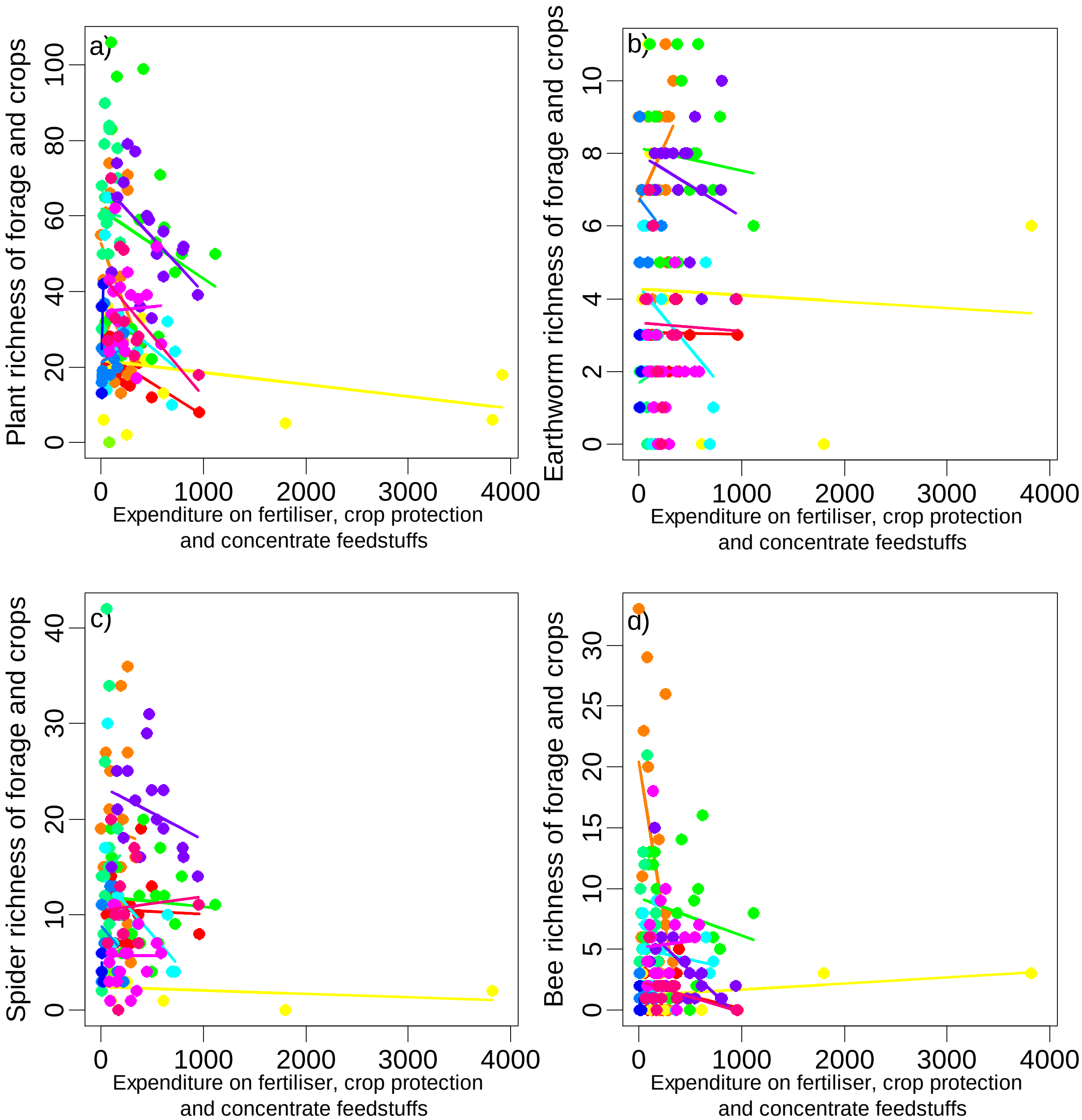


Figure 56: Gamma diversity of plants, earthworms, spiders and bees (overall species richness) collected or observed in cultivated forage and food crops, related to the Intensification/extensification expenditure on fertiliser, crop protection and concentrate feedstuffs (EUR / ha UAA)(D8) on farms in 12 BIOBIO case study regions. Colours of points and lines refer to the case studies as for Figure 54.

6.3 Intensification/Extensification – Expenditure on fertilizer, crop protection and concentrate feedstuffs

Annual expenditures on fertiliser, crop protection and concentrate feed stuff is measured in Euros (€) per ha UAA. It is a pressure indicator. This indicator was proposed to distinguish between farms of low, medium or high farming intensity in IRENA indicator No.15. The process of intensification of agricultural production is characterised by an increase in farm production (yields) based on higher inputs of fertiliser, crop protection, machinery, water and energy.

Expenditure on fertilizer, crop protection and concentrate feedstuffs (D8) showed that two farms in The Netherlands had very high values compared to the other BIOBIO case study regions (Bulgaria had also a high value in one farm but is not considered here)(Figure 56). The negative linear relationship was significant ($p < 0.05$) in case study regions of Austria and Germany for plant richness, nowhere for earthworm and spider richness, and in France and Germany for the bee richness. In a separate analysis within management systems, plant richness showed a significant negative relationship to the management indicator in non-organic olive plantations of Spain. The earthworm richness was significantly related to the expenditure on the non-organic farms of France. On organic farms in Germany and non-organic farms of Norway, the spider richness showed a significant relationship to the expenditures. There was no specific relationship of the bee richness with any management system in any case study.

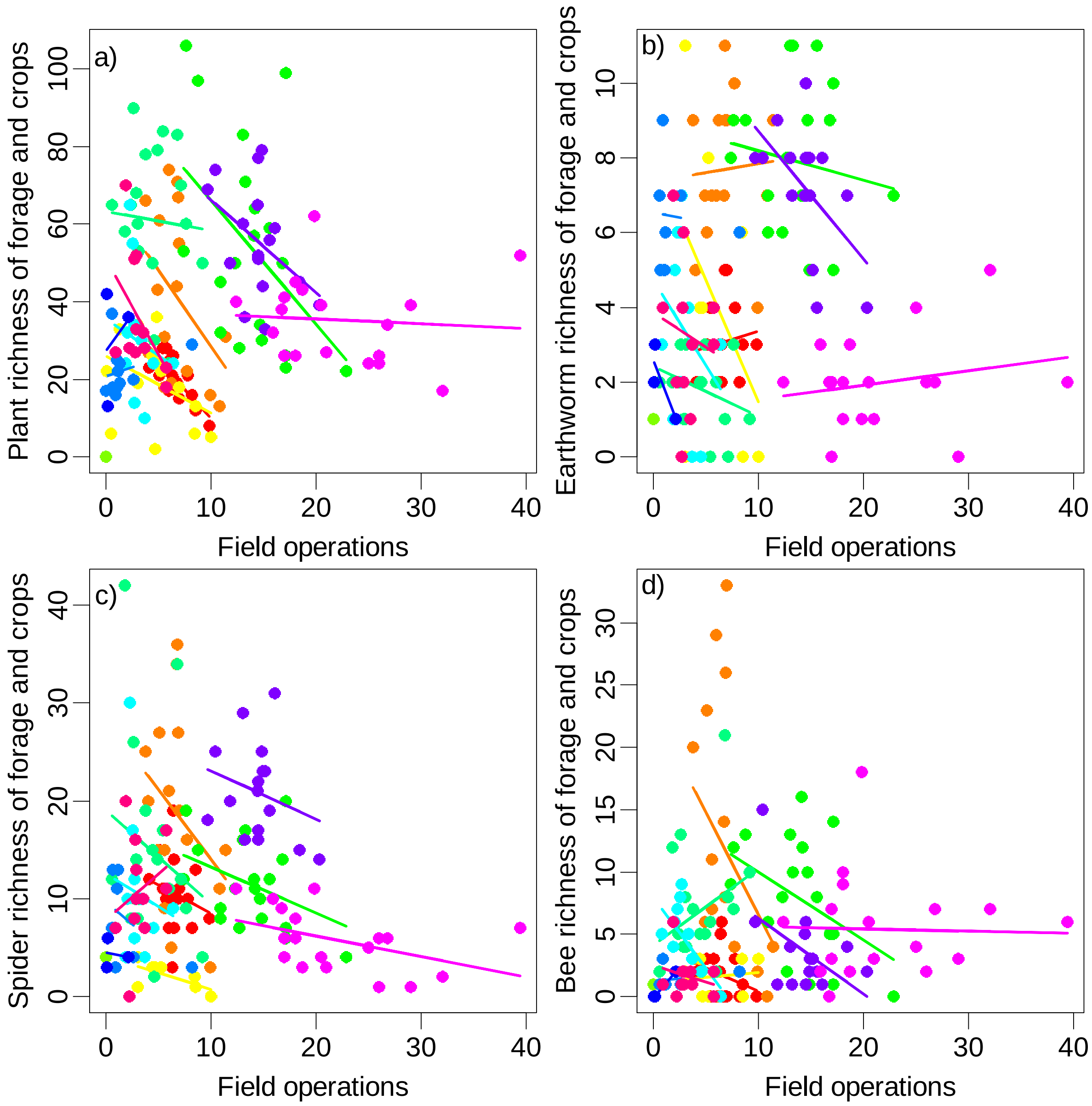


Figure 57: Gamma diversity of plants, earthworms, spiders and bees (overall species richness) collected or observed in cultivated forage and food crops, related to the field operations (D11.1) on farms in 12 BIOBIO case study regions. Colours of points and lines refer to the case studies as for Figure 54.

6.4 Field operations

The indicator quantifies the number of mechanised field operations in crop fields and grassland. The unit of measurement is total number of field operations. On farm-level the area-weighted average is calculated. It is a pressure indicator. Generally, trends in the intensification of production are strongly connected to processes that will increase the use of machinery on the parcels and the number of passages that is required in the cultivation of agricultural land. In grasslands, productivity increases with the number of cuts that are possible (1 to 2 cuts in extensive grassland, 5 to 6 cuts in intensively managed grassland). Equally in arable land or horticulture the number of operations from weeding, fertilisation or pesticide treatments increases.

The number of field operations (D11.1) had a significant negative relationship ($p < 0.05$) with the plant richness in the case study of Austria, the earthworm richness in Germany, and nowhere for spider richness or bee richness (Figure 57). Further separate analysis within organic and non-organic farms showed that the plant richness on non-organic farms of Austria and of Wales was significantly and negatively related to the field operations but not on the organic ones. A difference occurred also in the vineyards of the case study region of Italy where earthworm richness was significantly and negatively related to the field operations of non-organic farms. Spider richness was not related to the field operations in any case study, when all farms were considered but it was significantly and negatively related to the field operations in the non-organic farms of Italy. This can be explained by the fact that field operations in vineyards in Italy mainly consisted of pesticide application and occurred then in non-organic farms. Bee richness was not significantly dependent on the field operations in any case study regions when all farms were considered, but it was significantly and negatively related to the field operations in the non-organic farms of Norway.

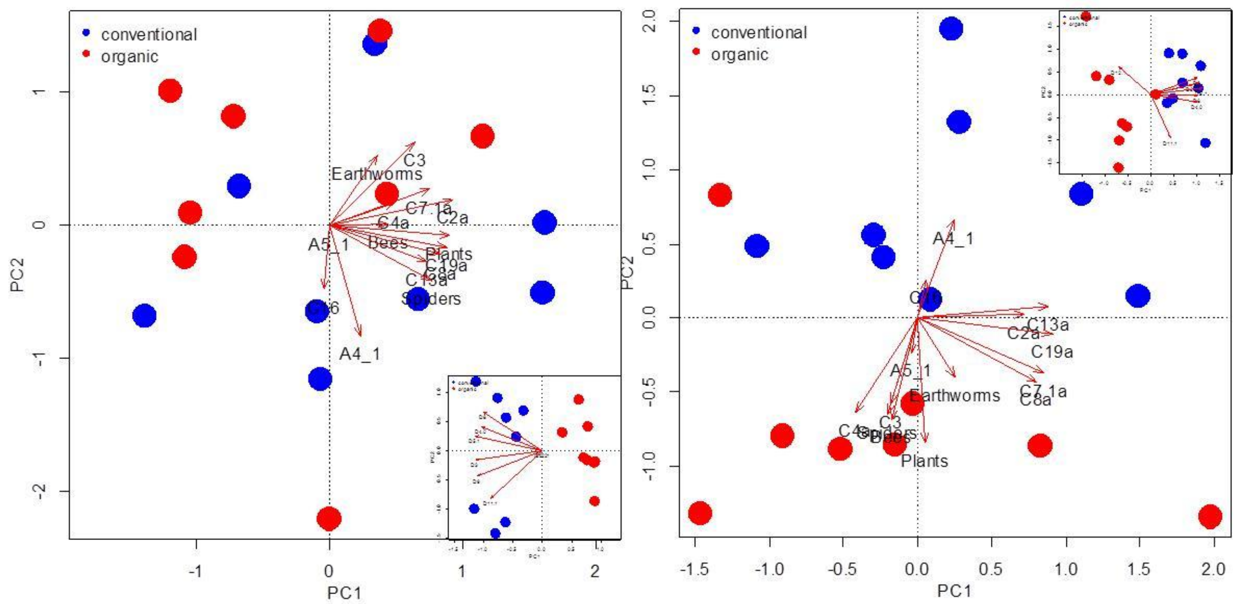


Figure 58: Principal component analysis of 16 BIOBIO French (left) and German (right) farms (points) according to the14 direct biodiversity indicators (arrows, big diagrams) and to the 8 indirect management indicators (arrows, small diagrams).

PCA diagrams of the case study regions in France and Germany showed two different situations with respect to the direct biodiversity and indirect management indicators (Figure 58). In both case studies, management indicators strongly differentiated organic from non-organic farms (small PCA graphs, organic farms on one side and non-organic ones on the other side), non-organic farms having commonly and on average higher values for almost all the management indicators, e.g. higher total nitrogen inputs (D4.0), higher total direct and indirect energy inputs (D5.1), higher number of field operations (D11.1). PCA of direct biodiversity indicators (large PCA graphs) shows, however, that there is no apparent difference between organic and non-organic farms in the French arable case study (both farm types are mixed on the diagram) while farms are well separated in the German mixed farming case study with usually higher values for most of the direct biodiversity indicators in organic farms. This suggests that management indicators have an impact on biodiversity indicators in the German but not in the French case study region.

Considering all farms in regions together on the one hand, or, organic and non-organic farms separated on the other hand, produced contrasting results regarding the impact of management indicators and species richness. This means that in some cases, taking all farms together can hide significant relationships in one or the other management system. In contrast, analysing farms by management system separately can hide more general relationship occurring at the region level.

7 Conclusions

In the FP7 research project "BIOBIO - Biodiversity indicators for organic and low-input farming systems" a core set of 23 indicators were identified: 8 habitat indicators, 4 species indicators, 3 indicators for the genetic diversity of crops and livestock and 8 farm management indicators. The indicator set is a result of thorough scientific screening and testing in 12 case study regions with various farm types and farming systems across Europe as well as of regular stakeholder consultation. The scientific screening showed that:

- The BIOBIO indicator set does not demonstrate any apparent pattern of correlations <u>within</u> the genetic, the species, the habitat and the farm management indicators across all the case study regions. Instead, pattern of correlations were case study specific;
- The BIOBIO indicator set does not demonstrate any apparent pattern of correlations <u>among</u> the four indicator groups; in some cases, habitat and management indicators correlated with species indicators but not in others;
- The BIOBIO indicator set could distinguish between the 12 case study regions; large gradients occurred between case study regions with respect to the number of plant, earthworm, spider and bee species as well as to the habitat indicators, the use of pesticide, the average stocking rate and the total nitrogen input;
- The BIOBIO indicator set could separate case study regions with permanent crops from other farm types, i.e. field crops and horticulture, specialist grazing livestock, and mixed crops and livestock, in particular because of the use of pesticide in these regions;
- The BIOBIO indicator set could not clearly distinguish between organic and non-organic farming systems but did distinguish the low-input systems, in particular the Dehesas in Spain;
- The number of species on the BIOBIO farms, i.e. plants, earthworms, spiders and bees, was on average higher on the organic farms compared to non-organic ones;
- The number of species occurring in low-input farms depended only rarely on the management indicators suggesting that at low level of farming intensity, differences between farms may be explained by other factors;
- The impact of semi-natural habitats on species richness of plants, spiders and bees was remarkable with commonly more than 60% of the species exclusively found there, earthworms were rather associated with fields;
- In 8 of 12 case study regions, positive associations were demonstrated between the species richness and the proportion of semi-natural habitat types occurring in farms but not with the

cover of semi-natural habitats as an area percentage, suggesting that numerous habitat types, even small, have a greater impact than a few but large ones;

- There were apparent patterns of species richness decreasing on farms with increasing management intensity reflected by the management indicators;
- Contrasting management in organic and non-organic farms as reflected by the management indicators did not conduct to an apparent impact on species richness in all the case study regions. However, sub-indicators of pesticide use or field operations may show management system effects.

The selected indicators are the minimum set necessary to represent the types, functions and scale of activity of different organisms and a further reduction in the number of indicators would lead to a substantial loss of information. Nevertheless, we do not know about the efficiency of the specific taxa compare to other possible combinations, e.g. plants and carabid beetles, butterflies and birds, which could have shown the same kind of relationships. Testing the indicators in 12 case study regions revealed a huge variability of indicator values within (variability between farms) and between regions (due to geographical differences and differences of farm types). The relations between indicators of the remaining indicator set also differ strongly between case studies. A further reduction of the number of indicators would therefore lead to a substantial loss of information that cannot be substituted by the other indicators. An aggregation of the indicators to a single index is not possible due to the different properties of the indicators (they vary across case study regions) and was explicitly discarded by the stakeholders due to difficulties with the interpretation of such an index. Instead, some of the indicators should be further specified by sub-indicators (e.g. rare species, species composition, Herbicide / Fungicide / Insecticide Use in the case of pesticide use) and further analyzed.

The purpose of this report was a coarse screening of the performance of the BIOBIO indicator set in order to substantiate the selection of the indicators. Further in-depth analysis is now needed:

- For individual case studies;
- At the level of farm types (across case studies; e.g. all grassland farm types);
- For individual indicators (e.g. earthworms on farmland across Europe);
- To investigate specific hypothesis (e.g. "we expect a negative correlation between the number of pesticide applications and the richness of crop varieties");
- To evaluate the correlations of indicators at plot level (notably farm management and species richness).

References

Bailey, D., Herzog, F., Bogers, M., Lüscher, G., Fjellstad, W. 2012. Habitat Indicators. In: Biodiversity Indicators for European Farming Systems. Editors: Herzog, F., Balazs, K., Dennis, P., Jeanneret, P., Jongman, R., Kainz, M., Pointereau, P. ART Publication Series 17.

Borcard D., Gillet F. & Legendre P. (2011). Numerical Ecology with R. Springer. 306 pp.

Bunce R. G. H., et al. (2008). A standardized procedure for surveillance and monitoring European habitats and provision of spatial data. Landscape Ecology, 23: 11-25.

Chambers J. M. (1992). Linear models. Chapter 4 of Statistical Models in S, eds J. M. Chambers and T. J. Hastie, Wadsworth & Brooks/Cole.

Chao A. (1987). Estimating the population size for capture-recapture data with unequal catchability. Biometrics 43, 783–791.

Dennis P., Arndorfer M., Balázs K. et al. (2009). Conceptual foundations for biodiversity indicator selection for organic and low-input farming systems. Aberystwyth, Deliverable 2.1 of the EU FP7 Project BIOBIO. ISBN 978-3-905733-16-7. http://www.biobio-indicator.org/deliverables.php.

Dennis P., Bogers M.M.B., Bunce R.G.H., Herzog F. & Jeanneret P. (2012). Biodiversity in organic and low-input farming systems. Wageningen, Deliverable 2.2 of the EU FP7 Project BIOBIO. ISBN 1566-7197. ALTERRA Report 2308. http://www.biobio-indicator.org/deliverables.php.

Herzog F., Balázs K., Dennis P., Friedel J., Jeanneret P., Geijzendorffer I., Kainz M., Pointereau P. (2012a). Biodiversity Indicators for European Farming Systems: A Guidebook. Deliverable 6.3 of the EU FP7 Project BIOBIO. http://www.biobio-indicator.org/deliverables.php.

Herzog F., Balázs K., Dennis P., Friedel J., Jeanneret P., Jongman R.G.H., Kainz M., Pointereau P. (2012b). Report on the final indicator set after stakeholder audit. Deliverable 4.3 of the EU FP7 Project BIOBIO. http://www.biobio-indicator.org/deliverables.php.

Hollander M. & Wolfe D.A. (1973), Nonparametric Statistical Methods. New York: John Wiley & Sons. Pages 185–194 (Kendall and Spearman tests).

Legendre P. & Legendre L. (1998). Numerical Ecology. Elsevier, Amsterdam. 853 pp.

Nascimbene J., Marini L. & Paoletti M. G. (2012). Organic Farming Benefits Local Plant Diversity in Vineyard Farms Located in Intensive Agricultural Landscapes. Environmental Management 49:1054-1060.

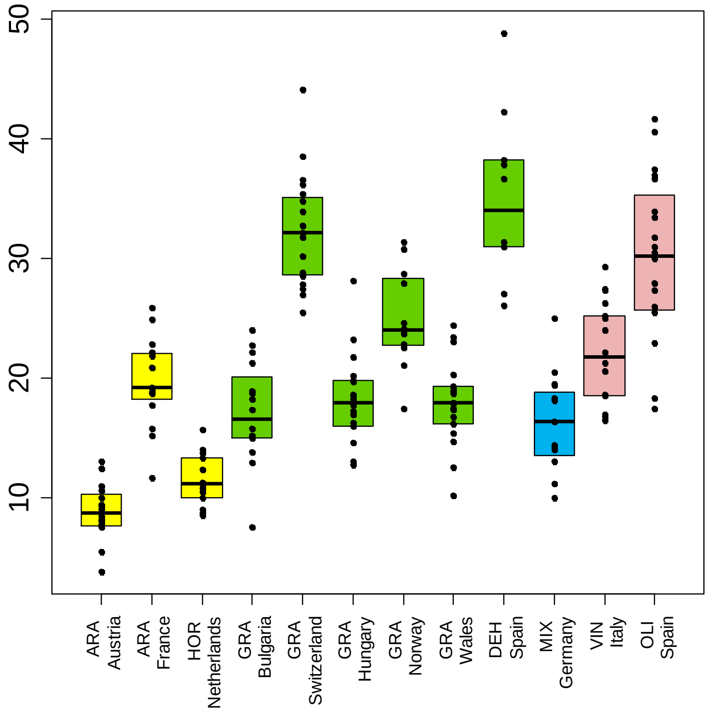
Oksanen J., Blanchet F. G., Kindt R., Legendre P., O'Hara B., Simpson G. L., Solymos P., Stevens M. H. H. & Wagner H. (2010). vegan: Community ecology package. R package Version 1.17-3.

R Development Core Team (2011). R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

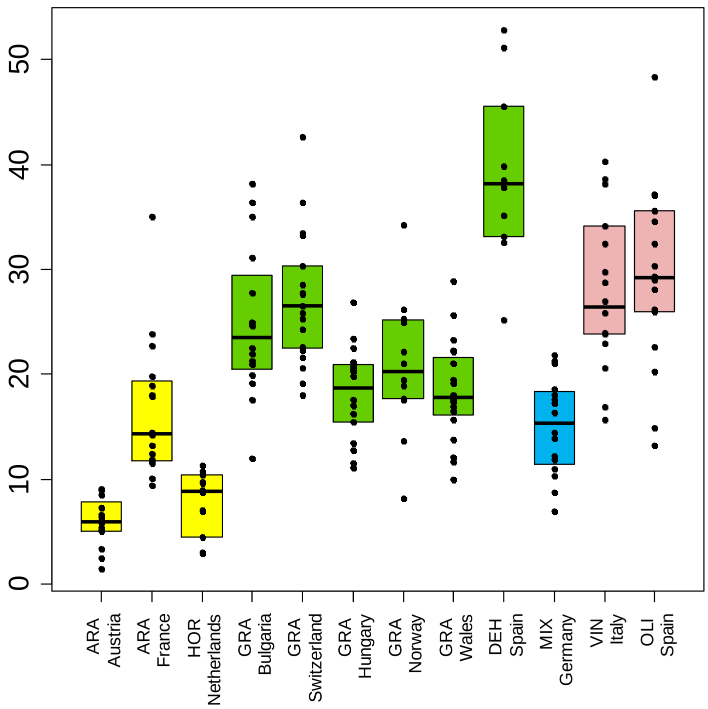
Targetti S., Viaggi D., Cuming D. (2011) Analysis of Costs and Efforts of BioBio Indicators of Biodiversity. Deliverable 3.3 of the EU FP7 Project BioBio (unpublished).

APPENDIX 1 – SPECIES RICHNESS INDICES

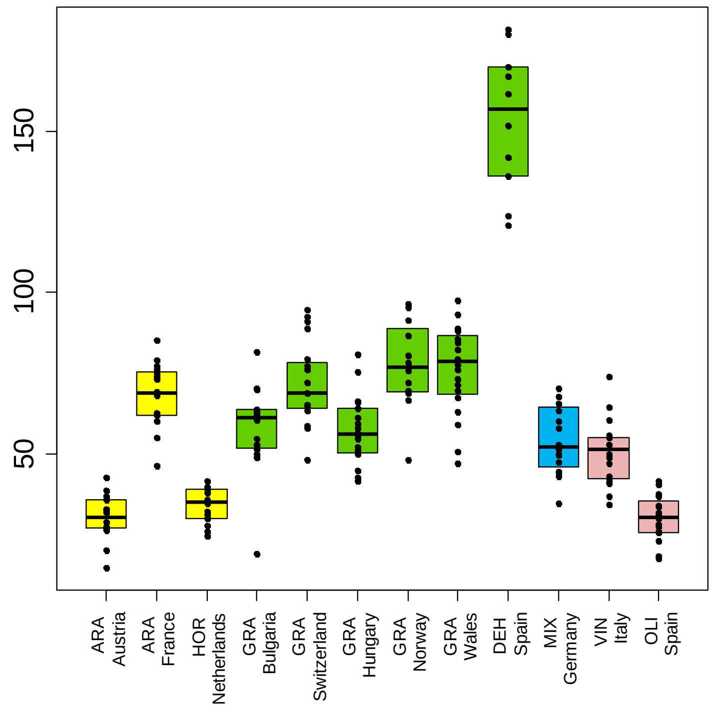
B2_1.2) *Alpha diversity* - plants of cultivated forage, food crops and semi-natural habitats.
Alpha diversity: average number of species over the habitats.



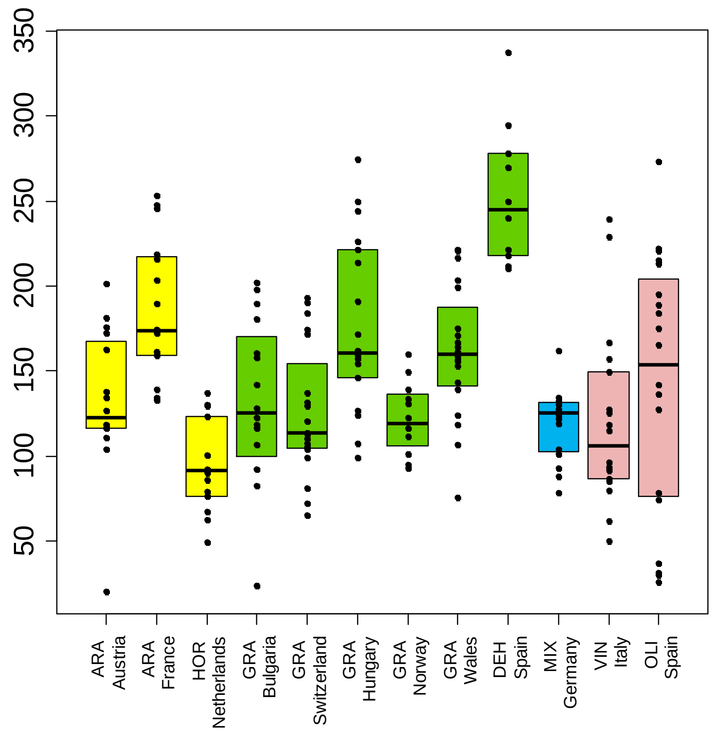
B2_1.3) *Area weighted diversity* - plants of cultivated forage, food crops and semi-natural habitats.
Area weighted diversity: number of species over the habitats weighted by the area of the habitats.



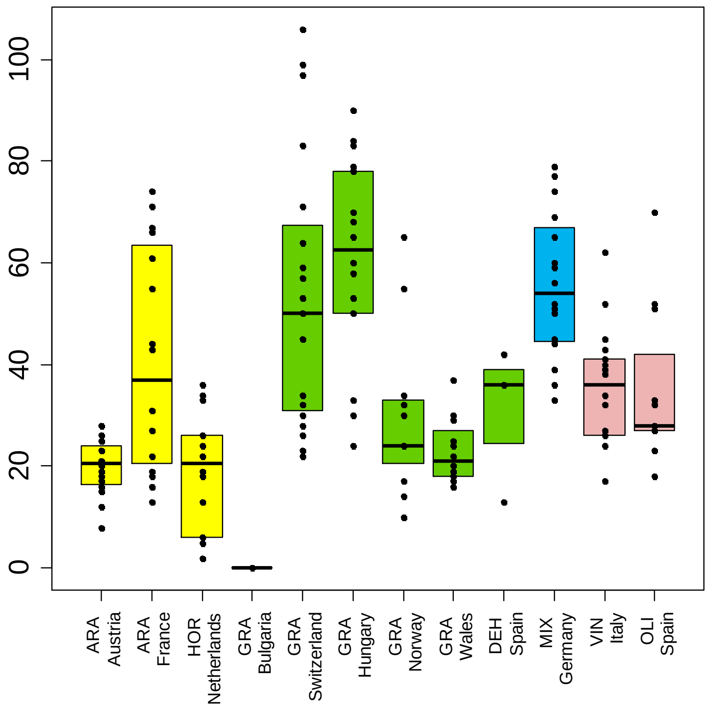
B2_1.4) *Rarefied richness* - plants of cultivated forage, food crops and semi-natural habitats.
Rarefied richness: average number of species over the smallest number of plots found in a farm.



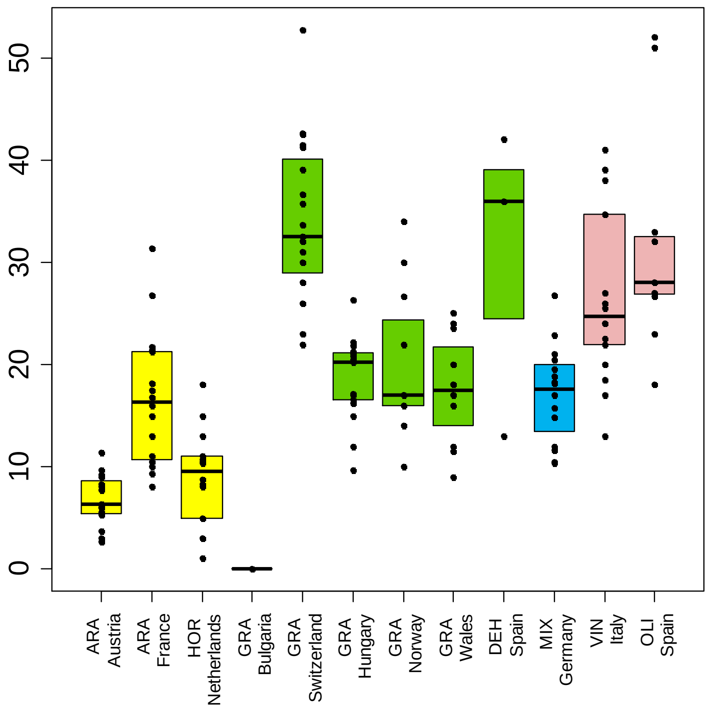
B2_1.5) *Chao estimated richness* - plants of cultivated forage, food crops and semi-natural habitats.
Chao estimated richness: extrapolated number of species based on the accumulated number of species found in plots.



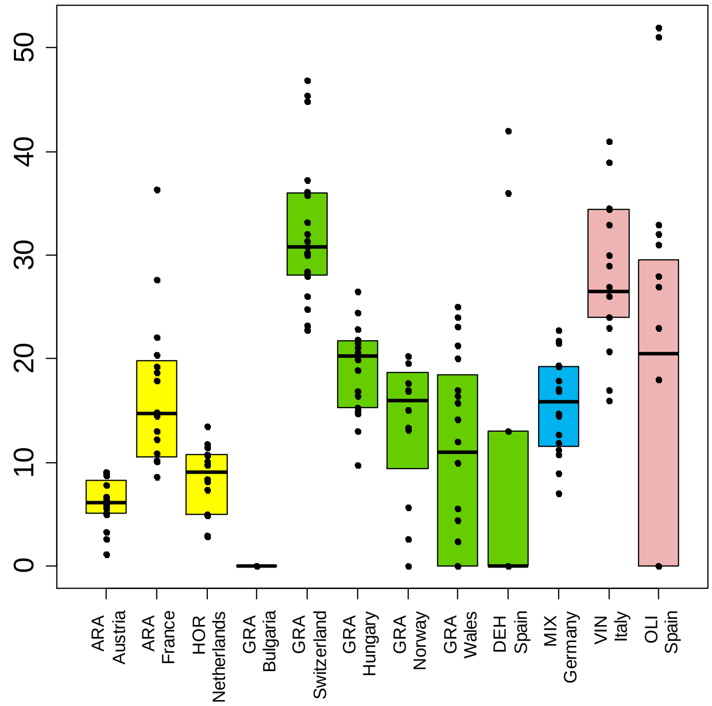
B2_2.1) *Gamma diversity* - plants of cultivated forage and food crops.



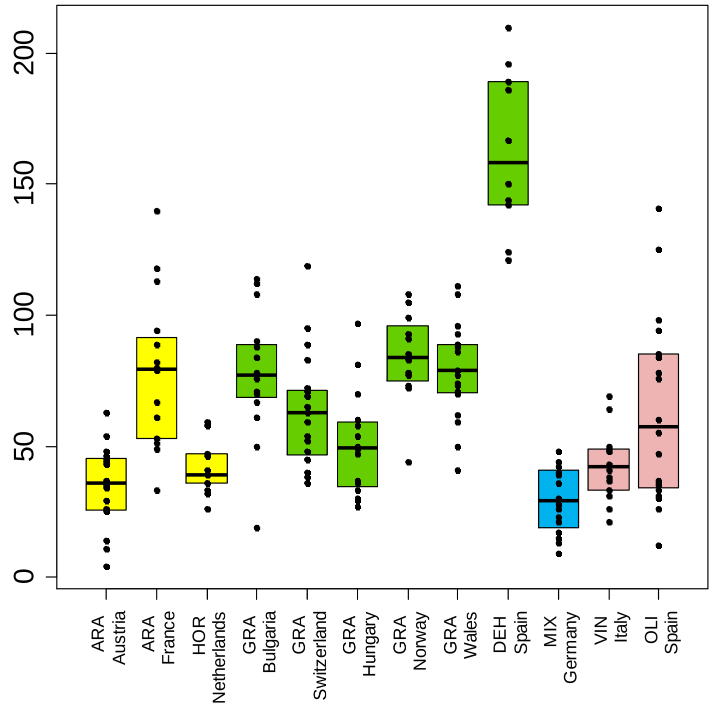
B2_2.2) *Alpha diversity* - plants of cultivated forage and food crops.



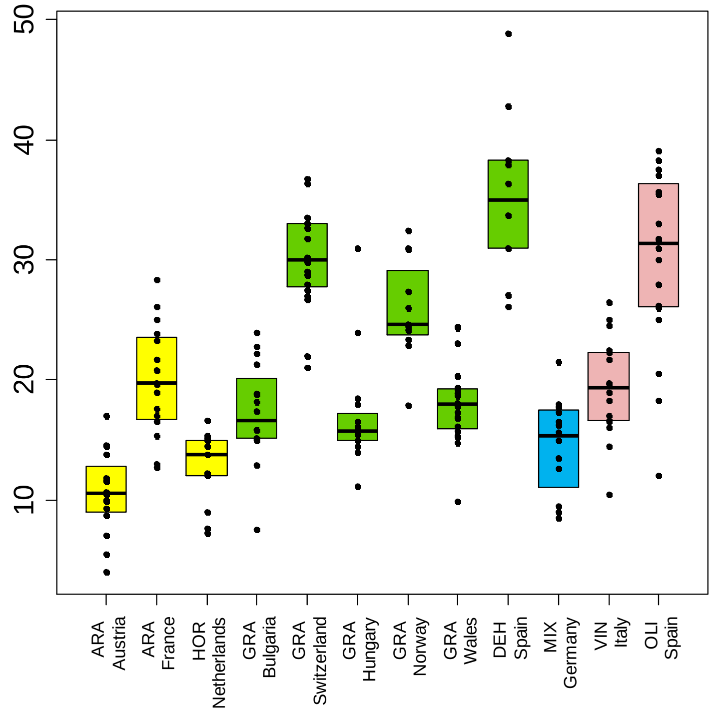
B2_2.3) *Area weighted diversity* - plants of cultivated forage and food.



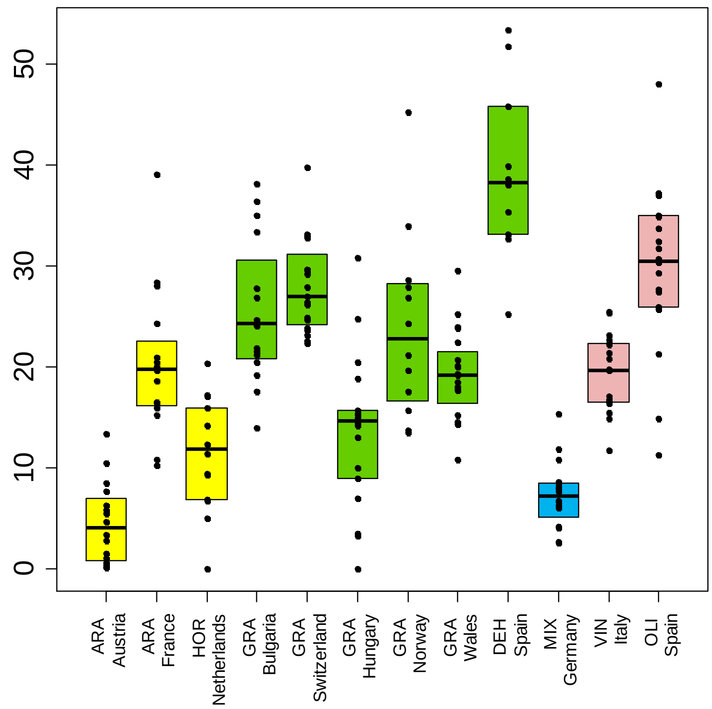
B2_3.1) *Gamma diversity* - plants of semi-natural habitats.



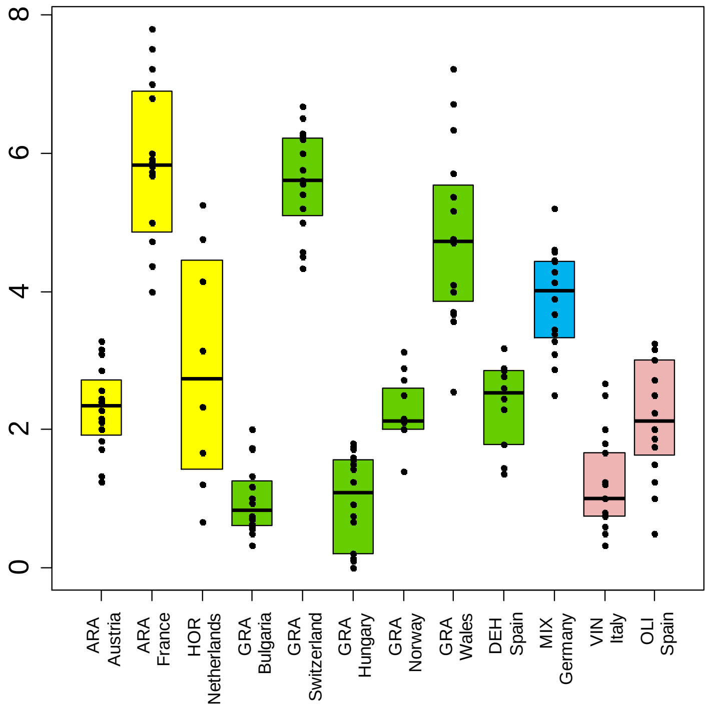
B2_3.2) *Alpha diversity* - plants of semi-natural habitats.



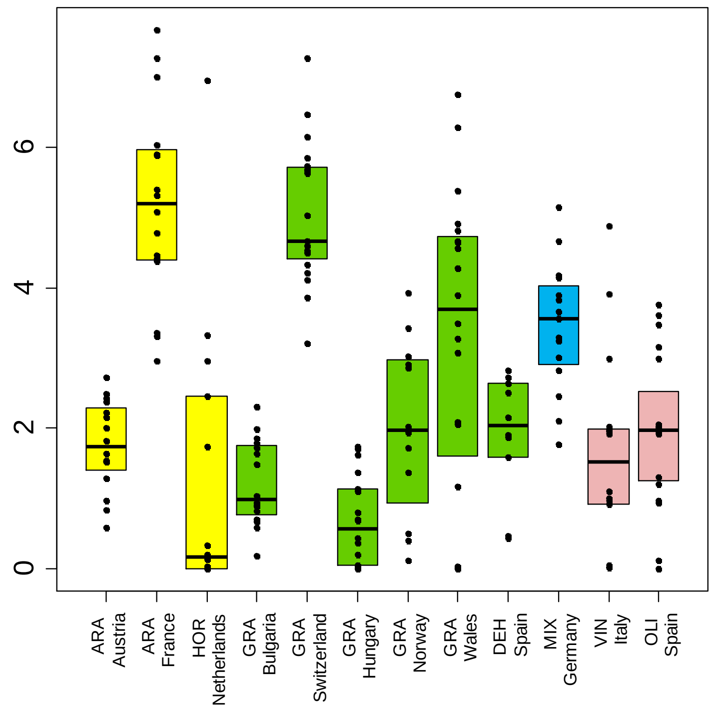
B2_3.3) *Area weighted diversity* - plants of semi-natural habitats.



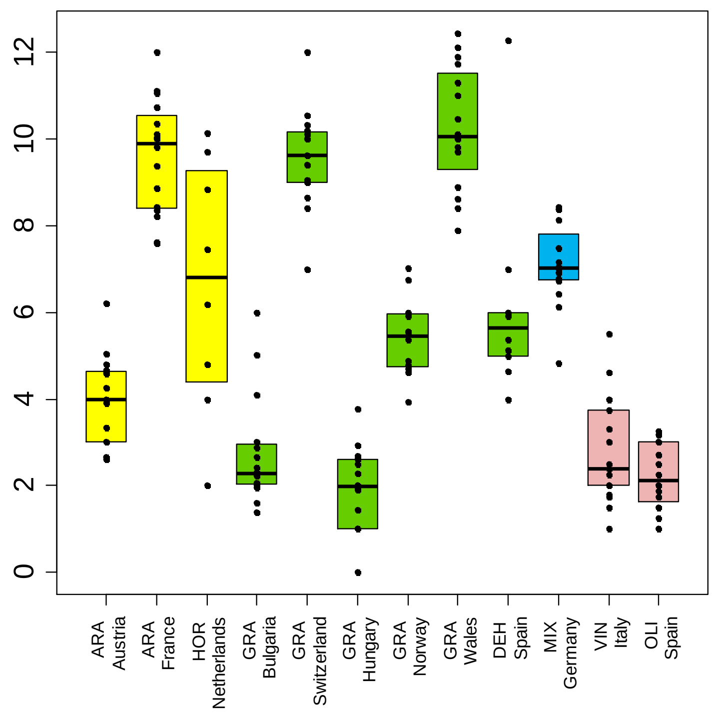
B4_1.2) *Alpha diversity* - earthworms of cultivated forage, food crops and semi-natural habitats.
Alpha diversity: average number of species over the habitats.



B4_1.3) *Area weighted diversity* - earthworms of cultivated forage, food crops and semi-natural habitats.
Area weighted diversity: number of species over the habitats weighted by the area of the habitats.

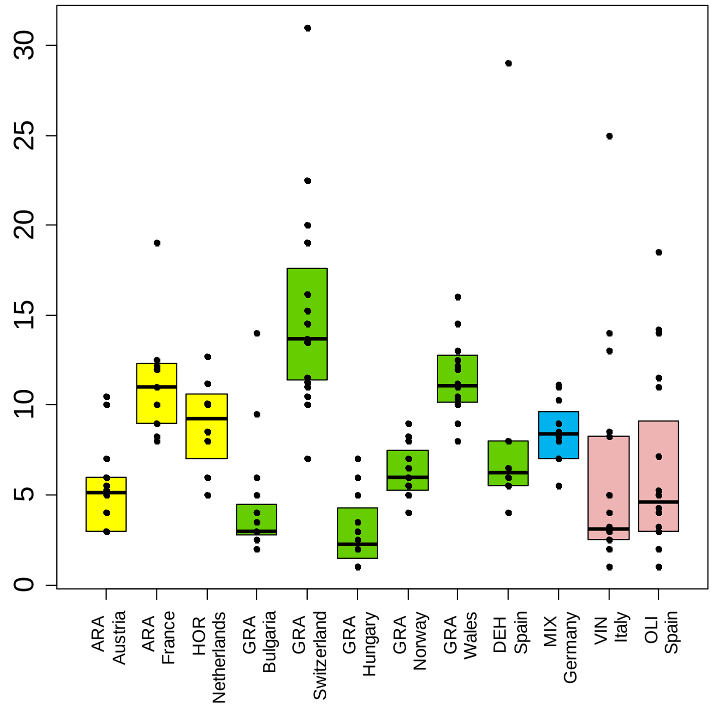


B4_1.4) *Rarefied richness* - earthworms of cultivated forage, food crops and semi-natural habitats. *Rarefied richness*: average number of species over the smallest number of plots found in a farm.

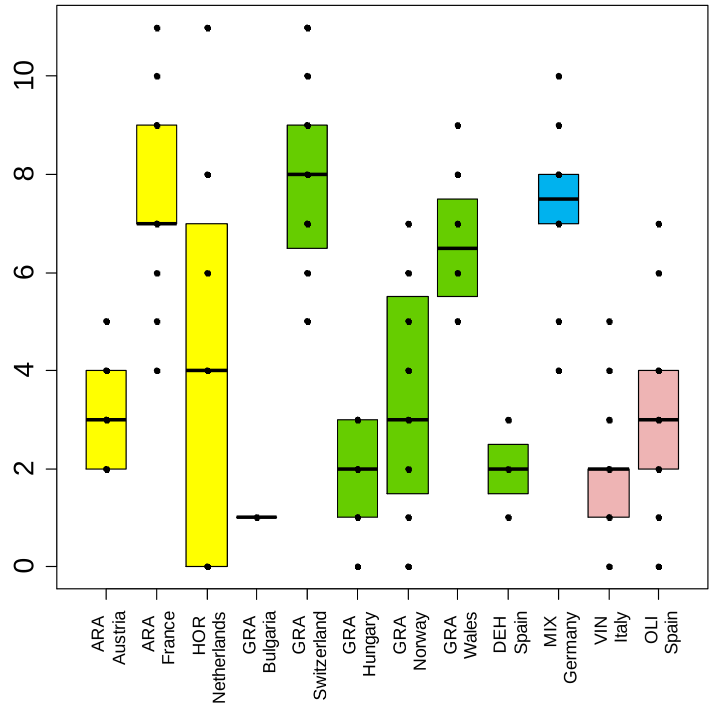


B4_1.5) *Chao estimated richness* - earthworms of cultivated forage, food crops and semi-natural habitats.

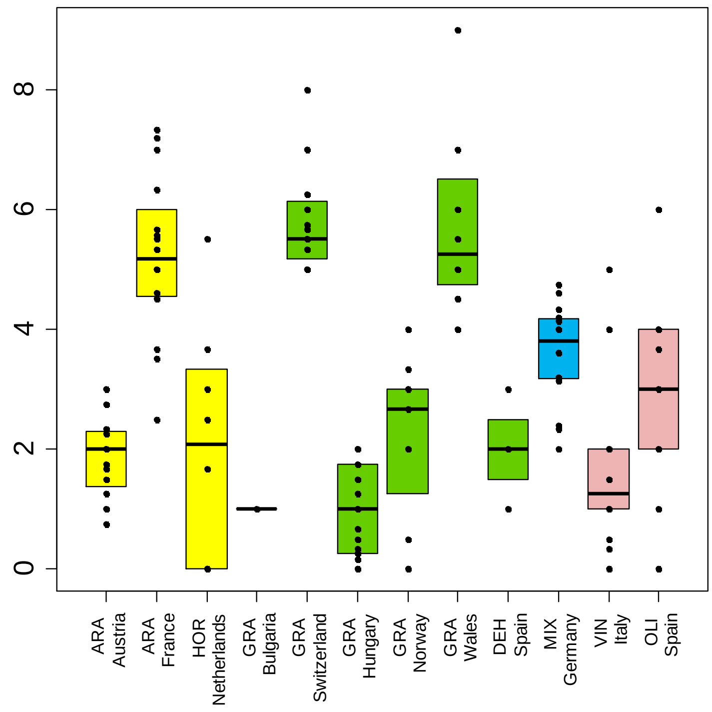
Chao estimated richness: extrapolated number of species based on the accumulated number of species found in plots.



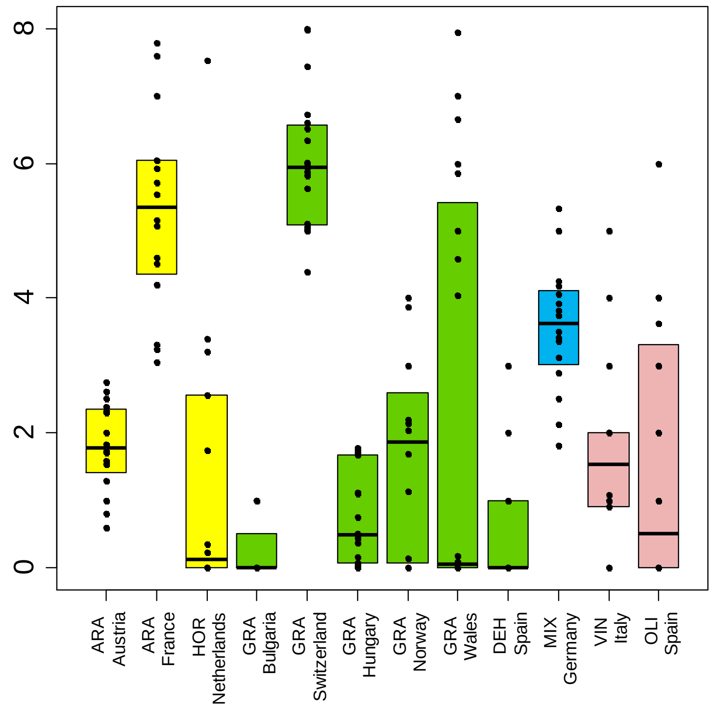
B4_2.1) *Gamma diversity* - earthworms of cultivated forage and food crops.



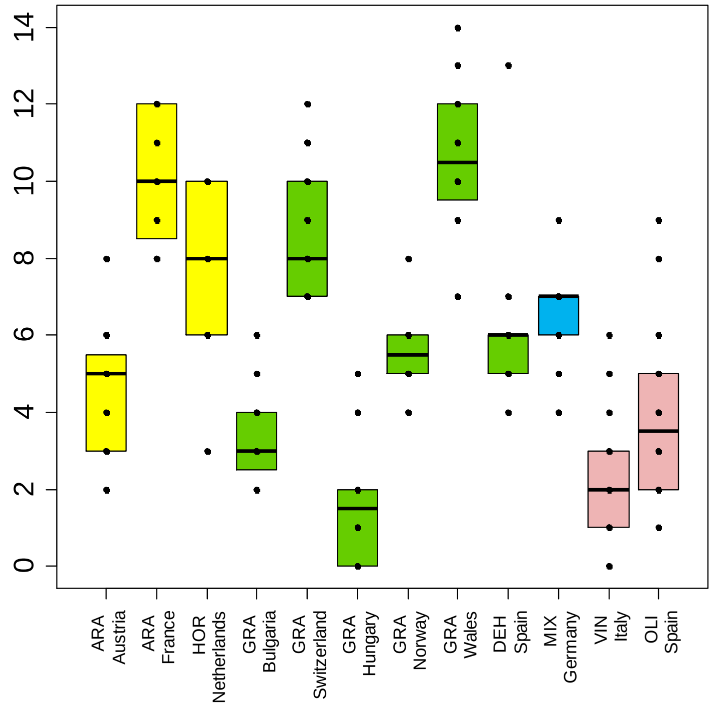
B4_2.2) *Alpha diversity* - earthworms of cultivated forage and food crops.



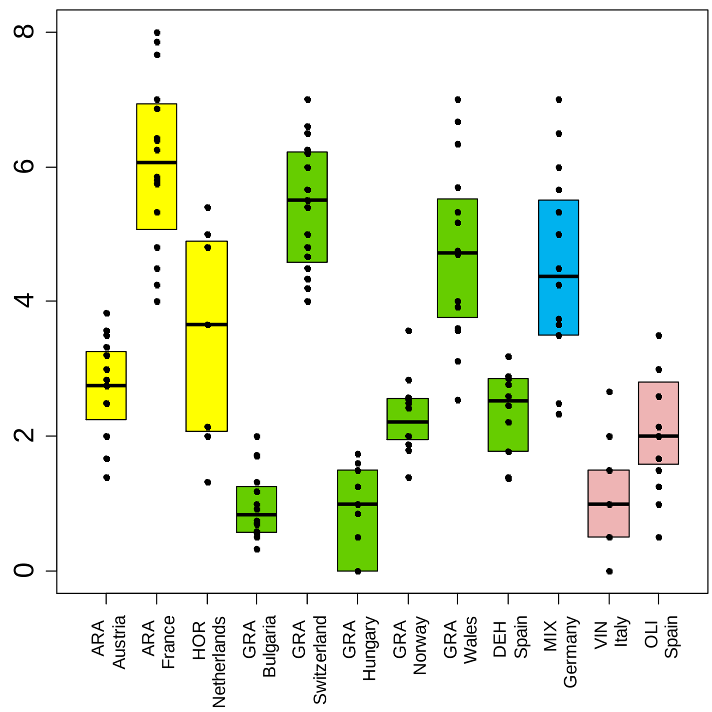
B4_2.3) *Area weighted diversity* - earthworms of cultivated forage and food.



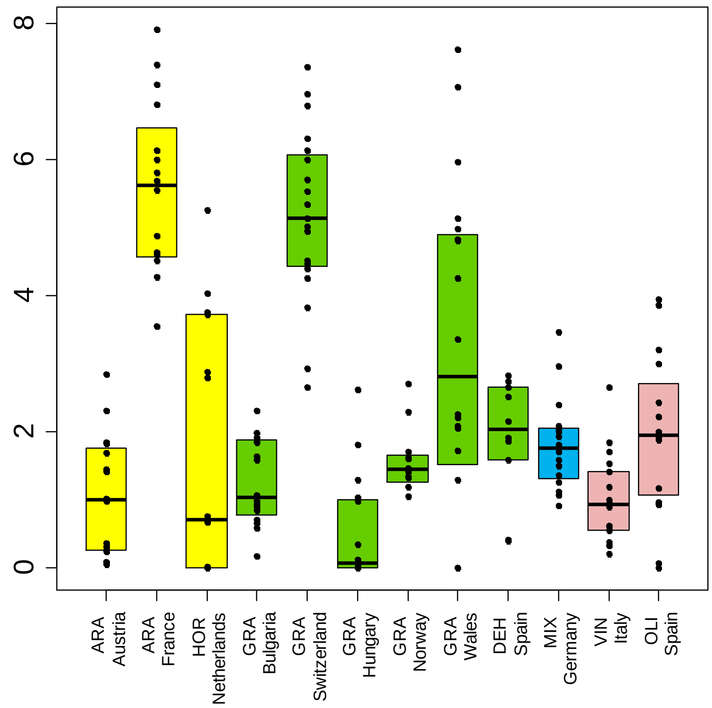
B4_3.1) *Gamma diversity* - earthworms of semi-natural habitats.



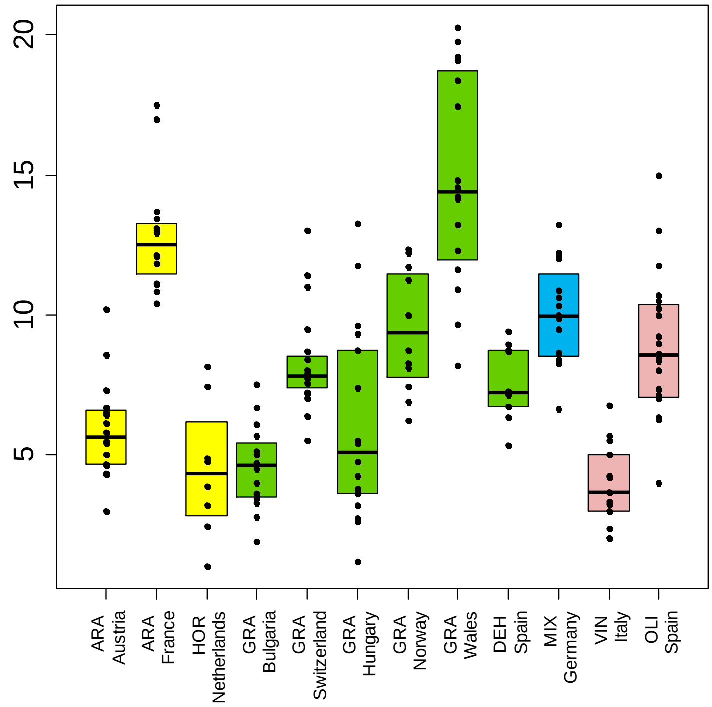
B4_3.2) *Alpha diversity* - earthworms of semi-natural habitats.



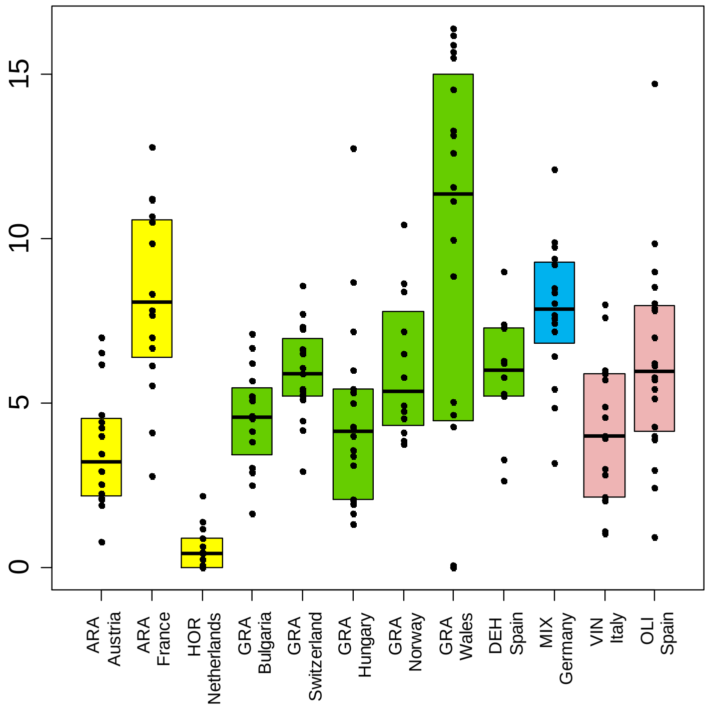
B4_3.3) *Area weighted diversity* - earthworms of semi-natural habitats.



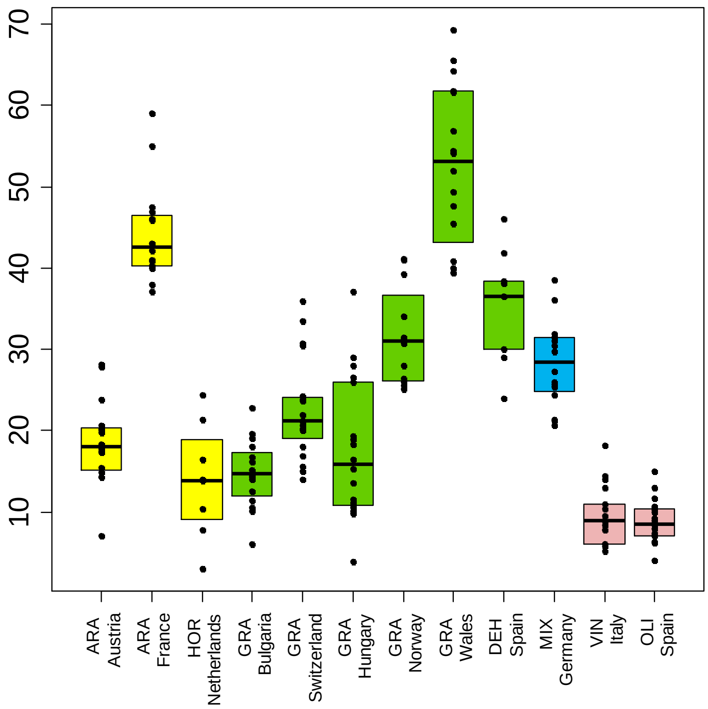
B8_1.2) *Alpha diversity* - spiders of cultivated forage, food crops and semi-natural habitats.
Alpha diversity: average number of species over the habitats.



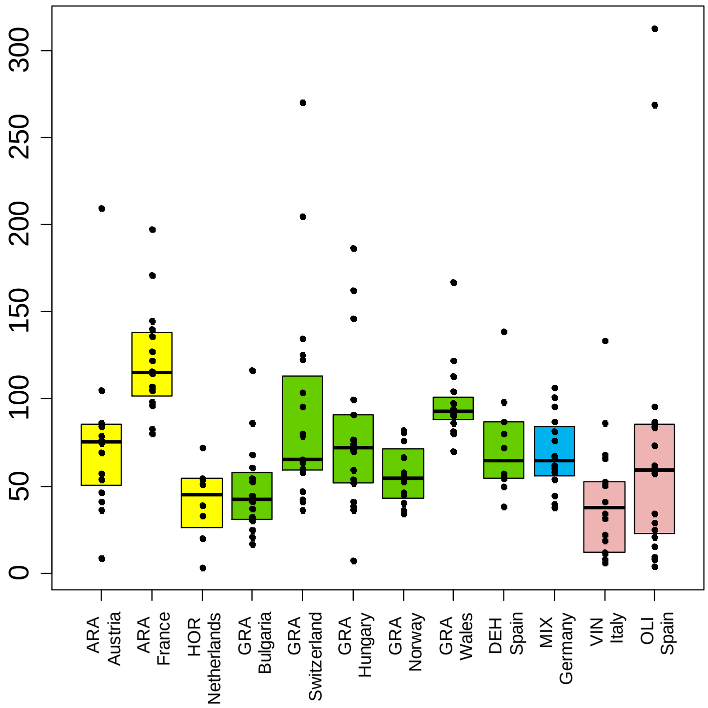
B8_1.3) *Area weighted diversity* - spiders of cultivated forage, food crops and semi-natural habitats.
Area weighted diversity: number of species over the habitats weighted by the area of the habitats.



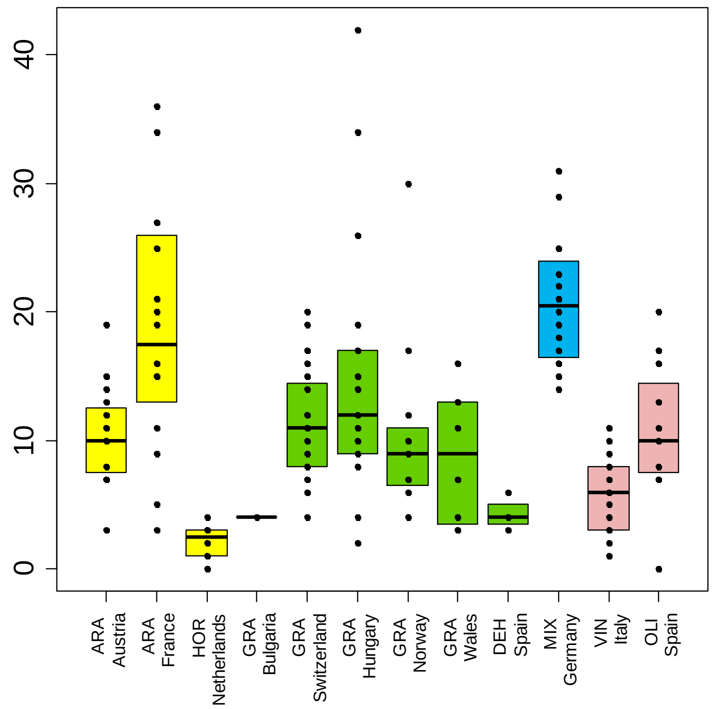
B8_1.4) *Rarefied richness* - spiders of cultivated forage, food crops and semi-natural habitats.
Rarefied richness: average number of species over the smallest number of plots found in a farm.



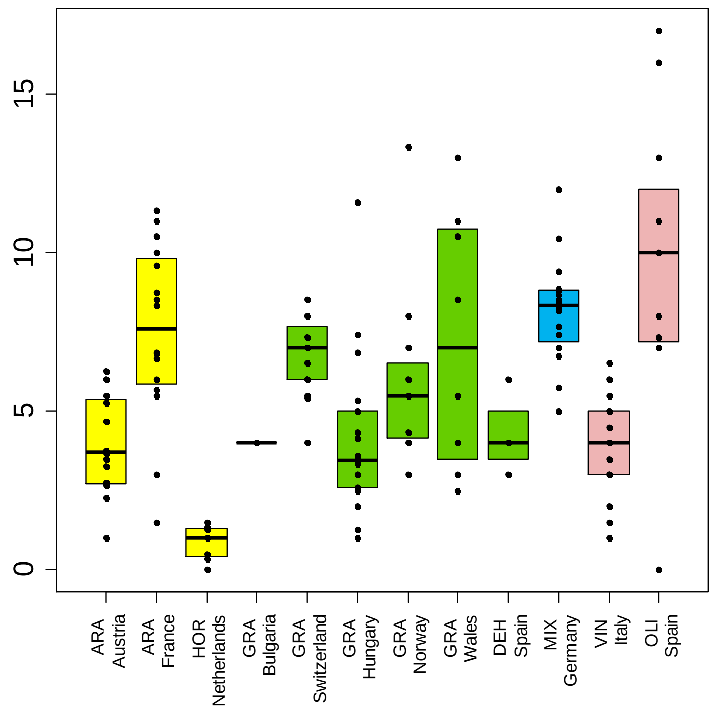
B8_1.5) *Chao estimated richness* - spiders of cultivated forage, food crops and semi-natural habitats.
Chao estimated richness: extrapolated number of species based on the accumulated number of species found in plots.



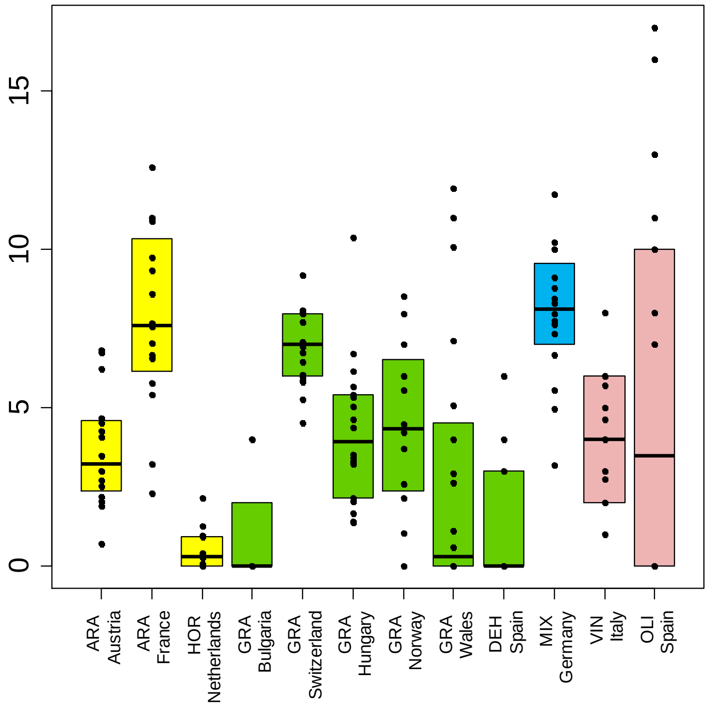
B8_2.1) *Gamma diversity* - spiders of cultivated forage and food crops.



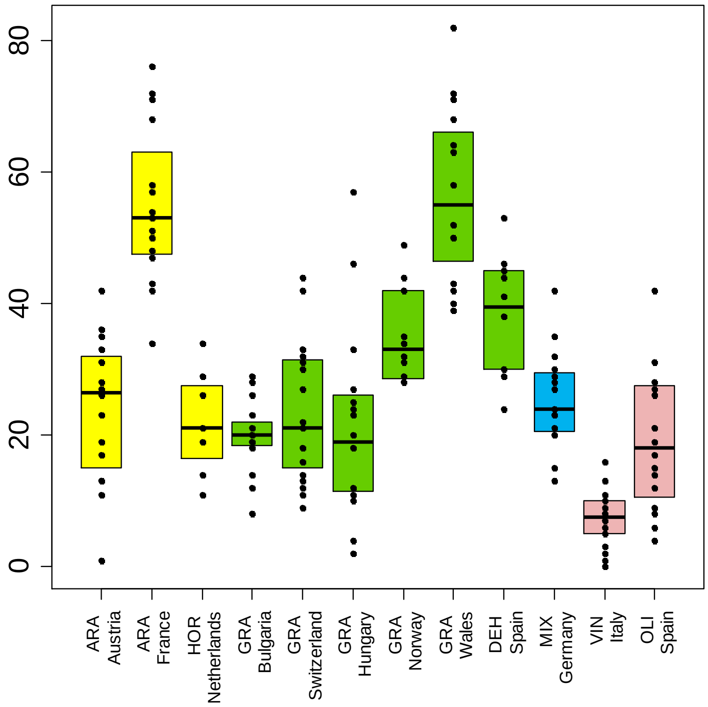
B8_2.2) *Alpha diversity* - spiders of cultivated forage and food crops.



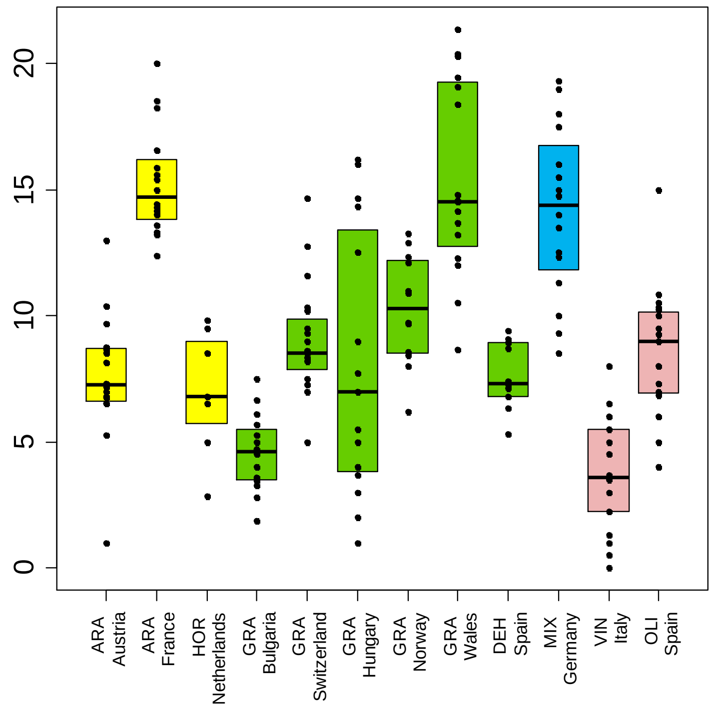
B8_2.3) *Area weighted diversity* - spiders of cultivated forage and food.



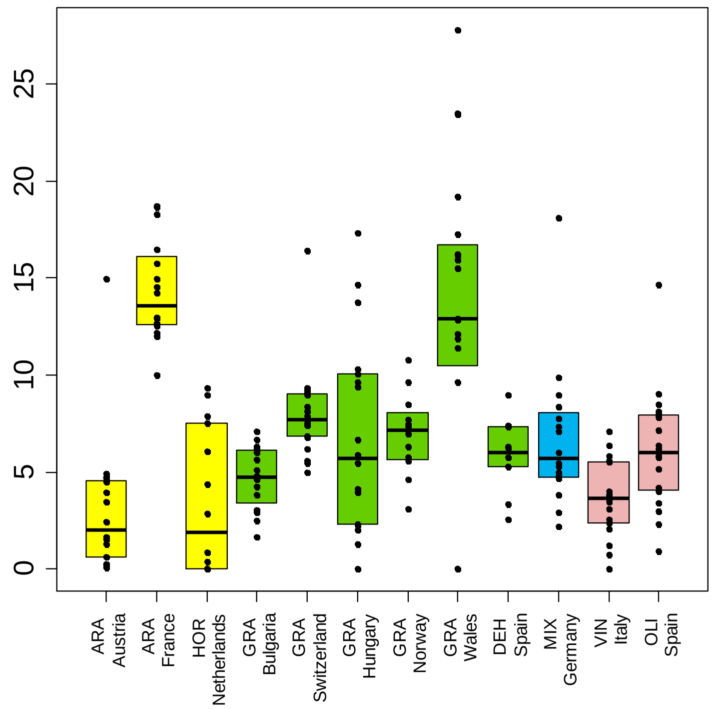
B8_3.1) *Gamma diversity* - spiders of semi-natural habitats.



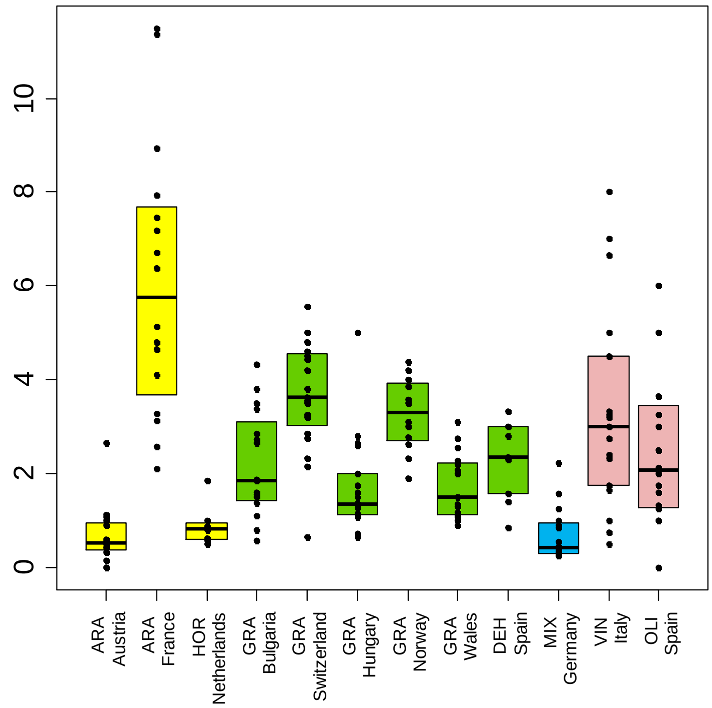
B8_3.2) *Alpha diversity* - spiders of semi-natural habitats.



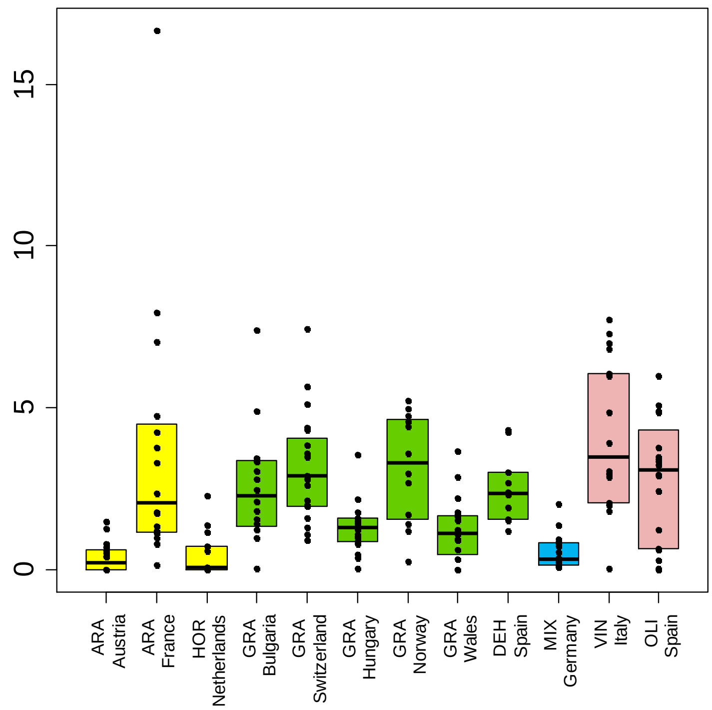
B8_3.3) *Area weighted diversity* - spiders of semi-natural habitats.



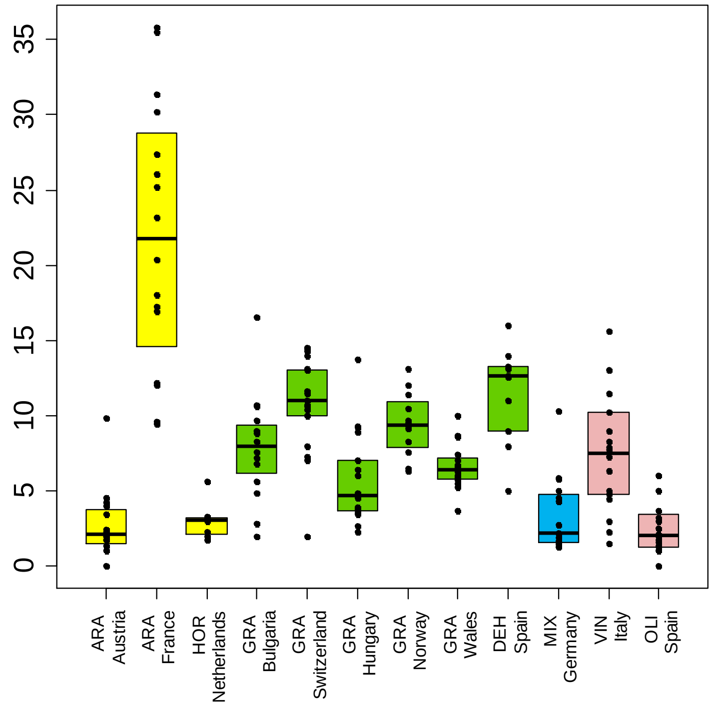
B9_1.2) *Alpha diversity* - bees of cultivated forage, food crops and semi-natural habitats.
Alpha diversity: average number of species over the habitats.



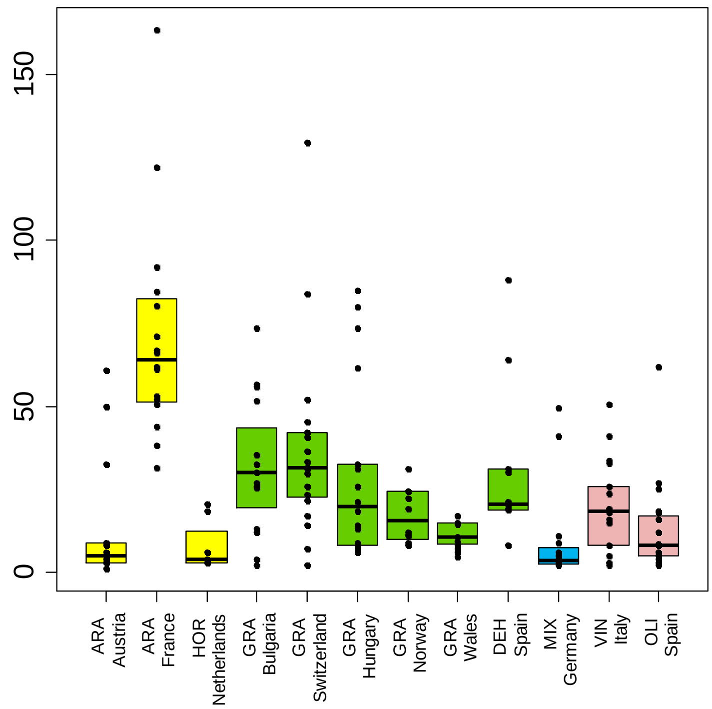
B9_1.3) *Area weighted diversity* - bees of cultivated forage, food crops and semi-natural habitats.
Area weighted diversity: number of species over the habitats weighted by the area of the habitats.



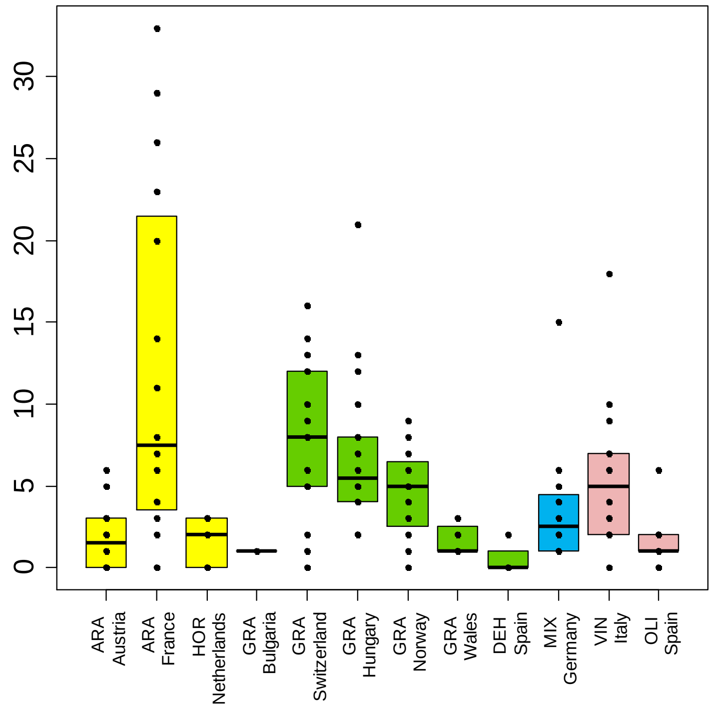
B9_1.4) *Rarefied richness* - bees of cultivated forage, food crops and semi-natural habitats.
Rarefied richness: average number of species over the smallest number of plots found in a farm.



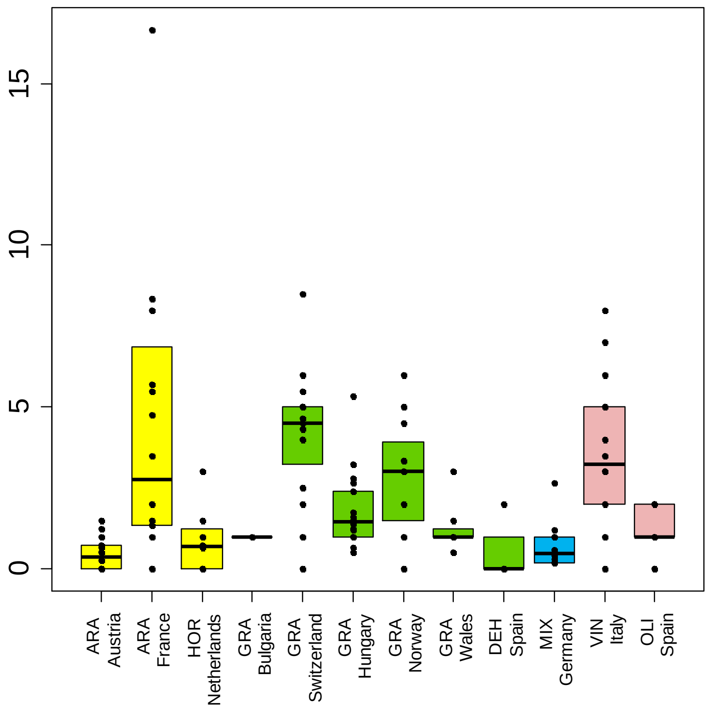
B9_1.5) *Chao estimated richness* - bees of cultivated forage, food crops and semi-natural habitats.
Chao estimated richness: extrapolated number of species based on the accumulated number of species found in plots.



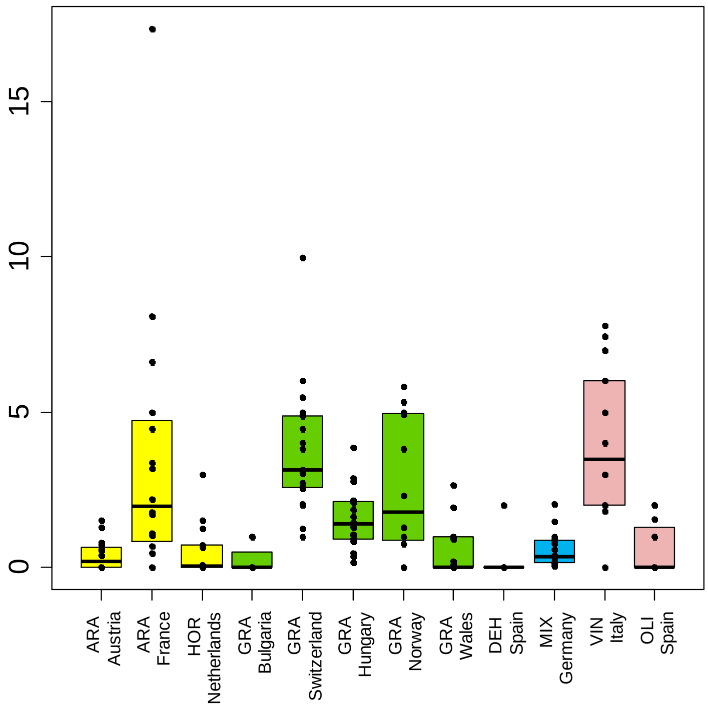
B9_2.1) *Gamma diversity* - bees of cultivated forage and food crops.



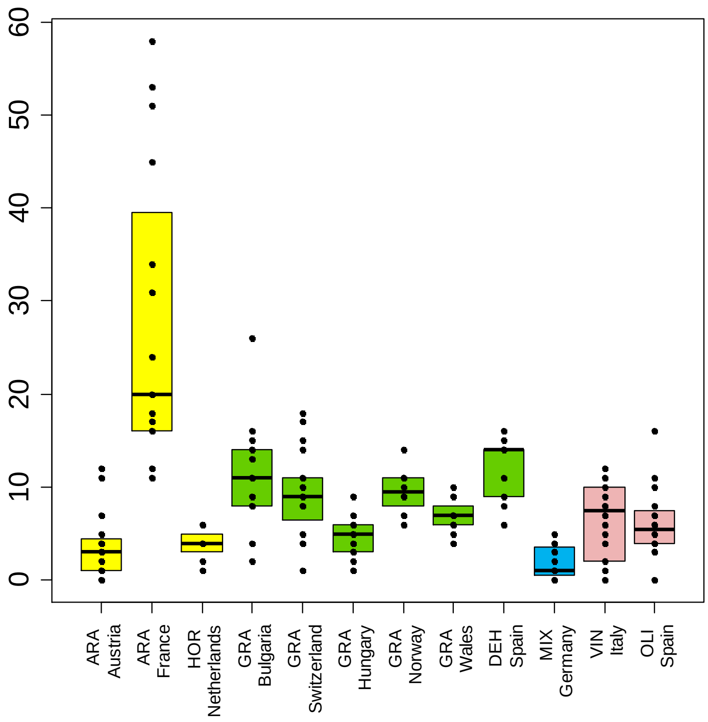
B9_2.2) *Alpha diversity* - bees of cultivated forage and food crops.



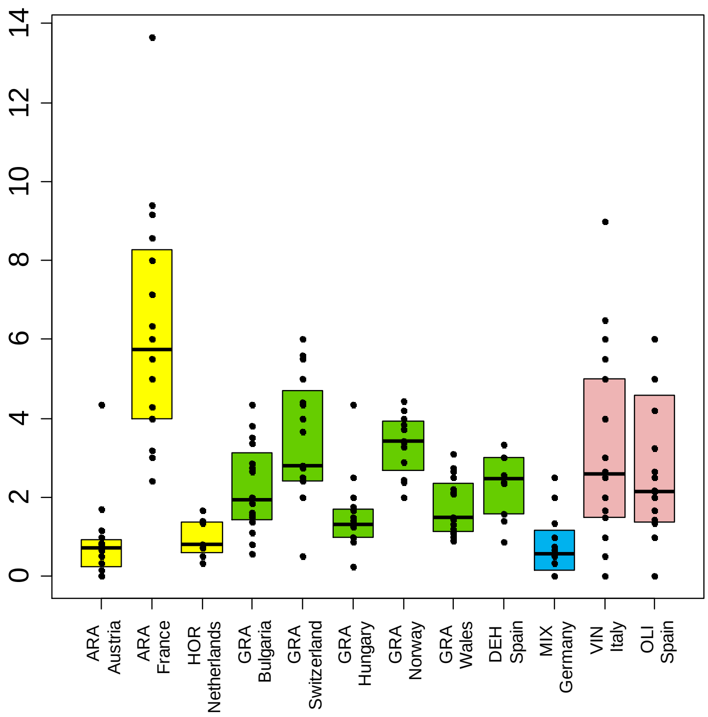
B9_2.3) *Area weighted diversity* - bees of cultivated forage and food.



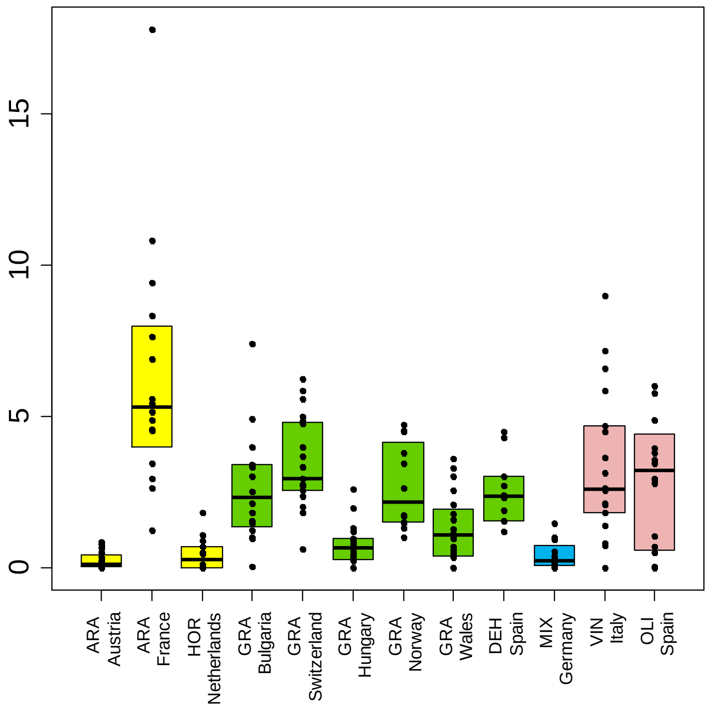
B9_3.1) *Gamma diversity* - bees of semi-natural habitats.



B9_3.2) *Alpha diversity* - bees of semi-natural habitats.

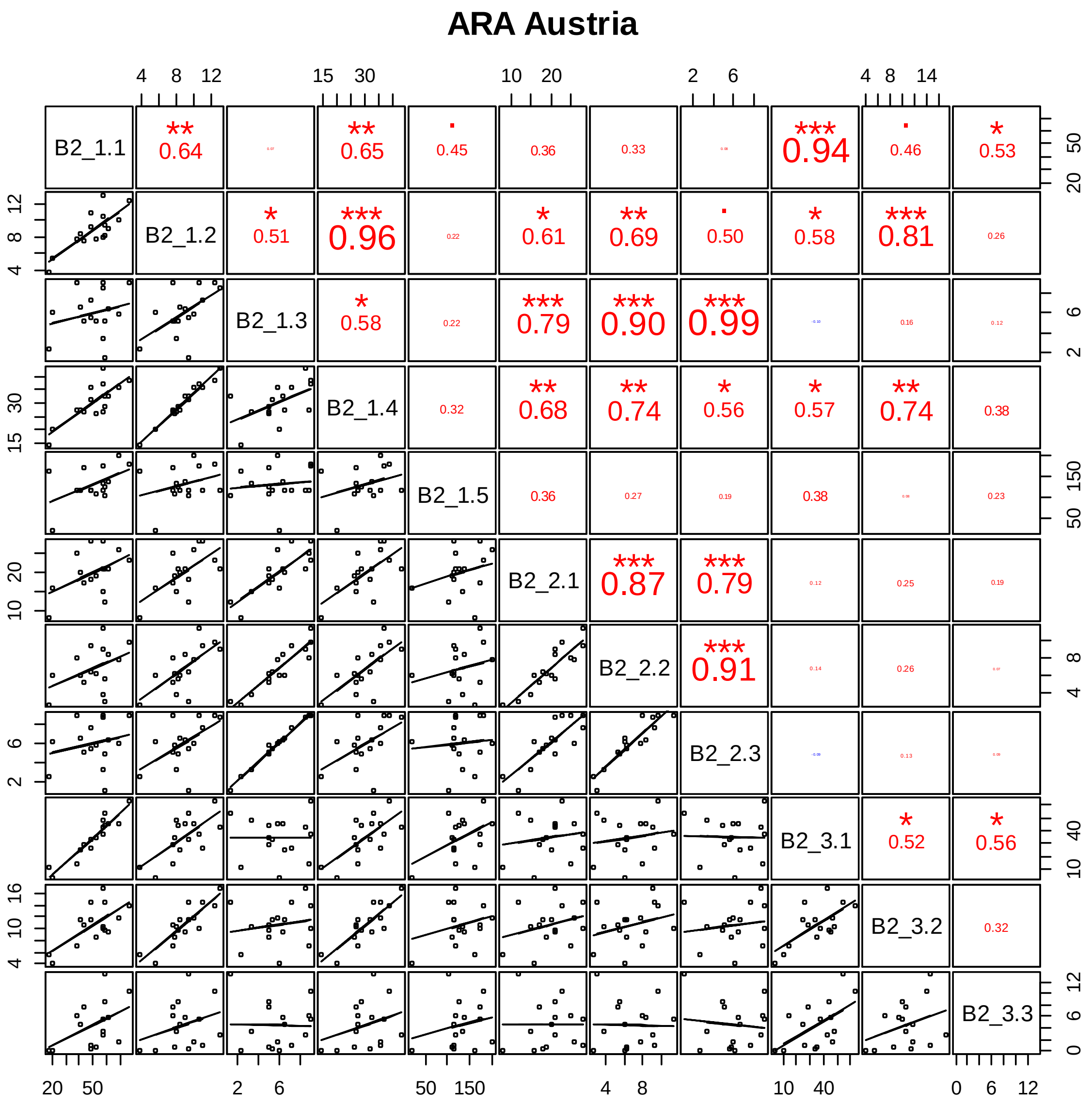


B9_3.3) *Area weighted diversity* - bees of semi-natural habitats.

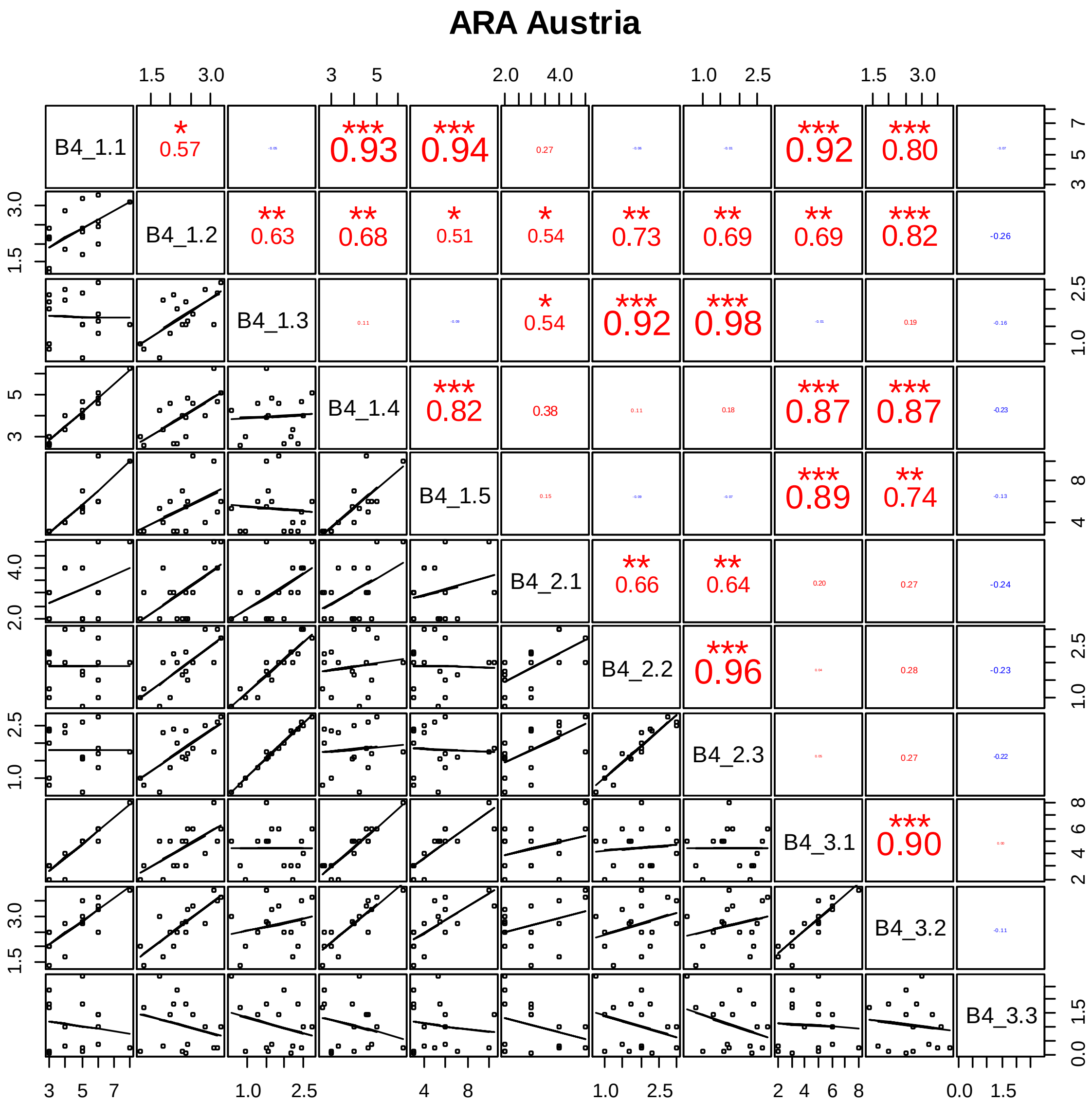


Appendix 2 – correlations between species richness indices within case study regions

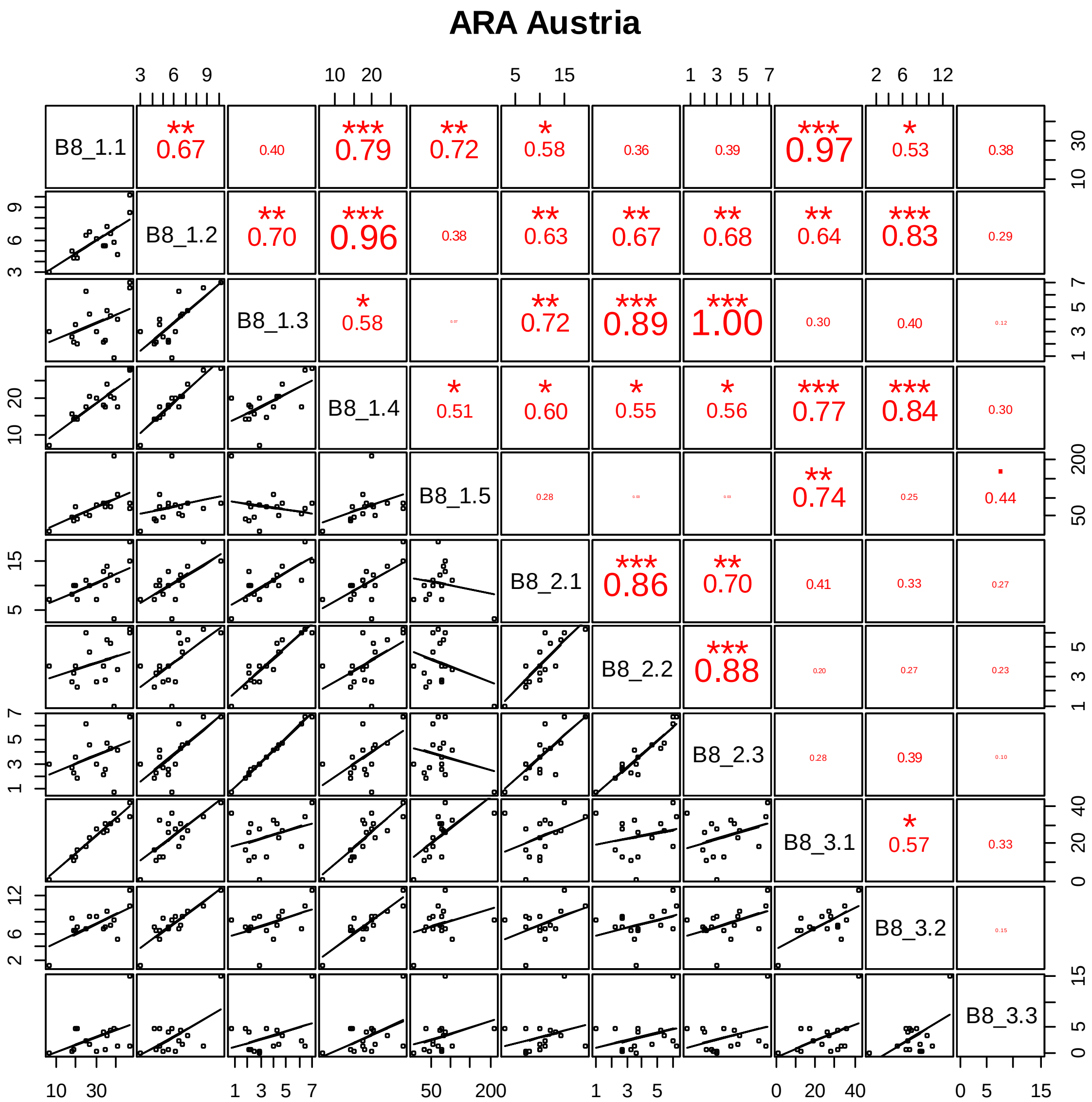
Plant diversity: Correlogram of species diversity indices of plant calculated from the species list with abundance. B2_1.1 = Gamma richness of plant in farms, B2_1.2 = Alpha richness, B2_1.3 = Area weighted richness, B2_1.4 = Rarefied richness, B2_1.5 = Chao estimated richness, B2_2.1 = Gamma richness of plant in cultivated forage and food crops, B2_2.2 = Alpha richness of plant in cultivated forage and food crops, B2_2.3 = Area weighted richness of plant in cultivated forage and food crops, B2_3.1 = Gamma richness of plant in semi-natural habitats, B2_3.2 = Alpha richness of plant in semi-natural habitats, B2_3.3 = Area weighted richness of plant in semi-natural habitats.



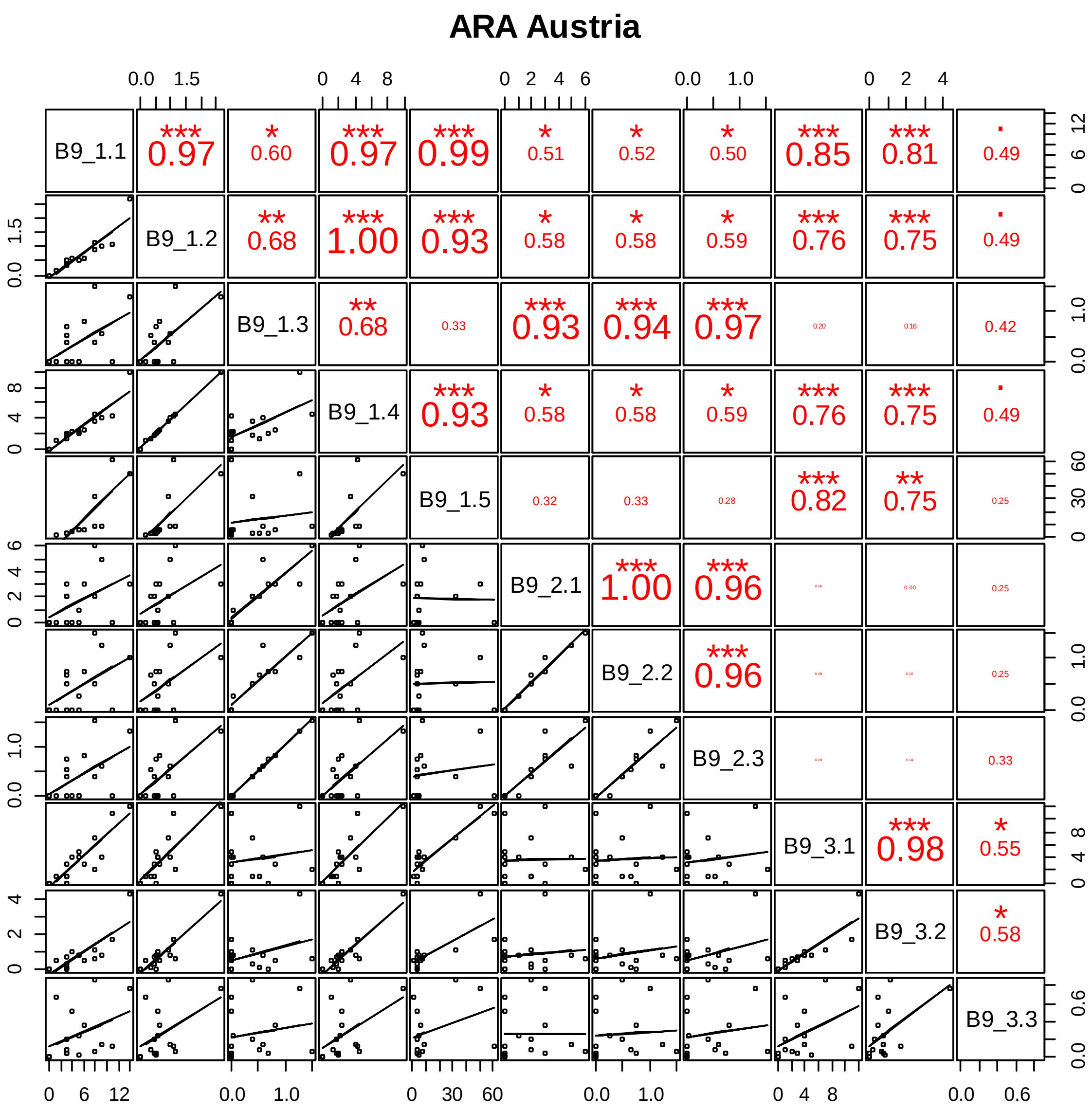
Earthworm diversity: Correlogram of species diversity indices of earthworms calculated from the species list with abundance. B4_1.1 = Gamma richness of earthworms in farms, B4_1.2 = Alpha richness, B4_1.3 = Area weighted richness, B4_1.4 = Rarefied richness, B4_1.5 = Chao estimated richness, B4_2.1 = Gamma richness of earthworms in cultivated forage and food crops, B4_2.2 = Alpha richness of earthworms in cultivated forage and food crops, B4_2.3 = Area weighted richness of earthworms in cultivated forage and food crops, B4_3.1 = Gamma richness of earthworms in semi-natural habitats, B4_3.2 = Alpha richness of earthworms in semi-natural habitats, B4_3.3 = Area weighted richness of earthworms in semi-natural habitats.



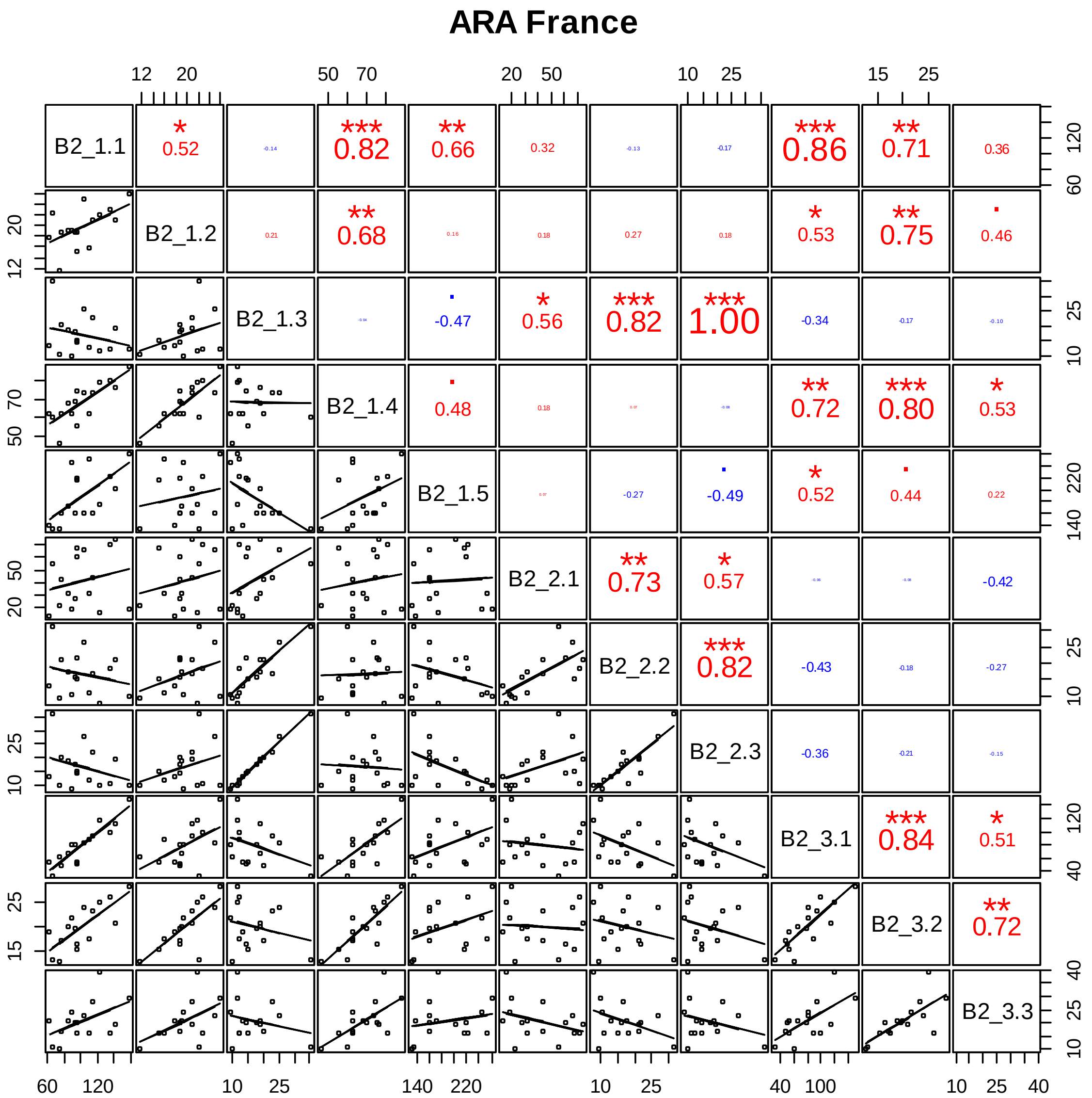
Spider diversity: Correlogram of species diversity indices of spiders calculated from the species list with abundance. B8_1.1 = Gamma richness of spiders in farms, B8_1.2 = Alpha richness, B8_1.3 = Area weighted richness, B8_1.4 = Rarefied richness, B8_1.5 = Chao estimated richness, B8_2.1 = Gamma richness of spiders in cultivated forage and food crops, B8_2.2 = Alpha richness of spiders in cultivated forage and food crops, B8_2.3 = Area weighted richness of spiders in cultivated forage and food crops, B8_3.1 = Gamma richness of spiders in semi-natural habitats, B8_3.2 = Alpha richness of spiders in semi-natural habitats, B8_3.3 = Area weighted richness of spiders in semi-natural habitats.



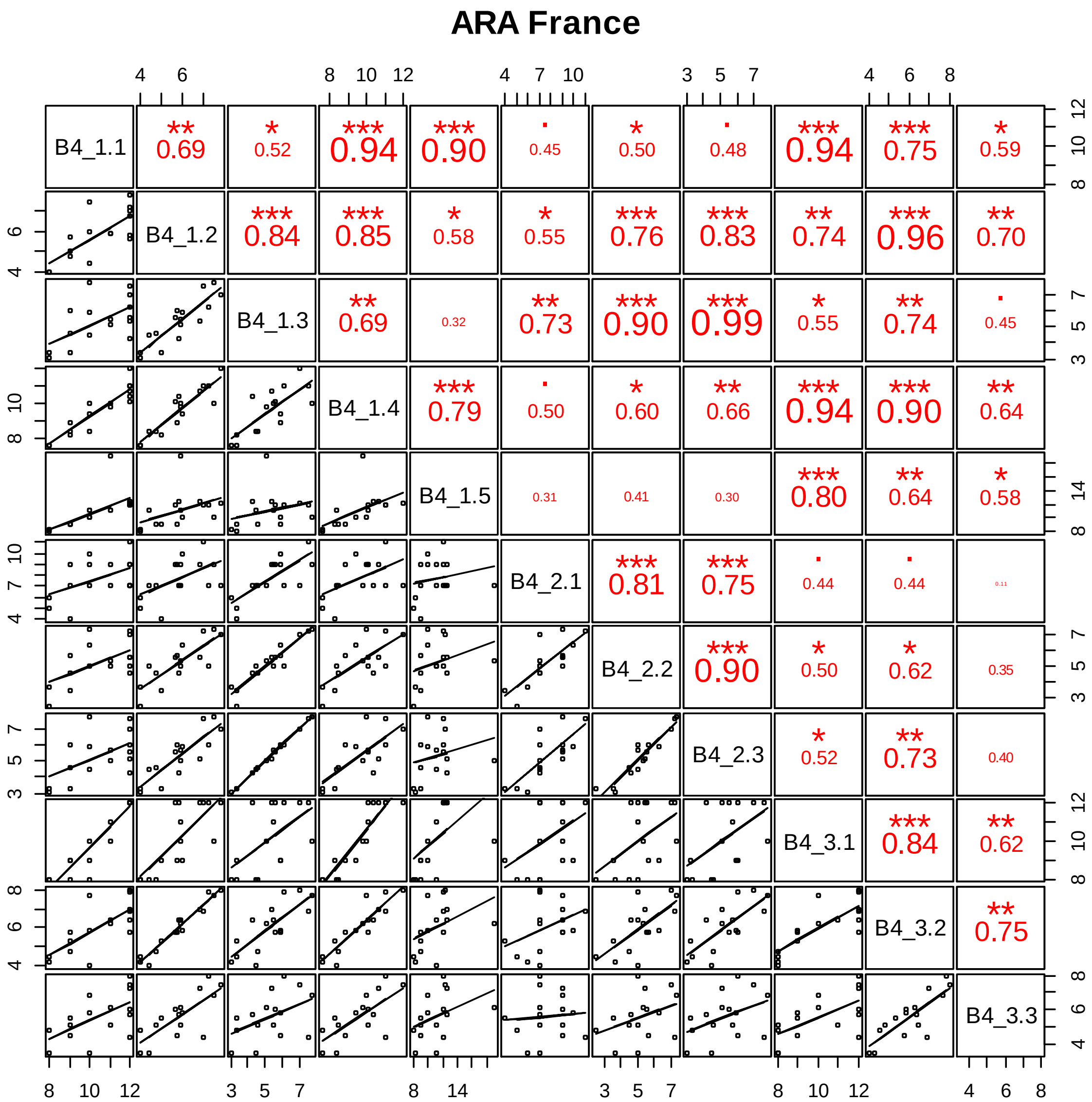
Bee diversity: Correlogram of species diversity indices of bees calculated from the species list with abundance. B9_1.1 = Gamma richness of bees in farms, B9_1.2 = Alpha richness, B9_1.3 = Area weighted richness, B9_1.4 = Rarefied richness, B9_1.5 = Chao estimated richness, B9_2.1 = Gamma richness of bees in cultivated forage and food crops, B9_2.2 = Alpha richness of bees in cultivated forage and food crops, B9_2.3 = Area weighted richness of bees in cultivated forage and food crops, B9_3.1 = Gamma richness of bees in semi-natural habitats, B9_3.2 = Alpha richness of bees in semi-natural habitats, B9_3.3 = Area weighted richness of bees in semi-natural habitats.



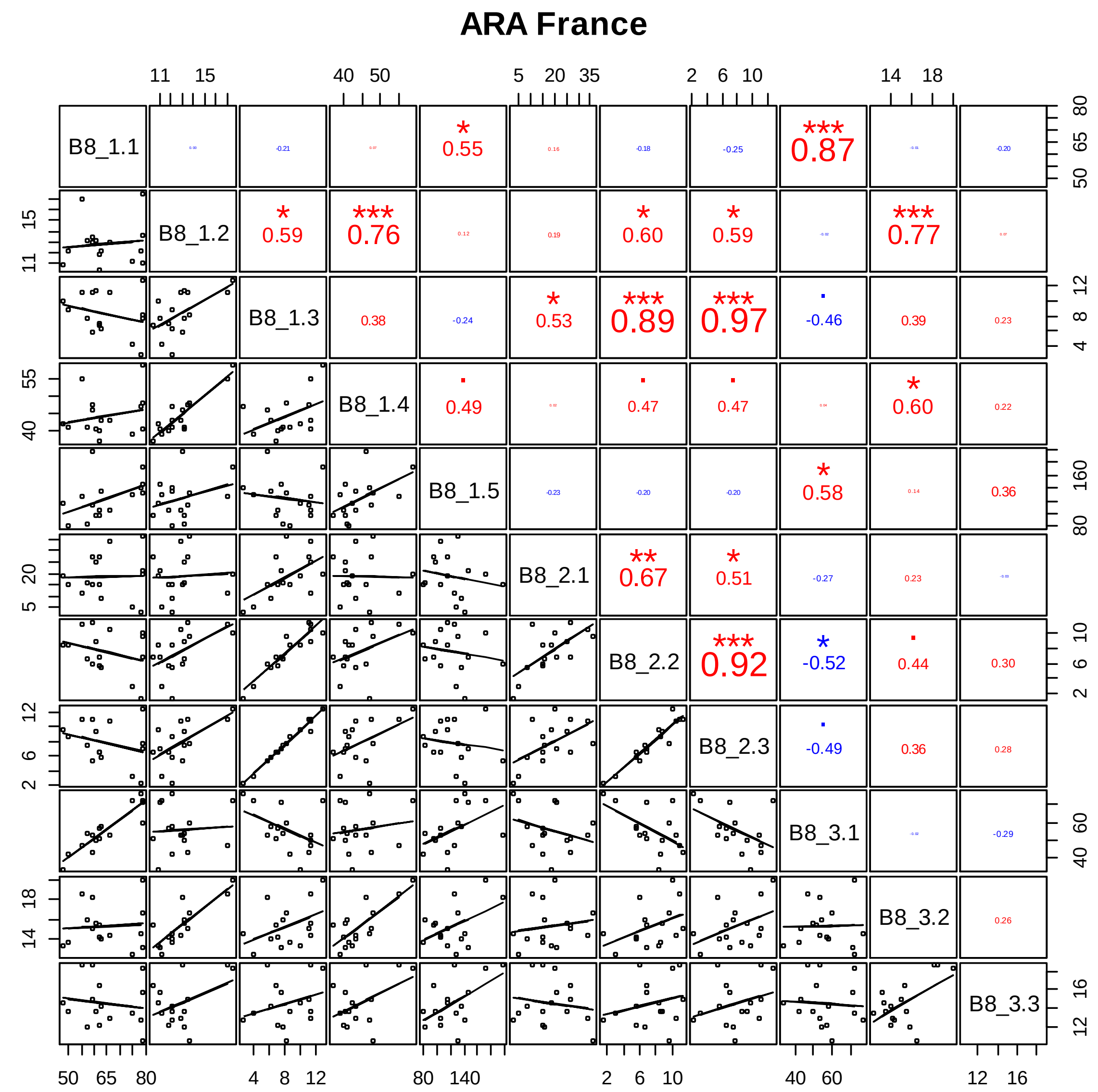
Plant diversity: Correlogram of species diversity indices of plant calculated from the species list with abundance. B2_1.1 = Gamma richness of plant in farms, B2_1.2 = Alpha richness, B2_1.3 = Area weighted richness, B2_1.4 = Rarefied richness, B2_1.5 = Chao estimated richness, B2_2.1 = Gamma richness of plant in cultivated forage and food crops, B2_2.2 = Alpha richness of plant in cultivated forage and food crops, B2_2.3 = Area weighted richness of plant in cultivated forage and food crops, B2_3.1 = Gamma richness of plant in semi-natural habitats, B2_3.2 = Alpha richness of plant in semi-natural habitats, B2_3.3 = Area weighted richness of plant in semi-natural habitats.



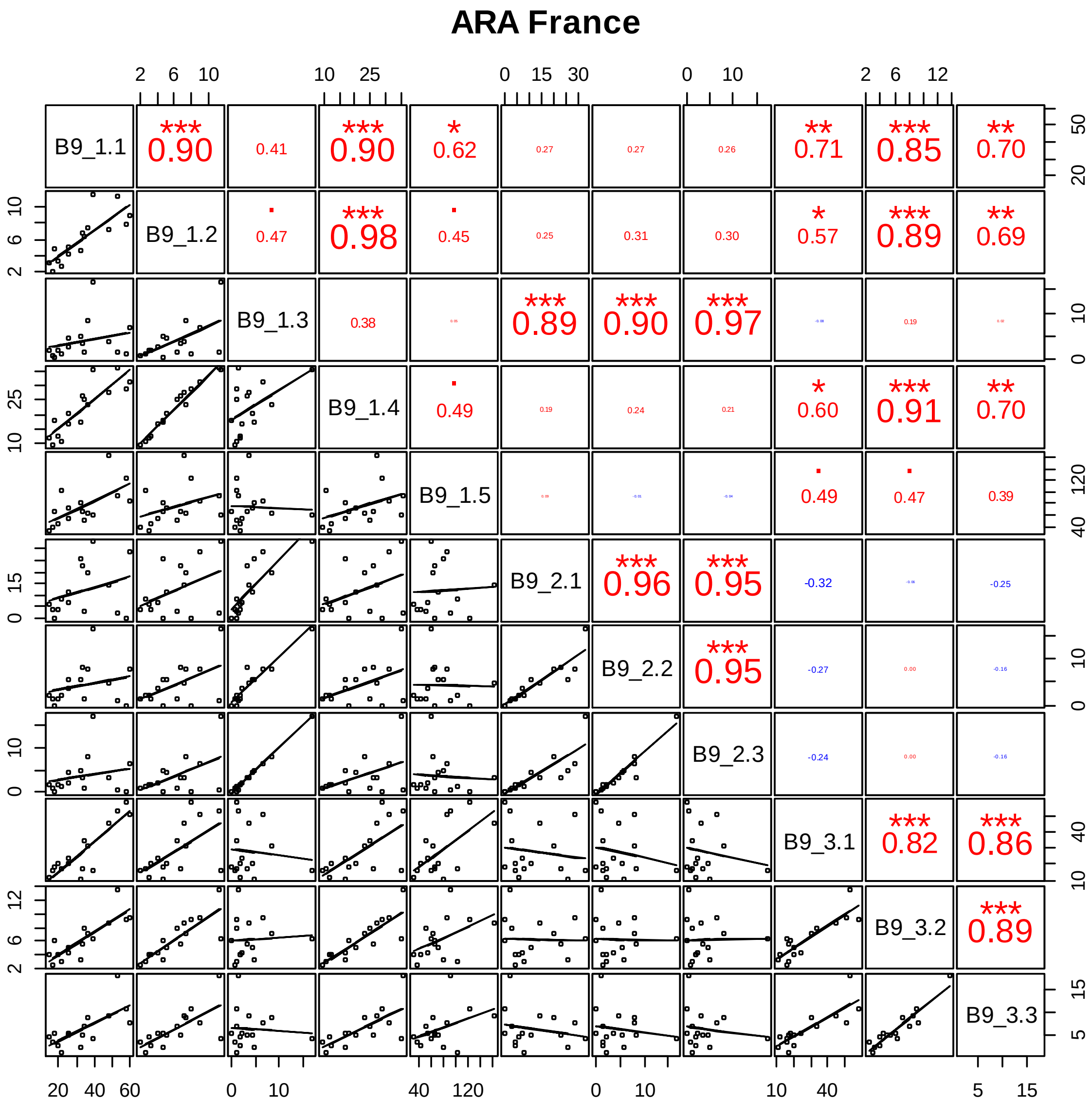
Earthworm diversity: Correlogram of species diversity indices of earthworms calculated from the species list with abundance. B4_1.1 = Gamma richness of earthworms in farms, B4_1.2 = Alpha richness, B4_1.3 = Area weighted richness, B4_1.4 = Rarefied richness, B4_1.5 = Chao estimated richness, B4_2.1 = Gamma richness of earthworms in cultivated forage and food crops, B4_2.2 = Alpha richness of earthworms in cultivated forage and food crops, B4_2.3 = Area weighted richness of earthworms in cultivated forage and food crops, B4_3.1 = Gamma richness of earthworms in semi-natural habitats, B4_3.2 = Alpha richness of earthworms in semi-natural habitats, B4_3.3 = Area weighted richness of earthworms in semi-natural habitats.



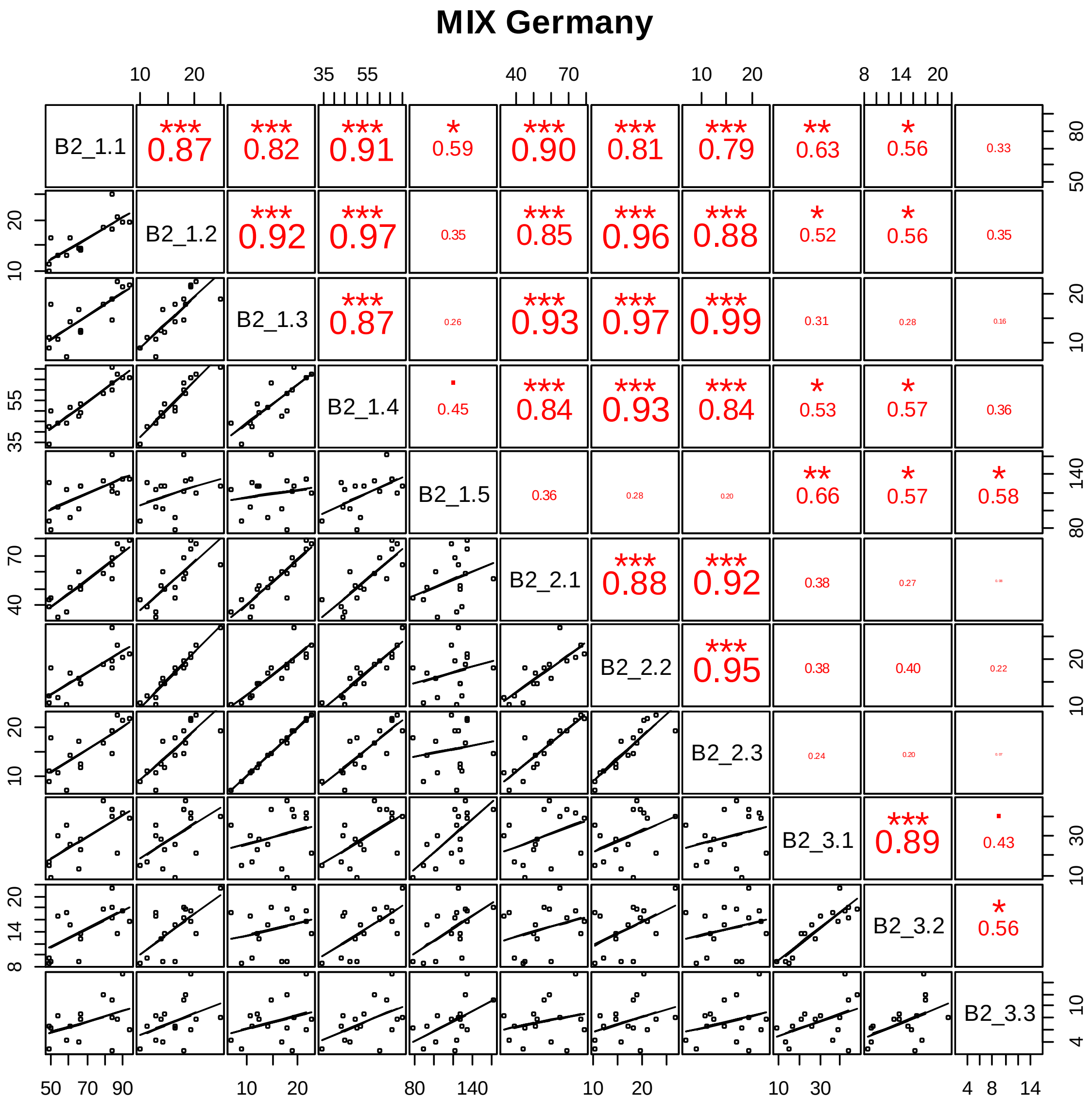
Spider diversity: Correlogram of species diversity indices of spiders calculated from the species list with abundance. B8_1.1 = Gamma richness of spiders in farms, B8_1.2 = Alpha richness, B8_1.3 = Area weighted richness, B8_1.4 = Rarefied richness, B8_1.5 = Chao estimated richness, B8_2.1 = Gamma richness of spiders in cultivated forage and food crops, B8_2.2 = Alpha richness of spiders in cultivated forage and food crops, B8_2.3 = Area weighted richness of spiders in cultivated forage and food crops, B8_3.1 = Gamma richness of spiders in semi-natural habitats, B8_3.2 = Alpha richness of spiders in semi-natural habitats, B8_3.3 = Area weighted richness of spiders in semi-natural habitats.



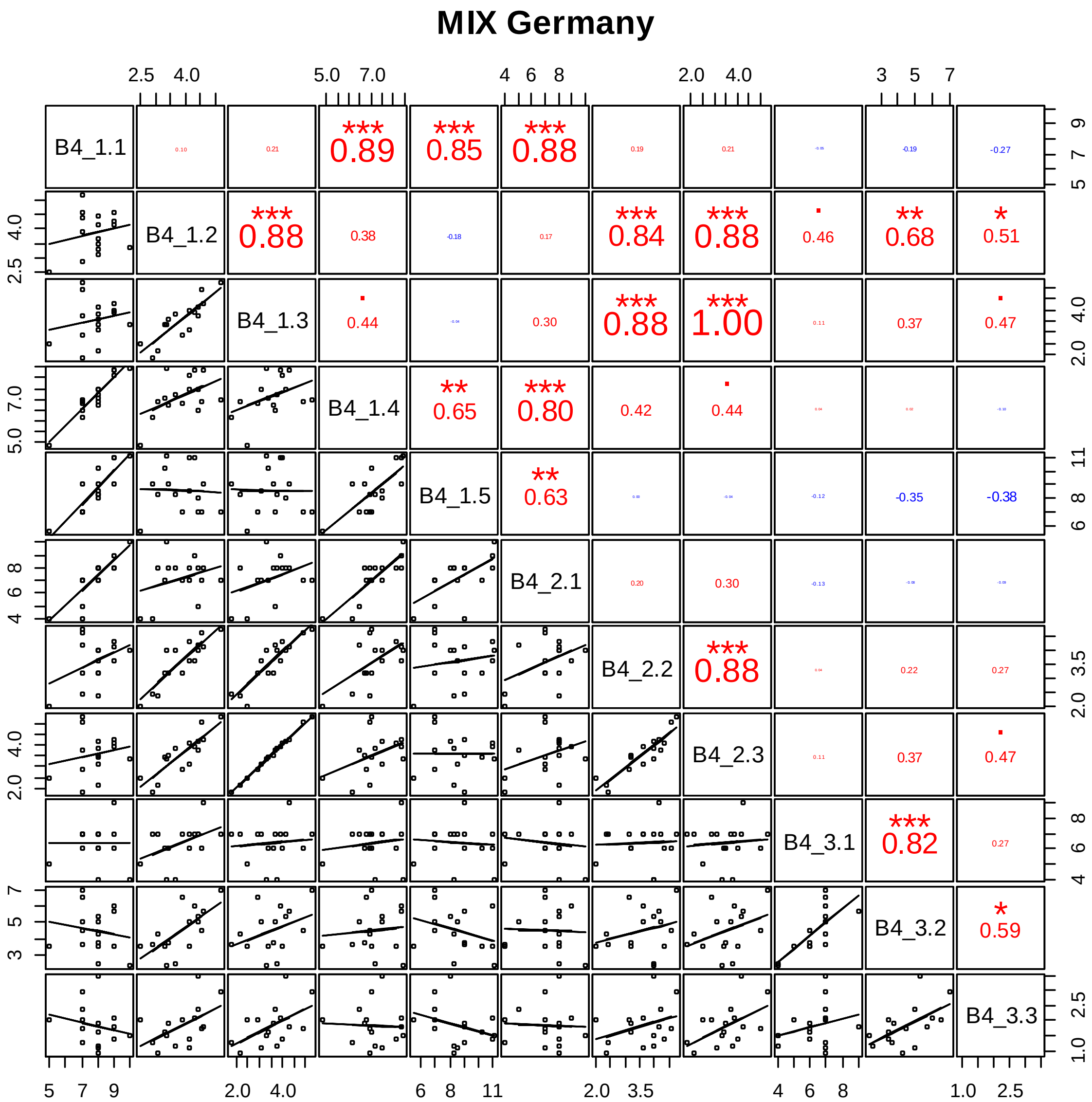
Bee diversity: Correlogram of species diversity indices of bees calculated from the species list with abundance. B9_1.1 = Gamma richness of bees in farms, B9_1.2 = Alpha richness, B9_1.3 = Area weighted richness, B9_1.4 = Rarefied richness, B9_1.5 = Chao estimated richness, B9_2.1 = Gamma richness of bees in cultivated forage and food crops, B9_2.2 = Alpha richness of bees in cultivated forage and food crops, B9_2.3 = Area weighted richness of bees in cultivated forage and food crops, B9_3.1 = Gamma richness of bees in semi-natural habitats, B9_3.2 = Alpha richness of bees in semi-natural habitats, B9_3.3 = Area weighted richness of bees in semi-natural habitats.



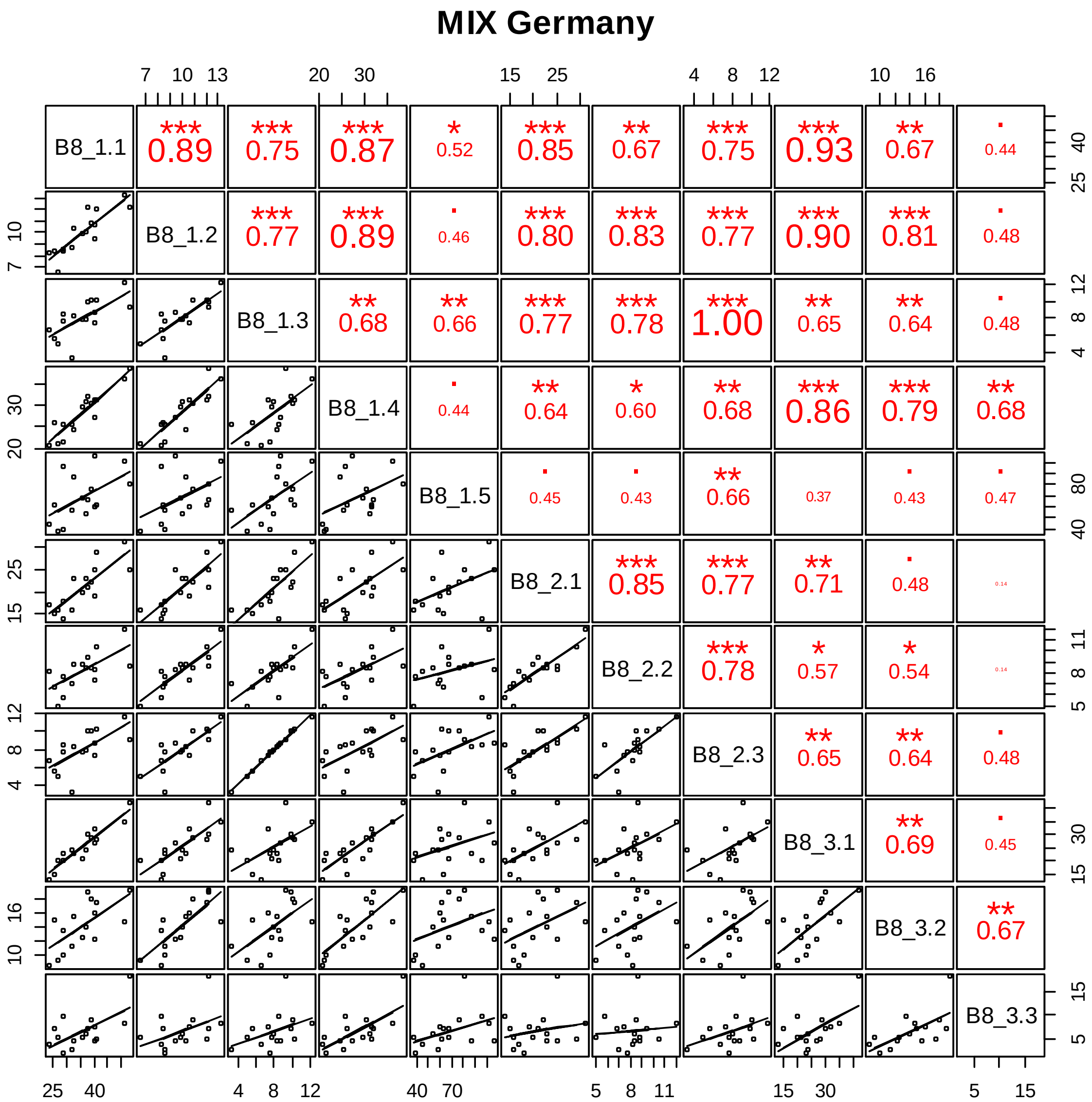
Plant diversity: Correlogram of species diversity indices of plant calculated from the species list with abundance. B2_1.1 = Gamma richness of plant in farms, B2_1.2 = Alpha richness, B2_1.3 = Area weighted richness, B2_1.4 = Rarefied richness, B2_1.5 = Chao estimated richness, B2_2.1 = Gamma richness of plant in cultivated forage and food crops, B2_2.2 = Alpha richness of plant in cultivated forage and food crops, B2_2.3 = Area weighted richness of plant in cultivated forage and food crops, B2_3.1 = Gamma richness of plant in semi-natural habitats, B2_3.2 = Alpha richness of plant in semi-natural habitats, B2_3.3 = Area weighted richness of plant in semi-natural habitats.



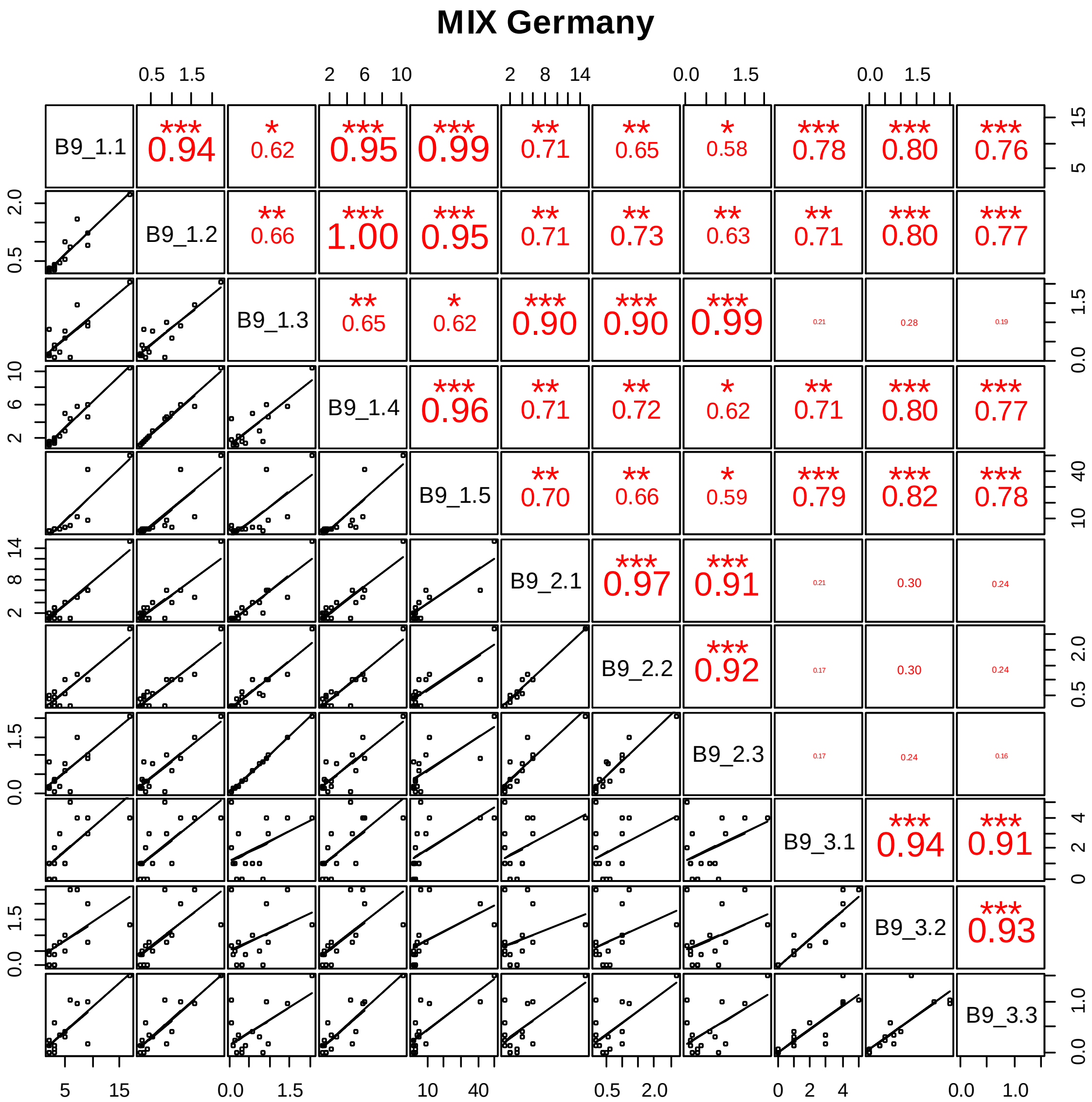
Earthworm diversity: Correlogram of species diversity indices of earthworms calculated from the species list with abundance. B4_1.1 = Gamma richness of earthworms in farms, B4_1.2 = Alpha richness, B4_1.3 = Area weighted richness, B4_1.4 = Rarefied richness, B4_1.5 = Chao estimated richness, B4_2.1 = Gamma richness of earthworms in cultivated forage and food crops, B4_2.2 = Alpha richness of earthworms in cultivated forage and food crops, B4_2.3 = Area weighted richness of earthworms in cultivated forage and food crops, B4_3.1 = Gamma richness of earthworms in semi-natural habitats, B4_3.2 = Alpha richness of earthworms in semi-natural habitats, B4_3.3 = Area weighted richness of earthworms in semi-natural habitats.



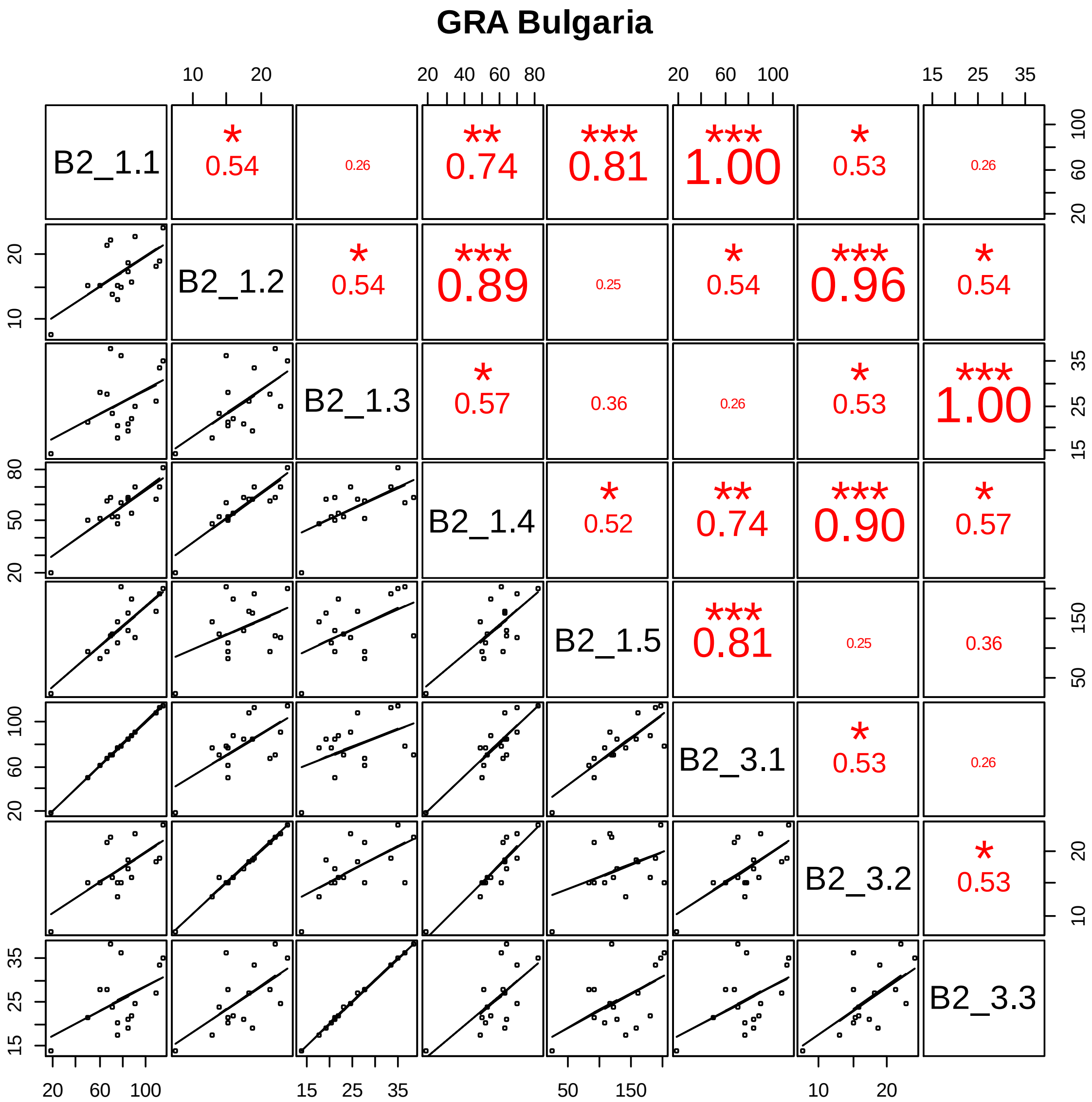
Spider diversity: Correlogram of species diversity indices of spiders calculated from the species list with abundance. B8_1.1 = Gamma richness of spiders in farms, B8_1.2 = Alpha richness, B8_1.3 = Area weighted richness, B8_1.4 = Rarefied richness, B8_1.5 = Chao estimated richness, B8_2.1 = Gamma richness of spiders in cultivated forage and food crops, B8_2.2 = Alpha richness of spiders in cultivated forage and food crops, B8_2.3 = Area weighted richness of spiders in cultivated forage and food crops, B8_3.1 = Gamma richness of spiders in semi-natural habitats, B8_3.2 = Alpha richness of spiders in semi-natural habitats, B8_3.3 = Area weighted richness of spiders in semi-natural habitats.



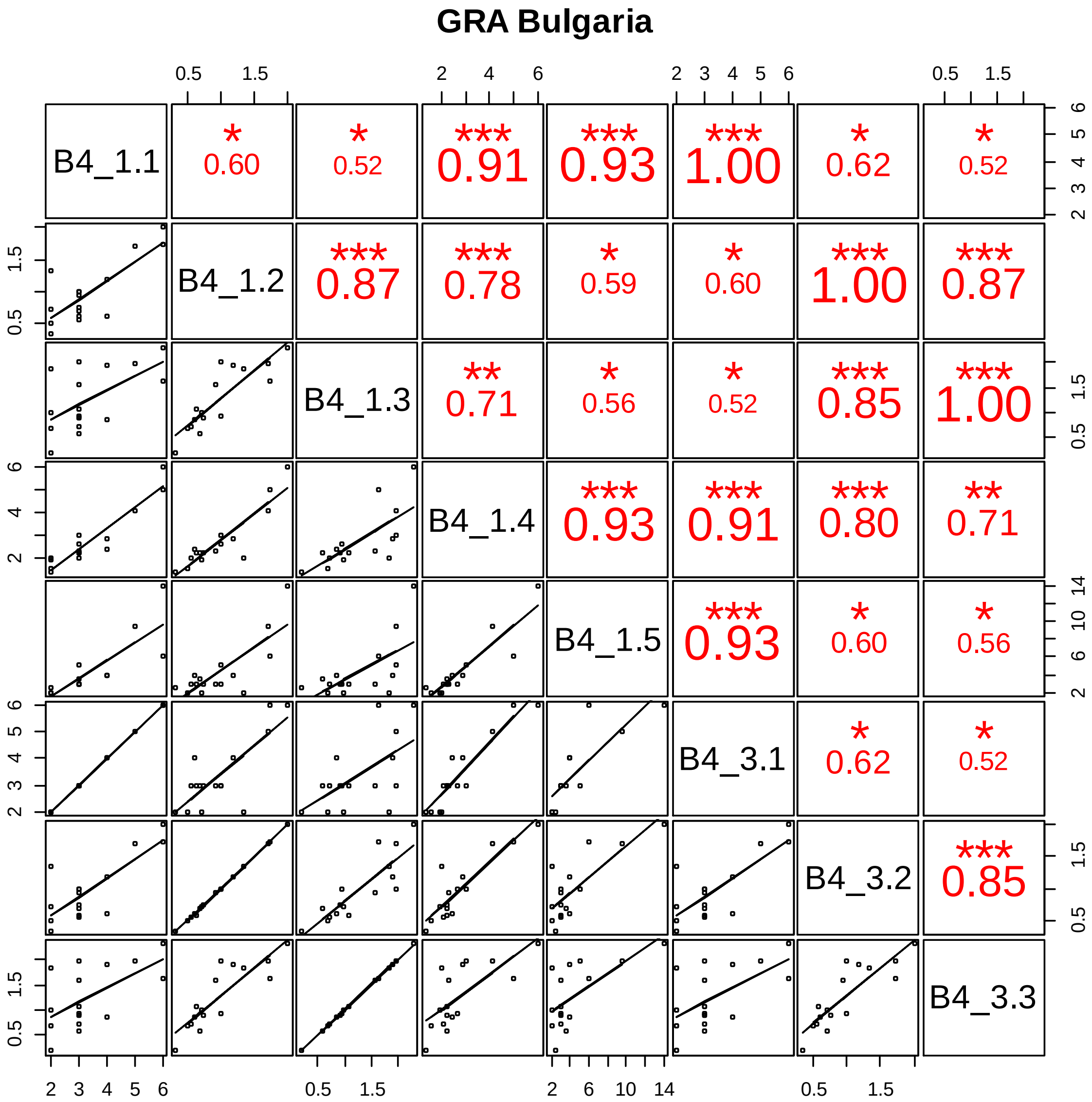
Bee diversity: Correlogram of species diversity indices of bees calculated from the species list with abundance. B9_1.1 = Gamma richness of bees in farms, B9_1.2 = Alpha richness, B9_1.3 = Area weighted richness, B9_1.4 = Rarefied richness, B9_1.5 = Chao estimated richness, B9_2.1 = Gamma richness of bees in cultivated forage and food crops, B9_2.2 = Alpha richness of bees in cultivated forage and food crops, B9_2.3 = Area weighted richness of bees in cultivated forage and food crops, B9_3.1 = Gamma richness of bees in semi-natural habitats, B9_3.2 = Alpha richness of bees in semi-natural habitats, B9_3.3 = Area weighted richness of bees in semi-natural habitats.



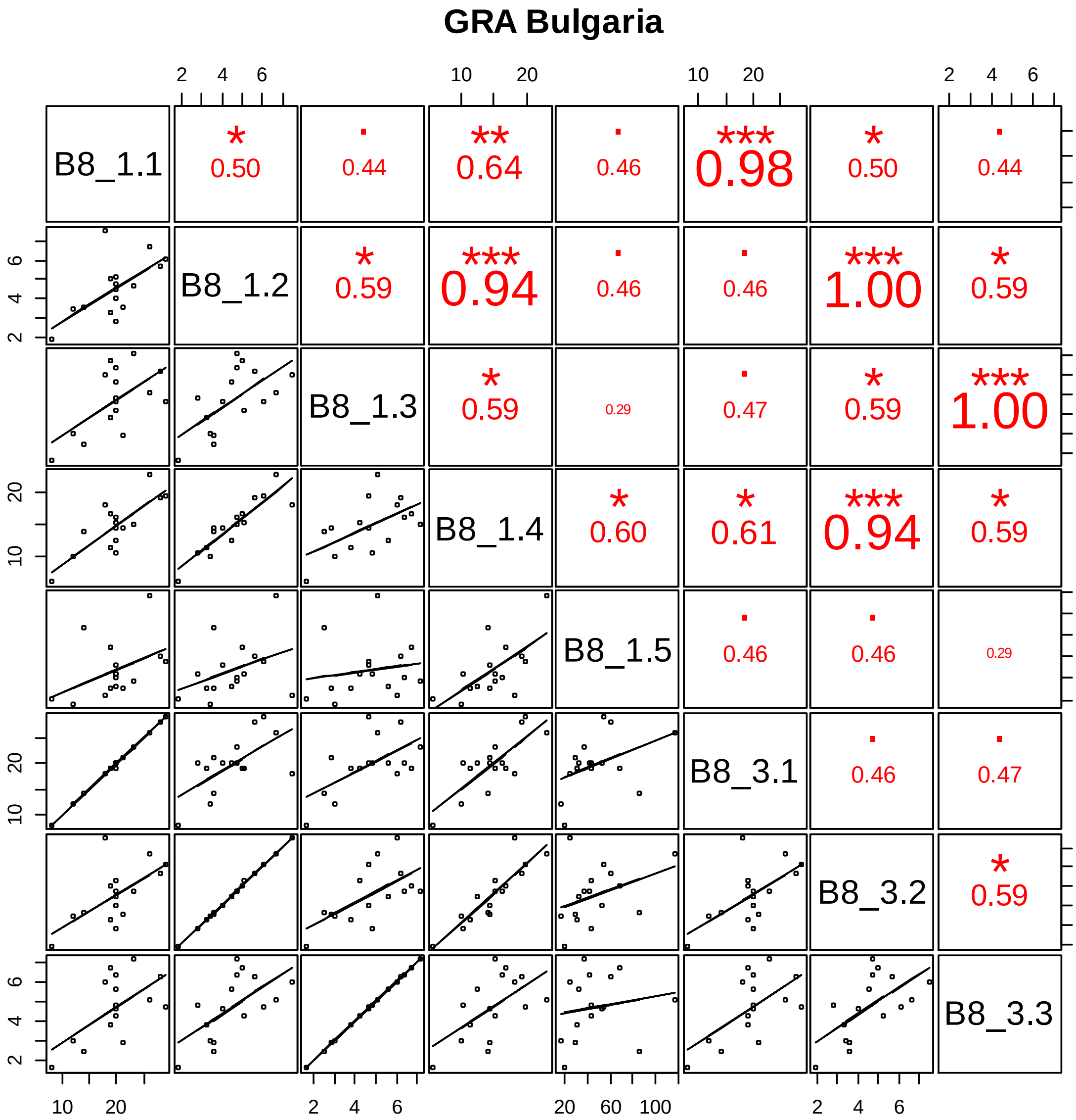
Plant diversity: Correlogram of species diversity indices of plant calculated from the species list with abundance. B2_1.1 = Gamma richness of plant in farms, B2_1.2 = Alpha richness, B2_1.3 = Area weighted richness, B2_1.4 = Rarefied richness, B2_1.5 = Chao estimated richness, B2_3.1 = Gamma richness of plant in semi-natural habitats, B2_3.2 = Alpha richness of plant in semi-natural habitats, B2_3.3 = Area weighted richness of plant in semi-natural habitats.



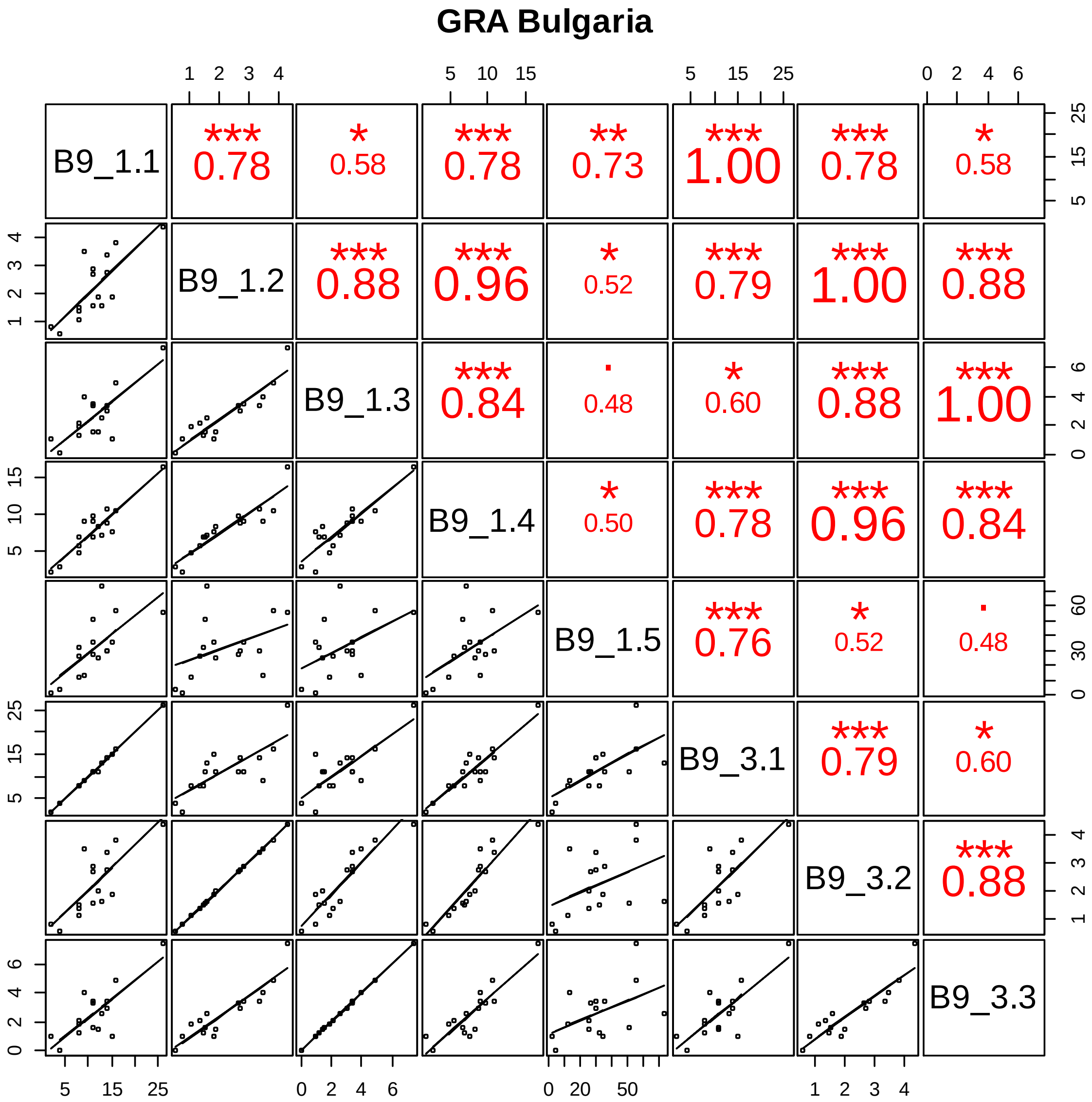
Earthworm diversity: Correlogram of species diversity indices of earthworms calculated from the species list with abundance. B4_1.1 = Gamma richness of earthworms in farms, B4_1.2 = Alpha richness, B4_1.3 = Area weighted richness, B4_1.4 = Rarefied richness, B4_1.5 = Chao estimated richness, B4_3.1 = Gamma richness of earthworms in semi-natural habitats, B4_3.2 = Alpha richness of earthworms in semi-natural habitats, B4_3.3 = Area weighted richness of earthworms in semi-natural habitats.



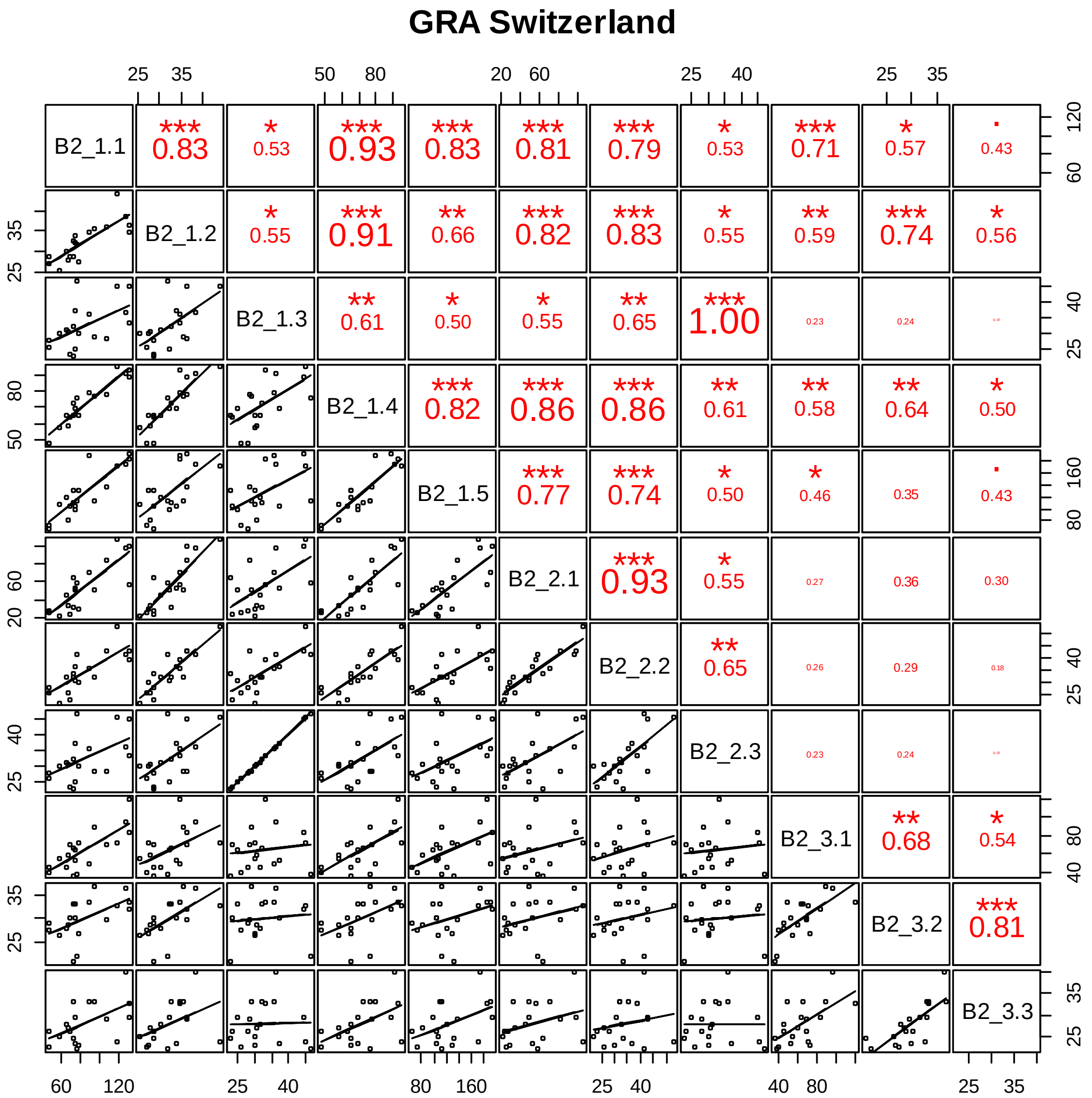
Spider diversity: Correlogram of species diversity indices of spiders calculated from the species list with abundance. B8_1.1 = Gamma richness of spiders in farms, B8_1.2 = Alpha richness, B8_1.3 = Area weighted richness, B8_1.4 = Rarefied richness, B8_1.5 = Chao estimated richness, B8_3.1 = Gamma richness of spiders in semi-natural habitats, B8_3.2 = Alpha richness of spiders in semi-natural habitats, B8_3.3 = Area weighted richness of spiders in semi-natural habitats.



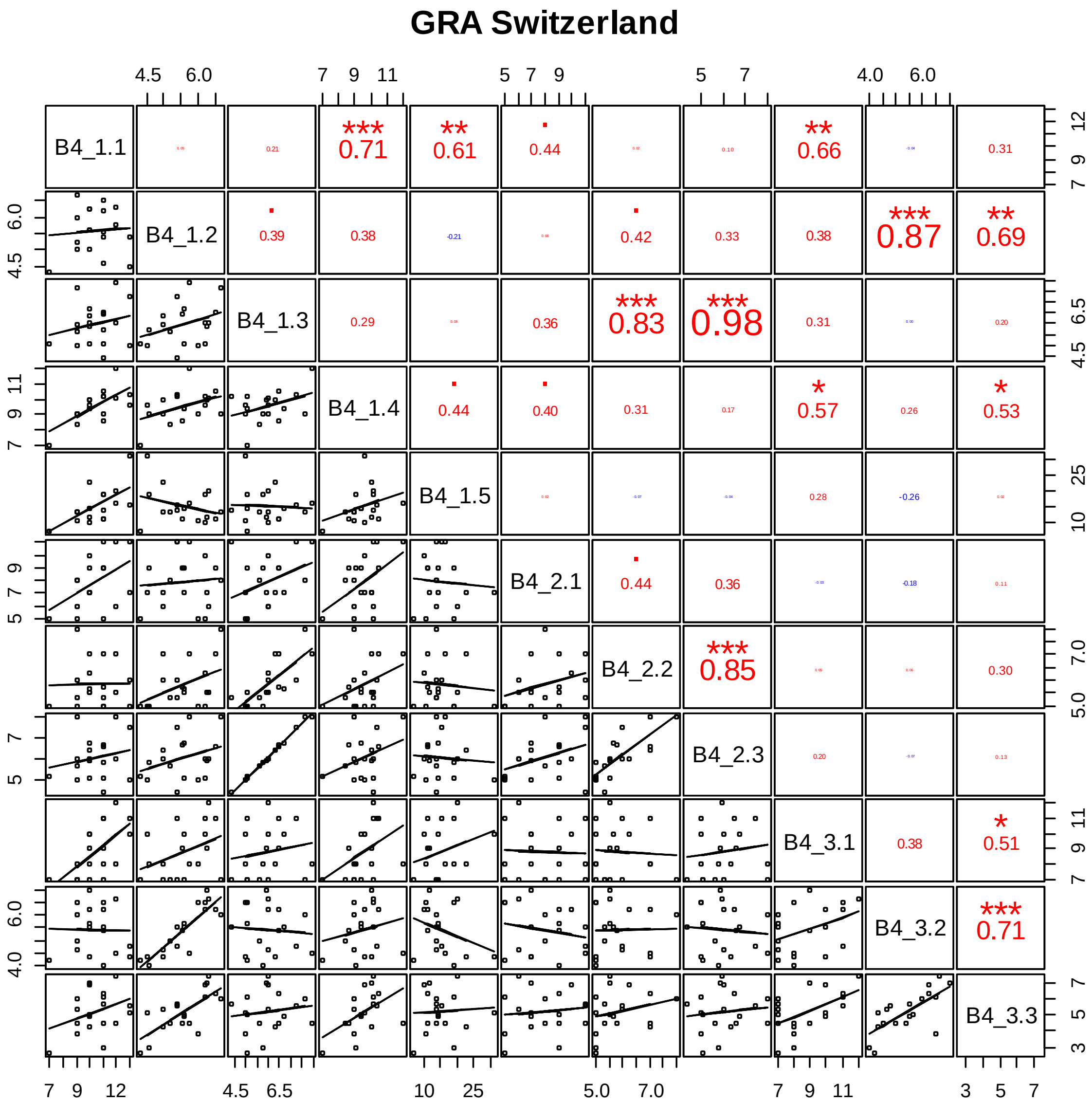
Bee diversity: Correlogram of species diversity indices of bees calculated from the species list with abundance. B9_1.1 = Gamma richness of bees in farms, B9_1.2 = Alpha richness, B9_1.3 = Area weighted richness, B9_1.4 = Rarefied richness, B9_1.5 = Chao estimated richness, B9_3.1 = Gamma richness of bees in semi-natural habitats, B9_3.2 = Alpha richness of bees in semi-natural habitats, B9_3.3 = Area weighted richness of bees in semi-natural habitats.



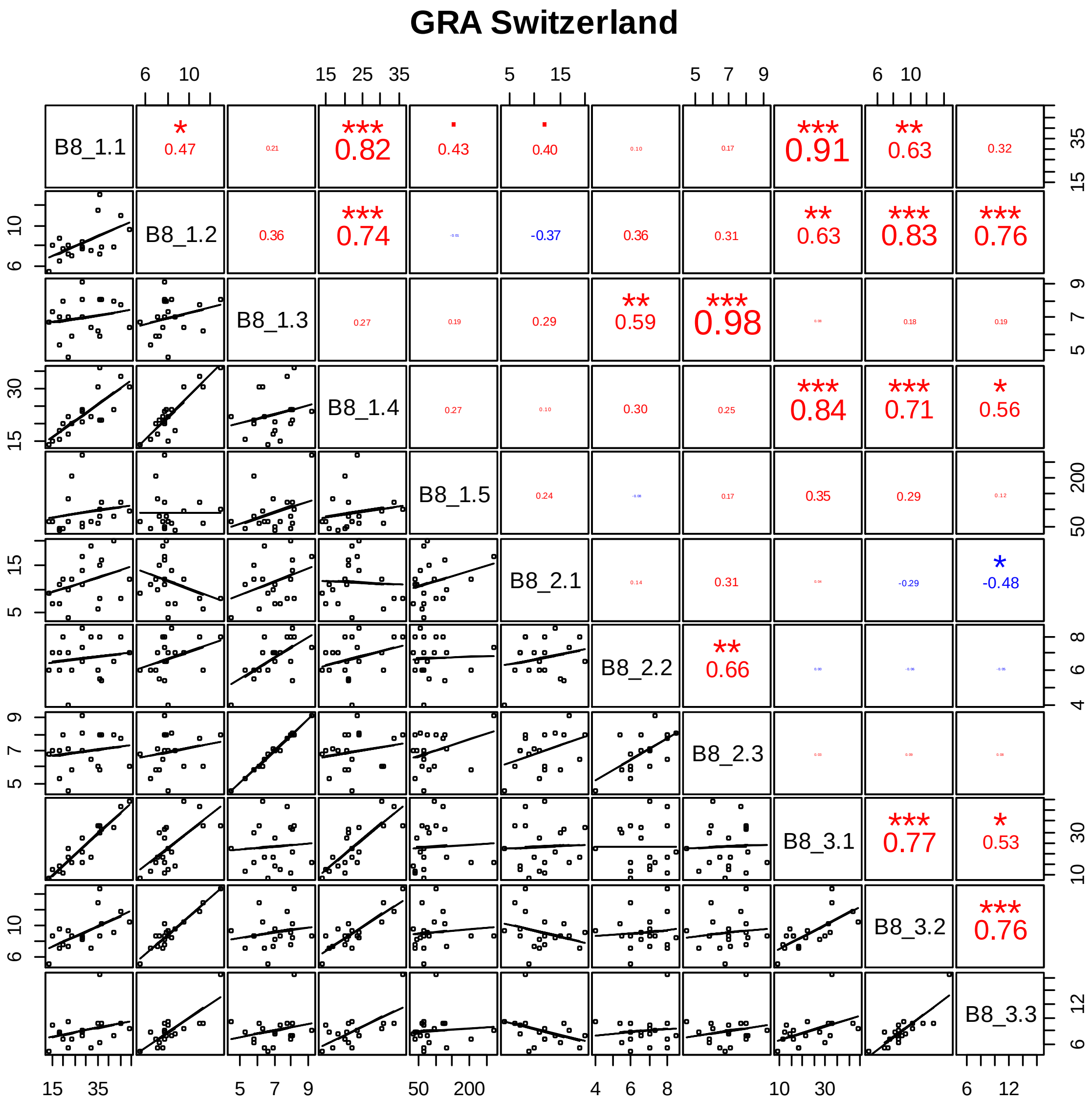
Plant diversity: Correlogram of species diversity indices of plant calculated from the species list with abundance. B2_1.1 = Gamma richness of plant in farms, B2_1.2 = Alpha richness, B2_1.3 = Area weighted richness, B2_1.4 = Rarefied richness, B2_1.5 = Chao estimated richness, B2_2.1 = Gamma richness of plant in cultivated forage and food crops, B2_2.2 = Alpha richness of plant in cultivated forage and food crops, B2_2.3 = Area weighted richness of plant in cultivated forage and food crops, B2_3.1 = Gamma richness of plant in semi-natural habitats, B2_3.2 = Alpha richness of plant in semi-natural habitats, B2_3.3 = Area weighted richness of plant in semi-natural habitats.



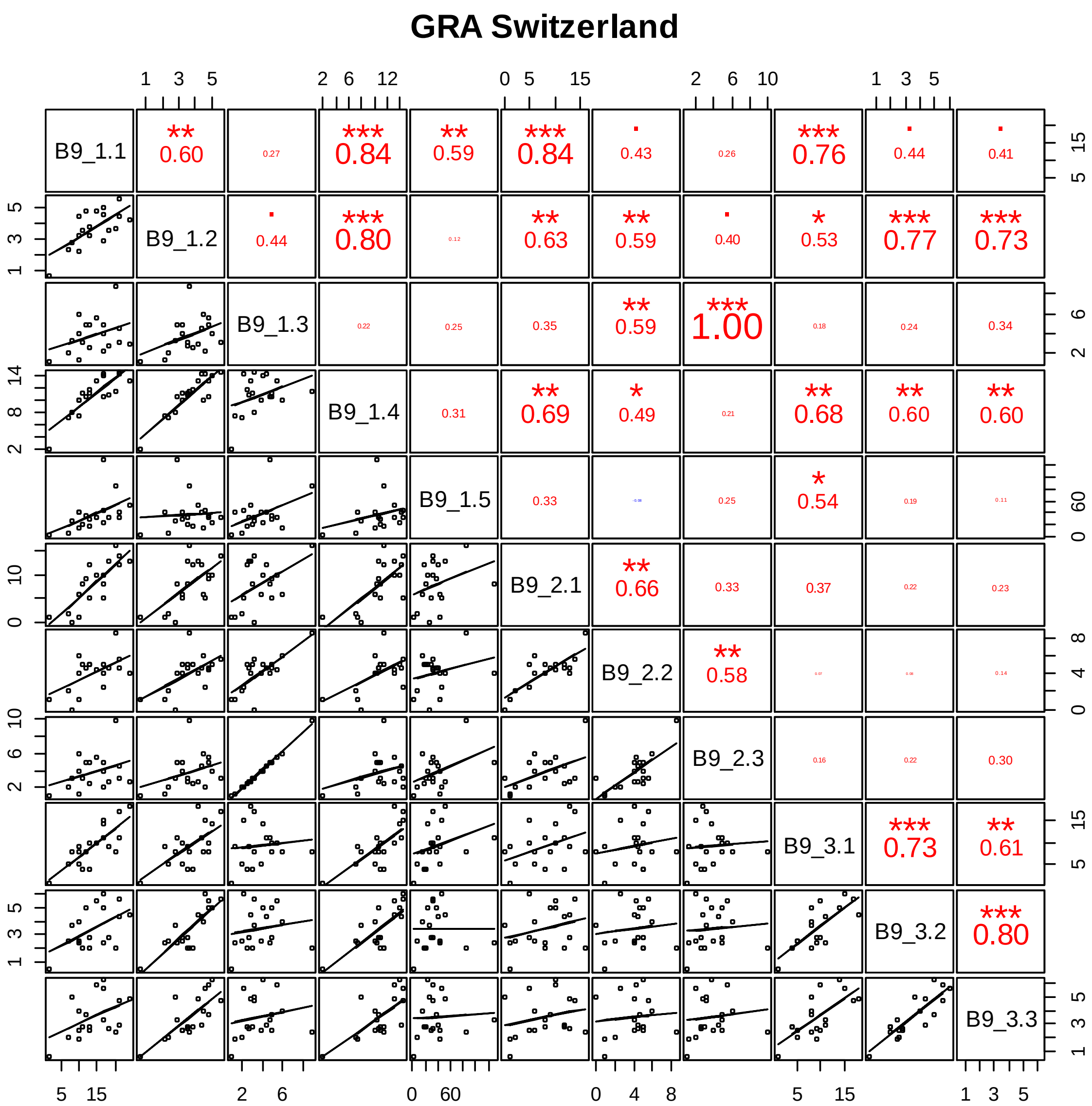
Earthworm diversity: Correlogram of species diversity indices of earthworms calculated from the species list with abundance. B4_1.1 = Gamma richness of earthworms in farms, B4_1.2 = Alpha richness, B4_1.3 = Area weighted richness, B4_1.4 = Rarefied richness, B4_1.5 = Chao estimated richness, B4_2.1 = Gamma richness of earthworms in cultivated forage and food crops, B4_2.2 = Alpha richness of earthworms in cultivated forage and food crops, B4_2.3 = Area weighted richness of earthworms in cultivated forage and food crops, B4_3.1 = Gamma richness of earthworms in semi-natural habitats, B4_3.2 = Alpha richness of earthworms in semi-natural habitats, B4_3.3 = Area weighted richness of earthworms in semi-natural habitats.



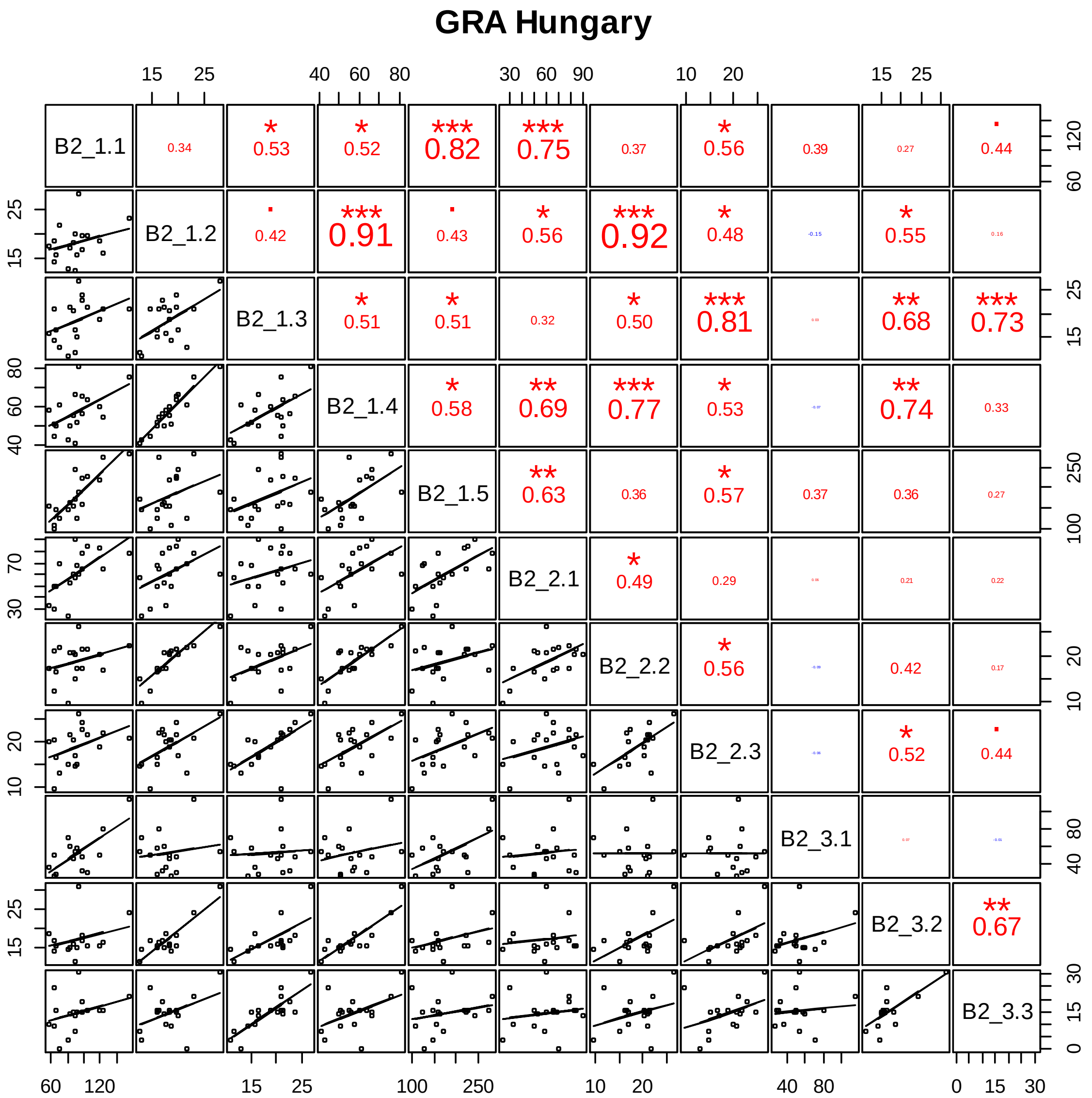
Spider diversity: Correlogram of species diversity indices of spiders calculated from the species list with abundance. B8_1.1 = Gamma richness of spiders in farms, B8_1.2 = Alpha richness, B8_1.3 = Area weighted richness, B8_1.4 = Rarefied richness, B8_1.5 = Chao estimated richness, B8_2.1 = Gamma richness of spiders in cultivated forage and food crops, B8_2.2 = Alpha richness of spiders in cultivated forage and food crops, B8_2.3 = Area weighted richness of spiders in cultivated forage and food crops, B8_3.1 = Gamma richness of spiders in semi-natural habitats, B8_3.2 = Alpha richness of spiders in semi-natural habitats, B8_3.3 = Area weighted richness of spiders in semi-natural habitats.



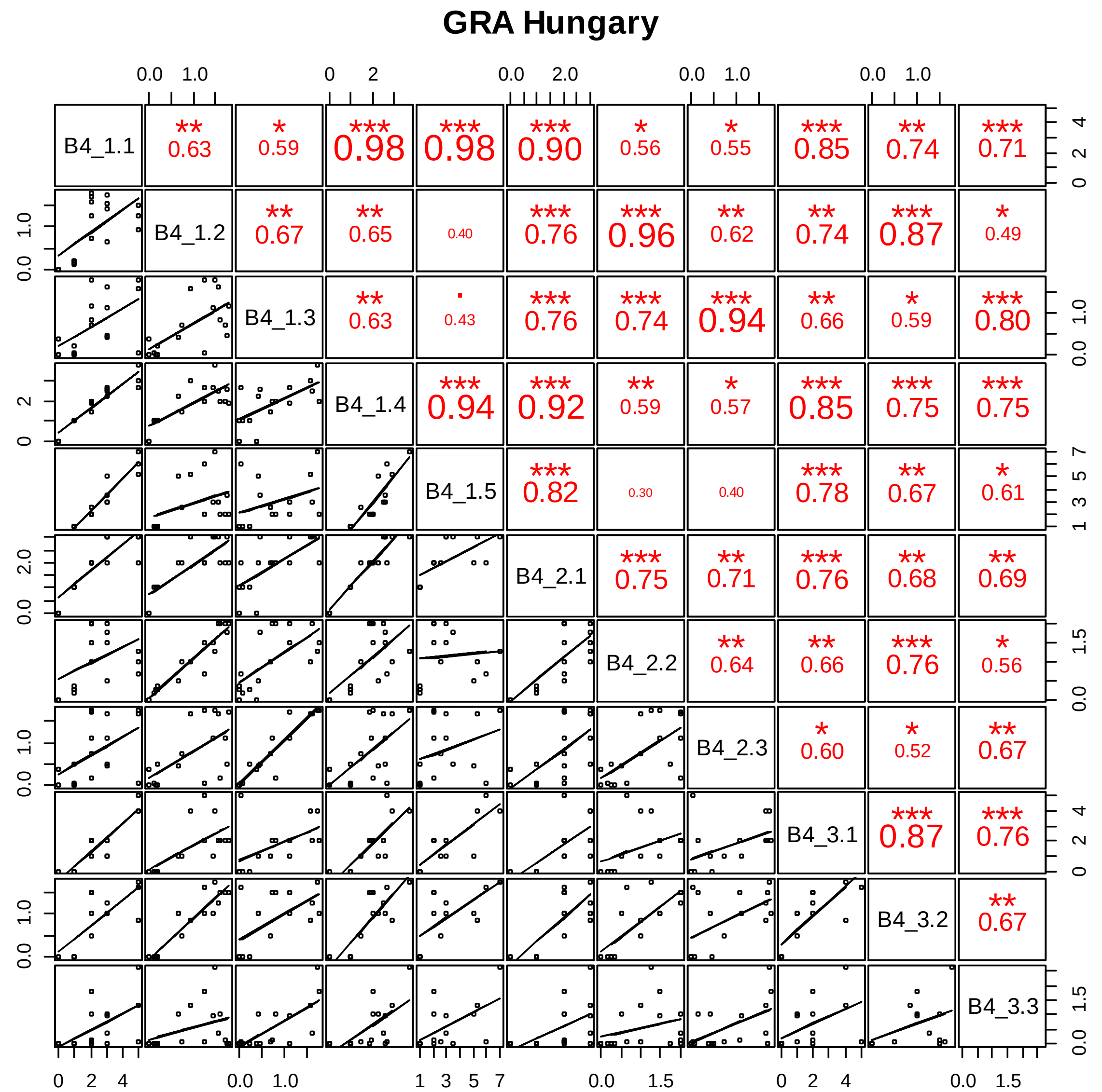
Bee diversity: Correlogram of species diversity indices of bees calculated from the species list with abundance. B9_1.1 = Gamma richness of bees in farms, B9_1.2 = Alpha richness, B9_1.3 = Area weighted richness, B9_1.4 = Rarefied richness, B9_1.5 = Chao estimated richness, B9_2.1 = Gamma richness of bees in cultivated forage and food crops, B9_2.2 = Alpha richness of bees in cultivated forage and food crops, B9_2.3 = Area weighted richness of bees in cultivated forage and food crops, B9_3.1 = Gamma richness of bees in semi-natural habitats, B9_3.2 = Alpha richness of bees in semi-natural habitats, B9_3.3 = Area weighted richness of bees in semi-natural habitats.



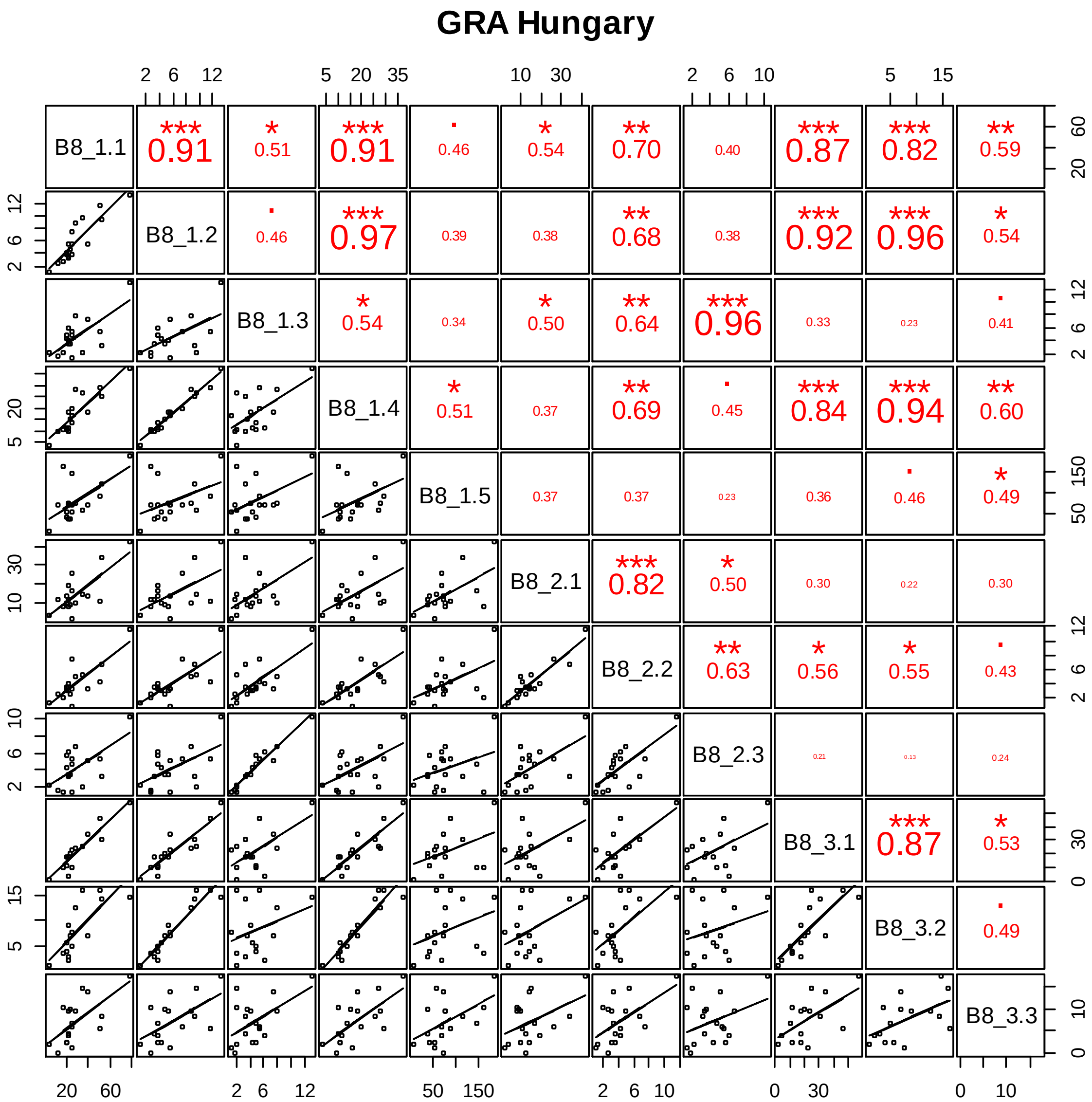
Plant diversity: Correlogram of species diversity indices of plant calculated from the species list with abundance. B2_1.1 = Gamma richness of plant in farms, B2_1.2 = Alpha richness, B2_1.3 = Area weighted richness, B2_1.4 = Rarefied richness, B2_1.5 = Chao estimated richness, B2_2.1 = Gamma richness of plant in cultivated forage and food crops, B2_2.2 = Alpha richness of plant in cultivated forage and food crops, B2_2.3 = Area weighted richness of plant in cultivated forage and food crops, B2_3.1 = Gamma richness of plant in semi-natural habitats, B2_3.2 = Alpha richness of plant in semi-natural habitats, B2_3.3 = Area weighted richness of plant in semi-natural habitats.



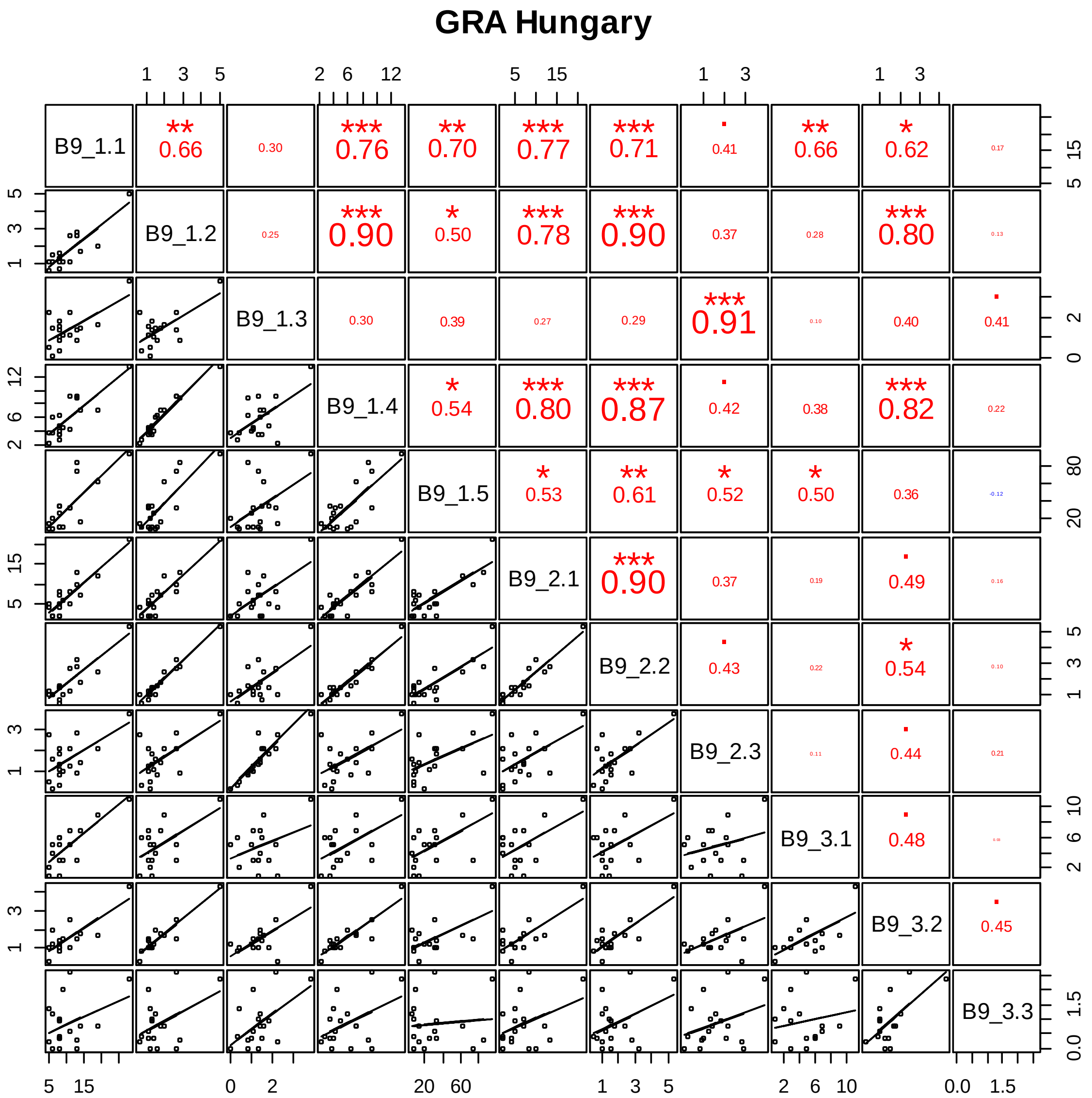
Earthworm diversity: Correlogram of species diversity indices of earthworms calculated from the species list with abundance. B4_1.1 = Gamma richness of earthworms in farms, B4_1.2 = Alpha richness, B4_1.3 = Area weighted richness, B4_1.4 = Rarefied richness, B4_1.5 = Chao estimated richness, B4_2.1 = Gamma richness of earthworms in cultivated forage and food crops, B4_2.2 = Alpha richness of earthworms in cultivated forage and food crops, B4_2.3 = Area weighted richness of earthworms in cultivated forage and food crops, B4_3.1 = Gamma richness of earthworms in semi-natural habitats, B4_3.2 = Alpha richness of earthworms in semi-natural habitats, B4_3.3 = Area weighted richness of earthworms in semi-natural habitats.



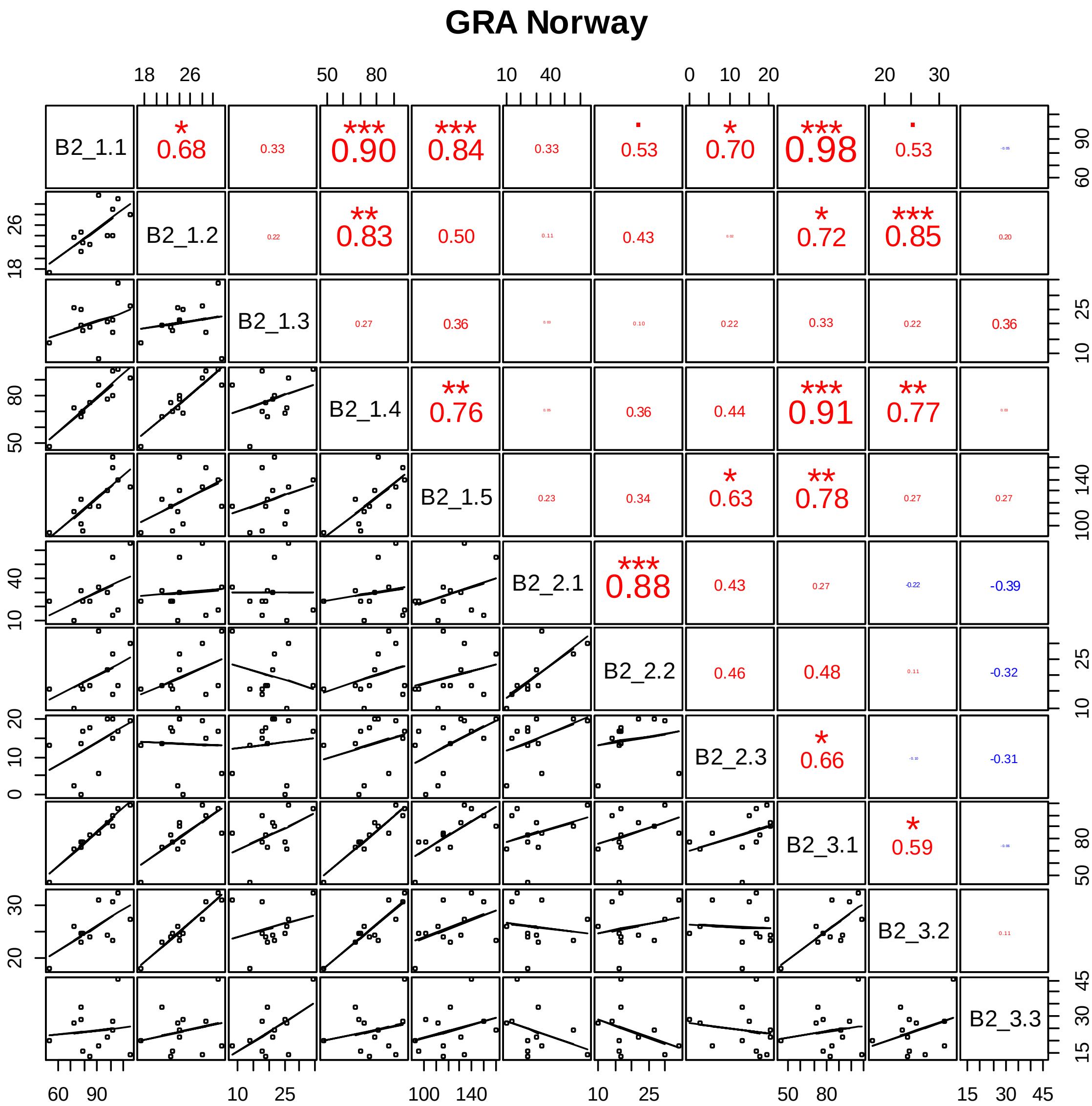
Spider diversity: Correlogram of species diversity indices of spiders calculated from the species list with abundance. B8_1.1 = Gamma richness of spiders in farms, B8_1.2 = Alpha richness, B8_1.3 = Area weighted richness, B8_1.4 = Rarefied richness, B8_1.5 = Chao estimated richness, B8_2.1 = Gamma richness of spiders in cultivated forage and food crops, B8_2.2 = Alpha richness of spiders in cultivated forage and food crops, B8_2.3 = Area weighted richness of spiders in cultivated forage and food crops, B8_3.1 = Gamma richness of spiders in semi-natural habitats, B8_3.2 = Alpha richness of spiders in semi-natural habitats, B8_3.3 = Area weighted richness of spiders in semi-natural habitats.



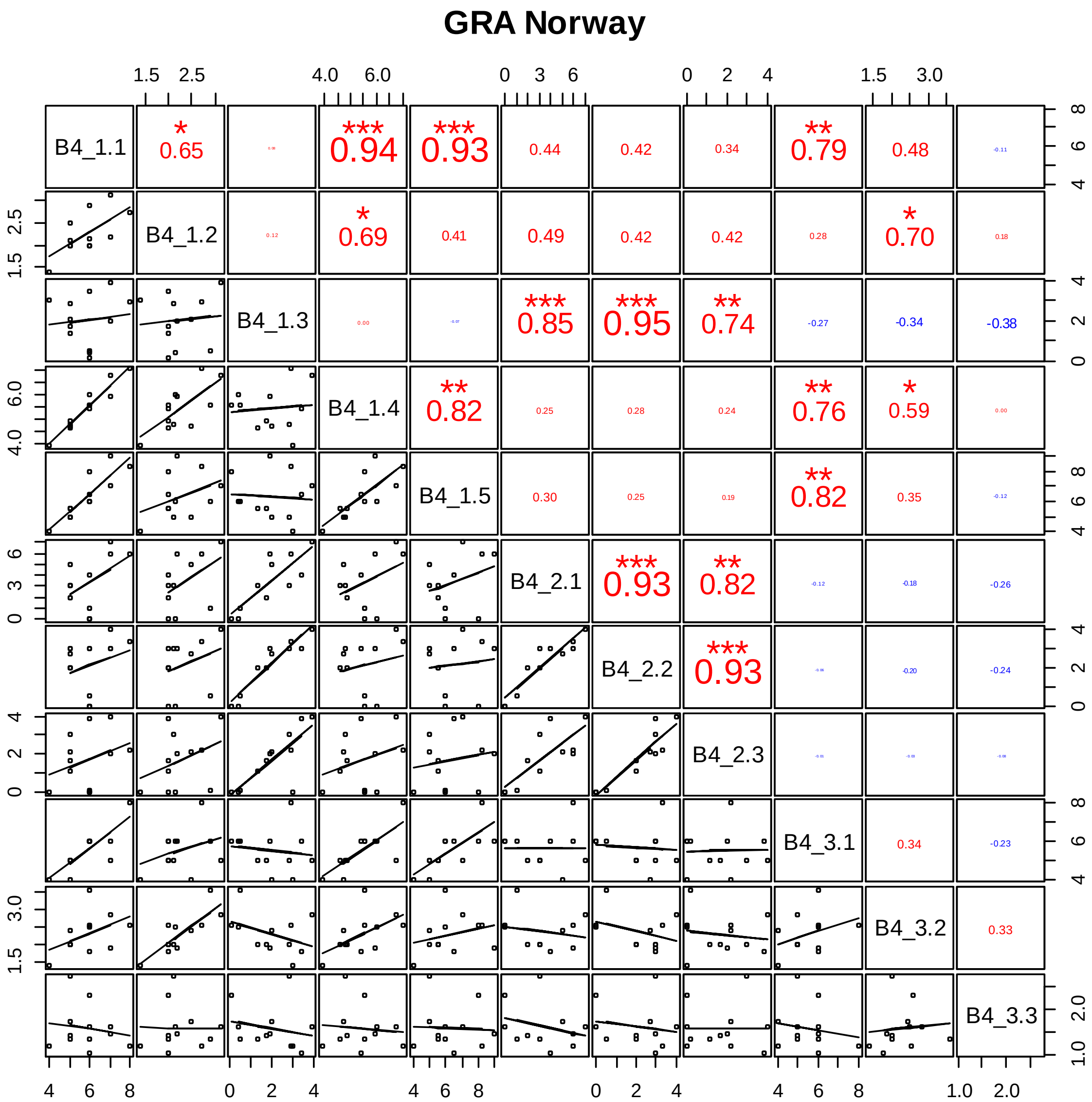
Bee diversity: Correlogram of species diversity indices of bees calculated from the species list with abundance. B9_1.1 = Gamma richness of bees in farms, B9_1.2 = Alpha richness, B9_1.3 = Area weighted richness, B9_1.4 = Rarefied richness, B9_1.5 = Chao estimated richness, B9_2.1 = Gamma richness of bees in cultivated forage and food crops, B9_2.2 = Alpha richness of bees in cultivated forage and food crops, B9_2.3 = Area weighted richness of bees in cultivated forage and food crops, B9_3.1 = Gamma richness of bees in semi-natural habitats, B9_3.2 = Alpha richness of bees in semi-natural habitats, B9_3.3 = Area weighted richness of bees in semi-natural habitats.



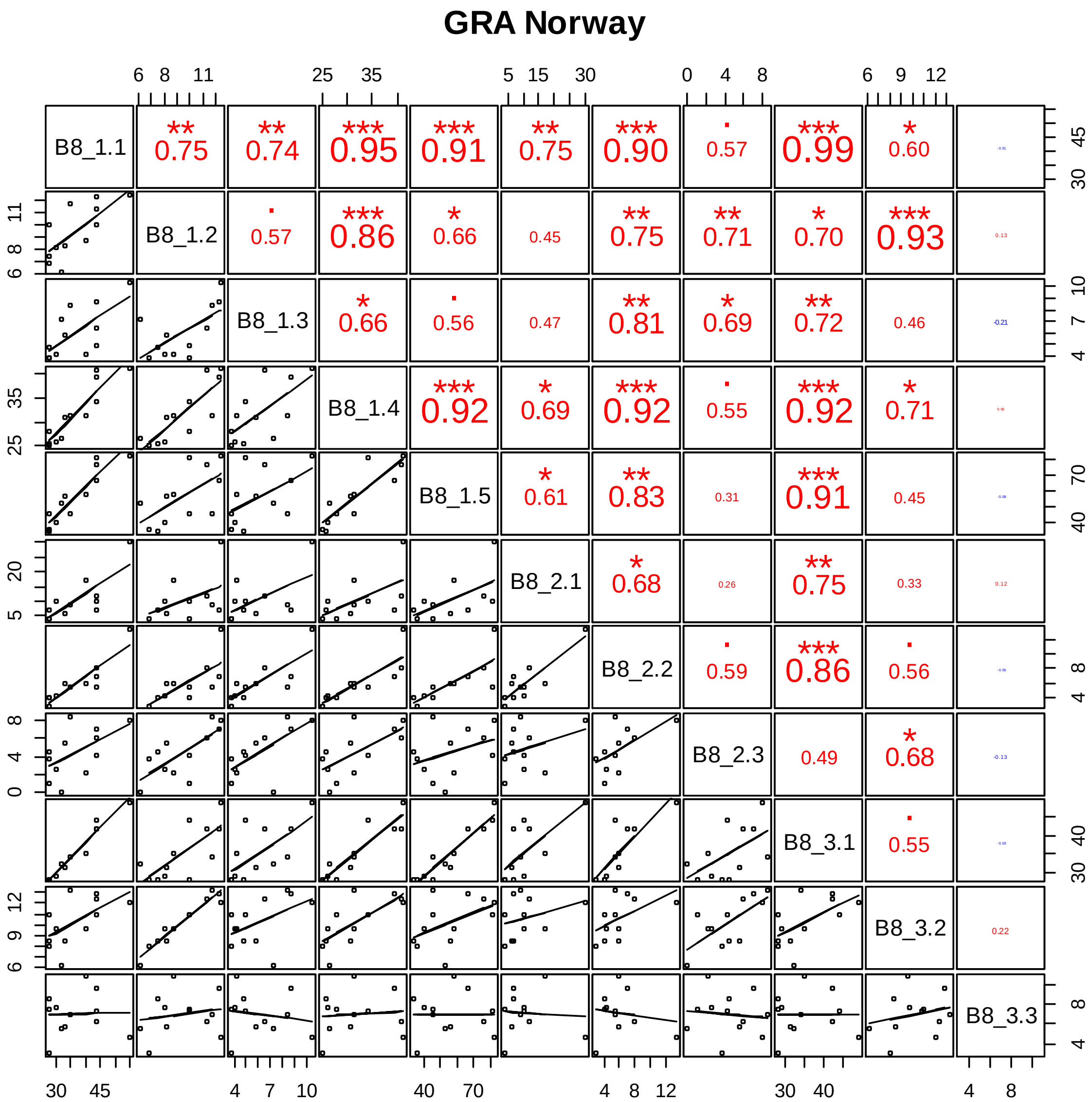
Plant diversity: Correlogram of species diversity indices of plant calculated from the species list with abundance. B2_1.1 = Gamma richness of plant in farms, B2_1.2 = Alpha richness, B2_1.3 = Area weighted richness, B2_1.4 = Rarefied richness, B2_1.5 = Chao estimated richness, B2_2.1 = Gamma richness of plant in cultivated forage and food crops, B2_2.2 = Alpha richness of plant in cultivated forage and food crops, B2_2.3 = Area weighted richness of plant in cultivated forage and food crops, B2_3.1 = Gamma richness of plant in semi-natural habitats, B2_3.2 = Alpha richness of plant in semi-natural habitats, B2_3.3 = Area weighted richness of plant in semi-natural habitats.



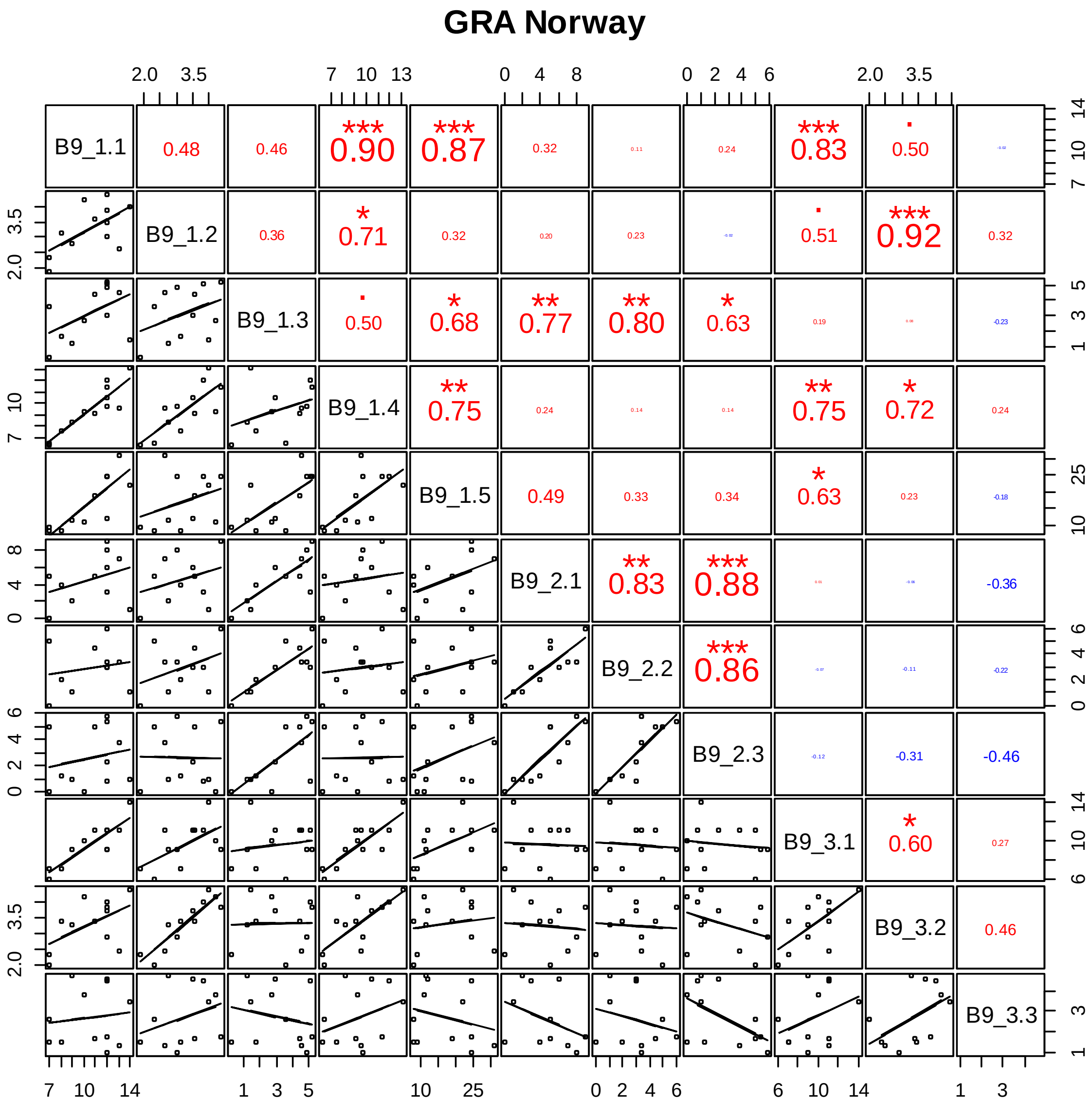
Earthworm diversity: Correlogram of species diversity indices of earthworms calculated from the species list with abundance. B4_1.1 = Gamma richness of earthworms in farms, B4_1.2 = Alpha richness, B4_1.3 = Area weighted richness, B4_1.4 = Rarefied richness, B4_1.5 = Chao estimated richness, B4_2.1 = Gamma richness of earthworms in cultivated forage and food crops, B4_2.2 = Alpha richness of earthworms in cultivated forage and food crops, B4_2.3 = Area weighted richness of earthworms in cultivated forage and food crops, B4_3.1 = Gamma richness of earthworms in semi-natural habitats, B4_3.2 = Alpha richness of earthworms in semi-natural habitats, B4_3.3 = Area weighted richness of earthworms in semi-natural habitats.



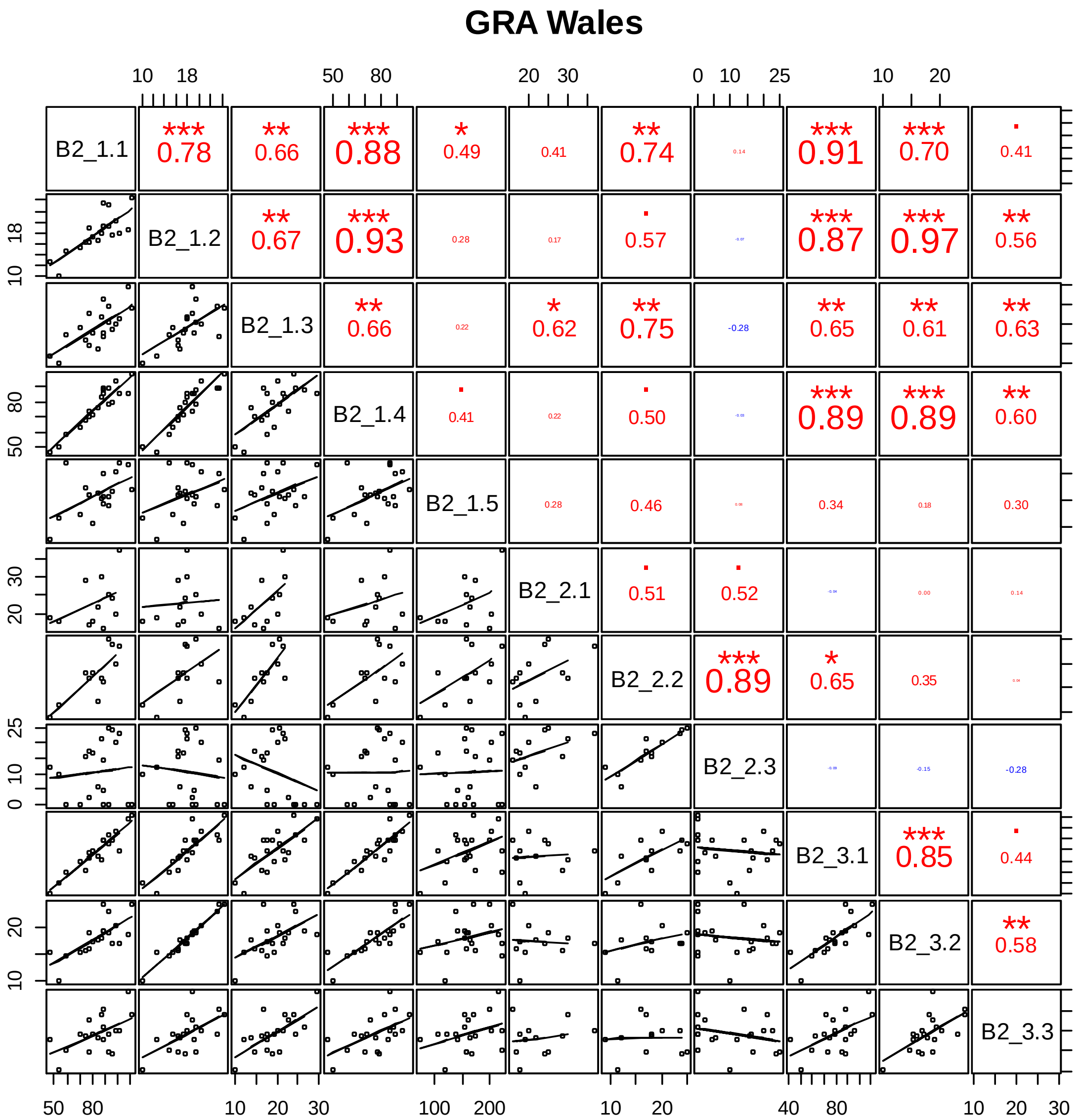
Spider diversity: Correlogram of species diversity indices of spiders calculated from the species list with abundance. B8_1.1 = Gamma richness of spiders in farms, B8_1.2 = Alpha richness, B8_1.3 = Area weighted richness, B8_1.4 = Rarefied richness, B8_1.5 = Chao estimated richness, B8_2.1 = Gamma richness of spiders in cultivated forage and food crops, B8_2.2 = Alpha richness of spiders in cultivated forage and food crops, B8_2.3 = Area weighted richness of spiders in cultivated forage and food crops, B8_3.1 = Gamma richness of spiders in semi-natural habitats, B8_3.2 = Alpha richness of spiders in semi-natural habitats, B8_3.3 = Area weighted richness of spiders in semi-natural habitats.



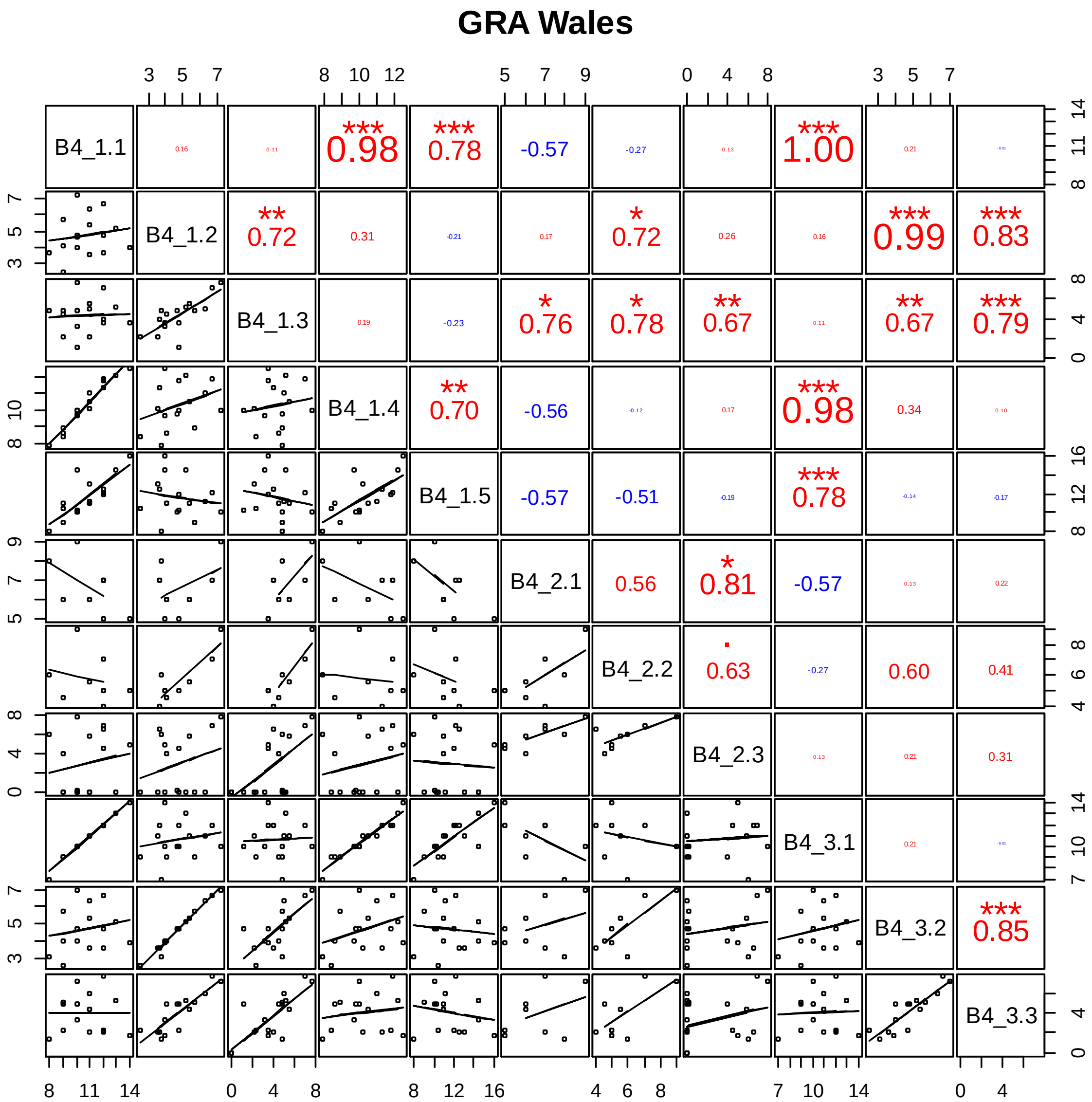
Bee diversity: Correlogram of species diversity indices of bees calculated from the species list with abundance. B9_1.1 = Gamma richness of bees in farms, B9_1.2 = Alpha richness, B9_1.3 = Area weighted richness, B9_1.4 = Rarefied richness, B9_1.5 = Chao estimated richness, B9_2.1 = Gamma richness of bees in cultivated forage and food crops, B9_2.2 = Alpha richness of bees in cultivated forage and food crops, B9_2.3 = Area weighted richness of bees in cultivated forage and food crops, B9_3.1 = Gamma richness of bees in semi-natural habitats, B9_3.2 = Alpha richness of bees in semi-natural habitats, B9_3.3 = Area weighted richness of bees in semi-natural habitats.



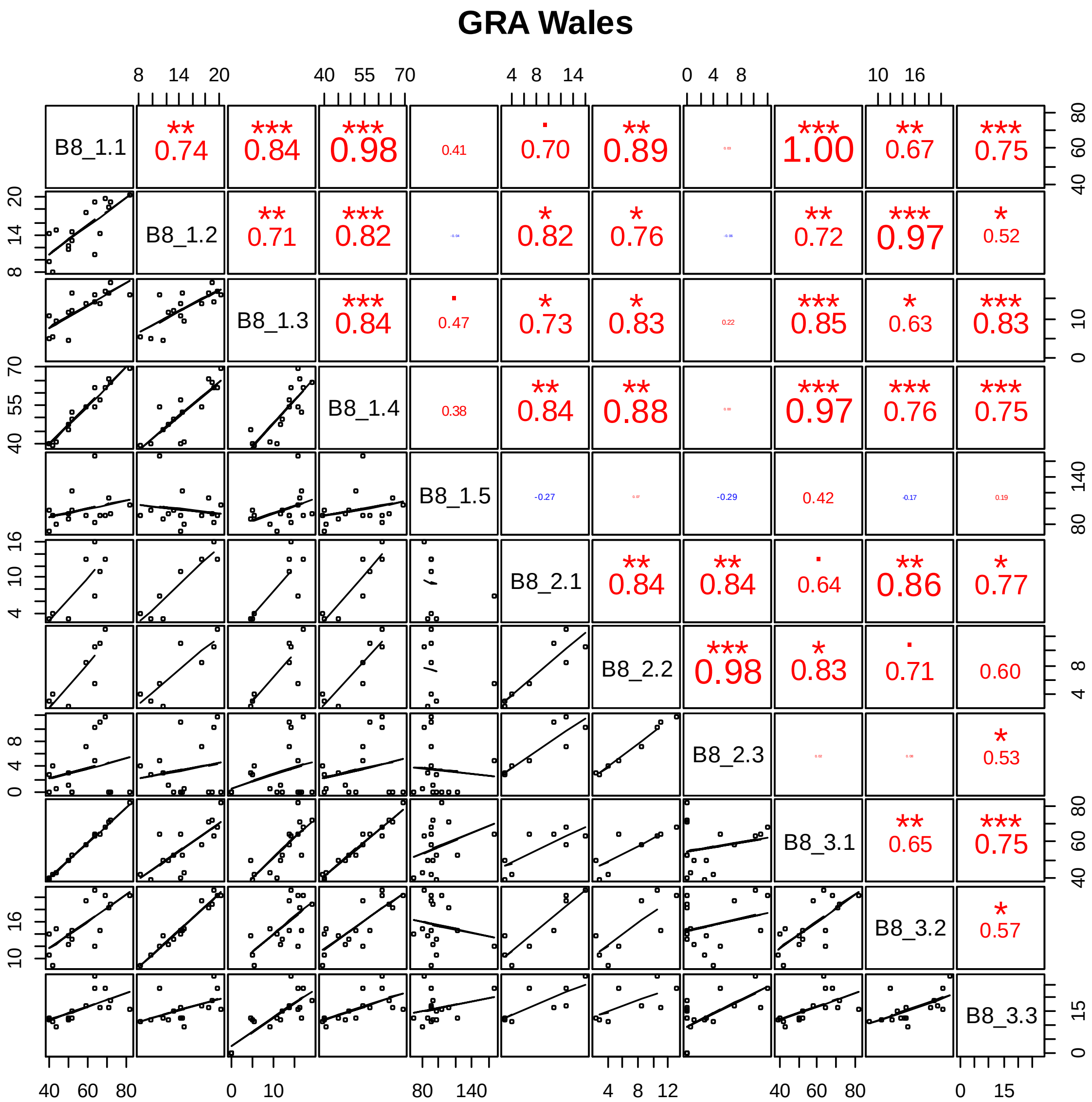
Plant diversity: Correlogram of species diversity indices of plant calculated from the species list with abundance. B2_1.1 = Gamma richness of plant in farms, B2_1.2 = Alpha richness, B2_1.3 = Area weighted richness, B2_1.4 = Rarefied richness, B2_1.5 = Chao estimated richness, B2_2.1 = Gamma richness of plant in cultivated forage and food crops, B2_2.2 = Alpha richness of plant in cultivated forage and food crops, B2_2.3 = Area weighted richness of plant in cultivated forage and food crops, B2_3.1 = Gamma richness of plant in semi-natural habitats, B2_3.2 = Alpha richness of plant in semi-natural habitats, B2_3.3 = Area weighted richness of plant in semi-natural habitats.



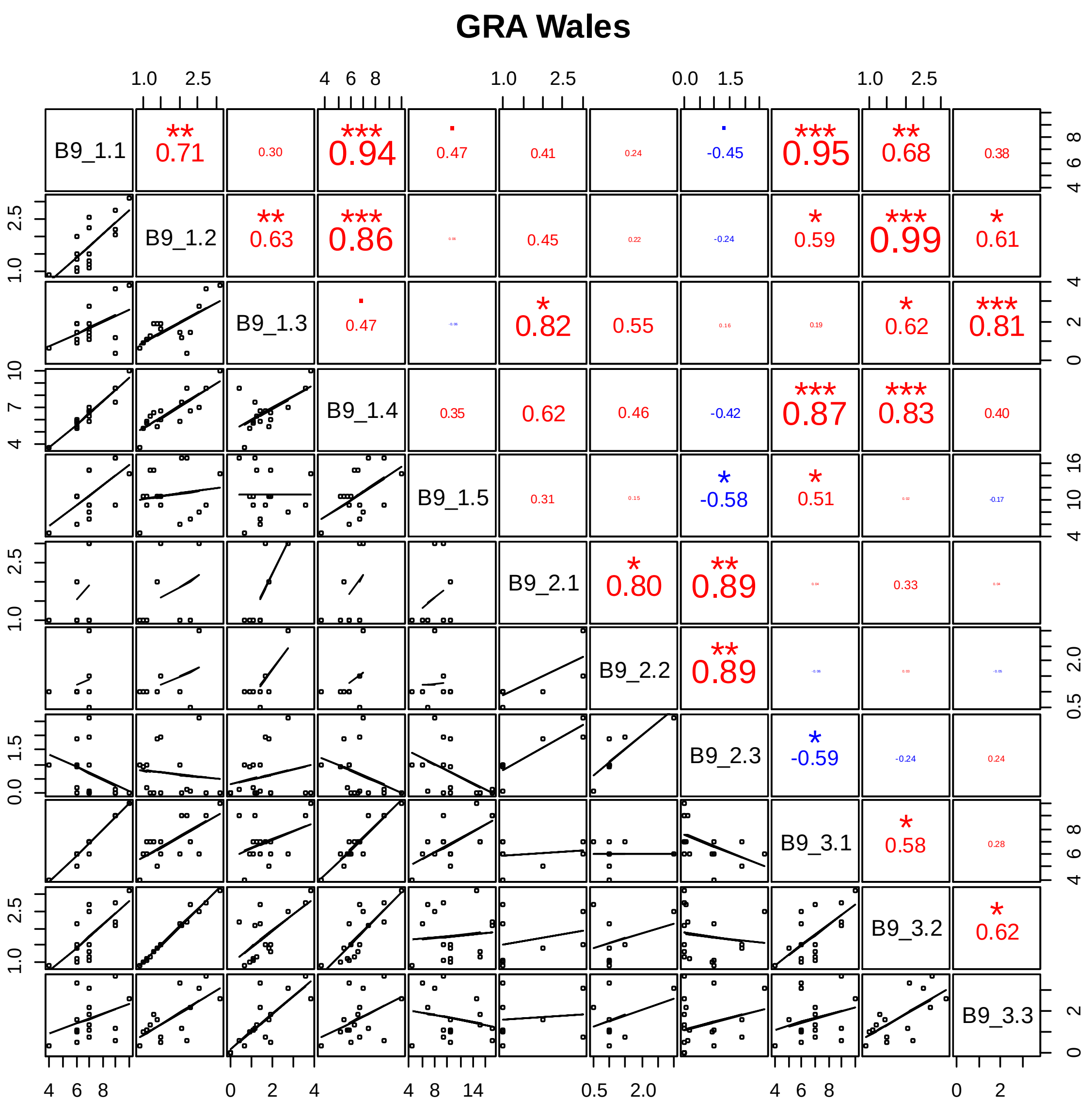
Earthworm diversity: Correlogram of species diversity indices of earthworms calculated from the species list with abundance. B4_1.1 = Gamma richness of earthworms in farms, B4_1.2 = Alpha richness, B4_1.3 = Area weighted richness, B4_1.4 = Rarefied richness, B4_1.5 = Chao estimated richness, B4_2.1 = Gamma richness of earthworms in cultivated forage and food crops, B4_2.2 = Alpha richness of earthworms in cultivated forage and food crops, B4_2.3 = Area weighted richness of earthworms in cultivated forage and food crops, B4_3.1 = Gamma richness of earthworms in semi-natural habitats, B4_3.2 = Alpha richness of earthworms in semi-natural habitats, B4_3.3 = Area weighted richness of earthworms in semi-natural habitats.



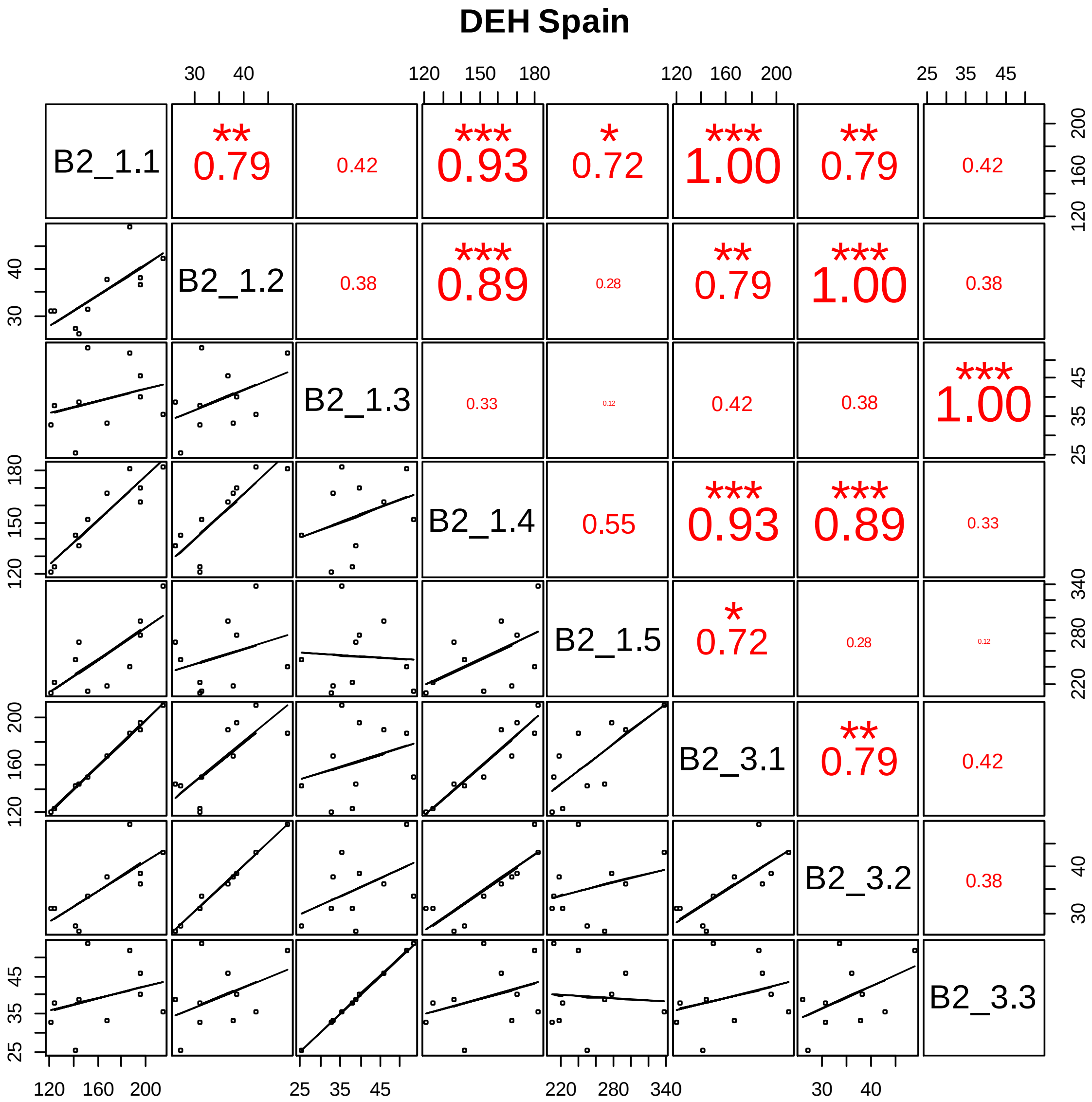
Spider diversity: Correlogram of species diversity indices of spiders calculated from the species list with abundance. B8_1.1 = Gamma richness of spiders in farms, B8_1.2 = Alpha richness, B8_1.3 = Area weighted richness, B8_1.4 = Rarefied richness, B8_1.5 = Chao estimated richness, B8_2.1 = Gamma richness of spiders in cultivated forage and food crops, B8_2.2 = Alpha richness of spiders in cultivated forage and food crops, B8_2.3 = Area weighted richness of spiders in cultivated forage and food crops, B8_3.1 = Gamma richness of spiders in semi-natural habitats, B8_3.2 = Alpha richness of spiders in semi-natural habitats, B8_3.3 = Area weighted richness of spiders in semi-natural habitats.



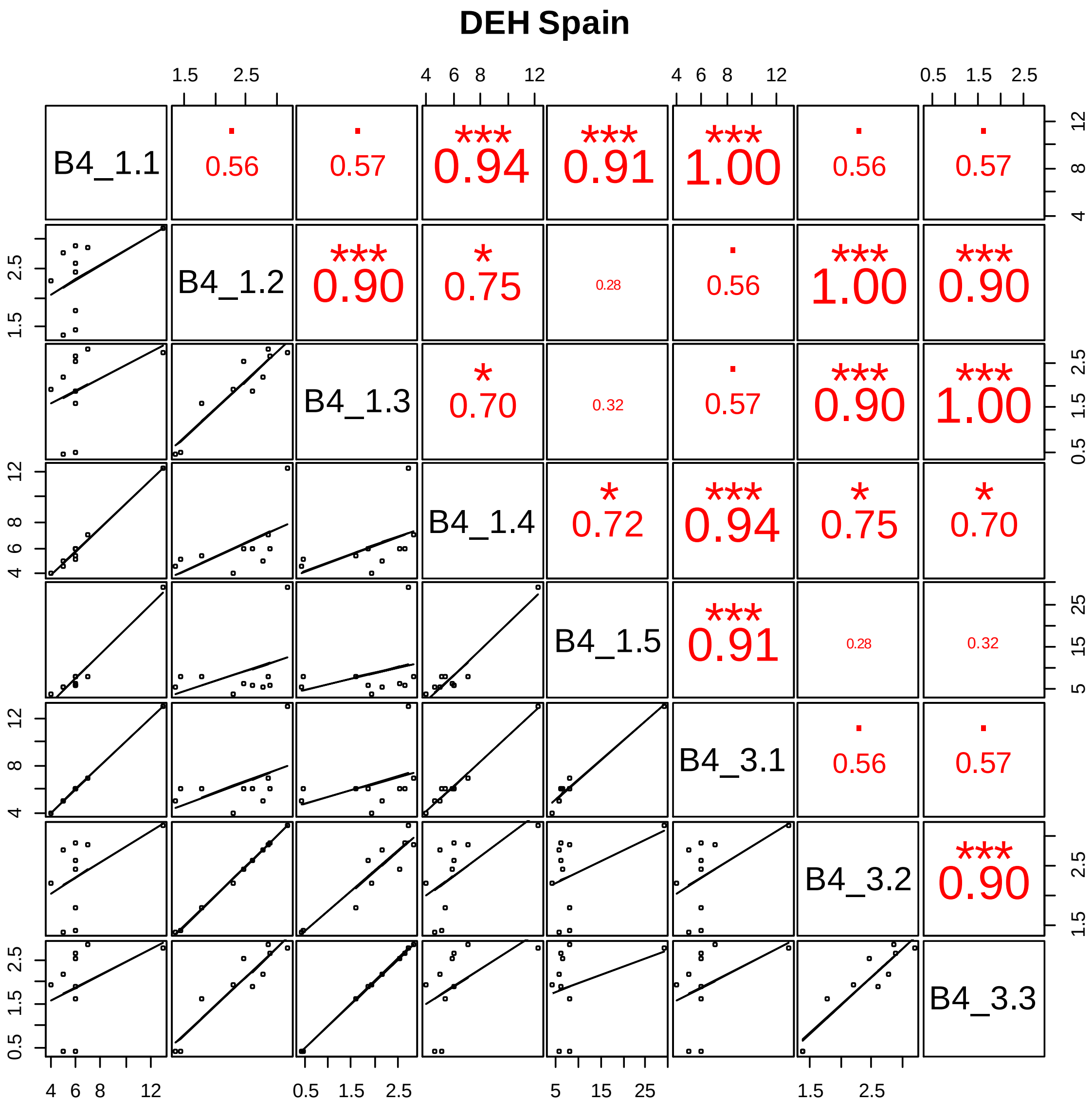
Bee diversity: Correlogram of species diversity indices of bees calculated from the species list with abundance. B9_1.1 = Gamma richness of bees in farms, B9_1.2 = Alpha richness, B9_1.3 = Area weighted richness, B9_1.4 = Rarefied richness, B9_1.5 = Chao estimated richness, B9_2.1 = Gamma richness of bees in cultivated forage and food crops, B9_2.2 = Alpha richness of bees in cultivated forage and food crops, B9_2.3 = Area weighted richness of bees in cultivated forage and food crops, B9_3.1 = Gamma richness of bees in semi-natural habitats, B9_3.2 = Alpha richness of bees in semi-natural habitats, B9_3.3 = Area weighted richness of bees in semi-natural habitats.



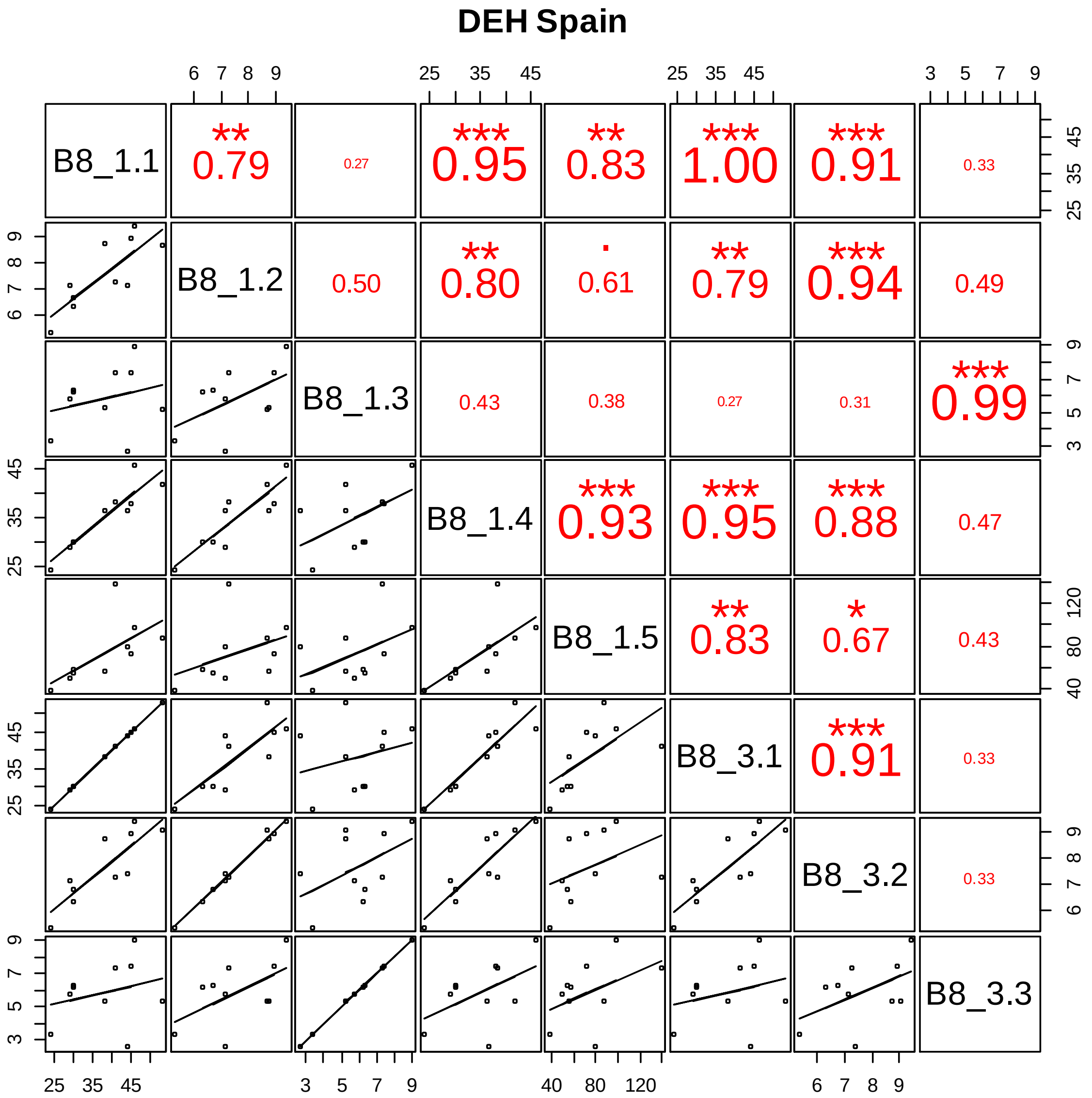
Plant diversity: Correlogram of species diversity indices of plants calculated from the species list with abundance. B2_1.1 = Gamma richness of plants in farms, B2_1.2 = Alpha richness, B2_1.3 = Area weighted richness, B2_1.4 = Rarefied richness, B2_1.5 = Chao estimated richness, B2_3.1 = Gamma richness of plants in semi-natural habitats, B2_3.2 = Alpha richness of plants in semi-natural habitats, B2_3.3 = Area weighted richness of plants in semi-natural habitats.



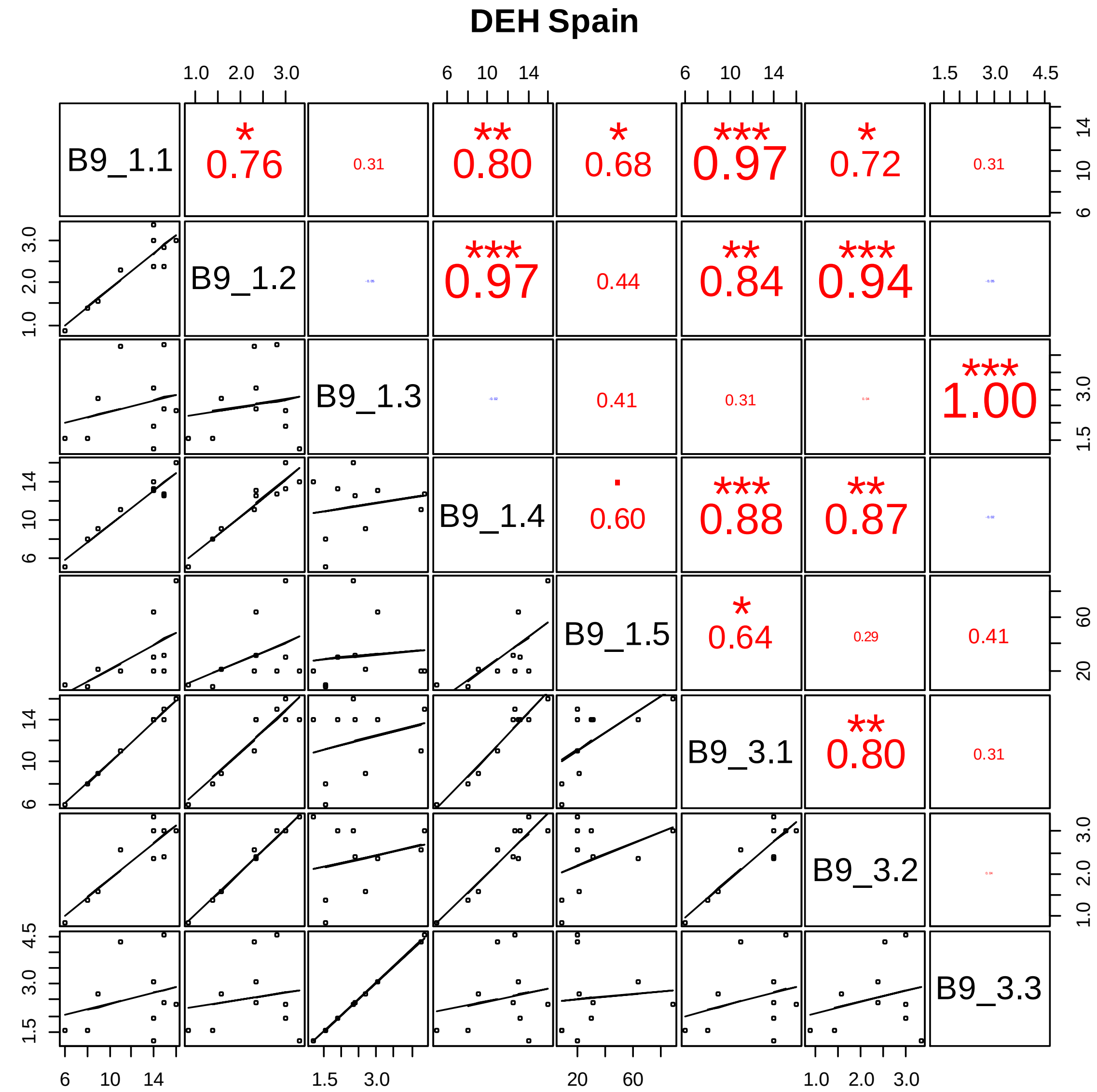
Earthworm diversity: Correlogram of species diversity indices of earthworms calculated from the species list with abundance. B4_1.1 = Gamma richness of earthworms in farms, B4_1.2 = Alpha richness, B4_1.3 = Area weighted richness, B4_1.4 = Rarefied richness, B4_1.5 = Chao estimated richness, B4_3.1 = Gamma richness of earthworms in semi-natural habitats, B4_3.2 = Alpha richness of earthworms in semi-natural habitats, B4_3.3 = Area weighted richness of earthworms in semi-natural habitats



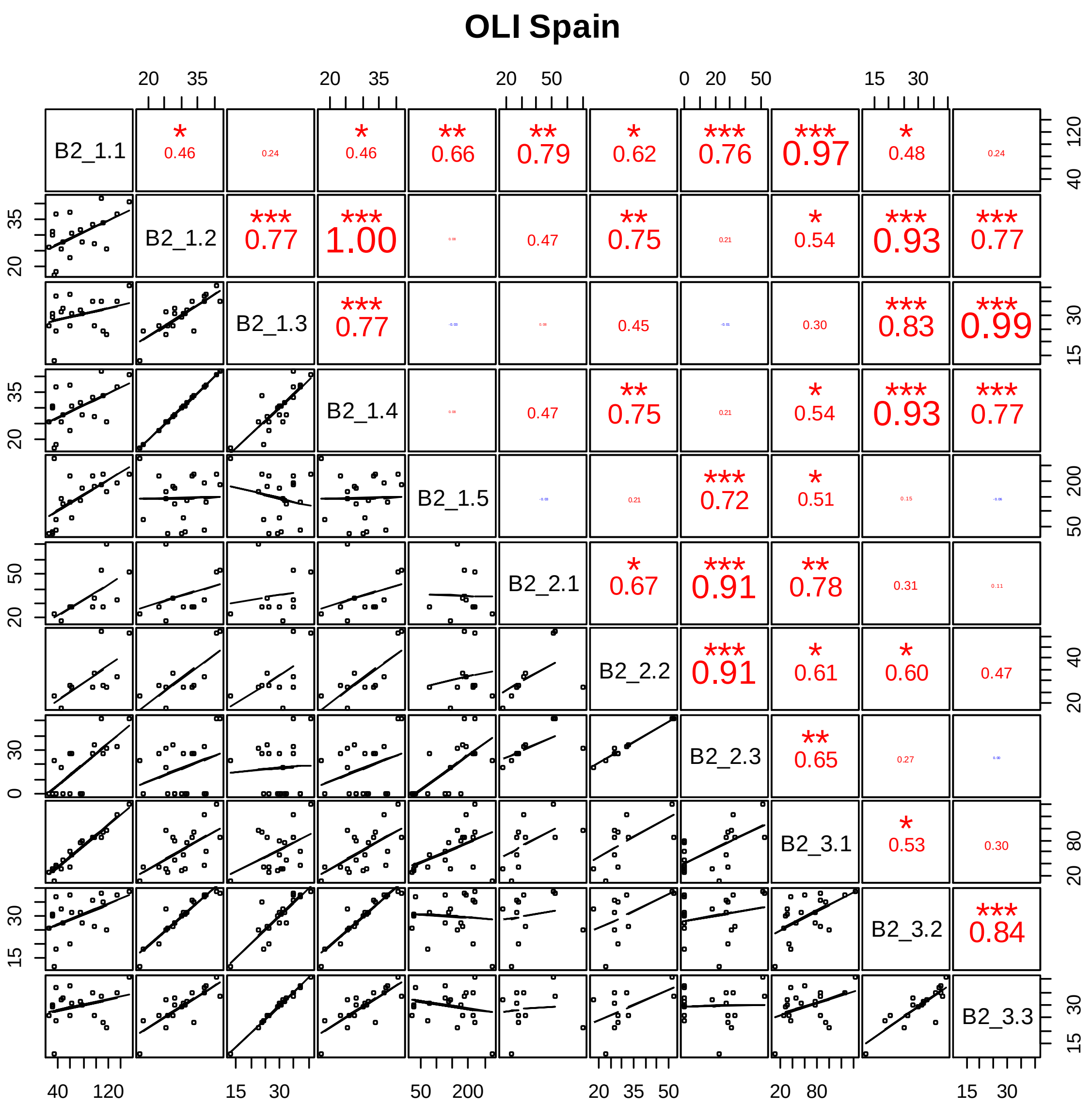
Spider diversity: Correlogram of species diversity indices of spiders calculated from the species list with abundance. B8_1.1 = Gamma richness of spiders in farms, B8_1.2 = Alpha richness, B8_1.3 = Area weighted richness, B8_1.4 = Rarefied richness, B8_1.5 = Chao estimated richness, B8_3.1 = Gamma richness of spiders in semi-natural habitats, B8_3.2 = Alpha richness of spiders in semi-natural habitats, B8_3.3 = Area weighted richness of spiders in semi-natural habitats.



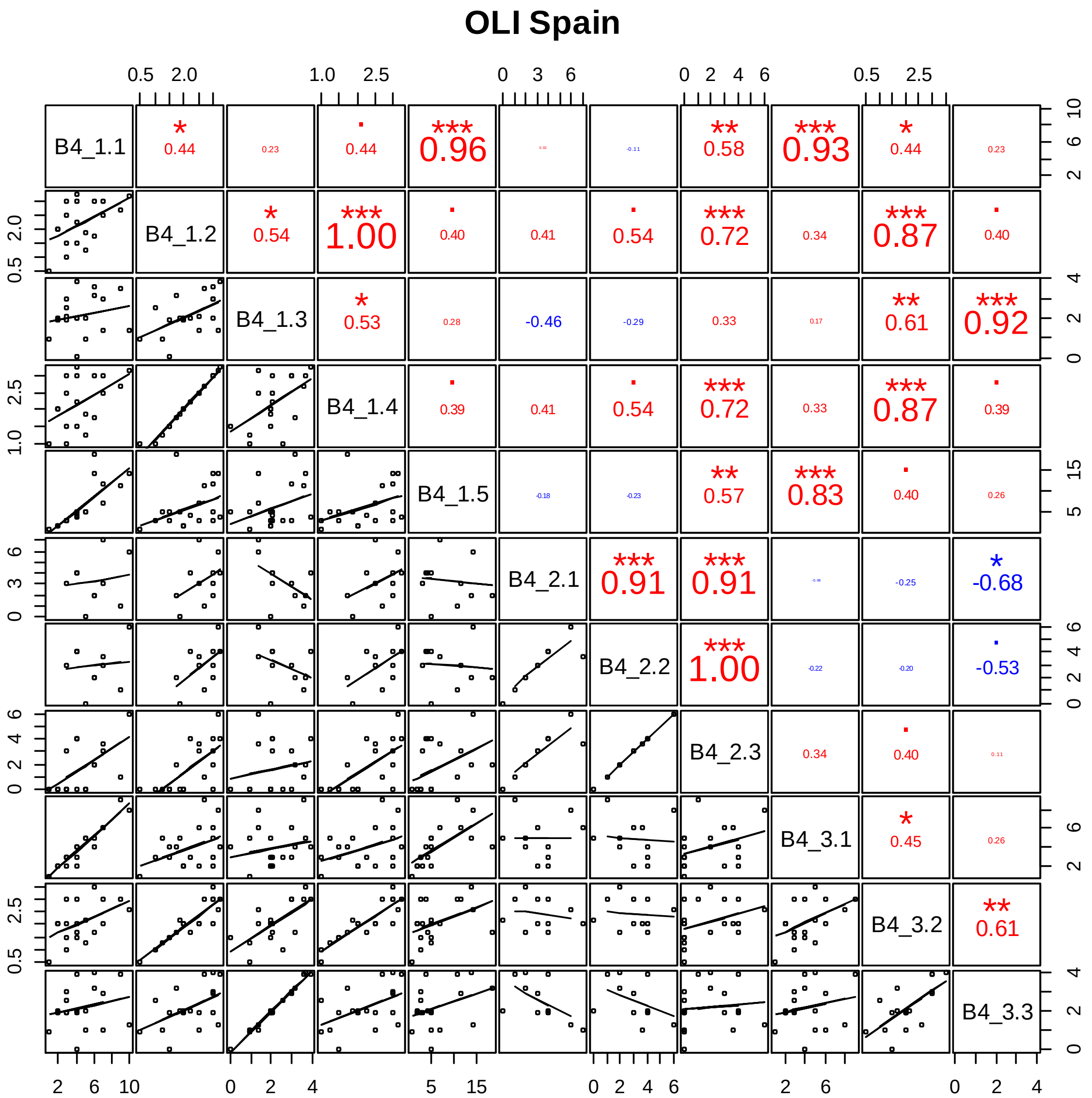
Bee diversity: Correlogram of species diversity indices of bees calculated from the species list with abundance. B9_1.1 = Gamma richness of bees in farms, B9_1.2 = Alpha richness, B9_1.3 = Area weighted richness, B9_1.4 = Rarefied richness, B9_1.5 = Chao estimated richness, B9_3.1 = Gamma richness of bees in semi-natural habitats, B9_3.2 = Alpha richness of bees in semi-natural habitats, B9_3.3 = Area weighted richness of bees in semi-natural habitats.



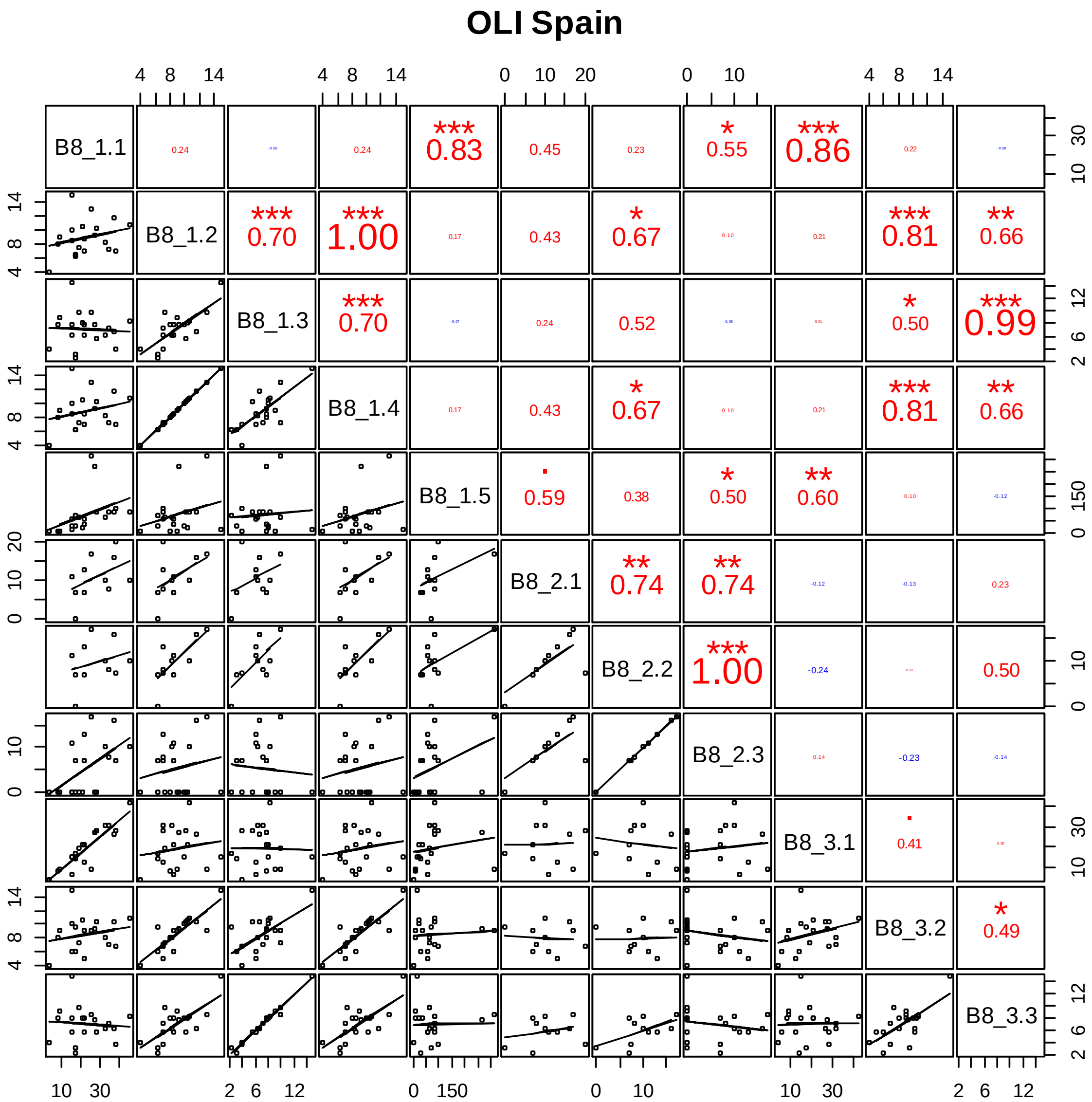
Plant diversity: Correlogram of species diversity indices of plant calculated from the species list with abundance. B2_1.1 = Gamma richness of plant in farms, B2_1.2 = Alpha richness, B2_1.3 = Area weighted richness, B2_1.4 = Rarefied richness, B2_1.5 = Chao estimated richness, B2_2.1 = Gamma richness of plant in cultivated forage and food crops, B2_2.2 = Alpha richness of plant in cultivated forage and food crops, B2_2.3 = Area weighted richness of plant in cultivated forage and food crops, B2_3.1 = Gamma richness of plant in semi-natural habitats, B2_3.2 = Alpha richness of plant in semi-natural habitats, B2_3.3 = Area weighted richness of plant in semi-natural habitats.



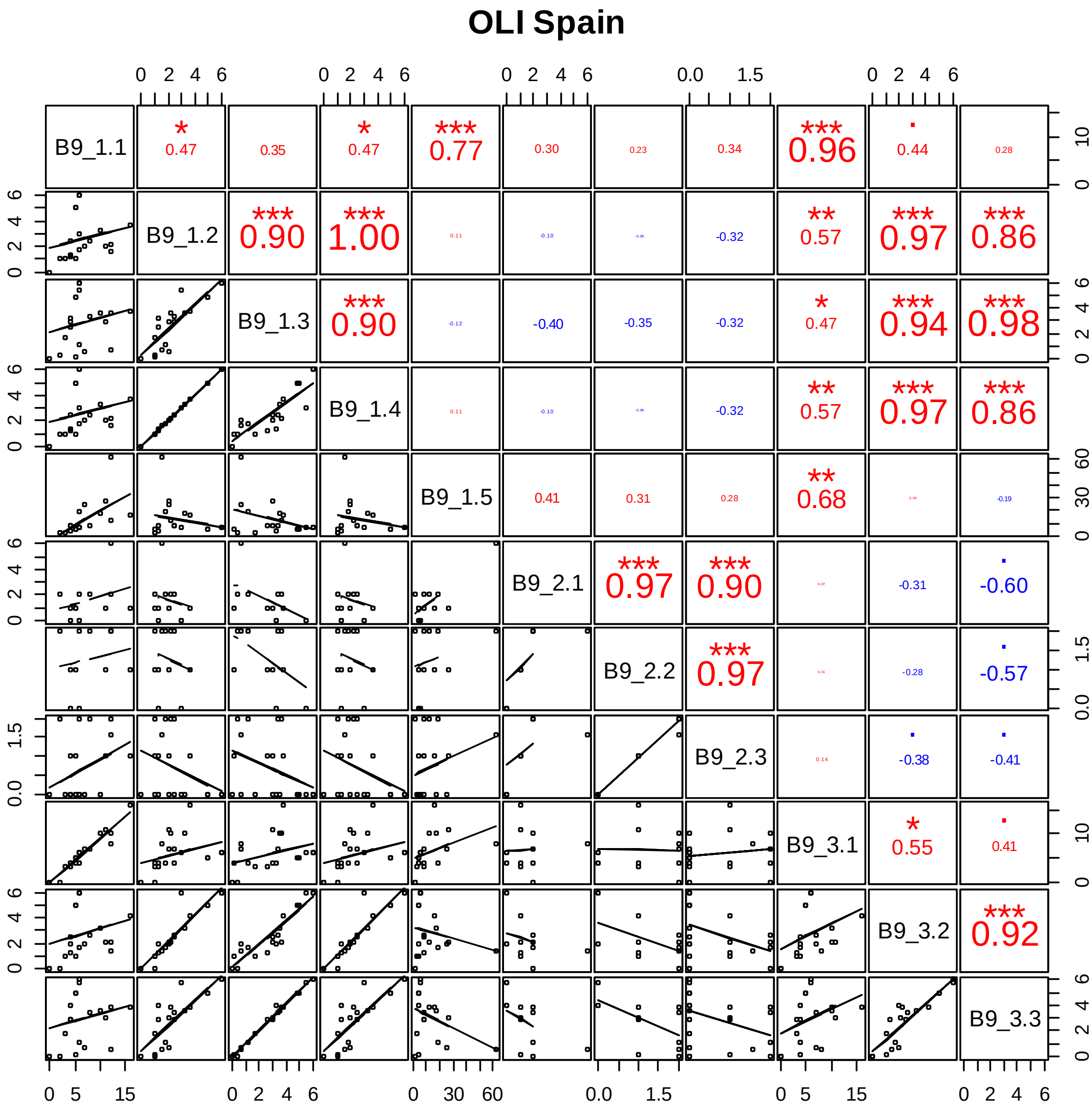
Earthworm diversity: Correlogram of species diversity indices of earthworms calculated from the species list with abundance. B4_1.1 = Gamma richness of earthworms in farms, B4_1.2 = Alpha richness, B4_1.3 = Area weighted richness, B4_1.4 = Rarefied richness, B4_1.5 = Chao estimated richness, B4_2.1 = Gamma richness of earthworms in cultivated forage and food crops, B4_2.2 = Alpha richness of earthworms in cultivated forage and food crops, B4_2.3 = Area weighted richness of earthworms in cultivated forage and food crops, B4_3.1 = Gamma richness of earthworms in semi-natural habitats, B4_3.2 = Alpha richness of earthworms in semi-natural habitats, B4_3.3 = Area weighted richness of earthworms in semi-natural habitats.



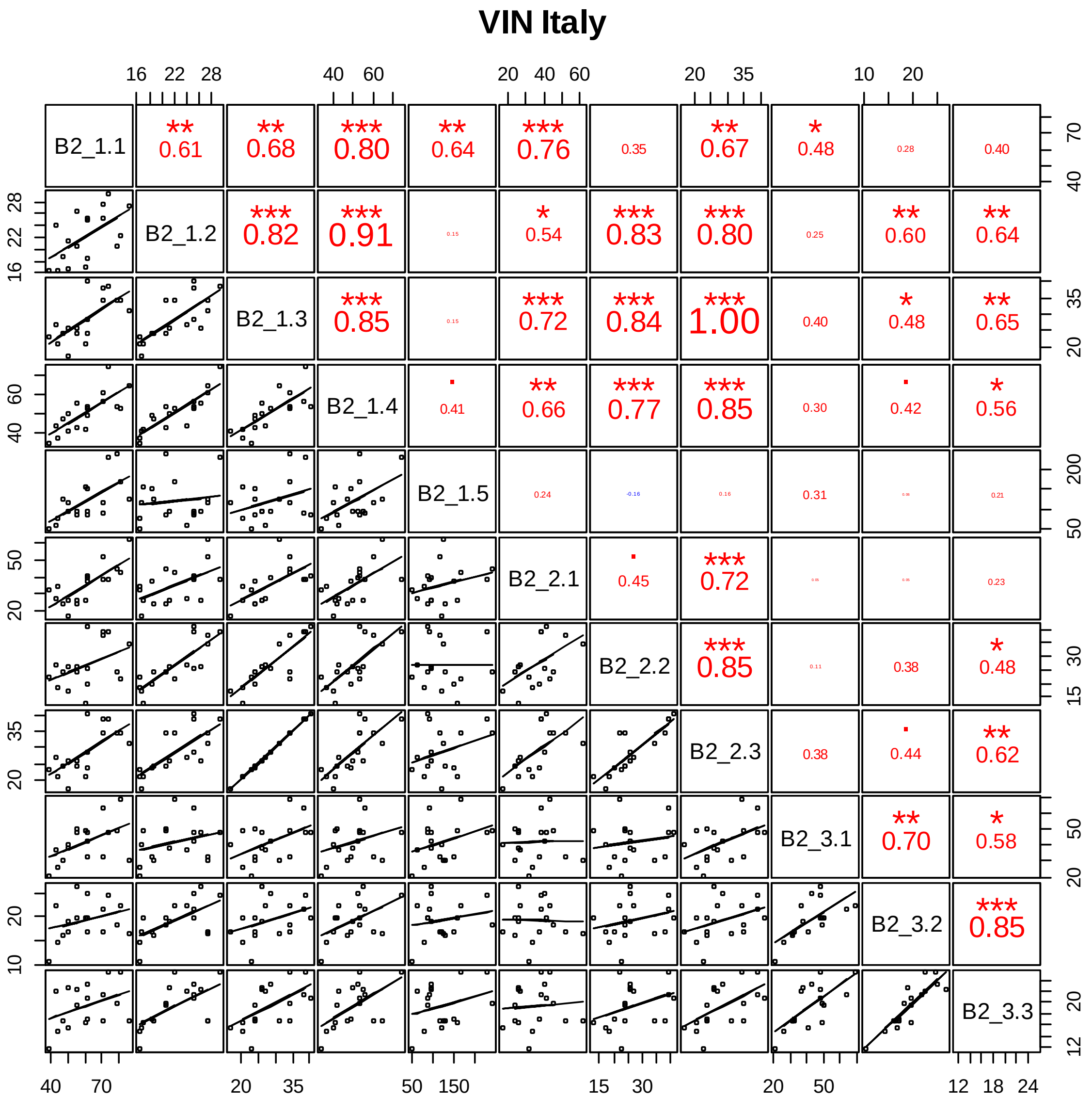
Spider diversity: Correlogram of species diversity indices of spiders calculated from the species list with abundance. B8_1.1 = Gamma richness of spiders in farms, B8_1.2 = Alpha richness, B8_1.3 = Area weighted richness, B8_1.4 = Rarefied richness, B8_1.5 = Chao estimated richness, B8_2.1 = Gamma richness of spiders in cultivated forage and food crops, B8_2.2 = Alpha richness of spiders in cultivated forage and food crops, B8_2.3 = Area weighted richness of spiders in cultivated forage and food crops, B8_3.1 = Gamma richness of spiders in semi-natural habitats, B8_3.2 = Alpha richness of spiders in semi-natural habitats, B8_3.3 = Area weighted richness of spiders in semi-natural habitats.



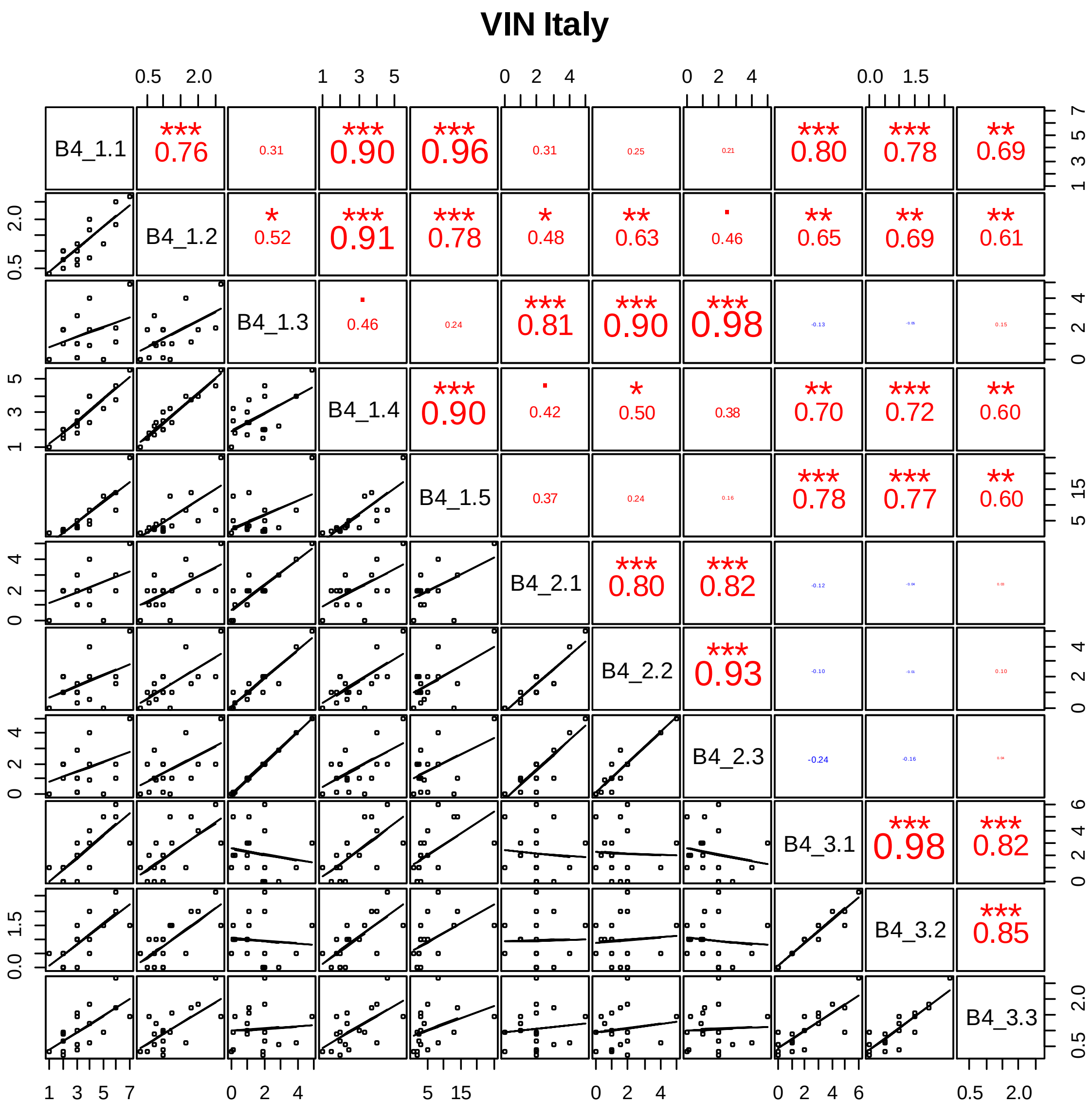
Bee diversity: Correlogram of species diversity indices of bees calculated from the species list with abundance. B9_1.1 = Gamma richness of bees in farms, B9_1.2 = Alpha richness, B9_1.3 = Area weighted richness, B9_1.4 = Rarefied richness, B9_1.5 = Chao estimated richness, B9_2.1 = Gamma richness of bees in cultivated forage and food crops, B9_2.2 = Alpha richness of bees in cultivated forage and food crops, B9_2.3 = Area weighted richness of bees in cultivated forage and food crops, B9_3.1 = Gamma richness of bees in semi-natural habitats, B9_3.2 = Alpha richness of bees in semi-natural habitats, B9_3.3 = Area weighted richness of bees in semi-natural habitats.



Plant diversity: Correlogram of species diversity indices of plant calculated from the species list with abundance. B2_1.1 = Gamma richness of plant in farms, B2_1.2 = Alpha richness, B2_1.3 = Area weighted richness, B2_1.4 = Rarefied richness, B2_1.5 = Chao estimated richness, B2_2.1 = Gamma richness of plant in cultivated forage and food crops, B2_2.2 = Alpha richness of plant in cultivated forage and food crops, B2_2.3 = Area weighted richness of plant in cultivated forage and food crops, B2_3.1 = Gamma richness of plant in semi-natural habitats, B2_3.2 = Alpha richness of plant in semi-natural habitats, B2_3.3 = Area weighted richness of plant in semi-natural habitats.

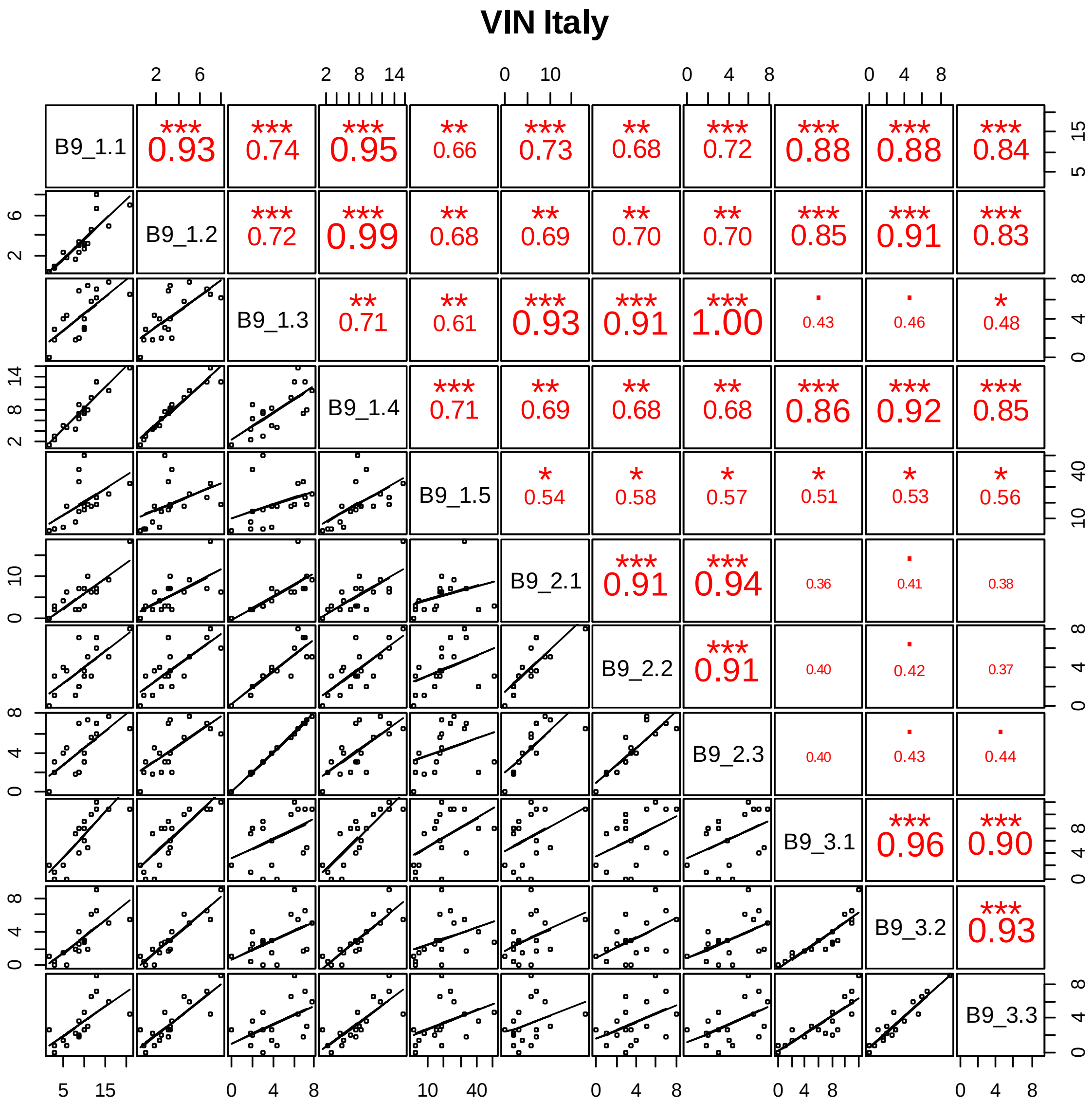


Earthworm diversity: Correlogram of species diversity indices of earthworms calculated from the species list with abundance. B4_1.1 = Gamma richness of earthworms in farms, B4_1.2 = Alpha richness, B4_1.3 = Area weighted richness, B4_1.4 = Rarefied richness, B4_1.5 = Chao estimated richness, B4_2.1 = Gamma richness of earthworms in cultivated forage and food crops, B4_2.2 = Alpha richness of earthworms in cultivated forage and food crops, B4_2.3 = Area weighted richness of earthworms in cultivated forage and food crops, B4_3.1 = Gamma richness of earthworms in semi-natural habitats, B4_3.2 = Alpha richness of earthworms in semi-natural habitats, B4_3.3 = Area weighted richness of earthworms in semi-natural habitats.

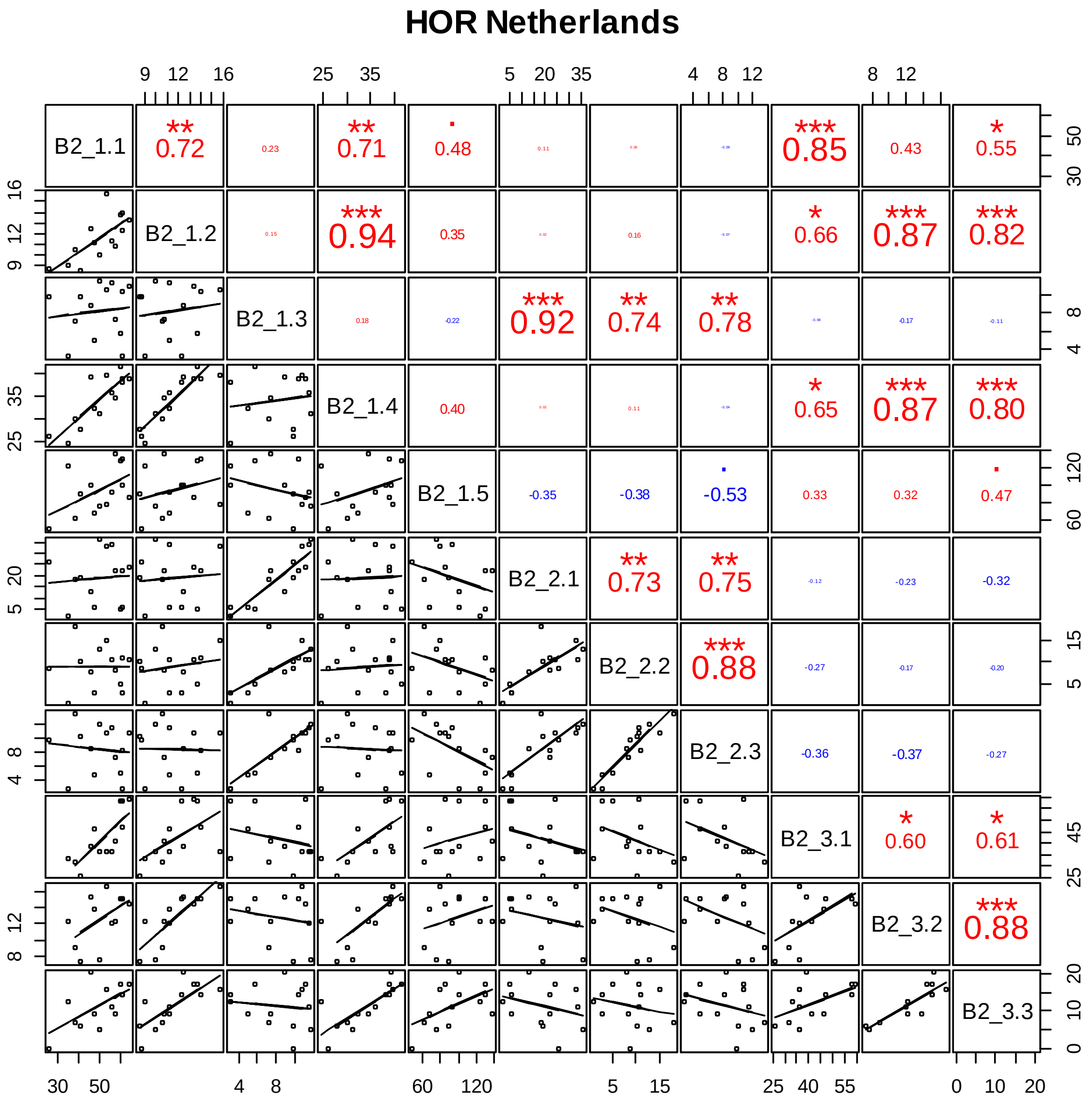


Spider diversity: Correlogram of species diversity indices of spiders calculated from the species list with abundance. B8_1.1 = Gamma richness of spiders in farms, B8_1.2 = Alpha richness, B8_1.3 = Area weighted richness, B8_1.4 = Rarefied richness, B8_1.5 = Chao estimated richness, B8_2.1 = Gamma richness of spiders in cultivated forage and food crops, B8_2.2 = Alpha richness of spiders in cultivated forage and food crops, B8_2.3 = Area weighted richness of spiders in cultivated forage and food crops, B8_3.1 = Gamma richness of spiders in semi-natural habitats, B8_3.2 = Alpha richness of spiders in semi-natural habitats, B8_3.3 = Area weighted richness of spiders in semi-natural habitats.

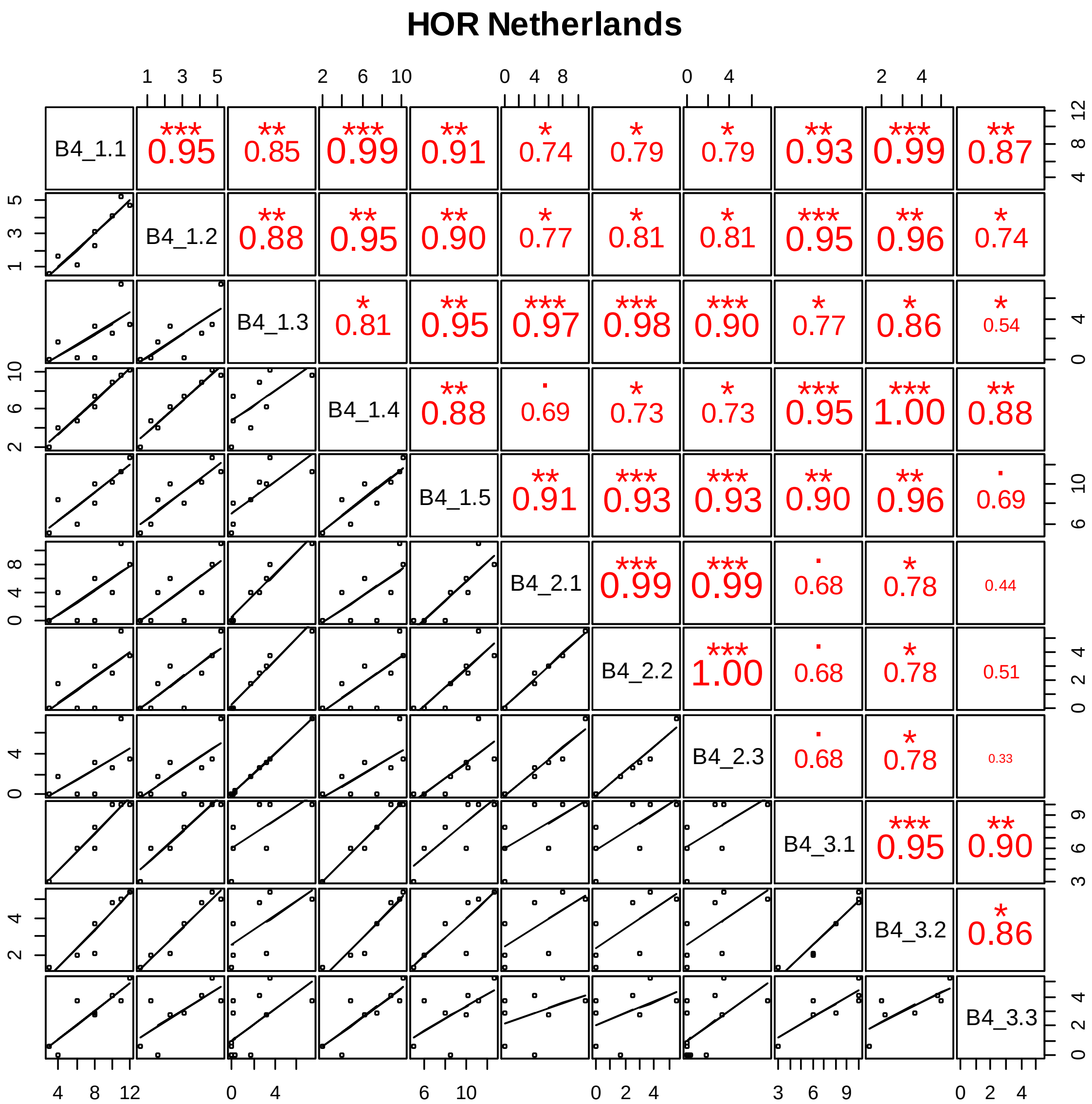
Bee diversity: Correlogram of species diversity indices of bees calculated from the species list with abundance. B9_1.1 = Gamma richness of bees in farms, B9_1.2 = Alpha richness, B9_1.3 = Area weighted richness, B9_1.4 = Rarefied richness, B9_1.5 = Chao estimated richness, B9_2.1 = Gamma richness of bees in cultivated forage and food crops, B9_2.2 = Alpha richness of bees in cultivated forage and food crops, B9_2.3 = Area weighted richness of bees in cultivated forage and food crops, B9_3.1 = Gamma richness of bees in semi-natural habitats, B9_3.2 = Alpha richness of bees in semi-natural habitats, B9_3.3 = Area weighted richness of bees in semi-natural habitats.



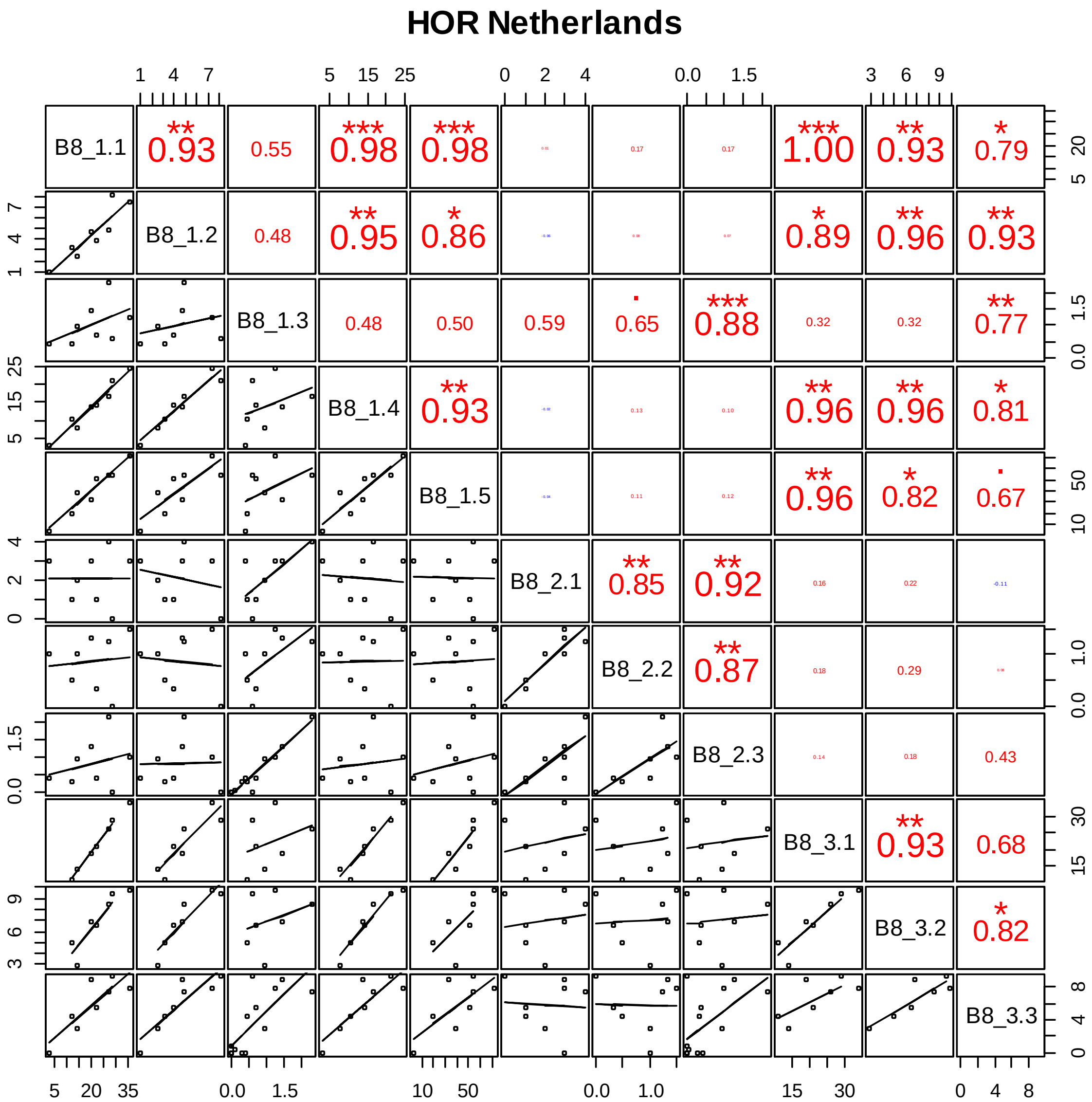
Plant diversity: Correlogram of species diversity indices of plant calculated from the species list with abundance. B2_1.1 = Gamma richness of plant in farms, B2_1.2 = Alpha richness, B2_1.3 = Area weighted richness, B2_1.4 = Rarefied richness, B2_1.5 = Chao estimated richness, B2_2.1 = Gamma richness of plant in cultivated forage and food crops, B2_2.2 = Alpha richness of plant in cultivated forage and food crops, B2_2.3 = Area weighted richness of plant in semi-natural habitats, B2_3.1 = Gamma richness of plant in semi-natural habitats, B2_3.2 = Alpha richness of plant in semi-natural habitats, B2_3.3 = Area weighted richness of plant in semi-natural habitats.



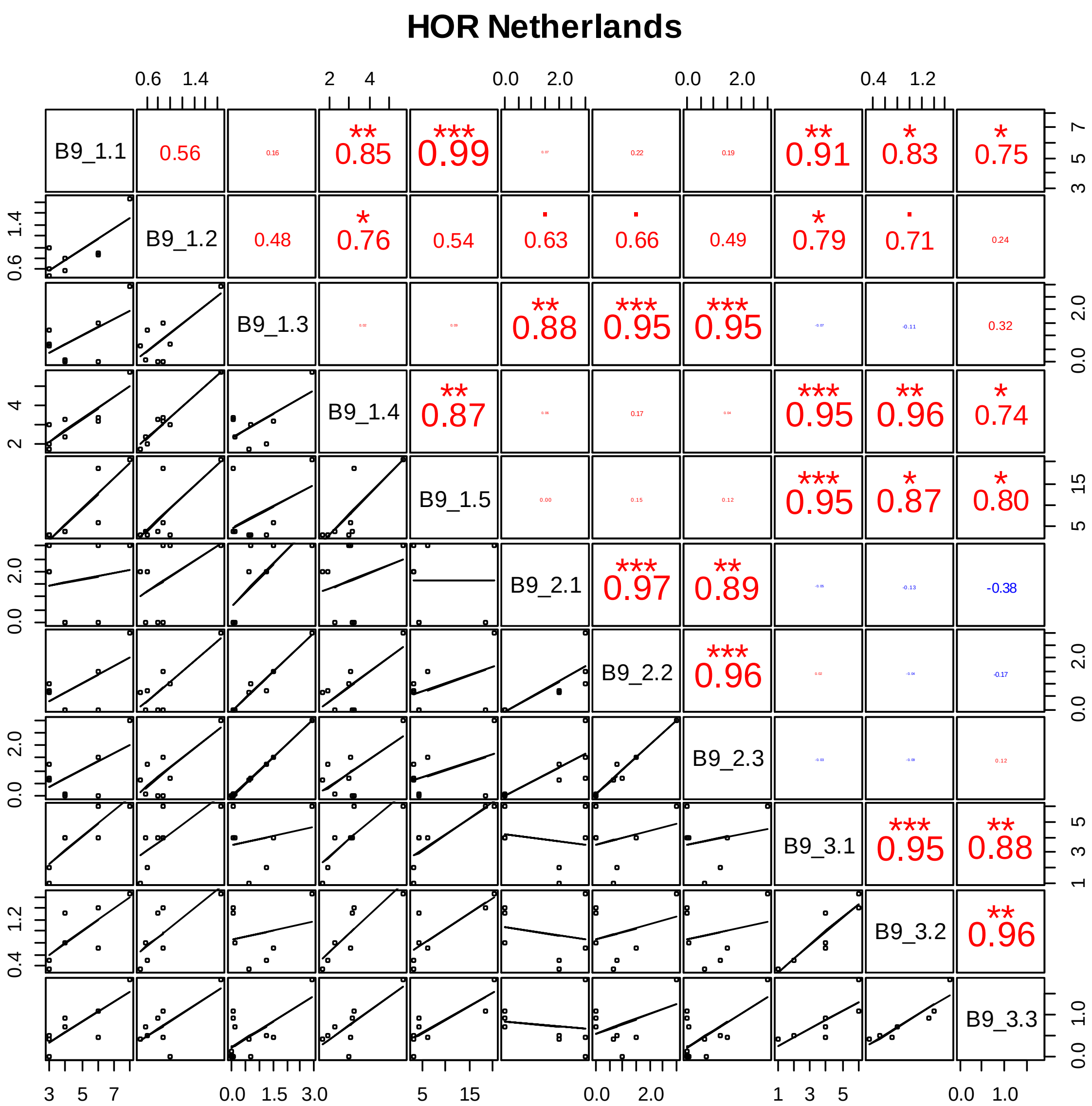
Earthworm diversity: Correlogram of species diversity indices of earthworms calculated from the species list with abundance. B4_1.1 = Gamma richness of earthworms in farms, B4_1.2 = Alpha richness, B4_1.3 = Area weighted richness, B4_1.4 = Rarefied richness, B4_1.5 = Chao estimated richness, B4_2.1 = Gamma richness of earthworms in cultivated forage and food crops, B4_2.2 = Alpha richness of earthworms in cultivated forage and food crops, B4_2.3 = Area weighted richness of earthworms in cultivated forage and food crops, B4_3.1 = Gamma richness of earthworms in semi-natural habitats, B4_3.2 = Alpha richness of earthworms in semi-natural habitats, B4_3.3 = Area weighted richness of earthworms in semi-natural habitats.



Spider diversity: Correlogram of species diversity indices of spiders calculated from the species list with abundance. B8_1.1 = Gamma richness of spiders in farms, B8_1.2 = Alpha richness, B8_1.3 = Area weighted richness, B8_1.4 = Rarefied richness, B8_1.5 = Chao estimated richness, B8_2.1 = Gamma richness of spiders in cultivated forage and food crops, B8_2.2 = Alpha richness of spiders in cultivated forage and food crops, B8_2.3 = Area weighted richness of spiders in cultivated forage and food crops, B8_3.1 = Gamma richness of spiders in semi-natural habitats, B8_3.2 = Alpha richness of spiders in semi-natural habitats, B8_3.3 = Area weighted richness of spiders in semi-natural habitats.

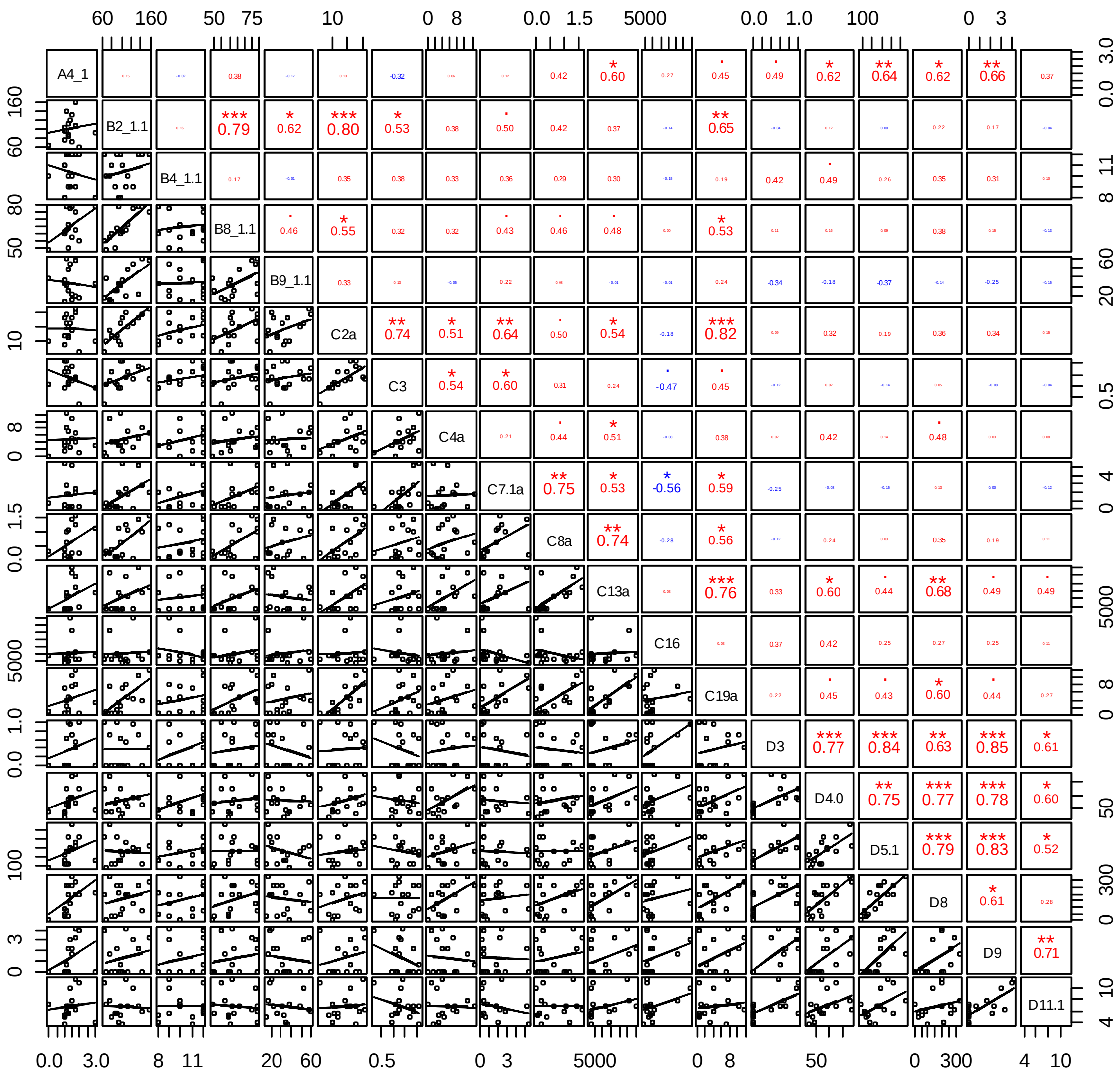


Bee diversity: Correlogram of species diversity indices of bees calculated from the species list with abundance. B9_1.1 = Gamma richness of bees in farms, B9_1.2 = Alpha richness, B9_1.3 = Area weighted richness, B9_1.4 = Rarefied richness, B9_1.5 = Chao estimated richness, B9_2.1 = Gamma richness of bees in cultivated forage and food crops, B9_2.2 = Alpha richness of bees in cultivated forage and food crops, B9_2.3 = Area weighted richness of bees in cultivated forage and food crops, B9_3.1 = Gamma richness of bees in semi-natural habitats, B9_3.2 = Alpha richness of bees in semi-natural habitats, B9_3.3 = Area weighted richness of bees in semi-natural habitats.

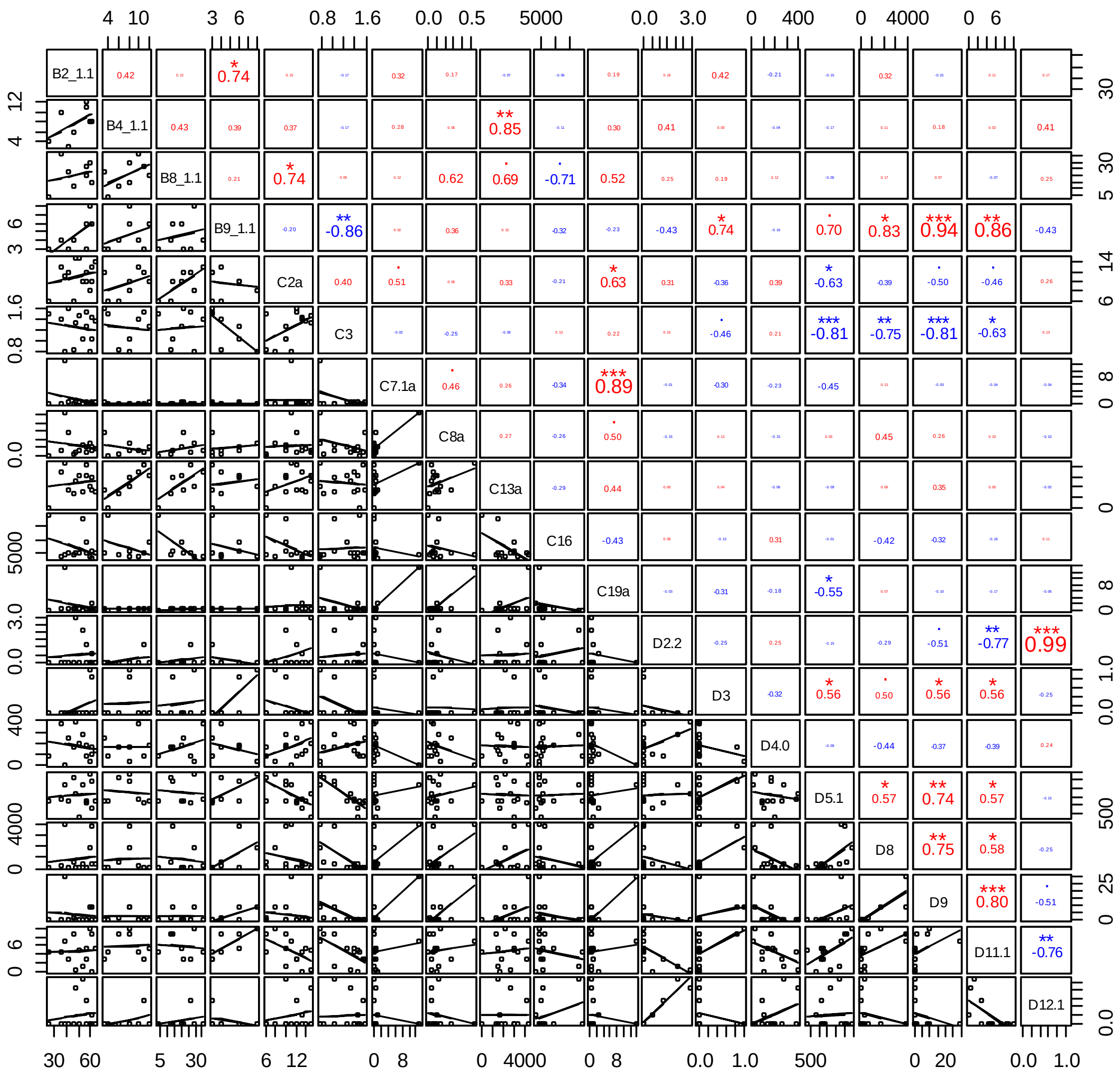


APPENDIX 3 – CORRELATIONS BETWEEN GENETIC, SPECIES, HABITAT AND FARM MANAGEMENT INDICATORS WITHIN CASE STUDY REGIONS

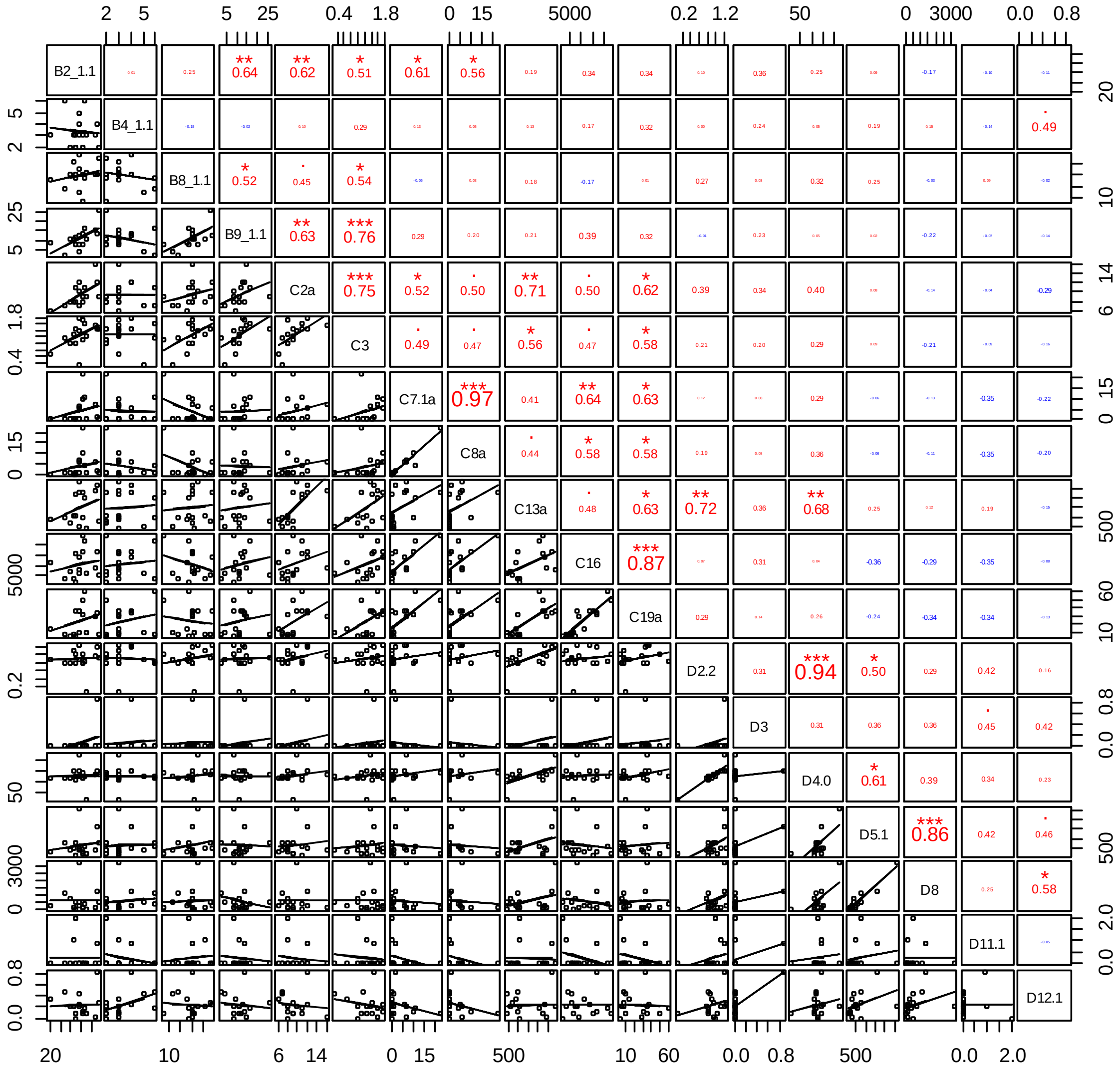
ARA France



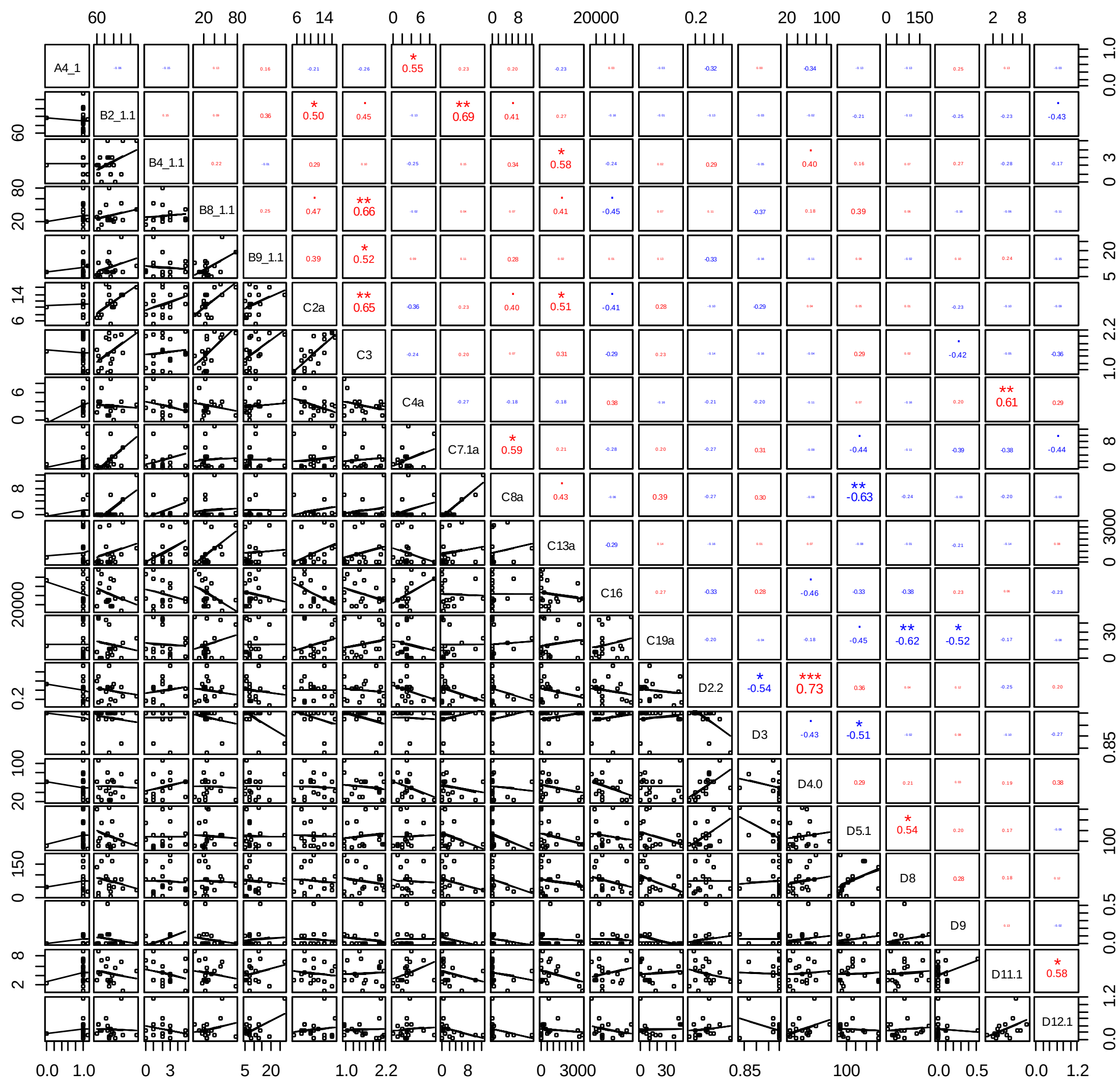
HOR The Netherlands



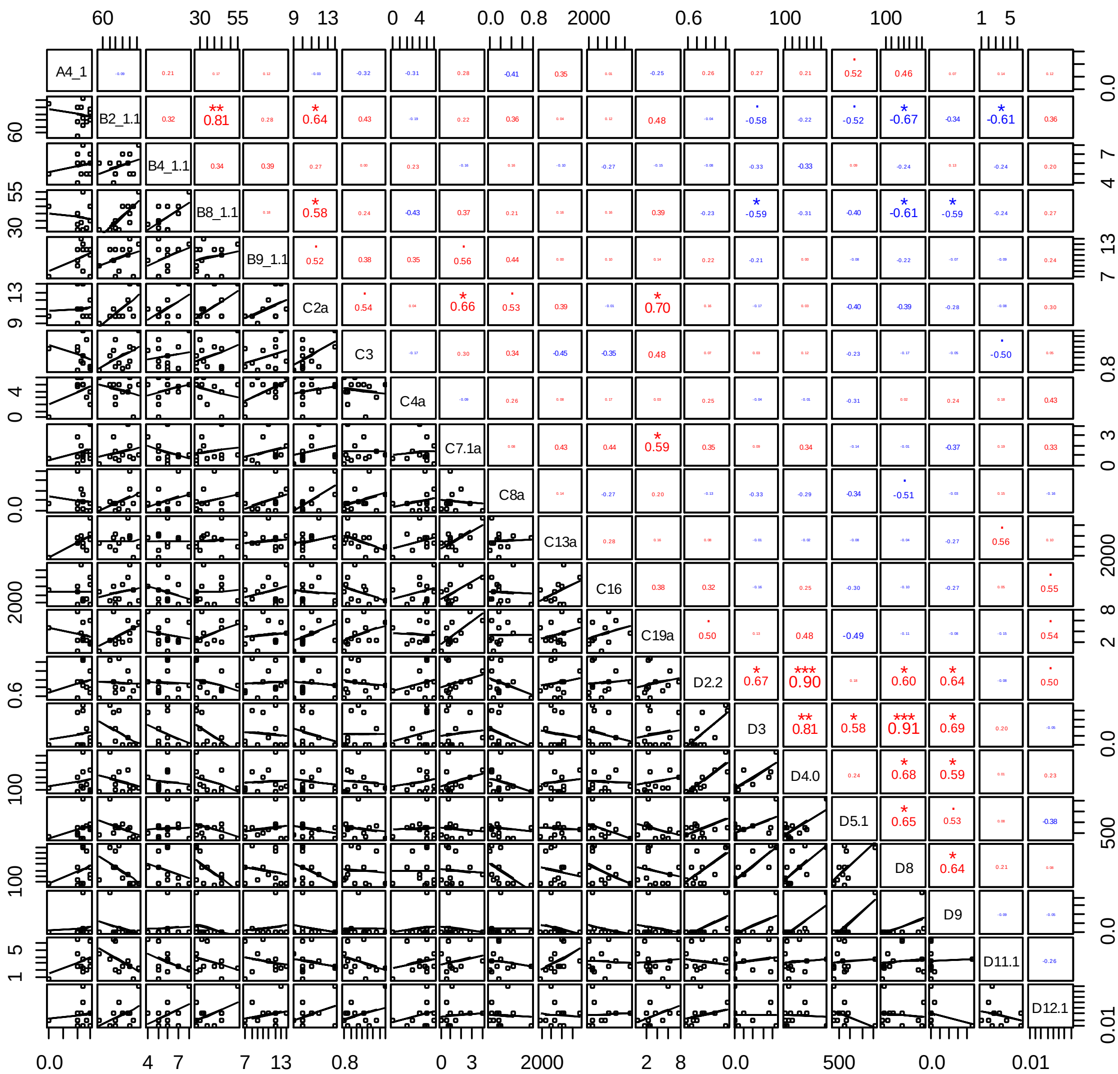
GRA Bulgaria



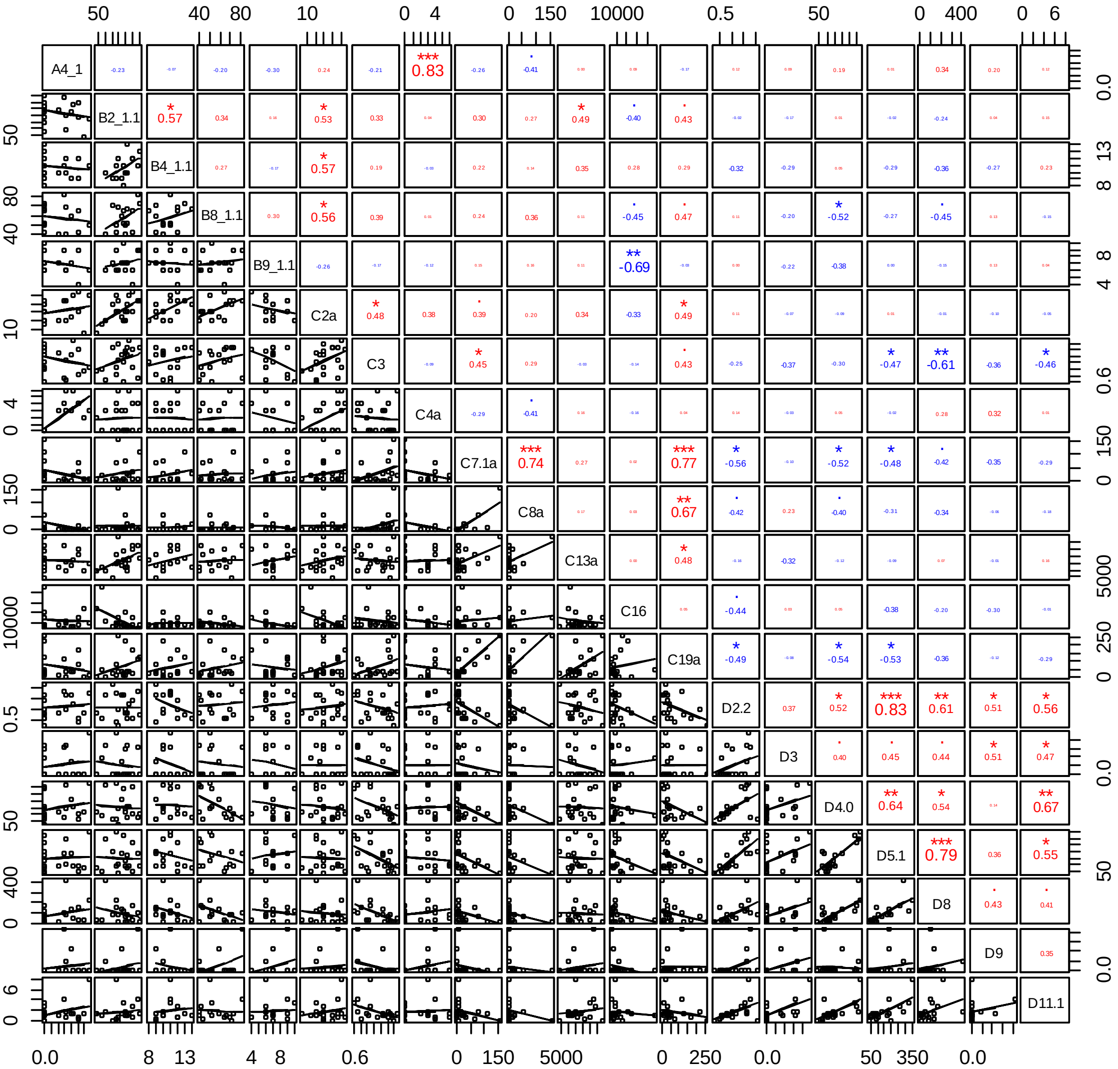
GRA Hungary


A4_1
B2_1.1
B4_1.1
B8_1.1
B9_1.1
C2a
C3
C4a
C7.1a
C8a
C13a
C16
C19a
D2.2
D3
D4.0
D5.1
D8
D9
D11.1
D12.1

GRA Norway

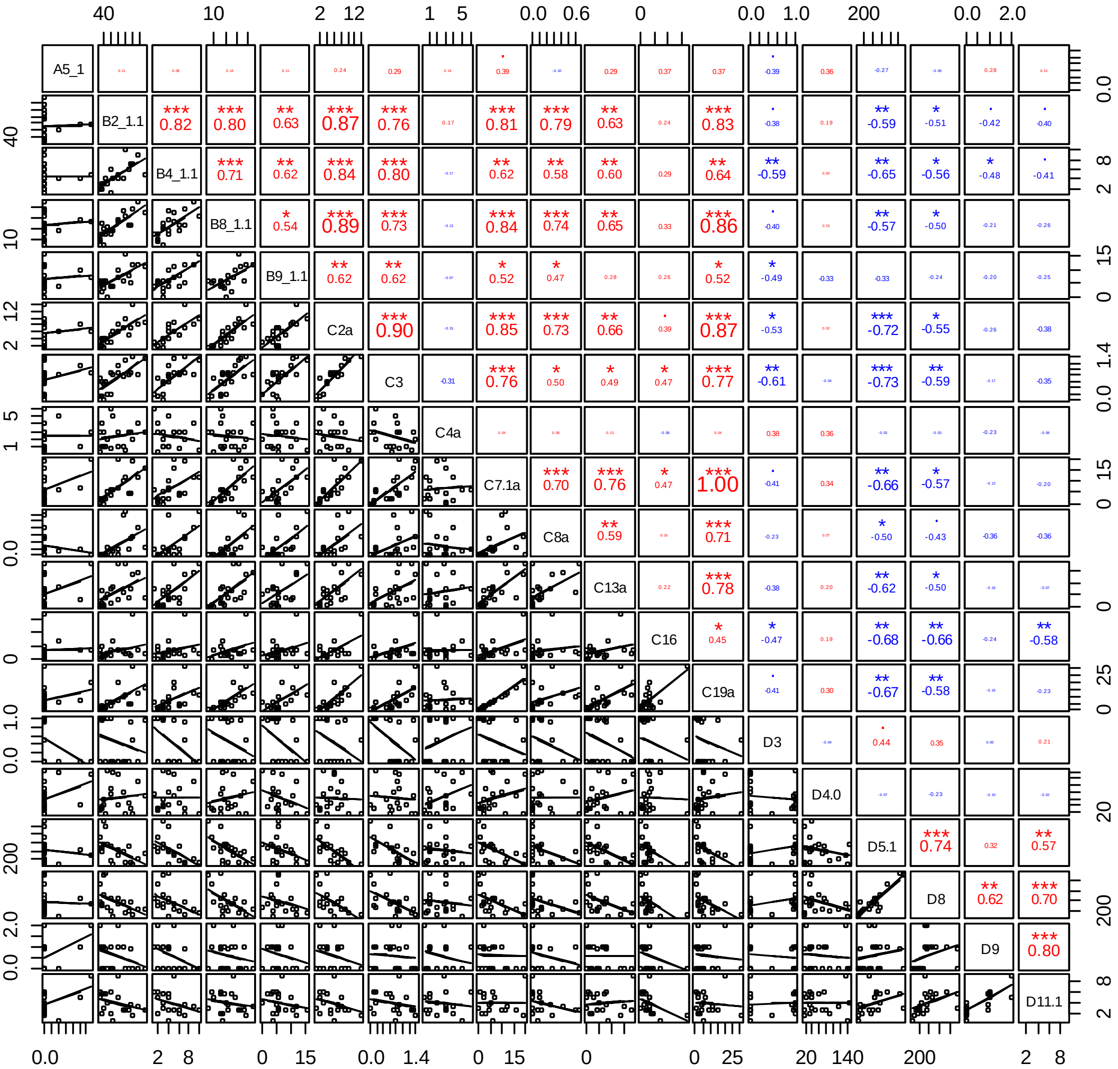

A4_1
B2_1.1
B4_1.1
B8_1.1
B9_1.1
C2a
C3
C4a
C7.1a
C8a
C13a
C16
C19a
D2.2
D3
D4.0
D5.1
D8
D9
D11.1
D12.1

GRA Wales



A4_1
B2_1.1
B4_1.1
B8_1.1
B9_1.1
C2a
C3
C4a
C7.1a
C8a
C13a
C16
C19a
D2.2
D3
D4.0
D5.1
D8
D9
D11.1

OLI Spain


A5_1
B2_1.1
B4_1.1
B8_1.1
B9_1.1
C2a
C3
C4a
C7.1a
C8a
C13a
C16
C19a
D3
D4.0
D5.1
D8
D9
D11.1

BIOBIO categorization of the farm habitats: Detailed classification using the EBONE (http://www.ebone.wur.nl) terminology

Debra Bailey, Felix Herzog (2012)

In the BIOBIO project the habitat mapping method follows the EBONE approach, which has been adapted to farm scale mapping. The method of habitat/land use classification is based on a generic system of habitat definitions, so called General Habitat Categories (GHC). In order to calculate the BIOBIO habitat indicators (see Bailey et al., 2012), the individual habitats were categorized. This section details how the EBONE terminology is applied to derive the categorization of the different farm habitats. For more details on the habitat mapping and EBONE methodology see Dennis et al. (2012) and Bunce et al. (2011). The EBONE handbook is available at www.ebone.wur.nl (Alterra Report 2154; Bunce et al., 2011) and the report from Dennis et al. (2012) is available at www.biobio-indicator.org (Deliverable 2.2).

Defining habitat types in BIOBIO

To identify the different habitat types that were present on the farms we used the EBONE terminology. The habitat types were used to locate sampling plots and to calculate some of the habitat indicators (Bailey et al., 2012). The terminology, General Habitat Category (GHC) as well as global and environmental qualifiers, are described further in Bunce et al. (2011). The subdivision of crops into BIOBIO crop types is explained in Dennis et al. (2012) and Bailey et al. (2012). The following rules are applied to identify different habitat types:

- Count single GHC as individual habitats
- Count GHC combinations as separate habitats, e.g. FPH/DEC/CON and LHE/CHE
- Count GHC habitats which were assigned with the global qualifier “OPE” (trees 1 to 10%) as separate habitats
- Count the grassland GHC’s (CHE, LHE, THE, GEO and combinations) assigned with different environmental qualifiers as individual habitats
- Divide the crop category (CRO) into four BIOBIO crop types: annuals, not entomophilic and/or bee attracting winter crops (CAN1) or spring crops (CAN2); annuals, entomophilic and/or bee attracting crops (CFL); perennials (CFO); see Dennis et al., 2012). Count each BIOBIO crop type as a separate habitats

Habitat Categorization

The individual habitat types identified on the farm are hierarchically structured into habitat categories as shown in Figure 1. Common lands, forest and aquatic habitat that are not used for agricultural purposes and urban habitats are excluded.

At the first hierarchical level the farm area is subdivided into (1) Intensively farmed land, including all crop fields and grasslands that are managed for the main purpose of agricultural production and (2) Semi-natural habitats. Intensively farmed land (1) is then subdivided into (1.1) *Crops and sown & productive permanent grassland* and (1.2) *Intensive agriculture involving trees*. The category (2), *Semi-natural habitats*, comprises all linear and areal habitats managed as farmland where the species composition of the vegetation reflects less intensive farming practices. Semi-natural habitats are further subdivided into (2.1) Semi-natural habitats without trees, (2.2) Semi-natural habitats with trees and (2.3) Aquatic habitats.

Following categorization of the individual habitats using the EBONE terminology, local experts were asked to validate and confirm the classification applied to their case study.

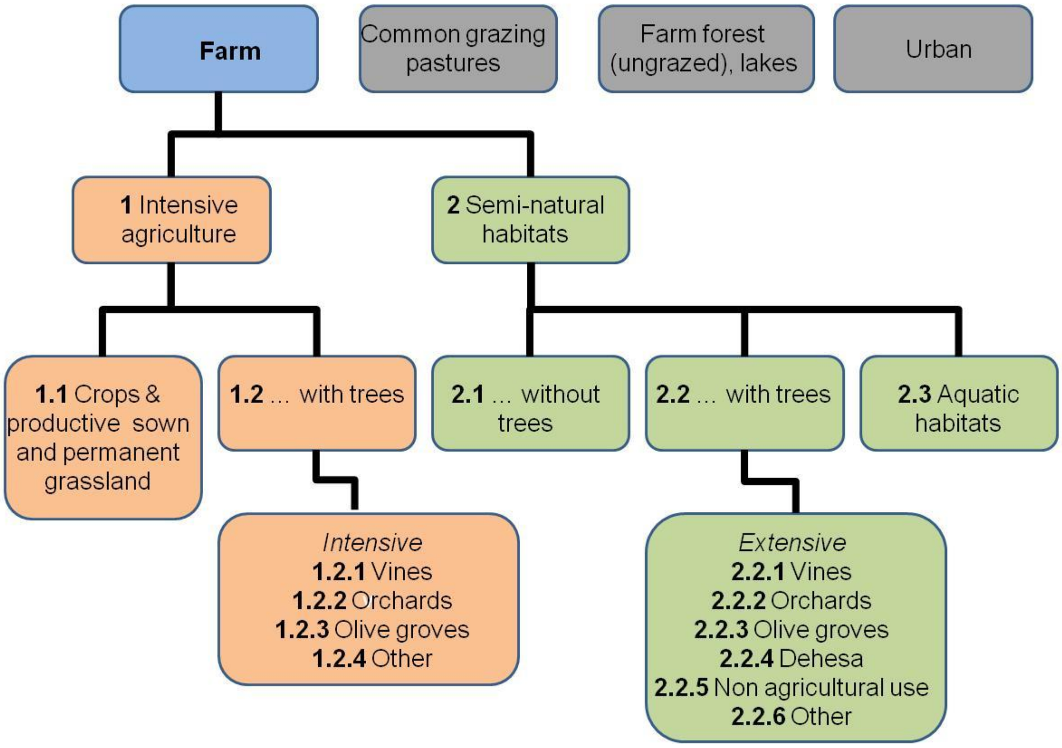


Figure 1: Habitat types of the farm are integrated into categories. The majority of the farmland of most farms consists of the category 1 "Intensive agriculture", interspersed with "Semi-natural habitats" (category 2), mostly linear elements with or without trees or shrubs. In some European regions, large areas of semi-natural grasslands or agroforestry systems dominate the farms.

Categories not considered as farm habitat in BIOBIO and excluded

Common grazing pastures, forests and aquatic habitat not used for agricultural purposes and urban areas are excluded from the farm according to the following EBONE and BIOBIO terminology (Table 1). For further details on the terms General Habitat Categories (GHC) and management qualifiers please refer to Bunce et al. (2011), for Farmland Class see Dennis et al. (2012).

Table 1: Habitats excluded as farm habitat

BIOBIO CATEGORY	Description	EBONE and BIOBIO Terminology
Common grazing pastures	Nature protection areas even if they are managed by the farmer but if they are no longer part of the UAA	Farmland Class: 8 General Habitat Categories (GHC): Leafy Hemicryptophytes (LHE), Caespitose Hemicryptophytes (CHE), Therophytes (THE), Geophytes (GEO), Herbaceous Chamaephytes (HCH) and combinations Management Qualifier: 2 (Semi-natural) !!Boundaries/location needs to be defined through interview with farmer and excluded from habitat mapping!!
	Commonly grazed lands such as summer pastures or outfields (Switzerland, Norway), out bye (Northern England), and ffridd (Wales), that cannot be assigned to a specific farm. The fodder produced (and consumed) on those areas needs, however, to be estimated and included in the calculation of some farm management indicators	Farmland Class: 4 GHC: Leafy Hemicryptophytes (LHE), Caespitose Hemicryptophytes (CHE), Therophytes (THE), Geophytes (GEO), Herbaceous Chamaephytes (HCH)and combinations Management Qualifier: 1 (Agricultural) or 2 (Semi-natural) !!Boundaries/location needs to be defined through interview with farmer and excluded from habitat mapping!!
Farm forest, ungrazed	Large forests (>800 m^2) even if managed by the farmer (because this is a different economic activity) !!To be excluded from farm prior to habitat mapping using aerial photography and topographic maps or subsequent to mapping!!	Farmland Class: 8 GHC: Forest Phanerophytes (>5 m) (FPH), Mega Forest Phanerophytes (>40 m) (GPH) The above GHC codes are used in combination with: Winter Deciduous (DEC), Evergreen (EVR), Coniferous (CON), Non-leafy Evergreen (NLE) Summer Deciduous (SUM), e.g. MPH/DEC, TPH/DEC/CON

		Management Qualifier: 3 (forestry)
Farm forest, ungrazed	Shrubby habitats (>800 m^2)	Farmland Class: 8 GHC: Dwarf Chamaephytes (<0.05 m) (DCH), Shrubby Chamaephytes (0.05-0.30 m) (SCH), Low Phanerophytes (0.30-0.6 m) (LPH), Mid Phanerophytes (0.6-2 m) (MPH), Tall Phanerophytes (2-5 m) (TPH) The above GHC codes are used in combination with: Winter Deciduous (DEC), Evergreen (EVR), Coniferous (CON), Non-leafy Evergreen (NLE), Summer Deciduous (SUM), e.g. MPH/DEC, TPH/DEC/CON Management Qualifier: 3 (forestry)
Aquatic	Aquatic habitats (>800 m^2)	Farmland Class: 0 or 8 GHC: Aquatic (AQU), Submerged Hydrophytes (SHY), Emergent Hydrophytes (EHY), Helophytes (HEL), SHY/EHY, SHY/HEL, EHY/HEL Management Qualifier: 6 (inland water) !!To be excluded from farm prior to habitat mapping using aerial photography and topographic maps or subsequent to mapping!!
Urban	Farmhouses and gardens	Farmland Class: 0 GHC: Artificial (ART), Non-Vegetated (NON), Vegetables (VEG), Herbaceous (GRA), Woody (TRE), Combinations, e.g. ART/NON Management Qualifier: 5 (urban) !!Excluded after habitat mapping!!

Categories included as farm habitat in BioBio

The following tables describe how the EBONE and BIOBIO terminology were applied to categorize the habitats mapped on the farm (see Figure 1). Further details regarding the General Habitat Category, management qualifier and global qualifier can be found in Bunce et al. (2011). Details on the Farmland Class are to be found in Dennis et al. (2012).

Table 2: Habitats included in Intensive Agriculture

	BioBio category	EBONE and BIOBIO Terminology
Categories included in the farm: Intensive Agriculture	**1 Intensive agriculture**	Farmland Class: 1 Management Qualifier: 1 GHC's from the super-category Crops (CUL): Cultivated Bare Ground (SPA), Cultivated Herbaceous Crops (CRO), Woody Crops (WOC), combinations GHC's from the super-category Vegetated Herbaceous (HER) *if classified as Farmland Class 1*: Leafy Hemicryptophytes (LHE), Caespitose Hemicryptophytes (CHE), Therophytes (THE), Geophytes (GEO), Herbaceous Chamaephytes (HCH), combinations
	1.1 Crops & productive sown and permanent grassland	GHC's from the super-category Crops (CUL): Cultivated Bare Ground (SPA), Cultivated Herbaceous Crops (CRO), Combinations GHC's from the super-category Vegetated Herbaceous (HER) *if classified as Farmland Class 1*: Leafy Hemicryptophytes (LHE), Caespitose Hemicryptophytes (CHE), Therophytes (THE), Geophytes (GEO), Herbaceous Chamaephytes (HCH), combinations The GHC's falling into this category that were given the Global Qualifier SCA (scattered trees <1 %)
	1.2 ….with trees	GHC: Woody Crops (WOC) and where the super-category crops (CUL) allows a 'with' and 'without tree' combination, i.e. CRO/WOC and SPA/WOC. In this case the woody crop is given priority. Management Qualifier: A1.13 GHC's from the super-category Vegetated Herbaceous (HER) *if classified as Farmland Class 1and assigned the global qualifier OPE (trees 1- 10 %)*: Leafy Hemicryptophytes (LHE), Caespitose Hemicryptophytes (CHE), Therophytes (THE), Geophytes (GEO), Herbaceous Chamaephytes (HCH), combinations

		1.2.1 Intensive vines	GHC: Woody Crops (WOC) Management Qualifier: A1.13.31 *Level of intensity to be clarified with regional experts*
		1.2.2 Intensive orchards	GHC: Woody Crops (WOC) Management Qualifier: A1.13. (33 through to 44) Modern fruit orchards with dwarf trees, no undercropping, > 100 trees per hectare
		1.2.3 Intensive olive groves	GHC: Woody Crops (WOC) Management Qualifier: A1.13.32 >200 trees per hectare
		1.2.4 Intensive other	GHC's from the super-category Vegetated Herbaceous (HER) *if classified as Farmland Class 1and assigned the global qualifier OPE (trees 1- 10 %)*: Leafy Hemicryptophytes (LHE), Caespitose Hemicryptophytes (CHE), Therophytes (THE), Geophytes (GEO), Herbaceous Chamaephytes (HCH), combinations

Table 3: Habitats included in Semi-natural habitats

	BIOBIO CATEGORY	EBONE and BIOBIO Terminology
Categories included in the farm: Semi-natural habitat	**2 Semi-natural habitats**	Farmland Classes: 2, 3, 4, 5, 6, 7 and 8 but NOT if given management qualifiers 4 (recreational) or 5 (urban). Farmland class 1 **ONLY** if element is categorized as a Woody Crop (GHC: WOC) and **IF** the crop meets the extensive orchard criteria outlined in 2.2.1 through to 2.2.3 (detailed in the sub-categories below) Management Qualifier: 2 (semi-natural) Habitat classified as Annex 1 Possible areal GHC *if fulfil one or more of the above criteria* are: GHC's from the super-category Sparsely Vegetated (SPV): Tidal (TID), Ice and Snow (ICE), Bare Rocks (ROC), Boulders (BOU), Stones (STO), Gravel (GRV), Sand (SAN), Earth (EAR), combinations GHC's from the super-category Vegetated Herbaceous (HER): Submerged Hydrophytes (SHY), Emergent Hydrophytes (EHY), Helophytes (HEL), Leafy Hemicryptophytes (LHE), Caespitose Hemicryptophytes (CHE), Therophytes (THE), Geophytes(GEO), Herbaceous Chamaephytes (HCH), Cryptograms (CRY), combinations GHC's from the super-category Vegetated Tree/Shrub (TRS): Dwarf Chamaephytes (<0.05 m) (DCH), Shrubby Chamaephytes (0.05-0.30 m) (SCH), Low Phanerophytes (0.30-0.6 m) (LPH), Mid Phanerophytes (0.6-2 m) (MPH), Tall Phanerophytes (2-5 m) (TPH), Forest Phanerophytes (>5 m) (FPH), Mega Forest Phanerophytes (>40 m) (GPH), Vegetated Tree/Shrub are used in combination with: Winter Deciduous (DEC), Evergreen (EVR), Coniferous (CON), Non-leafy Evergreen (NLE), Summer Deciduous (SUM), e.g. MPH/DEC, TPH/DEC/CON All linear GHC's: Pond (x), Walls (WAL), Water Edges (WAT), Lines of Scrub (LSC), Hedges (HED), Species Rich Hedges (SRH), Lines of Trees (LTR), Herbaceous Strips (HST), Grass Strips (GST), Private Track Grass Strips (TGS), Private Track Herbaceous Strips (THS) Size limitations for Tree, Shrub and Aquatic habitats: GHC's from the super-category Vegetated Tree/Shrub (TRS): DCH, SCH, LPH, MPH, TPH, FPH, GPH <800 m^2 included as semi-natural habitat DCH, SCH, LPH, MPH, TPH, FPH, GPH >800 m^2 included as semi-natural habitat if used for agricultural purposes, e.g. extensive grazing

			Aquatic (AQU) and Sea (SEA) <800 m^2 if they are used for agricultural purposes Exclude forest plots (FPH, GPH) >800 m^2, shrub plots (DCH, SCH, LPH, MPH, TPH) and Water plots (AQU, SEA) >800 m^2 if they are not used for agricultural purposes (see Table 1)
		2.1 …. without trees	Farmland Classes 2, 3, 4, 5, 6, 7 and 8 but NOT if given management qualifiers 3 (forestry) 4 (recreational) or 5 (urban) Management qualifier 2 (semi-natural management) Annex 1 habitat Possible GHC if *one or more of the above semi-natural habitat criteria are fulfilled* are: GHC's from the super-category Sparsely vegetated (SPV): Ice and Snow (ICE), Bare Rocks (ROC), Boulders (BOU), Stones (STO), Gravel (GRV), Sand (SAN), Earth (EAR), combinations GHC's from the super-category Vegetated herbaceous (HER): Leafy Hemicryptophytes (LHE), Caespitose Hemicryptophytes (CHE), Therophytes (THE), Geophytes(GEO), Herbaceous Chamaephytes (HCH), Cryptograms (CRY), Helophytes (HEL), combinations GHC's from super-categories sparsely vegetated (SPV) and vegetated herbaceous (HER) that were assigned the global qualifier SCA (trees <1 %) Linear habitats: walls (WAL), Herbaceous Strips (HST), Grass Strips (GST), Private Track Grass Strips (TGS), Private Track Herbaceous Strips (THS) Linear habitats Water Edges (WAT) and Pond (x) if not a separate aquatic category in the indicator. "Permanent grassland" is NOT generally considered semi-natural
		2.2 …. with trees	Farmland class 1 **ONLY** if element is categorized as a Woody Crop (GHC: WOC) and **IF** the crop meets the extensive orchard criteria outlined in 2.2.1 through to 2.2.3 (detailed in the sub-categories below) Farmland Classes 2, 3, 4, 5, 6, 7 **ONLY** if assigned to the sparsely vegetated (SPV) and vegetated herbaceous (HER) super-categories **AND** assigned the global qualifier OPE (trees 1-10 %) Farmland Class 8 but **NOT** if given management qualifiers 4 (recreational) or 5 (urban)

			Management qualifier 2 Annex 1 habitat Possible GHC if *one or more of the above semi-natural habitat criteria are fulfilled* are: Dwarf Chamaephytes (<0.05 m) (DCH), Shrubby Chamaephytes (0.05-0.30 m) (SCH), Low Phanerophytes (0.30-0.6 m) (LPH), Mid Phanerophytes (0.6-2 m) (MPH), Tall Phanerophytes (2-5 m) (TPH), Forest Phanerophytes (>5 m) (FPH), Mega Forest Phanerophytes (>40 m) (GPH), Vegetated Tree/Shrub are used in combination with: Winter Deciduous (DEC), Evergreen (EVR), Coniferous (CON), Non-leafy Evergreen (NLE), Summer Deciduous (SUM), e.g. MPH/DEC, TPH/DEC/CON GHC's from super-categories sparsely vegetated (SPV) and vegetated herbaceous (HER) that were assigned the global qualifier OPE (trees 1-10 %) Woody Crops (WOC) **IF** the crops meet the extensive orchard criteria outlined in 2.2.1 through to 2.2.3 (below) Linear habitats: Lines of Scrub (LSC), Hedges (HED), Species Rich Hedges (SRH), Lines of Trees (LTR) Size criteria: Forest plots (DCH, SCH, LPH, MPH, TPH, FPH, GPH) <800 m^2 Forest plots (DCH, SCH, LPH, MPH, TPH, FPH, GPH) >800 m^2 if they are used for agricultural purposes (extensive grazing)
		2.2.1 Extensive vines	Farmland Class 1 GHC: Woody Crops (WOC) Management Qualifier: A1.13.31 *Level of intensity to be clarified with regional experts*
		2.2.2 Extensive orchards	Farmland Class 1 GHC: Woody Crops (WOC) Management Qualifier: A1.13.33 through to 43 *Level of intensity to be clarified with regional experts* *Traditional fruit orchards with standard trees, usually on grazed and/or mown grassland, tree density should be <100 trees per hectare*
		2.2.3 Extensive olive groves	Farmland class 1 GHC: Woody Crops (WOC)

			Management Qualifier: A1.13.32 <200 trees per hectare
		2.2.4 Dehesa	Occurs in Spain and to be identified by local experts.
		2.2.5 Non-agricultural use	Possible GHC if *one or more of the following semi-natural habitat criteria are fulfilled (Farmland Class 8 but NOT if given management qualifiers 4 (recreational) or 5 (urban); management qualifier 2; Annex 1)* are: Dwarf Chamaephytes (<0.05 m) (DCH), Shrubby Chamaephytes (0.05-0.30 m) (SCH), Low Phanerophytes (0.30-0.6 m) (LPH), Mid Phanerophytes (0.6-2 m) (MPH), Tall Phanerophytes (2-5 m) (TPH), Forest Phanerophytes (>5 m) (FPH), Mega Forest Phanerophytes (>40 m) (GPH), Vegetated Tree/Shrub are used in combination with: Winter Deciduous (DEC), Evergreen (EVR), Coniferous (CON), Non-leafy Evergreen (NLE), Summer Deciduous (SUM), e.g. MPH/DEC, TPH/DEC/CON Linear habitats: Lines of Scrub (LSC), Hedges (HED), Species Rich Hedges (SRH), Lines of Trees (LTR) Forest plots (DCH, SCH, LPH, MPH, TPH, FPH, GPH) <800 m^2
		2.2.6 Exten-sive other	GHC's from super-categories sparsely vegetated (SPV) and vegetated herbaceous (HER) that were assigned the global qualifier OPE (trees 1-10 %) Forest plots (DCH, SCH, LPH, MPH, TPH, FPH, GPH) >800 m^2 if they are used for extensive agricultural purposes
		2.3 Aquatic	Aquatic (AQU) and Sea (SEA) <800 m^2 if they are used for agricultural purposes Aquatic (AQU) and Sea (SEA) >800 m2 if they are used for agricultural purposes GHC's from the super-category Sparsely vegetated (SPV): Tidal (TID) GHC's from the super-category Vegetated herbaceous (HER): Submerged Hydrophytes (SHY), Emergent Hydrophytes (EHY) Linear habitats: Pond (x) and Water Edges (WAT)

Feedback to categorization methodology

Particularly assigning the label 'semi-natural' habitat is not a straightforward process. For example in gradient landscapes it is not always clear where to place the boundary between an intensive agricultural and semi-natural habitat. Furthermore, habitats categorized as agricultural at the European level may at the national or regional scale be considered semi-natural (see Bailey et al., 2012). Such considerations should be accounted for when implementing a monitoring programme.

Generally, when mapping the habitats in the field, it is essential for the researcher to fully understand and implement the EBONE terminology. This requires adequate training of all the persons involved in the mapping and preferably training sessions on-site in the locations to be mapped. Particular attention should be paid to understanding not only the General Habitat Categories but also to assigning the environmental, global, site and management qualifiers. This is because correct assignment of these GHC's and qualifiers are essential for the identification and categorization afterwards of semi-natural habitats. The management qualifiers are very useful for the identification of semi-natural habitats and elements that are considered to fall under this category should be appropriately indicated with the code (2) (semi-natural) by the person doing the mapping. It is further suggest by BIOBIO that the management qualifiers in the EBONE handbook be extended to allow for a more accurate management description of the semi-natural habitat. A further possibility would be to extend and use the site qualifiers for this purpose. Site qualifiers were not used in the categorization described above due to frequent misunderstanding by the people in the field. This has now been improved in the current version of the EBONE handbook.

In BIOBIO global, site, environmental and management qualifiers were only recorded for areal elements and a more restricted method was applied for the linear elements. However, if further information in terms of linear elements is required, this can be gained by implementing the extended linear mapping method as outlined in the EBONE handbook.

References:

Bailey, D., Herzog, F., Bogers, M., Lüscher, G., Fjellstad, W. 2012. Habitat Indicators. In: Herzog, F., Balázs, K., Dennis, P., Friedel, J.K., Jeanneret, P., Geijzendorffer, I., Kainz, M., Pointereau, P., Biodiversity Indicators for European Farming Systems: A Guidebook, ART Publication Series 17, August 2012.

Bunce, R.G.H., Bogers, M.M.B., Roche, P., Walczak, M., Geijzendorffer, I.R., Jongman, R.H.G. 2011Manual for Habitat and Vegetation Surveillance and Monitoring: Temperate, Mediterranean and Desert Biomes. First edition. Wageningen, Alterra report 2154 . 106 pp.; 15 fig.; 14 tab.; 35 ref

P. Dennis, M.M.B. Bogers, R.G.H. Bunce, F. Herzog and P. Jeanneret, 2012. Biodiversity in organic and low-input farming systems. Handbook for recording key indicators. Wageningen, Alterra, Alterra-Report 2308. 92 pp.; 17 fig.; 24 tab.; 30 ref.

APPENDIX 5 – NAMES AND ADDRESSES OF TAXONOMISTS FOR SPECIES IDENTIFICATION

Case study	Plants	Earthworms	Spiders	Bees
ARA-Austria	**M.-L. Oschatz** BOKU Institut für Botanik Gregor-Mendel-Straße 33 A-1180 Wien	**S. Papaja-Hülsbergen** Technische Universität München, Arcisstr. 21 D-80333 München	**X. Heer** Gerbegasse 15 CH-5036 Oberentfelden	**J. Neumayer** Obergrubstrasse 18 A-5161 Elixhausen
ARA France	**L. Mouney** 40, Rue des Saules Résidence La Charmeraie Bât. F, Apt. 138 F-31 400 Toulouse **A. Simmons** 33 rue Pasteur, F-91 120 Palaiseau	**C. Pelosi** INRA-PESSAC Bâtiment 6 RD 10 F-78026 Versailles Cedex	**S. Déjean** Conservatoire Régional des Espaces Naturels de Midi-Pyrénées 75 voie du TOEC F-31076 Toulouse Cedex 03, BP 57611	**D. Genoud** DGE (Diagnostic, Gestion, Expertise) 10, rue du Président Fallières F-11000 Carcassonne
HOR The Netherlands	**B. Bunce** Alterra, Wageningen UR PO Box 47 6700 AA Wageningen the Netherlands	**A. van der Hout** Alterra, Wageningen UR PO Box 47 6700 AA Wageningen the Netherlands	**Christoph Muster** Neukamp 29 D-18581 Putbus, Germany	**R. van Kats** Alterra, Wageningen UR PO Box 47 6700 AA Wageningen the Netherlands
GRA Bulgaria	**J. Guteva** Institute of Plant Genetic Resources, 2 Druzhba str.., B-4122 Sadovo	**A. Vassilev** Sofia University, Faculty of Biology 8 Dragan Tsankov Blvd., B-1164 Sofia,	**Ch. Delchev** Institute of Biodiversity and Ecosystem research at the Bulgarian Academy of Science 2 Gagarin Street, B-1113 Sofia	**T. Ljubomirov** Institute of Biodiversity and Ecosystem research at the Bulgarian Academy of Science 2 Gagarin Street, B-1113 Sofia
GRA Switzerland	**N. Richner & S. Buholzer**, Agroscope Reckenholz-Tänikon ART Reckenholzstr. 191 CH-8046 Zurich	**G. Cuendet**, ZOOCONTROL Ch. de la Croix 26, CH-1675 Vauderens	**X. Heer** Gerbegasse 15 CH-5036 Oberentfelden	**G. Lüscher** Agroscope Reckenholz-Tänikon ART Reckenholzstr. 191 CH-8046 Zurich
GRA Hungary	**E. Falusi-Saláta** Károly Penksza, Szent Istvan University, Pater K. u. 1, H-2100 Godolloe	**T. Szederjesi** Eotvos Lorand University, Pázmány Péter sétány 1/A., H-1117 Budapest	**O. Szalkovszki** University of Debrecen, Egyetem tér 1. H-4032 Debrecen	**A. Kovács** Ecosystem Services Research Group, Institute of Ecology and Botany, HAS, Alkotmány út 2-4, H-2163 Vácrátót
GRA Norway	**G. Engan** Norwegian Forest and Landscape Institute, P.O. Box 115, NO-1431 Ås Olav Balle Nylundsgata 12, NO-4014 Stavanger	**R. Pommeresche** Bioforsk Organic Food and Farming Division, NO-6630 Tingvoll	**O. Finch** Tierökologische Fachbeiträge, Achtern Nordpol 8 D - 26180 Rastede, Germany	**F. Ødegaard** Norwegian Institute for Nature Research, P.O. Box 5685 Sluppen, NO-7485 Trondheim A. Mjelde Randsfjordvegen 1526, NO-2866 Enger
GRA Wales	**J. Vale, L. Lemaire & J. Warren**: Institute of Biological, Environmental and Rural Sciences, Aberystwyth University, SY23 3FG UK **D. Schwenk**: Independent ecology consultant	**G. Jerkovich & J. Scullion**: Institute of Biological, Environmental and Rural Sciences, Aberystwyth University, SY23 3FG UK	**A. Whittington**: FlyEvidence, UK **G. Jerkovich**: Institute of Biological, Environmental and Rural Sciences, Aberystwyth University, SY23 3FG UK	**P. Dennis & P. Gillingham**: Institute of Biological, Environmental and Rural Sciences, Aberystwyth University, SY23 3FG UK **M. Pavett**: Entomology Section, National Museum Cardiff, Cathays Park,

				Cardiff, CF10 3NP, UK
DEH Spain	**J. Baonza Diaz** Botanist Ctra. Valdemanco, 28. Bustarviejo E-28729 Madrid	**D. Díaz Cosín** Departamento de Zoología, Facultad de Biología, Universidad Complutense de Madrid, Calle José Antonio Novais 2, E-28040 Madrid	**J. Benhadi Marín** Departamento de Biodiversidad y Gestión Ambiental, Área de Zoología, Universidad de León, E-24071 León	**G. González Bornay** Grupo de Investigación Forestal, Universidad de Extremadura, E-Plasencia 10600 **F. Javier Ortiz Sánchez** Grupo de Investigación "Transferencia de I+D en el Área de Recursos Naturales", Universidad de Almería, E-04120 La Cañada, Almería
MIX Germany	**H. Albrecht, M. Ruff & J. Brenner** Technische Universität München, Arcisstr. 21 D-80333 München	**S. Papaja-Hülsbergen** Technische Universität München, Arcisstr. 21 D-80333 München	**T. Blick** Heidloh 8 D-95503 Hummeltal	**K. Mandery** Hermann-Löns-Str. 16 D-96106 Ebern
VIN Italy	**J.Nascimbene & R. Trevisan** Dept. of Biology, Padova University	**M.G. Paoletti & T. Zanetti** Dept of Biology, Padova University	**I. Paschetta** Dept of Biology Torino University	**D. Sommaggio** Dept of Biology, Padova University
OLI Spain	**J. Baonza Diaz** Botanist Ctra. Valdemanco, 28. Bustarviejo E-28729 Madrid	**D. Díaz Cosín** Departamento de Zoología, Facultad de Biología, Universidad Complutense de Madrid, Calle José Antonio Novais 2, E-28040 Madrid	**J. Benhadi Marín** Departamento de Biodiversidad y Gestión Ambiental, Área de Zoología, Universidad de León, E-24071 León	**G. González Bornay** Grupo de Investigación Forestal, Universidad de Extremadura, E-Plasencia 10600 **F. Javier Ortiz Sánchez** Grupo de Investigación "Transferencia de I+D en el Área de Recursos Naturales", Universidad de Almería, E-04120 La Cañada, Almería

Titel: Cost-efficiency of measuring earthworm diversity in a German case study at farm-scale: lessons for application in monitoring and agricultural practice

Authors: Sebastian Wolfrum[1], Stefano Targetti[2], Davide Viaggi[2], Norman Siebrecht[1], Susanne Papaja-Hülsbergen[1], Gisela Lüscher[3], Maximilian Kainz[1], Philippe Jeanneret[3], Kurt-Jürgen Hülsbergen[1], Felix Herzog[3]

[1]Chair for Organic Agriculture and Agronomy, Technical University of Munich, Freising, D-85350, Germany

[2]Dept. of Agricultural Economics and Engineering. University of Bologna, Bologna, I-40127, Italy

[3]Agroscope ART, Zurich, CH-8046, Switzerland

Agricultural practices and farmland biodiversity are strongly related. Agriculture influences biodiversity through management activities whilst at the same time benefiting from services provided by biodiversity components. Knowledge about the level of biodiversity present on farmland and the relations to management is therefore crucial regarding both conservation and service provision. Measuring biodiversity is, however, a complex and laborious issue. One solution for this problem is the use of indicators. Still, biodiversity indicators for the level of the farm, which is the actual instance where most biodiversity relevant decisions are made, are lacking. The EU project BioBio (www.biobio-indicator.org) aims at closing this gap. In addition to scientific considerations, the cost-effectiveness of indicators is assessed. This is in particular important for indicator groups like earthworms, which can provide useful information on soils to farmers but which are also quite costly.

Given the fact that resources for monitoring are limited, the availability of cost data concerning the measurement of biodiversity indicators is of significant importance. Only few studies exist which provide empirical evidence about the cost of biodiversity indicator measurement.

Here we present data from the German case study, which focused on mixed dairy farms. The 16 selected farms are located in South Bavaria. In total 127 plots were sampled. Earthworms were assessed with a combination of an expellant solution and hand sorting procedure according to the BioBio standards (Dennis et al. 2010). All adult specimens found were identified to species level.

We used EstimateS software to calculate for each farm 100 resamplings (without replacement) of observed species numbers for each level of sampling effort. To estimate the expected total earthworm species number at the farm level we used the Chao 1 estimator. We calculate measures of bias, precision and accuracy according to Walther und Moore (2005) for the resampling results. The quantity and change in information gained from different sampling levels was calculated as the relation of species found to species expected on the farm and from accuracy measures.

The costs of the measurement were measured as the sum of monetary costs of resources consumed to undertake the measurement of the indicator and processing of data. These costs were estimated through direct information collection regarding resource use and unitary costs. The main cost categories were: i) labour; ii) equipment; iii) travel; iv) consumables; v) taxonomic identification. Each category considers specific resources and unitary costs and data collection was organised in order to trace the costs related to each day of the survey and to each single farm. Cost data collection was organised on a weekly basis during the field sampling activities.

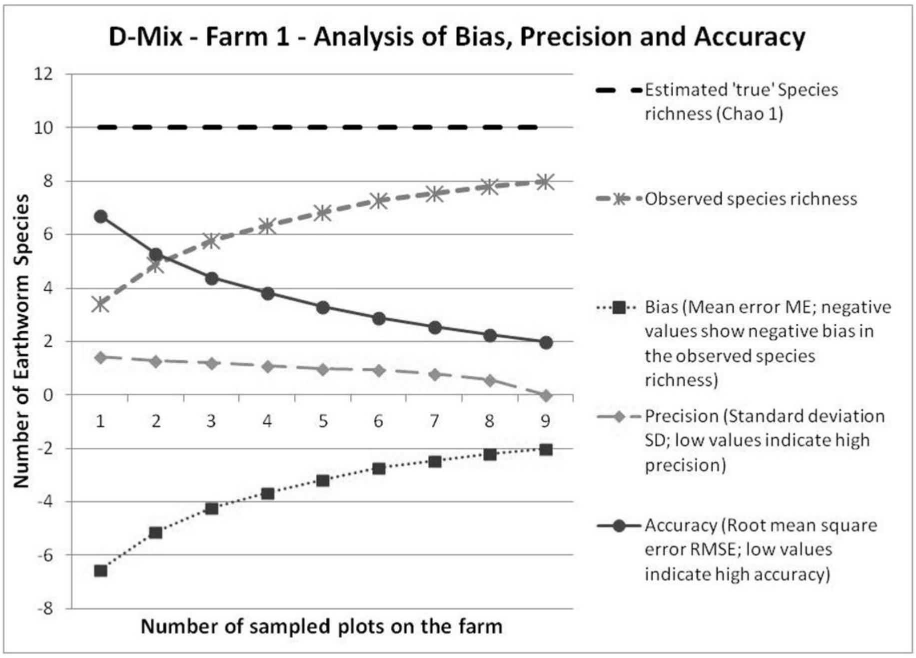
Results show that the observed species richness is always negatively biased when compared to the "true" species richness estimated by Chao 1. With increasing number of samples bias is reduced and accuracy of the measurement is increased. Compared to the latter, precision seems to increases only slowly with growing numbers of samples per farm. With one sample per farm about 35 % of information on species richness could be retrieved. By sampling three or five plots per farm, the information gained would raise to about 60 % or 70 % respectively. With three samples the most efficient allocation of money could be reached. Five samples could on the one hand improve bias, precision and accuracy without a big loss in efficiency and on the other hand would provide a reasonable number of samples for sample based rarefaction to compare species richness between farms.

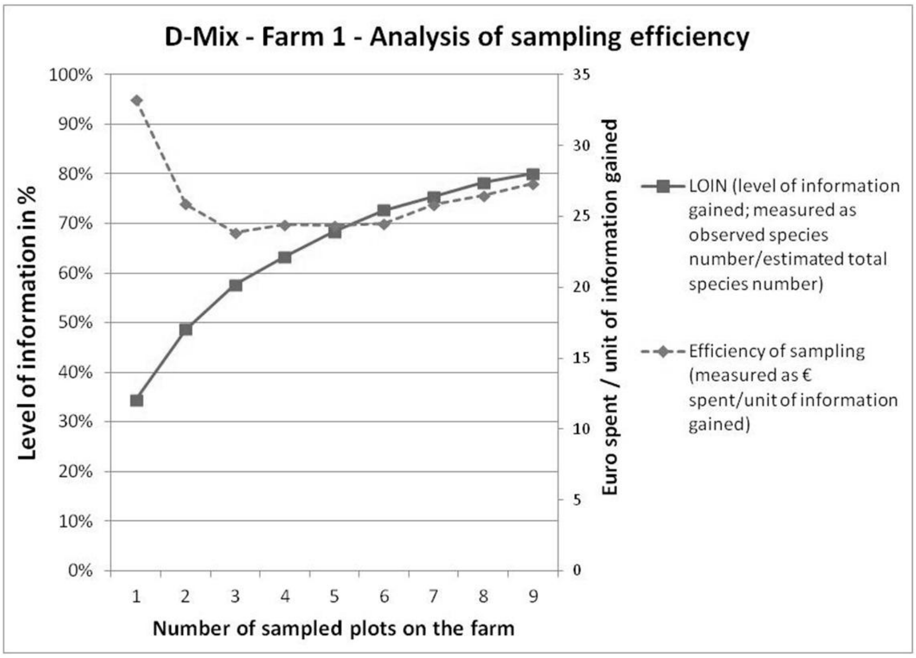
In this work we combine costs and information analysis in order to identify optimal levels of sampling effort, which is defined as the optimal level of € per percent of information. Costs are assessed through an empirical-based data collection and the percent of information is assessed through a nonparametric species richness estimation method. Our results aim at the identification of the optimal ratio between monetary investment and information gained and on identifying necessary funds to reach a certain level of accuracy when using indicators to assess biodiversity in practice. The analysis, which is applied to earthworms in this case, can be used for other species, too. However, for other species the optimal number of samples in terms of accuracy or cost-efficiency may be different. The result may also depend on the estimator used for defining the "true" species richness as a reference for the analysis of bias and accuracy and on the quality of single measurements. Therefore, in a second step the sensitivity of the obtained results to these points should be assessed.

Acknowledgement: Part of this research was funded under the European FP7 project BioBio.

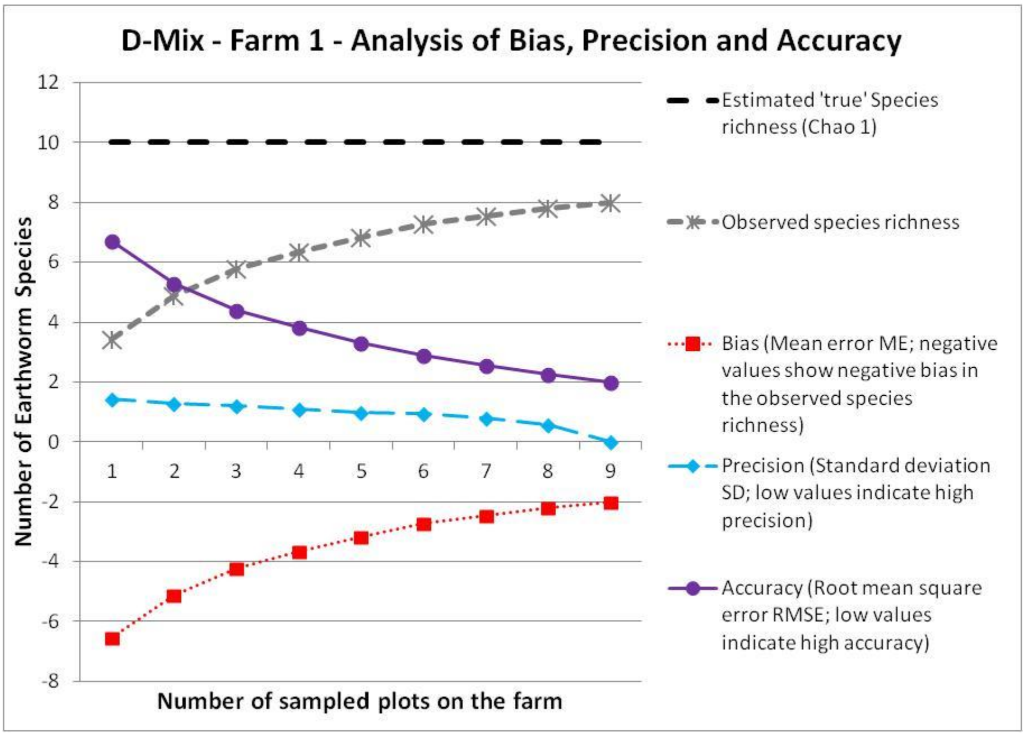
Dennis, P. et al. (2010): BIOBIO: Indicators for biodiversity in organic and low-input farming systems. D2.2. Selection and field validation of candidate biodiversity indicators, including field manual. Aberystwyth.

Walther, B. A.; Moore, J. L. (2005): The concepts of bias, precision and accuracy, and their use in testing the performance of species richness estimators, with a literature review of estimator performance. *Ecography* 28 (6), 815–829.

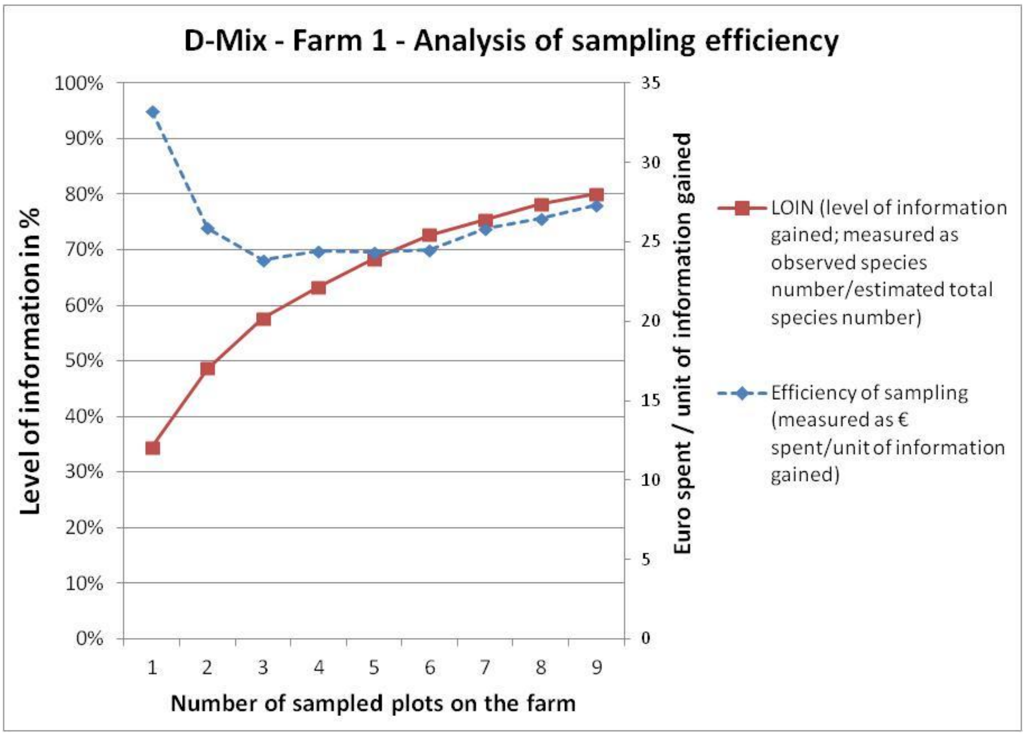




D-Mix - Farm 1 - Analysis of sampling efficiency
Level of information in %
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
Euro spent / unit of information gained
35
30
25
20
15
10
5
0
1
2
3
4
5
6
7
8
9
Number of sampled plots on the farm
LOIN (level of information gained; measured as observed species number/estimated total species number)
Efficiency of sampling (measured as € spent/unit of information gained)



D-Mix - Farm 1 - Analysis of Bias, Precision and Accuracy
Number of Earthworm Species
12
10
8
6
4
2
0
-2
-4
-6
-8
1
2
3
4
5
6
7
8
9
Number of sampled plots on the farm
Estimated 'true' Species richness (Chao 1)
Observed species richness
Bias (Mean error ME; negative values show negative bias in the observed species richness)
Precision (Standard deviation SD; low values indicate high precision)
Accuracy (Root mean square error RMSE; low values indicate high accuracy)



D-Mix - Farm 1 - Analysis of sampling efficiency
Level of information in %
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
35
30
25
20
15
10
5
0
Euro spent / unit of information gained
1
2
3
4
5
6
7
8
9
Number of sampled plots on the farm
LOIN (level of information gained; measured as observed species number/estimated total species number)
Efficiency of sampling (measured as € spent/unit of information gained)

Plant, earthworm, spider and bee diversity in agricultural fields of grazing and field crop farming systems in eight regions across Europe

Lüscher, Gisela[1]; Arndorfer, M.[2]; Balazs, K.[3]; Bernhardt, K. G.[2]; Bogers, M.[4]; Bunce, R. G. H.[4]; Choisis, J.-P.[5]; Dennis, P.[6]; Dramstad, W.[7]; Eiter, S.[7]; Engan, G.[7]; Fjellstad, W.[7]; Frank, T.[2]; Friedel, J. K.[2]; Geijzendorffer, I.[4]; Gillingham, P.[6]; Herzog, F.[8]; Hülsbergen, K.-J.[9]; Jeanneret, P.[8]; Jerkovich, G.[6]; Jongman, R.[4]; Kainz, M.[9]; Oschatz, M. L.[2]; Papaja-Hülsbergen, S.[9]; Pointereau, P.[10]; Sarthou, J.-P.[5]; Schneider, M. K.[8]; Siebrecht, N.[9]; Wolfrum, S.[9]

[1]ART Agroscope Reckenholz-Taenikon, SWITZERLAND;
[2]BOKU, Vienna, A-1180, AUSTRIA;
[3]Szent Istvan University, Godollo, HU-2100, HUNGARY;
[4]ALTERRA, Wageningen, NL-6700AA, NETHERLANDS;
[5]INRA/ENSAT, Castanet Tolosan, F-31326, FRANCE;
[6]Aberystwyth University, SY23 5AL, UNITED KINGDOM;
[7]NFLI, Aas, N-1431, NORWAY;
[8]Agroscope ART, Zurich, CH-8046, SWITZERLAND;
[9]Munich Technical University, Freising, D-85350, GERMANY;
[10]SOLAGRO, Toulouse, F-31076, FRANCE

Diversity of wild species living in agricultural fields is influenced by management practices and landscape characteristics. Factors acting on species diversity have contrasting effects on different species groups due to various dispersal abilities and resource requirements (Clough, Holzschuh et al. 2007).

The dataset of the EU-FP7 project BioBio was used to evaluate main drivers for plant, earthworm, spider and bee diversity in agricultural fields. In BioBio indicators for biodiversity in farmland were developed. The four species groups were selected as biodiversity indicators at the species level. Each species group fulfills distinct functions in the agricultural ecosystems. Plants act as primary producers and provide the food resource for all herbivores. Earthworms belong to the group of soil detritivores. Spiders are predators and have a potential role in the control of agricultural pests. Bees perform pollination (Kremen, Williams et al. 2007). In this study, data from eight case studies are investigated: specialist livestock grazing in Hungary, Norway, Switzerland and Wales, field crop and horticulture farming systems in Austria, France and the Netherlands and a mixed farming system in Germany. All four species groups were surveyed in a total of 385 agricultural fields. Based on questionnaires, management information was provided by the farmers. Hence, the pesticide use, the nitrogen input and the number of mechanical operations were recorded for each agricultural field. Additionally, field characteristics were assessed. Furthermore, the landscape composition in a buffer of 250 m was estimated for each field from aerial photographs. These explanatory variables will be included in models to explain the species assemblages of plants, earthworms, spiders and bees on agricultural fields. Species assemblages are applied as response variable since it takes into account both species richness and species abundance (Fig. 1). If just species richness is considered as diversity measurement, the contribution of frequent and rare species to the diversity is counted equally. However, focusing on species assemblages also takes account of the distribution of the species and enables us to detect more detailed patterns.

Results based on analysis of the diverse farming systems and regions will reveal whether nitrogen input, herbicide use and the number of mechanical operations act on plant diversity as expected. While management variables of fields are assumed to be the main drivers for earthworm diversity, landscape features may play an important additional role for spiders, which are known to use perennial vegetation outside the field for overwintering (Schmidt and Tscharntke 2005). Similarly, we will test whether bee diversity is more related to the landscape composition in the surroundings of fields or to small scale field characteristics and management practices (e.g. insecticide use).

The findings of the study will show the main drivers for plant, earthworm, spider and bee species assemblages in agricultural fields with respect to various farming systems. Such detailed investigations of driving factors for biodiversity in farming landscapes are necessary to implement effective measures in agro-environmental schemes.

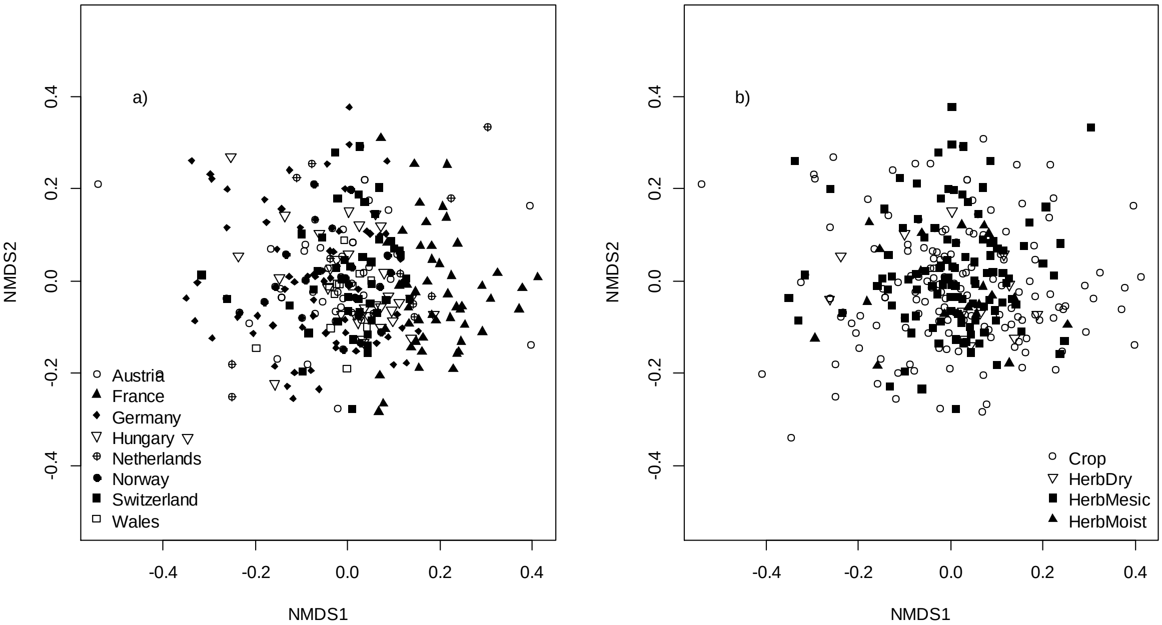


Fig. 1: One example of species assemblages. Non-metric multidimensional scaling of earthworm species, grouped by case studies (a) and main habitat types (b).

Clough, Y., A. Holzschuh, et al. (2007). "Alpha and beta diversity of arthropods and plants in organically and conventionally managed wheat fields." Journal of Applied Ecology 44(4): 804-812.

Kremen, C., N. M. Williams, et al. (2007). "Pollination and other ecosystem services produced by mobile organisms: a conceptual framework for the effects of land-use change." Ecology Letters 10(4): 299-314.

Schmidt, M. H. and T. Tscharntke (2005). "The Role of Perennial Habitats for Central European Farmland Spiders." Agriculture Ecosystems & Environment 105(1-2): 235-242.

Indicateurs de biodiversité dans les exploitations agricoles biologiques et conventionnelles du cas d'étude français « Vallées et Coteaux de Gascogne » du projet européen BioBio.

Stagiaire :

Alexandre Amossé

M2I Modélisation des Systèmes Ecologiques 2011-2012

Université Paul Sabatier Toulouse III

Maîtres de stage :

Jean-Pierre Sarthou

Maître de Conférences à l'INP/ENSAT

Université de Toulouse

UMR 1248 AGIR INRA-INP/ENSAT/EIP

Jean-Philippe Choisis

Ingénieur de recherche INRA

UMR 1201 DYNAFOR INRA-INP/ENSAT/EIP

Remerciements

Je tiens à remercier Jean-Pierre Sarthou et Jean-Philippe Choisis, pour leur encadrement pédagogique, la confiance qu'il m'ont accordée en me laissant mener l'analyse des résultats du projet européen BioBio, et surtout pour la bonne humeur quotidienne qu'ils ont apportés à mon stage.

Je remercie également Michel Goulard, pour ses conseils de statisticien, ainsi que Sylvie Ladet, pour son aide dans l'utilisation de Systèmes d'Information Géographique.

Je remercie aussi toute l'équipe de l'UMR 1201 DYNAFOR, pour leur intérêt à mon égard ayant contribué à mon excellente intégration dans cette unité.

Enfin, je remercie mes enseignants du Master MSE, qui m'ont fournis les outils nécessaires à l'analyse de données en écologie, ligne directrice de mon stage et, je l'espère, de mon avenir professionnel.

Liste des abréviations

ACP : analyse en composantes principales

BIOMAX : indicateur de biodiversité maximale

BIOMOY : indicateur de biodiversité moyenne

BROUS : taillis et broussailles

CE : cultures entomophiles

CNE : cultures non entomophiles

DEN : densité

EA : exploitation agricole

EBONE : European Biodiversity Observation Network

FORET : couverts ligneux de plus de 5m de hauteur

GLM : Modèle Linéaire Généralisé

H : indice de diversité de Shannon

HAIE : linéaires d'arbres et de buissons de moins de 5m de largeur

HERB : bandes herbeuses de moins de 5m de largeur

HCU : habitats cultivés

HSN : habitats semi-naturels

Ibio : indice de biodiversité d'un plot

Ja : indice de diversité de Piélou alpha

Jg : indice de diversité de Piélou gamma

LOO-CV : Leave-One-Out Cross-Validation

PNAT : prairies naturelles

PTEM : prairies temporaires

RSa : richesse spécifique alpha

RSb : richesse spécifique beta

RSg : richesse spécifique gamma

Sommaire

1 Introduction 1

1.1 Les indicateurs de biodiversité : rôle et importance 1

1.2 Le projet européen BioBio 2

1.2.1 Présentation du projet 2

1.2.2 Protocole de recueil des données 3

1.3 Les objectifs du stage 4

2 Matériel et Méthodes 4

2.1 Mise au point d'un indicateur à partir des variables « espèces » 4

2.1.1 Mise en forme des variables « espèces » 4

2.1.2 Etude de corrélation 5

2.1.3 Comparaison entre EA biologiques et conventionnelles 6

2.1.4 Elaboration de modèles prédictifs supervisés 7

2.1.5 Création d'un indicateur de biodiversité à l'échelle de l'EA 7

2.2 Etude des relations entre les variables « espèces » et les variables « gestion » et « habitat » 9

2.2.1 Mise en forme des variables « gestion » et « habitat » 9

2.2.2 Modélisation de la biodiversité dans les agroécosystèmes 10

2.2.3 Modélisation de la diversité des quatre taxons étudiés 11

3 Résultats 11

3.1 Mise au point d'un indicateur à partir des variables « espèces » 11

3.1.1 Etude de corrélation 11

3.1.2 Comparaison entre EA biologiques et conventionnelles 13

3.1.3 Elaboration de modèles prédictifs supervisés 16

3.1.4 Création d'un indicateur de biodiversité à l'échelle de l'EA 16

3.2 Etude des relations entre les variables « espèces » et les variables « gestion » et « habitat » 20

3.2.1 Modélisation de la biodiversité dans les agroécosystèmes 20

3.2.2 Modélisation de la diversité des quatre taxons étudiés 21

4 Discussion 23
4.1 Qualité des résultats obtenus 23
4.2 Amélioration de l'indicateur de biodiversité proposé 25
4.3 Réduction des coûts de la méthode 26
4.4 Conclusion 27

Bibliographie 27

Annexes 30

1 Introduction

1.1 Les indicateurs de biodiversité : rôle et importance

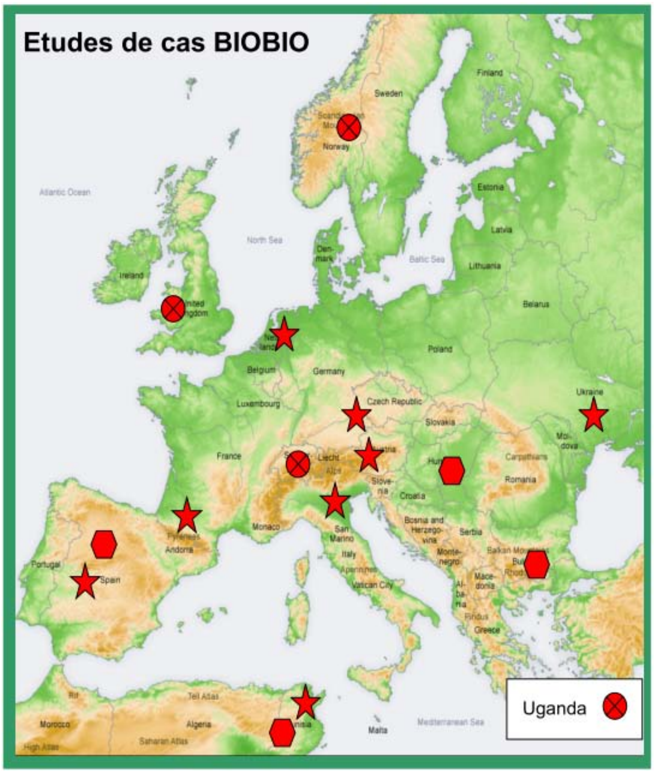
La surface terrestre est une immense mosaïque d'écosystèmes et d'habitats dont la structure, la composition et le fonctionnement subissent continuellement des changements sous l'influence de facteurs biotiques et abiotiques, naturels et anthropiques. L'intervention humaine, aussi courte soit-elle à l'échelle des temps géologiques, est une cause majeure de ces changements (Turner *et al.,* 2007). Les agroécosystèmes constituent actuellement l'un des principaux paysages (*i.e.* assemblages d'écosystèmes) structurant l'environnement terrestre puisqu'ils occupent 40% de la surface des terres émergées (Turner *et al.*, 2007). Mais ces changements d'utilisation du sol par l'Homme, notamment le développement des activités agricoles et urbaines au détriment des forêts, ont altéré les écosystèmes au niveau de leur fonctionnement et des services écosystémiques qu'ils procurent, du fait d'une érosion de la biodiversité (Foley *et al.,* 2005). La composante fonctionnelle de cette biodiversité est elle aussi affectée (Geiger *et al.,* 2010). Ainsi, une diminution d'intensité de certains services écosystémiques, dont la régulation, a été observée (Winqvist *et al.,* 2011). La Convention sur la Diversité Biologique de 2003 avait pour objectif l'arrêt de l'érosion de la biodiversité pour 2010 (UNEP/CBD/COP7, 2003), mais les indicateurs utilisés ne permettant pas de déterminer les causes des tendances d'évolution de la biodiversité, les objectifs de cette convention n'ont pas été atteints. Il est donc indispensable de développer des outils que sont les indicateurs de biodiversité et de services écosystémiques afin d'une part d'évaluer et suivre les tendances d'évolution de la biodiversité dans les agroécosystèmes, et d'autre part d'en déterminer les causes.

Selon Bockstaller (2008), un indicateur est "*une grandeur qui fournit une information au sujet d'une variable plus difficile d'accès ou d'un système plus complexe, afin d'aider un utilisateur dans son action (prise de décision, construction d'un programme d'action, modélisation, etc.).* Pour aider à la prise de décisions en matière de gestion corrective, Dale et Bayeler (2001) expliquent qu'un indicateur doit d'une part détecter les changements et d'autre part en déterminer les causes. Cet indicateur doit également respecter d'autres critères : être simple d'utilisation, sensible aux stress environnementaux, répondre de manière précoce, prédictible et constante, et être sensible à une large gamme de facteurs. D'après Duelli et Obrist (2003), un indicateur peut être composé de plusieurs sous-indicateurs évaluant des processus indépendants et donc, non corrélés entre eux. De plus, il doit permettre d'évaluer trois « valeurs écologiques », aussi bien dans les systèmes naturels que cultivés : la conservation, la résilience et la capacité de contrôle biologique. Les deux définitions de ces auteurs (Dale et Bayeler, 2001 ; Duelli et Obrist, 2003) sur les indicateurs et leurs critères d'utilisation sont complémentaires. D'autres critères peuvent être trouvés dans la littérature (*e.g.* Feld *et al.*, 2010), mais nous nous limiterons à ces définitions qui sont en phase avec les attentes du projet européen BioBio. Il apparait ainsi que globalement, la priorité pour l'avenir est de développer et d'appliquer des indicateurs directs de biodiversité et de services écosystémiques, qui soient applicables à plusieurs échelles d'espace ou qui soient composés de sous-ensembles d'indicateurs utilisables à plusieurs échelles et interconnectables (Feld *et al.*, 2009 ; Feld *et al.*, 2010 ; Perrings *et al.*, 2011).

1.2 Le projet européen BioBio

1.2.1 Présentation du projet

Le projet européen BioBio, lancé en 2009, a pour objectif d'identifier des indicateurs de biodiversité dans les systèmes d'agriculture biologique et à faible niveau d'intrants. Les indicateurs sélectionnés sont en cours d'évaluation à travers seize études de cas, dont douze en Europe, deux en Tunisie, une en Ukraine et une en Ouganda (Fig. 1). Quatre groupes d'indicateurs sont étudiés : des indicateurs génétiques, des indicateurs d'espèces, des indicateurs d'habitat et des indicateurs de gestion. Des indicateurs sont sélectionnés au sein de chacun de ces groupes et doivent respecter trois critères majeurs : ils doivent être scientifiquement fiables, valables à travers l'Europe et utiles et pertinents pour les différents acteurs concernés (professionnels agricoles, associations de protection de la nature, cadres administratifs…). Pour le groupe « indicateurs d'espèces », plantes vasculaires, abeilles, araignées et vers de terre ont ainsi été présélectionnés en 2009 en tant qu'indicateurs directs de biodiversité, puis échantillonnés en 2010 et sont actuellement en phase d'évaluation, notamment en regard des indicateurs indirects relatifs à la gestion des exploitations agricoles (EA).



Agriculture biologique : elle repose sur la rotation des cultures, les engrais verts, la lutte biologique, le compostage et un important travail du sol, dans le but d'augmenter la fertilité du sol sans utiliser d'engrais de synthèse et de protéger les cultures sans recourir aux pesticides.
Agriculture à faible niveau d'intrants : elle repose sur les mêmes principes agronomiques que l'agriculture biologique à ceci près qu'elle ne s'interdit pas un usage très modéré d'engrais de synthèse et de pesticides, elle est souvent pratiquée dans des régions au relief accidenté comme en zone de montagnes, où l'intensification est impossible.
Agriculture biologique et agriculture à faible niveau d'intrants vont souvent de pair.

Figure 1 : Répartition géographique des études de cas du projet BioBio.

Ce projet réunit 14 pays, dont la France représentée par l'Institut National de Recherche Agronomique (INRA) et SOLAGRO, une entreprise associative qui cherche de nouvelles voies pour l'énergie, l'agriculture et la gestion des ressources naturelles à long terme. Le cas d'étude français, qui représente le cadre de mon stage, rassemble seize exploitations agricoles : huit exploitations biologiques et huit exploitations conventionnelles, qui siègent toutes dans une structure paysagère originale : les « Vallées et Coteaux de Gascogne ».

1.2.2 Protocole de recueil des données

Un protocole de récolte des données a été élaboré pour l'ensemble des cas d'études. Il consiste en la réalisation de trois tâches :

(i) La soumission d'un questionnaire aux agriculteurs. Cela afin de recueillir des données relatives à la gestion de l'EA et aux variétés cultivées.

(ii) L'expertise de la diversité d'habitats par EA. Tous les habitats linéaires (entre 0,5m et 5m de largeur, et plus de 30m de long) et tous les habitats surfaciques (plus de 5m de largeur et plus de 400m² de surface) de chaque EA sont décrits selon la méthode BioHab, méthode inspirée de la méthode EBONE (http://www.ebone.wur.nl/UK).

(iii) La collecte des données relatives aux quatre taxons présélectionnés en tant qu'indicateurs « espèces ». Une parcelle ou ligne continue de chaque type d'habitat recensé par EA est sélectionnée et est alors définie comme un étant un « plot ». Chacun des 270 plots ainsi recensés dans les « Vallées et Coteaux de Gascogne », ont été échantillonnés selon une méthode propre à chaque taxon.

- Plantes vasculaires : communautés décrites à travers le pourcentage de recouvrement de chaque espèce identifiée dans une zone de 10m sur 10m pour un plot surfacique ou dans une ligne de 10m de long sur 1m de large maximum pour un plot linéaire. Un seul relevé par plot est réalisé.
- Abeilles : communautés décrites par chasse à vue au filet à insecte pendant 15 minutes, trois passages par plot sont réalisés.
- Araignées : communautés décrites par capture à l'aveugle par aspiration sur une zone de 0,1m². Trois passages de 5 relevés par plot sont réalisés.
- Vers de terre : communautés décrites par capture suite à l'épandage d'une solution irritante sur une zone de 30cm sur 30cm. Un passage de 3 relevés par plot est réalisé.

Pour ne pas biaiser les résultats, l'espèce d'abeille domestique *Apis mellifera* n'a pas été prise en compte dans les analyses.

1.3 Les objectifs du stage

L'objectif principal est de mettre au point un indicateur de biodiversité par habitat, qui sera constitué de quatre sous-indicateurs correspondant à un indicateur pour chacun des quatre taxons échantillonnés. Ces indicateurs de biodiversité des différents habitats seront alors interconnectables au sein d'une EA et seront utilisables à la fois à l'échelle de l'habitat et à l'échelle de l'EA dans les « Vallées et Coteaux de Gascogne ». Ces indicateurs de biodiversité devront respecter les critères du projet BioBio : fiabilité scientifique et utilité pour les acteurs concernés. Le choix des sous-indicateurs devra se faire suivant deux directives du projet BioBio : (i) un sous-indicateur doit permettre la meilleure distinction possible entre exploitations biologiques et conventionnelles, et (ii) il ne doit pas être redondant avec les autres sous-indicateurs.

Le coût de mise en œuvre de ces indicateurs entre ensuite en compte : il doit être le plus faible possible, aussi bien au niveau temporel que financier. Or comme certaines études l'expliquent (Qi *et al.*, 2008 ; Targetti *et al.*, 2011), les indicateurs indirects coûtent moins cher que les indicateurs directs, mais en contrepartie ils sont également moins précis et donc moins fiables. Le second objectif sera donc de remplacer les indicateurs directs d'espèces sélectionnés pour notre indicateur de biodiversité par des indicateurs indirects que sont les indicateurs de gestion, et par des indicateurs directs moins onéreux : les indicateurs d'habitat. Au final, notre indicateur de biodiversité sera constitué par les sous-indicateurs offrant le meilleur compromis entre pertinence des résultats et coûts de mise en œuvre.

2 Matériel et Méthodes

2.1 Mise au point d'un indicateur à partir des variables « espèces »

2.1.1 Mise en forme des variables « espèces »

Pour chacun des taxons animaux (abeilles, araignées et vers de terre), les espèces et le nombre d'individus par espèce par plot ont été recensés. A partir de ces données, un maximum d'indices ont été calculés : les richesses spécifiques alpha, gamma et beta (respectivement RSa, RSg et RSb), la densité (DEN), l'indice de diversité de Shannon (H), les indices de diversité de Piélou alpha et gamma (respectivement Ja et Jg). Pour le taxon plantes, les espèces et leur pourcentage de recouvrement par plot ont été recensés, ainsi seules les RSa, RSg et RSb ont été calculées.

- <u>Calcul des richesses spécifiques (RSa, RSg, RSb).</u>

La richesse spécifique alpha correspond au nombre d'espèces recensées par plot, la richesse spécifique gamma correspond au nombre d'espèces recensées par EA et la richesse spécifique beta, calculée par plot, est la différence du nombre d'espèces entre le plot et l'exploitation :

$$RSb = RSg - RSa$$

- <u>Calcul de la densité (DEN).</u>

Réalisé pour chacun des 270 plots, il dépend de la méthode d'échantillonnage qui est différente pour chacun des taxons.

Pour les abeilles, l'efficacité de capture de la chasse au filet étant faible, le nombre d'individus capturés pour le total des 3 passages a donc été utilisé. Après digitalisation des plots sous ArcGis, la surface (en m²) de chaque plot a été calculée avec l'outil « Calculer la géométrie… » de la table attributaire.

$$DEN_{abeilles}\,(nb.m^{-2}) = \frac{\sum(Nombre\ d'individus\ des\ 3\ passages)}{Surface\ (en\ m^2)}$$

Pour les araignées, le nombre moyen d'individus capturés par passage a été ramené au m².

$$DEN_{araignées}\,(nb.m^{-2}) = \frac{2 * \sum(Nombre\ d'individus\ des\ 3\ passages)}{3}$$

Pour les vers de terre, le total des individus capturés a été ramené au m².

$$DEN_{vers}\,(nb.m^{-2}) = \frac{\sum(Nombre\ d'individus\ des\ 3\ relevés)}{3 * 0{,}3 * 0{,}3}$$

- <u>Calcul de l'indice de diversité de Shannon (H).</u>

$$H = -\sum pi * \frac{\log(pi)}{\log(2)} \qquad avec\ \ pi = \frac{Ni}{N}$$

Ni = Nombre d'individus dans un plot d'une espèce donnée i

N = Nombre total d'individus dans un plot

- <u>Calcul de l'indice de diversité de Piélou (J).</u>

$$J = \frac{H}{\log(S)} * \log(2)$$

S = RSa pour J alpha et S = RSg pour J gamma

2.1.2 Etude de corrélation

Afin d'identifier les variables les plus corrélées entre elles, des corrélogrammes ont été réalisés sous R à l'aide du package « corrgram ». Dans ces corrélogrammes apparaissent les coefficients de corrélation rhô de Spearman, avec leur significativité déterminée par un test de conformité entre rhô et la valeur nulle. Pour déterminer les variables les plus représentatives par taxon et qui diffèrent le plus par rapport aux autres taxons, un score de représentativité (Scor) a été calculé pour chaque variable. Ce score est basé sur la différence entre la somme des coefficients de corrélation entre une variable donnée et les autres variables du même taxon, et la somme des coefficients de corrélation entre la variable donnée et les variables de même indice des autres taxons.

- <u>Calcul du score de représentativité :</u>

$$Scor_{spe1.var1} = \sum_{X=2}^{n} rhô(spe1.var1, spe1.varX) - \sum_{Y=2}^{n} rhô(spe1.var1, speY.var1)$$

$$avec\ speY.varX = variable\ X\ du\ taxon\ Y$$

2.1.3 Comparaison entre EA biologiques et conventionnelles

Les quarante habitats décrits suivant la méthode BioHab, n'ayant pour certains pas assez de plots pour réaliser des analyses statistiques pertinentes, ont été regroupés en huit groupes d'habitats :

- HERB : bandes herbeuses de moins de 5m de largeur.
- HAIE : linéaires d'arbres et de buissons de moins de 5m de largeur
- BROUS : taillis et broussailles
- FORET : couverts ligneux de plus de 5m de hauteur
- CNE : cultures non entomophiles
- CE : cultures entomophiles
- PTEM : prairies temporaires
- PNAT : prairies naturelles

Pour chacun des huit groupes d'habitats et chacune des 4 variables « espèces » les plus représentatives selon la méthode décrite en 2.1.2, les hypothèses suivantes ont été testées sous le logiciel R :

- H0 : Il n'y a pas de différence entre les plots biologiques et conventionnels.
- H1 : Une différence existe entre les plots biologiques et conventionnels.

Les tests de comparaison réalisés sont des tests non paramétriques de Mann-Whitney car sur les échantillons non appariés testés deux à deux, au moins un contenait moins de 30 valeurs. Ces hypothèses ont également été testées pour chacune des 4 variables « espèces » sur l'ensemble des plots sans distinction d'habitat par des tests paramétriques de Welch.

Pour identifier un gradient de biodiversité global, des analyses en composantes principales (ACP) à partir des 4 variables « espèces », pour chaque habitat, ont été réalisées sous R avec la fonction dudi.pca() du package « ade4 ». Pour s'assurer que la répartition des plots biologiques et conventionnels sur l'ACP n'est pas due au hasard, des tests de Monte-Carlo avec 10 000 répétitions ont été réalisés. Sur l'axe de chaque ACP distinguant le plus pertinemment possible EA biologiques et conventionnelles selon un gradient de biodiversité globale, un test de comparaison de Mann-Whitney a été effectué selon les mêmes hypothèses nulle et alternative que précédemment. Lorsque ni l'axe 1 ni l'axe 2 de l'ACP ne se sont révélés pertinents pour visualiser cette distinction entre types d'exploitation, un nouvel axe (axeN) a été créé. La projection orthogonale des points de coordonnées (axe1, axe2) sur cet axeN a été calculée ainsi :

$$axeN = axe1 * \cos\theta + axe2 * \sin\theta \qquad \text{avec } \theta = \frac{3\pi}{4} \text{ pour BROUS et CE.}$$

2.1.4 Elaboration de modèles prédictifs supervisés

Pour chaque groupe d'habitats dont l'ACP permet de distinguer EA biologiques et conventionnelles selon leur biodiversité globale, un modèle de prédiction a été élaboré. Basé sur l'ACP, le modèle consiste en la prédiction d'une valeur de biodiversité (Ibio) comprise entre 0 et 5 d'un plot à partir des 4 variables « espèces ». Cette prédiction est réalisée sous R grâce à la fonction suprow() du package « ade4 » qui projète de nouveaux points suivant le modèle d'une ACP déjà existante, sous condition de renseigner pour chaque nouveau point les mêmes variables que celles ayant permis de créer l'ACP sur laquelle ils sont projetés. Les valeurs de l'Ibio ont été calculées en fonction du sens du gradient de biodiversité le long de l'axe de l'ACP comme défini dans le tableau 1. Chaque modèle a été évalué en calculant son pourcentage d'erreur de prédiction après Leave-One-Out Cross-Validation (LOO-CV). Cette méthode de LOO-CV consiste à retirer un à un chaque plot et ses données, à reconstruire une ACP à partir des données des autres plots, puis à prédire la position sur l'ACP du plot retiré.

Tableau 1 : Modalités de calcul de l'indice de biodiversité (Ibio)

gradient de biodiversité	cas 1	cas 2	Ibio
+	MAX ≤ x	x ≤ MIN	5
	MAX/2 ≤ x <MAX	MIN < x ≤ MIN/2	4
	0 ≤ x < MAX/2	MIN/2 < x ≤ 0	3
	MIN/2 ≤ x < 0	0 < x ≤ MAX/2	2
	MIN ≤ x < MIN/2	MAX/2 < x ≤ MAX	1
-	x < MIN	MAX < x	0

cas 1 : les valeurs x de l'axe sont proportionnelles au gradient de biodiversité

cas 2 : les valeurs x de l'axe sont inversement proportionnelles au gradient de biodiversité

2.1.5 Création d'un indicateur de biodiversité à l'échelle de l'EA

Pour construire un indicateur de biodiversité à l'échelle de l'EA, il est nécessaire de prendre en compte tous les habitats d'une EA. Pour les groupes d'habitats pour lesquels un modèle prédictif a été créé et validé en 2.1.4, nous disposons des Ibios, mais pas pour les groupes sans modèle prédictif. On a donc calculé un Ibio, d'une valeur comprise entre 0 et 5, pour ces groupes, à partir de la somme des richesses spécifiques alpha des quatre taxons :

$$Ibio = \frac{totRSa * 5}{MAX(totRSa)} \qquad avec\ totRSa = abeRSa + aracRSa + verRSa + plaRSa$$

A partir des Ibios, deux indicateurs à l'échelle de l'EA ont pu être calculés :

(i) L'indicateur de biodiversité maximale : BIOMAX, qui correspond à l'Ibio le plus élevé des plots d'un même groupe d'habitats d'une EA.

(ii) L'indicateur de biodiversité moyenne : BIOMOY, qui correspond à la moyenne des Ibios des plots d'un même groupe d'habitats d'une EA.

Ces indicateurs de biodiversité des huit groupes d'habitats, dont la valeur est comprise entre 0 et 5, sont interconnectables. Ainsi, en additionnant les BIOMAX ou les BIOMOY d'une EA, on obtient un indicateur de biodiversité à l'échelle de l'EA qui a une valeur comprise entre 0 et 40. L'impact anthropique étant différent suivant les habitats, deux sous-ensembles d'habitats se distinguent au sein d'une EA : les habitats semi-naturels (HSN) que sont les groupes HERB, HAIE, BROUS et FORET ; et les habitats cultivés (HCU) que sont les groupes CNE, CE, PTEM et PNAT. Un indicateur de biodiversité, d'une valeur comprise entre 0 et 20, a été calculé pour les HSN et les HCU des seize EA en additionnant leurs BIOMAX ou BIOMOY respectifs.

Nous avons ensuite voulu vérifier que nos indicateurs permettent de mieux rendre compte d'une différence d'impact sur la biodiversité entre gestions biologique et conventionnelle que les données observées globalement (RSg). Pour cela, ont été élaborées des régressions linéaires de la richesse spécifique gamma, de l'indicateur BIOMAX et de l'indicateur BIOMOY en fonction du nombre d'habitats Biohab identifiés par EA. Ces régressions ont été réalisées à l'aide de la fonction lm() sous le logiciel R, pour les EA biologiques et les EA conventionnelles. La richesse spécifique gamma correspond à la somme des richesses spécifiques gamma des quatre taxons et représente donc la diversité d'espèces observées dans une EA.

Pour mieux comprendre les causes de cette différence, des tests de comparaison non paramétriques ont été réalisés entre les indicateurs des huit EA biologiques et des huit EA conventionnelles, pour les indicateurs BIOMAX et BIOMOY des HSN, des HCU et de l'EA dans son ensemble. Des tests de Wilcoxon ont été réalisés entre les indicateurs des HSN et des HCU des EA biologiques ou conventionnelles, car ils reposent sur des échantillons appariés. Pour toutes les autres comparaisons, réalisées sur des échantillons non appariés, des tests de Mann-Whitney ont été appliqués.

2.2 Etude des relations entre les variables « espèces » et les variables « gestion » et « habitat »

Dans les agroécosystèmes, deux groupes de variables environnementales influencent les populations : les variables relatives à la gestion des exploitations agricoles par les agriculteurs que l'on nomme variables « gestion » ; et les variables relatives à l'environnement naturel nommées variables « habitat ».

2.2.1 Mise en forme des variables « gestion » et « habitat »

A partir des réponses des agriculteurs aux questionnaires, dix-sept variables « gestion » utilisables à l'échelle du plot ont pu être extraites (Tab.2). Les cinq variables « habitat » décrites dans le tableau 2 ont quant à elles été obtenues à partir des relevés effectués pendant la collecte des données relatives aux quatre taxons (abeilles, araignées, vers de terre et plantes), notamment les variables Flower, Tree, PH et HUM. La variable Area a été calculée avec le logiciel ArcGis, comme décrit en 2.1.1, § « calcul de la densité ».

Tableau 2 : Recensement des variables « gestion » et « habitat »

	Nom	Description	Données
Variables gestion	FoSeed	Nombre d'opérations de préparation du lit de semence sur l'année	valeurs discrètes
	undersow	Utilisation d'un semis sous couvert	0 = non 1 = oui
	FoMeWeed	Nombre d'opérations de désherbage mécanique sur l'année	valeurs discrètes
	FoFertil	Nombre d'épandage d'engrais et/ou de fumier sur l'année	valeurs discrètes
	herbi	Nombre de pulvérisations d'herbicides sur l'année	valeurs discrètes
	insecti	Nombre de pulvérisations d'insecticides sur l'année	valeurs discrètes
	fungi	Nombre de pulvérisations de fongicides sur l'année	valeurs discrètes
	mollu	Nombre de pulvérisations de molluscicides sur l'année	valeurs discrètes
	FoPest	Nombre de pulvérisations de pesticides sur l'année = herbi + insecti + fungi + mollu	valeurs discrètes
	FoHarvest	Nombre d'opérations de fauche et de récolte sur l'année	valeurs discrètes
	FoTot	Nombre total d'opérations sur l'année = FoSeed + FoMeWeed + FoFertil + FoPest + FoHarvest	valeurs discrètes
	Cut	Nombre de fauches sur l'année	valeurs discrètes
	residues	Gestion de la végétation après fauche	0 = non fauchée 1 = fauchée et laissée au sol 2 = fauchée et retirée
	winter	Etat du couvert végétal durant l'hiver	0 = sol nu 1 = résidus de culture 2 = culture d'hiver
	age	Age de la prairie depuis le dernier semis	valeurs discrètes
	NoCut	Nombre d'années depuis la dernière taille	valeurs discrètes
	Man	Gestion globale du plot	0 = aucune gestion 1 = abandonné/négligé 2 = gestion active
Variables habitat	Area	Surface du plot en m²	valeurs continues
	Flower	Pourcentage de fleurs sur la parcelle	pourcentages
	Tree	Présence/absence d'arbres sur le plot	0 = absence 1 = présence
	PH	Acidité du sol	2 = acide 3 = neutre 4 = basique
	HUM	Humidité du sol	4 = humide 5 = semi-humide 6 = sec

2.2.2 Modélisation de la biodiversité dans les agroécosystèmes

Pour pouvoir substituer les indicateurs espèces par des indicateurs de gestion et d'habitat, il est nécessaire de déterminer quelles variables « gestion » et « habitat » sont les plus explicatives de la biodiversité. La réponse des Ibios aux variables décrites dans le tableau 2 a donc été modélisée pour chaque habitat comme spécifié dans le tableau 3. Les Ibios des groupes d'habitats BROUS, CNE, CE et PNAT étant des valeurs discrètes, ils ont été remplacés par les valeurs des axes des ACP, représentant les gradients de biodiversité, pour gagner en précision. Les modèles linéaires ont été construits sous le logiciel R avec la fonction lm(). Les variables explicatives ont été sélectionnées par stepwise grâce à la fonction stepAIC() du package « MASS ». L'influence significative de ces variables sur le modèle a ensuite été vérifiée en regardant les résultats des tests F de Fisher dans la table d'anova du modèle, donnée par la fonction anova(). La validité des modèles repose sur la normalité des résidus, testée *via* le test de Shapiro-Wilk, ainsi que sur l'homoscédasticité, l'indépendance et la linéarité des résidus qui ont été estimées graphiquement avec la fonction plot(). La significativité du coefficient de détermination R^2 obtenu pour chaque modèle est testée par comparaison au modèle nul avec un test de Fisher dont la p-value est donnée par la fonction summary().

Tableau 3 : Variables utilisées pour la construction des modèles suivant le groupe d'habitats

		HERB	HAIE	BROUS	FORET	CNE	CE	PTEM	PNAT
Variables gestion	FoSeed					X	X	X	
	undersow					X			
	FoMeWeed					X			
	FoFertil					X	X		
	herbi					X	X		
	insecti					X			
	fungi					X			
	mollu					X			
	FoPest					X	X		
	FoHarvest							X	X
	FoTot					X	X	X	X
	Cut	X						X	X
	residues	X				X		X	X
	winter					X	X		
	age							X	
	NoCut		X						
	Man			X	X				X
Variables habitat	Area	X	X	X	X	X	X	X	X
	Flower	X	X	X	X	X	X	X	X
	Tree								X
	PH			X	X	X			X
	HUM			X	X				X

Les X indiquent les variables utilisées dans le groupe d'habitats correspondant.

2.2.3 Modélisation de la diversité des quatre taxons étudiés

Pour identifier les variables environnementales ayant un effet propre à chaque taxon, on a modélisé la réponse de chacune des quatre variables « espèces » utilisées pour calculer les Ibios dans chaque groupe d'habitats. Les variables testées dans chacun des groupes d'habitats sont les mêmes que celles indiquées dans le tableau 3. Les modèles ont été élaborés sous R en fonction de la distribution des données des variables à expliquer : si la distribution est normale, un modèle linéaire a été construit avec la fonction lm() ; si la distribution suit une loi de Poisson, un modèle linéaire généralisé (GLM) a été construit avec la fonction glm(). Les variables explicatives ont été sélectionnées par stepwise grâce à la fonction stepAIC() du package « MASS ». L'influence significative de ces variables sur le modèle a ensuite été vérifiée en regardant les résultats des tests F de Fisher dans la table d'anova des modèles linéaires, et en se basant sur les résultats des tests de Chi2 dans la table d'anova des GLM. La validité des modèles repose sur la normalité des résidus, testée par le test de Shapiro-Wilk, ainsi que sur l'homoscédasticité, l'indépendance et la linéarité des résidus qui ont été estimées graphiquement avec la fonction plot(). La significativité du coefficient de détermination R^2 obtenu pour les modèles linéaires, ou de la part de déviance expliquée D^2 obtenue pour les GLM, est testée par comparaison au modèle nul avec un test de Fisher dont la p-value est donnée par la fonction summary().

Dans certains cas, la courbe de réponse à modéliser a une allure asymptotique. On a donc créé un modèle avec la fonction nls() basé sur l'équation de Michaelis-Menten. L'équation du modèle est alors de la forme : $y = \frac{Vm*x}{K_M+x}$, avec Vm la valeur y maximale et K_M une constante. Un R^2 a été calculé et sa significativité a été testée par comparaison au modèle nul par un test de Chi2.

3 Résultats

3.1 Mise au point d'un indicateur à partir des variables « espèces »

3.1.1 Etude de corrélation

Le corrélogramme réunissant toutes les variables « espèces » (Fig.2) révèle que les richesses spécifiques gamma et beta (RSg et RSb) des quatre taxons sont bien corrélées entre elles, mais qu'elles sont totalement indépendantes des autres variables. Etant donné que ces variables sont indépendantes des autres, qu'elles sont surtout informatives à l'échelle de l'EA et que la mise au point de notre indicateur de biodiversité doit être réalisée à l'échelle du plot, nous les avons écartées pour la suite de nos analyses. Les courbes de corrélations entre les richesses spécifiques gamma et beta sont présentées en annexe 1.

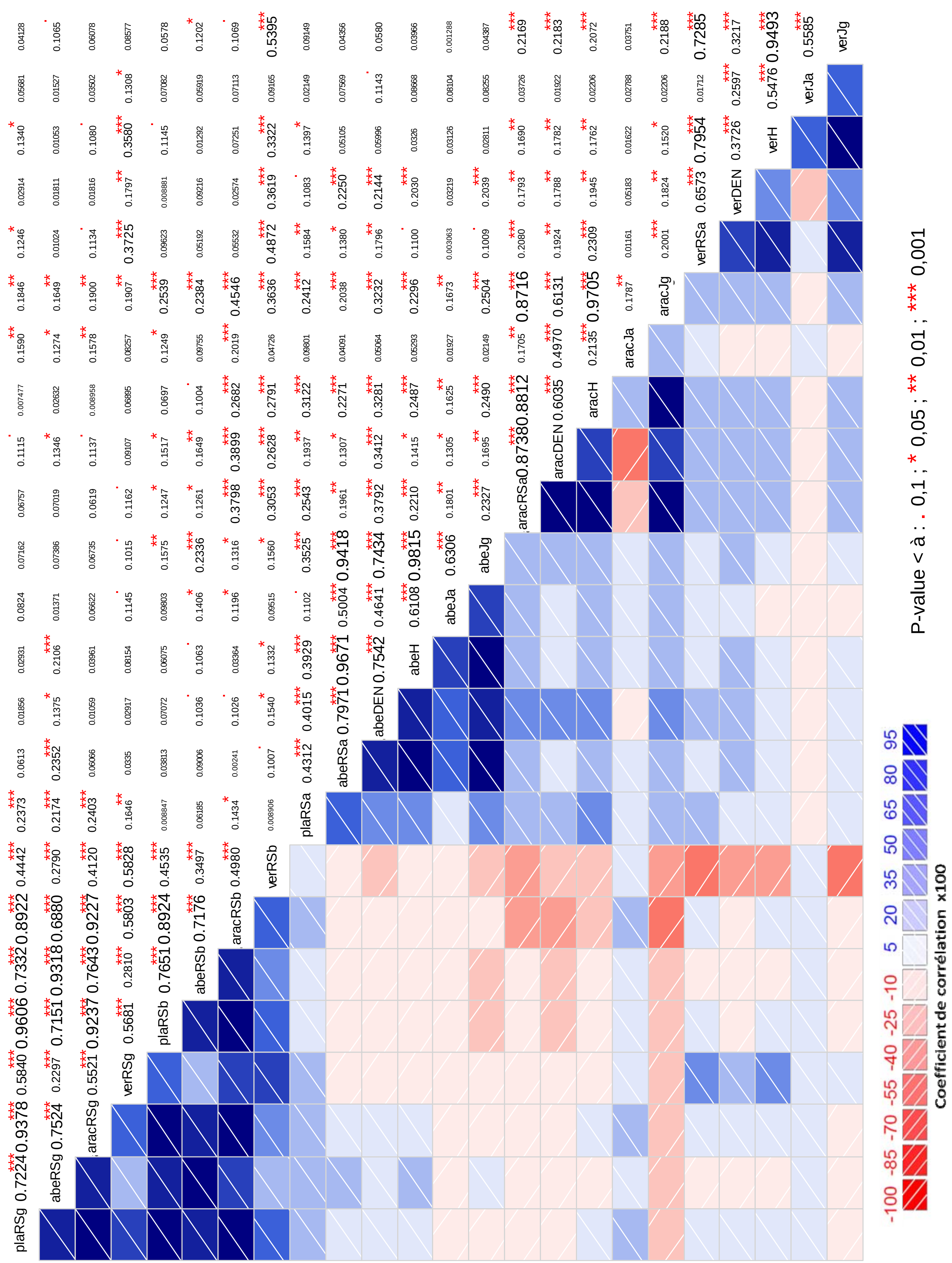


Figure 2 : Corrélogramme des variables « espèces »

pla=plantes, abe=abeilles, arac=araignées, ver=vers de terre, RSg=richesse spécifique gamma, RSb=richesse spécifique beta, RSa=richesse spécifique alpha, DEN=densité (nb/m²), H=indice de Shannon, Ja=indice de Piélou alpha, Jg=indice de Piélou gamma. Les coefficients de corrélation ont été calculés selon la corrélation de Spearman.

Pour les autres variables, celles relatives à un même taxon sont plus corrélées entre elles que celles relatives à un même indice entre taxons (Fig.2). Les courbes de corrélation sont présentées en annexe 2. Les scores de représentativité (Tab.4) indiquent que pour les abeilles et les vers de terre, la variable la plus représentative est l'indice de diversité de Shannon (scores respectifs de 3,03 et 2,46), alors qu'il s'agit de la richesse spécifique alpha pour les araignées (2,39). Ainsi notre indicateur intégrerait la richesse spécifique alpha des plantes et des araignées et l'indice de diversité de Shannon des abeilles et des vers de terre.

Tableau 4 : Scores de représentativité des variables « espèces » des taxons abeilles, araignées et vers de terre.

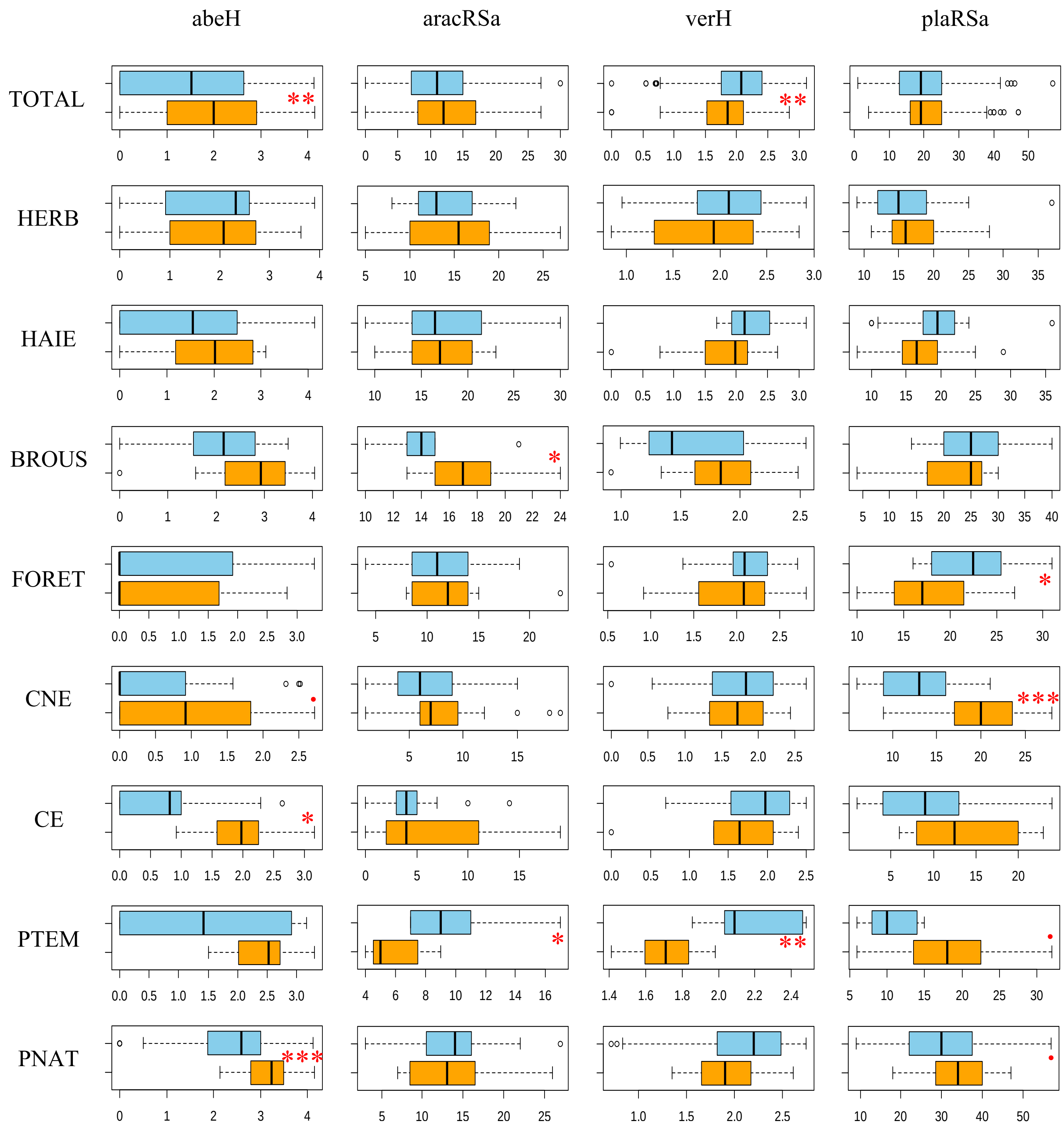
	abeilles	araignées	vers de terre	total
Richesse spécifique alpha	2,8723	2,3930	1,8524	7,1177
Densité (nb/m²)	2,2033	2,0674	1,2181	5,4889
Diversité de Shannon H	3,0323	2,2438	2,4562	7,7322
Diversité de Piélou J alpha	2,1056	1,0126	1,2740	4,3922
Diversité de Piélou J gamma	3,0030	2,1647	2,2953	7,4631

3.1.2 Comparaison entre EA biologiques et conventionnelles

A l'échelle de l'exploitation, les diversités de Shannon en abeilles et en vers de terre permettent de différencier exploitations biologiques et conventionnelles : les EA biologiques auraient une diversité en abeilles plus élevée et une diversité en vers de terre plus faible que les EA conventionnelles (Fig.3). A l'échelle de l'habitat, la même tendance est visible pour la diversité en abeilles, qui est significative pour les groupes CNE, CE et PNAT. Mais pour la diversité en vers de terre, la tendance est significativement plus importante uniquement dans les prairies temporaires conventionnelles.

Les richesses spécifiques en araignées et en plantes ne permettent pas d'établir une différence entre biologique et conventionnel à l'échelle de l'exploitation. En revanche des différences significatives entre les deux types de gestion apparaissent suivant le groupe d'habitats. Les EA biologiques possèdent la plus grande richesse spécifique en araignées dans le groupe BROUS, et en plantes dans les groupes CNE, PTEM et PNAT. Les EA conventionnelles ont quant à elles des prairies temporaires plus riches en espèces d'araignées et des forêts plus riches en espèces de plantes.

Aucune des 4 variables ne permet d'établir une différence significative entre gestions biologique et conventionnelle dans les groupes HERB et HAIE (Fig.3). Même en combinant ces 4 variables dans une ACP (Fig.4), les deux types de gestion restent indifférenciés dans ces groupes d'habitats. En effet les résultats des tests de Monte-Carlo sont de 0,6476 pour le groupe HERB et de 0,1573 pour le groupe HAIE, et les résultats des tests de Mann-Whitney



P-value < à : . 0,1 ; * 0,05 ; ** 0,01 ; *** 0,001

Figure 3 : Boîtes à moustaches des variables « espèces » dans les différents habitats des systèmes agricoles des « Vallées et Coteaux de Gascogne ». En bleu : plots conventionnels ; en orange : plots biologiques. abeH = indice de diversité de Shannon en abeilles, aracRSa = richesse spécifique alpha en araignées, verH = indice de diversité de Shannon en vers de terre, plaRSa = richesse spécifique alpha en plantes. TOTAL = tous les habitats confondus, HERB = bandes herbeuses, HAIE = lignes de buissons et d'arbres, BROUS = taillis et broussailles, FORET = couverts ligneux de plus de 5m de hauteur, CNE = cultures non entomophiles, CE = cultures entomophiles, PTEM = prairies temporaires, PNAT = prairies naturelles.

sur l'axe représentant le gradient de biodiversité ne traduisent pas de différence entre gestions biologique et conventionnelle (p-value > 0,05). Les mêmes résultats sont observés sur l'ACP du groupe FORET, donc dans ces 3 groupes d'habitats, les EA biologiques et conventionnelles ont une biodiversité équivalente. Pour les groupes CNE, PTEM et PNAT, la différence de biodiversité entre les deux gestions agricoles est en revanche confirmée par les résultats des tests de Monte-Carlo et des tests de Mann-Whitney (Fig.4). Pour les groupes

BROUS et CE, la différence de biodiversité entre agricultures biologique et conventionnelle est également révélée par les résultats des tests de Mann-Whitney (p-values respectives de 0,0251 et 0,0125 <0,05), alors qu'elle ne l'est pas d'après les résultats des tests de Monte-Carlo (respectivement de 0,1162 et 0,0563 > 0,05).

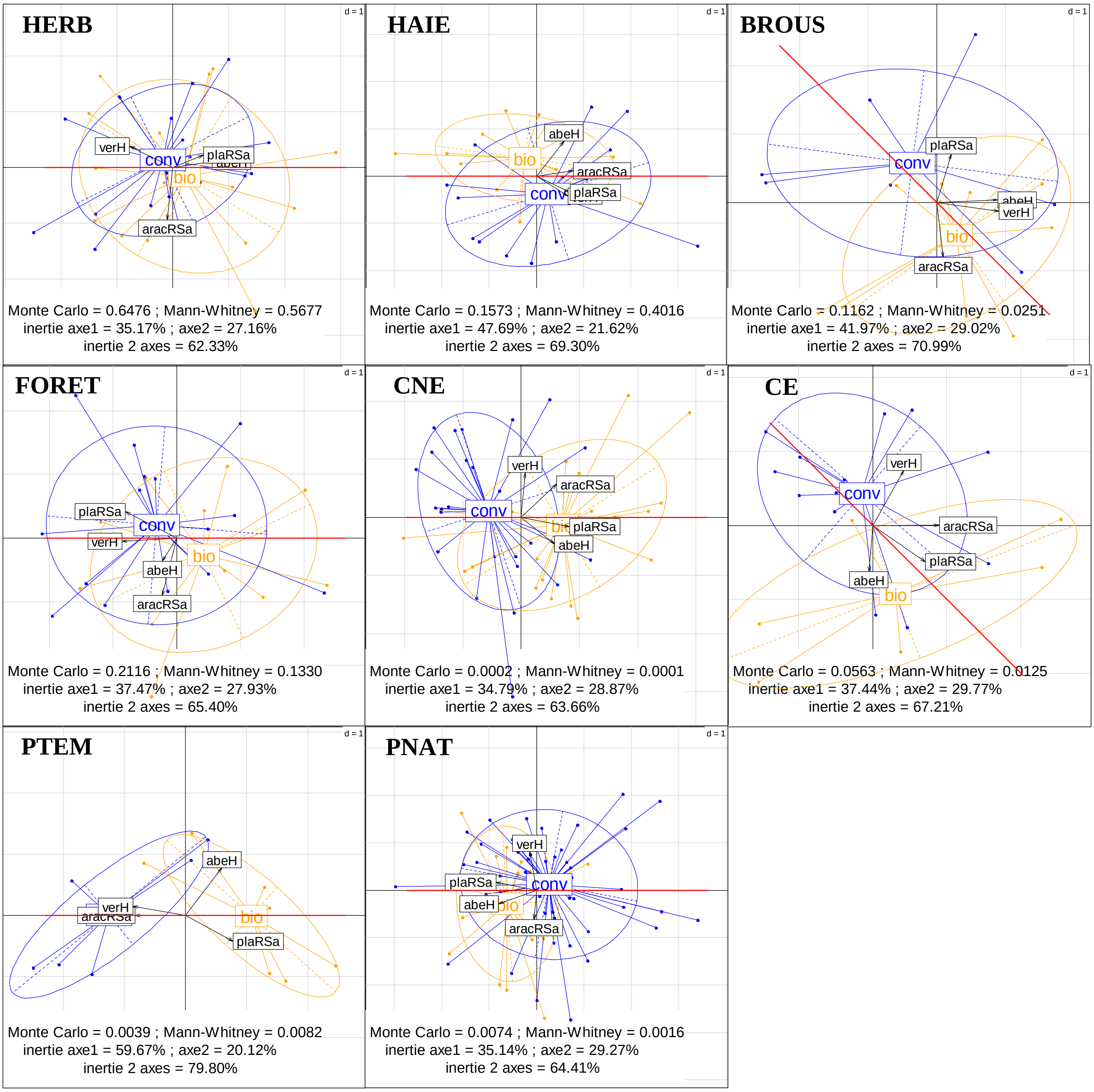


Figure 4 : Représentation graphique des Analyses en Composantes Principales des variables « espèces » pour les différents habitats des systèmes agricoles des « Vallées et Coteaux de Gascogne ». HERB = bandes herbeuses, HAIE = lignes de buissons et d'arbres, BROUS = taillis et broussailles, FORET = couverts ligneux de plus de 5m de hauteur, CNE = cultures non entomophiles, CE = cultures entomophiles, PTEM = prairies temporaires, PNAT = prairies naturelles. abeH = indice de diversité de Shannon en abeilles, aracRSa = richesse spécifique alpha en araignées, verH = indice de diversité de Shannon en vers de terre, plaRSa = richesse spécifique alpha en plantes. bio = plots biologiques, conv = plots conventionnels. Les tests de Mann-Whitney ont été réalisés sur les axes rouges.

Globalement, pour les groupes d'habitats dont le gradient de biodiversité permet de voir une différence entre gestions biologique et conventionnelle, ce sont les plots biologiques qui ont la biodiversité la plus élevée (Fig.4). Seule le groupe PTEM ne va pas dans ce sens : les prairies temporaires biologiques ont une plus grande biodiversité en abeilles et en plantes, alors que les prairies temporaires conventionnelles ont une plus grande biodiversité en araignées et en vers de terre. Il est donc impossible de lire un gradient de biodiversité dans le cas des prairies temporaires. Des modèles de prédiction ne peuvent donc être construits qu'à partir des ACP des groupes BROUS, CNE, CE et PNAT.

3.1.3 Elaboration de modèles prédictifs supervisés

Un modèle prédictif supervisé a été construit pour les groupes d'habitats BROUS, CNE, CE et PNAT respectivement nommés Mbrous, Mcne, Mce et Mpnat. Les prédictions des modèles Mbrous, Mcne et Mpnat sont relativement fiables car leur pourcentage d'erreur de prédiction par LOO-CV ne dépasse pas les 10 % (Fig.5).

Le modèle Mce est quant à lui moins efficace avec 31,58 % d'erreur de prédiction par LOO-CV. En regardant la position des plots mal prédits sur la figure 5, on s'aperçoit qu'ils sont les plus éloignés de l'origine. Cela montre que si l'on retire un plot « extrême », alors le modèle sous-jacent à l'ACP est très modifié. Pour minimiser cette influence des plots « extrêmes », il faut augmenter le nombre de plots pour ainsi améliorer la robustesse du modèle de l'ACP.

Les modèles Mbrous, Mcne et Mpnat sont donc validés, mais pas le modèle Mce dont la robustesse est trop faible.

3.1.4 Création d'un indicateur de biodiversité à l'échelle de l'EA

A l'échelle de l'EA, l'indicateur BIOMAX varie de 16,73 à 33,64 pour les EA biologiques et de 10,11 à 26,10 pour les EA conventionnelles (Tab.5). L'indicateur BIOMOY varie lui de 14,98 à 27,11 pour les EA biologiques et de 9,56 à 20,13 pour les EA conventionnelles. Que l'on s'intéresse à la biodiversité maximale ou à la biodiversité moyenne, la même tendance est donc observée : la biodiversité est globalement plus importante dans les EA biologiques que dans les EA conventionnelles, avec cependant un recouvrement important des intervalles. Les régressions linéaires des indicateurs BIOMAX et BIOMOY en fonction du nombre de plots par EA conduisent aux mêmes conclusions. Qu'il s'agisse de la richesse spécifique gamma ou des indicateurs, la biodiversité estimée est proportionnelle au nombre d'habitats Biohab recensés dans les EA (Fig. 6). Mais en comparant les régressions linéaires des indicateurs à celles de la richesse spécifique, la différence de biodiversité estimée entre EA biologiques et conventionnelles est plus marquée pour les indicateurs. Cela est dû au fait que pour un même nombre d'habitats, la biodiversité

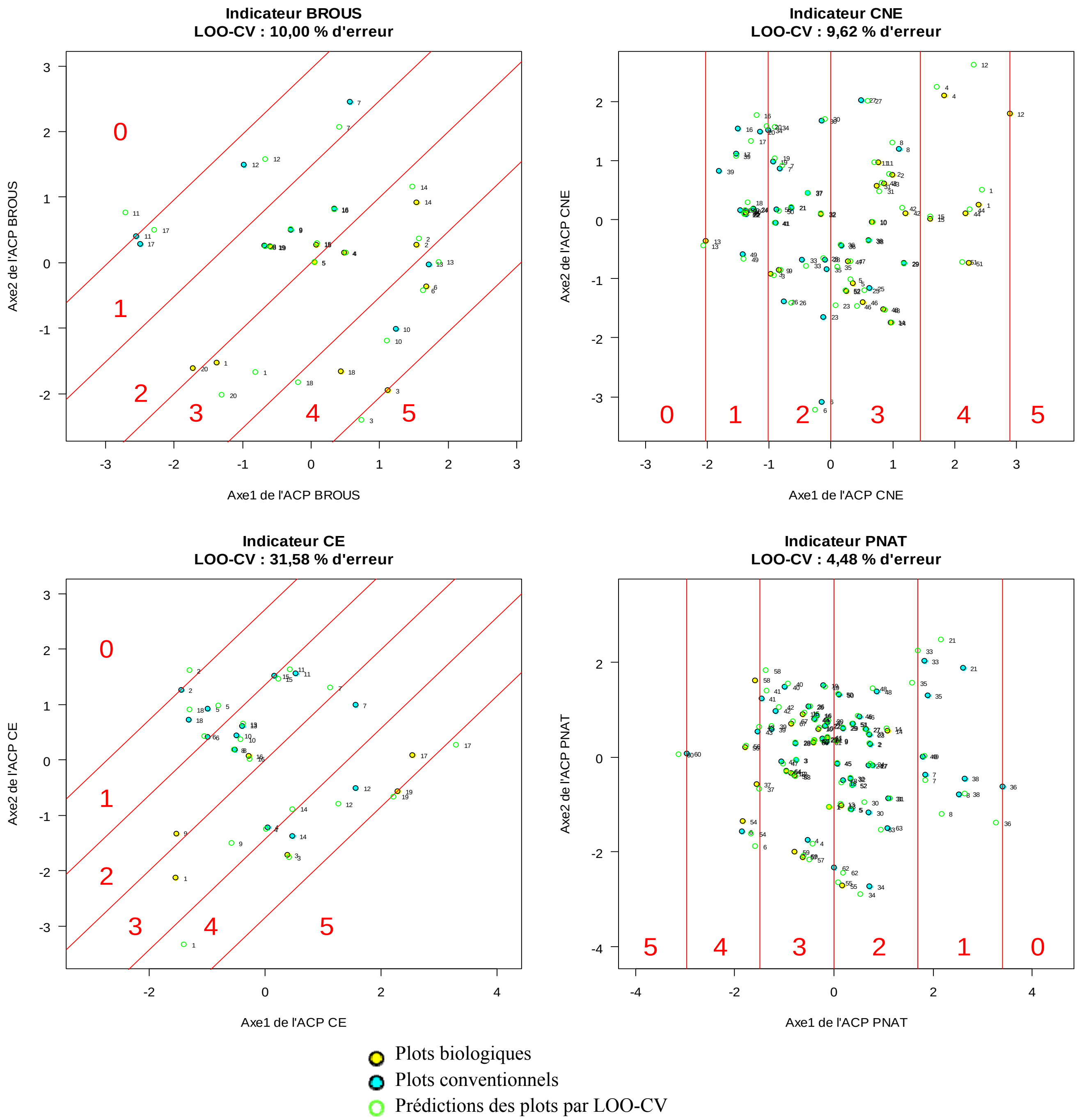


Figure 5 : Représentation graphique des plots et de leur prédiction par Leave-One-Out Cross-Validation. BROUS = taillis et broussailles, CNE = cultures non entomophiles, CE = cultures entomophiles, PNAT = prairies naturelles. En rouge sont définies les zones correspondantes aux valeurs des indices de biodiversité : 0=faible biodiversité, 5=biodiversité élevée.

estimée par nos indicateurs est 1,33 fois plus grande dans une EA biologique que dans une EA conventionnelle. Pour les modèles biologique et conventionnel expliquant la richesse spécifique gamma, les pentes sont, en effet, quasiment identiques (coefficients directeurs respectifs de 8,87 et 9,15), alors que pour les deux indicateurs, le coefficient directeur des modèles de gestion biologique est multiplié par 2 par rapport à celui des modèles de gestion conventionnelle (Fig. 6). Bien que la biodiversité tend à être plus élevée dans les EA biologiques, quelle que soit la manière dont elle est estimée, les tests de comparaison réalisés

Tableau 5 : Indicateurs BIOMAX et BIOMOY des habitats semi-naturels (HSN), des habitats cultivés (HCU) et des exploitations agricoles (EA). Les exploitations sont classées selon l'ordre décroissant des valeurs de l'indicateur à l'échelle de l'EA.

		HSN						HCU						EA
	n°EA	HERB	HAIE	BROUS	FORET	Total /20	n°EA	CNE	CE	PTEM	PNAT	Total /20	n°EA	Total /40
BIOMAX	F05	3,52	4,12	4	5	16,64	F05	5	4	5	3	17	F05	33,64
	F14	4,34	2,57	3	3,9	13,81	F14	4	4	4,04	4	16,04	F14	29,85
	F16	4,34	4,32	4	3,31	15,97	F16	4	5	NA	3	12	F16	27,97
	F12	4,43	3,92	4	4,75	17,1	F12	3	2	NA	4	9	F12	26,1
	F07	4,84	5	4	4,66	18,5	F07	2	1	NA	3	6	F07	24,5
	F08	3,36	3,31	NA	4,66	11,33	F08	3	3	3,94	2	11,94	F08	23,27
	F11	3,2	3,31	NA	2,97	9,48	F11	3	4	4,26	1	12,26	F11	21,74
	F02	5	3,45	3	NA	11,45	F02	4	3	NA	3	10	F02	21,45
	F15	4,26	4,39	2	2,29	12,94	F15	2	1	NA	5	8	F15	20,94
	F01	4,34	3,24	NA	3,64	11,22	F01	4	NA	4,26	NA	8,26	F01	19,48
	F10	4,26	2,97	NA	2,71	9,94	F10	3	2	NA	4	9	F10	18,94
	F13	3,28	2,43	NA	3,56	9,27	F13	3	NA	3,62	3	9,62	F13	18,89
	F03	4,02	2,23	NA	3,14	9,39	F03	3	1	NA	4	8	F03	17,39
	F06	3,2	3,78	NA	3,22	10,2	F06	1	3	NA	3	7	F06	17,2
	F04	3,28	2,91	5	2,54	13,73	F04	NA	NA	NA	3	3	F04	16,73
	F09	3,52	4,59	NA	NA	8,11	F09	2	NA	NA	NA	2	F09	10,11

		HSN						HCU						EA
	n°EA	HERB	HAIE	BROUS	FORET	Total /20	n°EA	CNE	CE	PTEM	PNAT	Total /20	n°EA	Total /40
BIOMOY	F05	3,09	3,75	3,5	3,45	13,79	F05	3	4	3,99	2,33	13,32	F05	27,11
	F14	3,61	2,3	2,5	3,35	11,76	F14	2,8	3	4,04	3,33	13,17	F14	24,93
	F16	3,61	3,65	2,67	2,83	12,76	F16	3,5	5	NA	3	11,5	F16	24,26
	F02	4,21	3,42	3	NA	10,63	F02	3,5	3	NA	3	9,5	F02	20,13
	F08	3,19	2,77	NA	4,66	10,62	F08	2,29	2,5	3,09	1,63	9,51	F08	20,13
	F07	4,55	3,99	2,25	3,84	14,63	F07	1,5	1	NA	2,5	5	F07	19,63
	F12	3,94	3,28	2	3,71	12,93	F12	1,8	2	NA	2,5	6,3	F12	19,23
	F13	3,2	2,43	NA	3,56	9,19	F13	3	NA	3,62	3	9,62	F13	18,81
	F11	2,95	2,74	NA	2,97	8,66	F11	2	2,8	3,83	1	9,63	F11	18,29
	F10	3,53	2,67	NA	2,71	8,91	F10	2,5	2	NA	4	8,5	F10	17,41
	F01	4,06	2,8	NA	3,64	10,5	F01	3	NA	3,76	NA	6,76	F01	17,26
	F15	3,91	3,99	1,5	2,29	11,69	F15	1,5	1	NA	3	5,5	F15	17,19
	F06	2,87	3,18	NA	3,22	9,27	F06	1	3	NA	2,33	6,33	F06	15,6
	F04	3,28	2,74	3,67	2,54	12,23	F04	NA	NA	NA	2,75	2,75	F04	14,98
	F03	3,31	2	NA	3,01	8,32	F03	2,33	1	NA	2,29	5,62	F03	13,94
	F09	3,44	4,12	NA	NA	7,56	F09	2	NA	NA	NA	2	F09	9,56

exploitations biologiques
exploitations conventionnelles

ne permettent pas de le confirmer, car aucune différence significative n'apparaît (Fig.7). La biodiversité étant proportionnelle au nombre d'habitats, la disparité du nombre d'habitats recensés dans les EA étudiées est trop importante pour pouvoir détecter une telle différence.

Dans les HSN, l'indicateur BIOMAX varie de 9,27 à 16,64 dans les EA biologiques et de 8,11 à 18,50 dans les EA conventionnelles (Tab.5). L'indicateur BIOMOY varie de 8,91 à 13,79 dans les EA biologiques et de 7,56 à 14,63 dans les EA conventionnelles. Il n'y a donc pas de différence significative de biodiversité entre les HSN des EA biologiques et des EA conventionnelles (Fig.7).

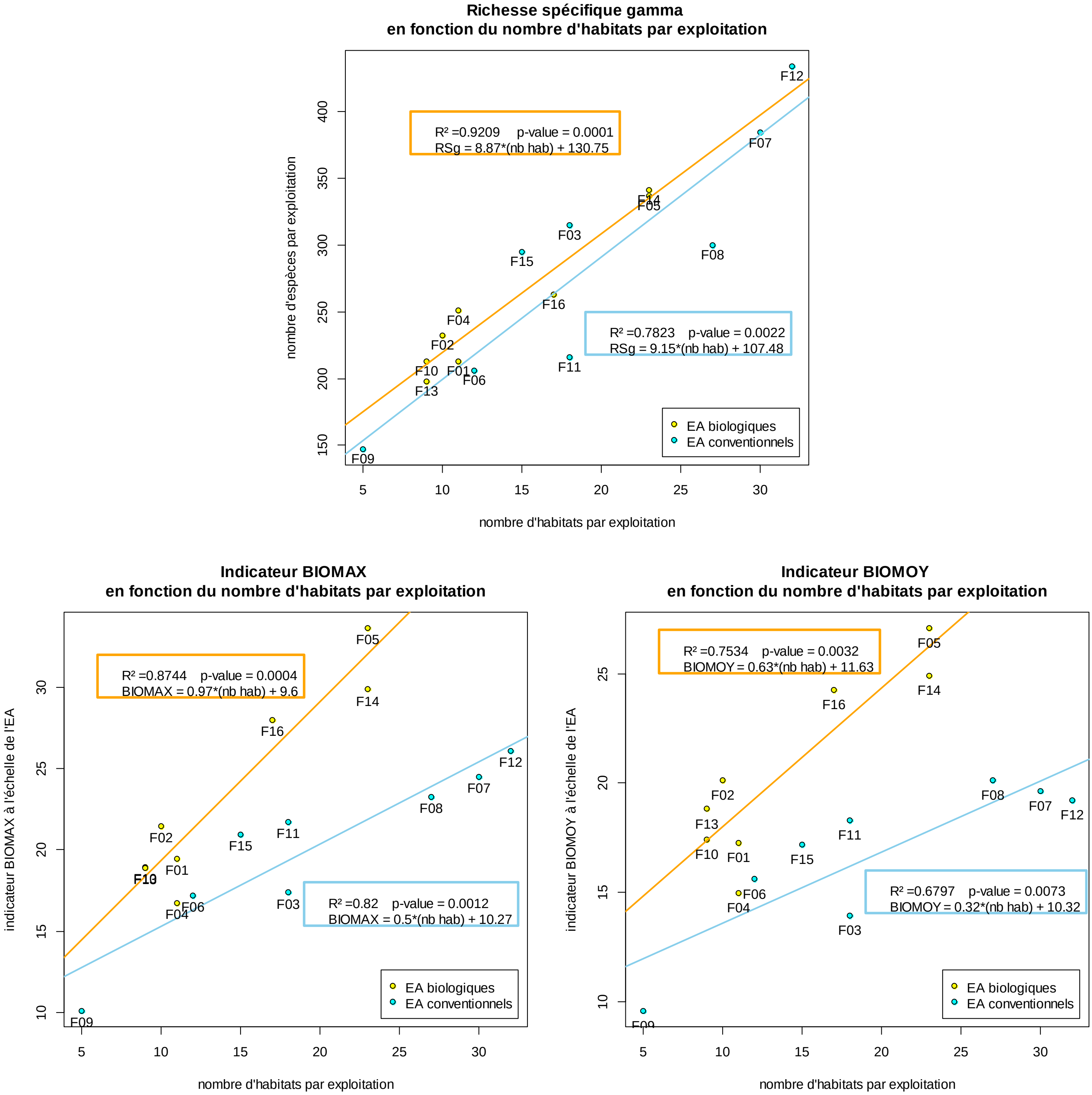


Figure 6 : Régression linéaire de la richesse spécifique gamma et des indicateurs BIOMAX et BIOMOY en fonction du nombre d'habitats Biohab recensés dans les exploitations agricoles (EA). En bleu, les modèles correspondant aux EA conventionnelles ; en orange, ceux correspondant aux EA biologiques.

Dans les HCU, l'indicateur BIOMAX varie de 3 à 17 dans les EA biologiques et de 2 à 12,26 dans les EA conventionnelles (Tab.5). La biodiversité maximale semble donc plus importante dans les HCU des EA biologiques que des EA conventionnelles. La même tendance est visible pour l'indicateur BIOMOY qui varie de 2,75 à 13,32 pour les EA biologiques et de 2 à 9,63 dans les EA conventionnelles. Mais cette tendance n'est, là encore, pas confirmée par les tests de comparaison qui indiquent seulement une différence significative avec un risque α de 10% pour l'indicateur BIOMOY (Fig.7).

En comparant la biodiversité des HSN et des HCU, l'indicateur BIOMOY dans les HCU des EA conventionnelles est significativement plus faible que dans les HCU biologiques, les HSN biologiques et les HSN conventionnels (Fig.7). Bien que la différence ne soit pas significative entre indicateurs BIOMAX des HCU des EA biologiques et conventionnelles, la même tendance est visible sur la figure 7. Le type de gestion a donc un impact plus fort sur la biodiversité des HCU que sur celle des HSN, mais la biodiversité des HCU subit une répression moins forte avec les pratiques agricoles biologiques que conventionnelles.

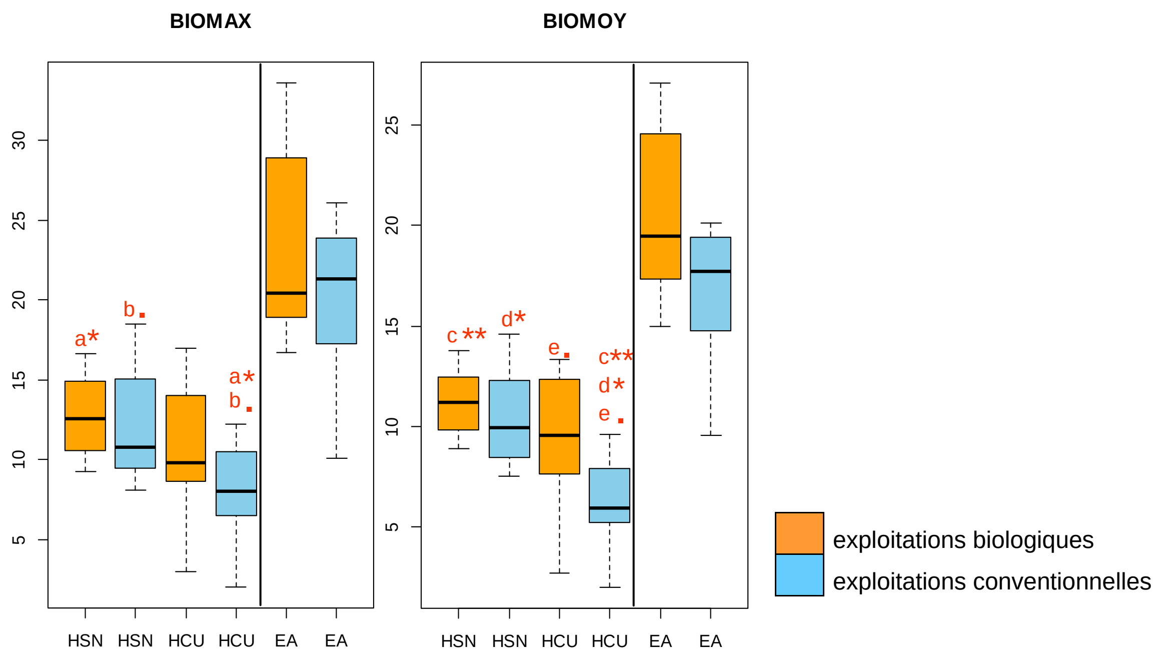


Figure 7 : Boîtes à moustaches des indicateurs BIOMAX et BIOMOY dans les habitats semi-naturels (HSN), les habitats cultivés (HCU) et les exploitations agricoles (EA).
Les lettres rouges indiquent une différence entre deux habitats significative à : . 10 % ; * 5 % ; ** 1 %.

3.2 Etude des relations entre les variables « espèces » et les variables « gestion » et « habitat »

3.2.1 *Modélisation de la biodiversité dans les agroécosystèmes*

Un modèle linéaire prédictif de la biodiversité a pu être construit pour chaque groupe d'habitats (Tab.6). La plupart sont significativement différents du modèle nul avec un risque α de 5% ; seul le modèle du groupe PTEM est significativement différent du modèle nul avec un risque α de 6%. Bien que les variables influent significativement sur la biodiversité, la variance expliquée ne dépasse pas les 46,64% pour les BROUS. Les variables « gestion » et « habitat » utilisées ne sont donc pas suffisantes pour remplacer les indicateurs de biodiversité.

Les variables les plus influentes sur la biodiversité semblent être le pourcentage de fleurs et le pH du sol dans les HSN (HERB, HAIE, FORET) et les prairies naturelles (PNAT). Dans les habitats à plus fort impact anthropique, telles les cultures (CE et CNE) et les prairies temporaires (PTEM), pesticides, récolte et préparation du lit de semence seraient les variables les plus explicatives de la biodiversité. Les taillis et broussailles (BROUS) qui sont des HSN, sont très impactés par l'Homme, puisque leur biodiversité dépend principalement de l'intensité de gestion. Ces huit modèles, bien qu'ayant une grande part de variance résiduelle, permettent ainsi de montrer que dans les habitats les moins soumis aux actions anthropiques, la biodiversité est plutôt sous l'influence des variables « habitat », alors que dans les habitats plus exploités/gérés par l'Homme, la biodiversité se trouve sous l'influence des variables « gestion ».

Tableau 6 : Modèles de prédiction de la biodiversité élaborés pour chaque groupe d'habitats

Habitat	modèle	R^2	p-value	
HERB	lm(Ibio ~ Flower)	0,2110	0,0019	**
HAIE	lm(Ibio ~ Flower)	0,3499	0,0002	***
BROUS	lm(Ibio ~ Man)	0,4664	0,0005	***
FORET	lm(Ibio ~ PH + Flower)	0,3424	0,0020	**
CNE	lm(Ibio ~ PH + FoSeed + Flower)	0,1610	0,0095	**
CE	lm(Ibio ~ FoPest)	0,1997	0,0315	*
PTEM	lm(Ibio ~ residues + FoHarvest)	0,3185	0,0591	.
PNAT	lm(Ibio ~ Flower)	0,2495	< 0,0001	***

significatif à : . 10% ; * 5% ; ** 1% ; *** 0,1%

en bleu : coefficients positifs
en rouge : coefficients négatifs

3.2.2 Modélisation de la diversité des quatre taxons étudiés

Pour prédire la diversité en abeilles, un modèle significativement différent du modèle nul, avec un risque α de 5 %, a pu être créé pour chaque groupe d'habitats (Tab.7). La variable qui apparait être la plus explicative de cette diversité est le pourcentage de fleurs de l'habitat. Les modèles construits avec cette variable ont en effet variance/déviance expliquée qui va de 28,5% dans les CNE ($D^2 = 0{,}2850$) à 81,96% dans les PTEM ($R^2 = 0{,}8196$). Dans les CE, ce sont les herbicides qui auraient une forte influence sur cette diversité ($D^2 = 0{,}4386$), alors que dans les BROUS, c'est l'intensité de gestion et la surface du plot qui ressortent légèrement ($R^2 = 0{,}2208$).

Les modèles de prédiction de la diversité en araignées sont beaucoup moins efficaces que ceux des abeilles. En effet, sur les six modèles significativement différents du modèle nul avec un risque α de 5%, la variance/déviance expliquée est comprise entre 10,98 % pour les HAIE ($R^2 = 0{,}1098$) et 24,52% pour les PTEM ($D^2 = 0{,}2452$) (Tab.7). Les variables explicatives sont essentiellement des variables de gestion, telles que les insecticides, les herbicides, les opérations de récolte et l'intensité de gestion. La seule variable « habitat » ayant une influence sur la diversité en araignées est le pourcentage de fleurs, qui intervient dans les modèles des groupes HAIE, CE et PTEM.

Tableau 7 : Modèles de prédiction de chaque taxon dans chaque groupe d'habitats

Habitat	taxon	modèle	R^2	D^2	p-value	
HERB	abeilles	glm(abeRSa ~ Flower+ Area, family=poisson)		0,5004	< 0,0001	***
	araignées	--				
	vers de terre	--				
	plantes	--				
HAIE	abeilles	glm(abeRSa ~ Flower+ Flower^2, family=poisson)		0,4013	< 0,0001	***
	araignées	lm(aracRSa ~ Flower)	0,1098		0,0360	*
	vers de terre	--				
	plantes	lm(plaRSa ~Flower)	0,1942		0,0067	**
BROUS	abeilles	lm(abeH ~ Man + Area)	0,2208		0,0466	*
	araignées	lm(aracRSa ~ Man)	0,1994		0,0278	*
	vers de terre	lm(verH ~ Man + HUM)	0,4873		0,0013	**
	plantes	lm(plaRSa ~ PH + PH^2)	0,1890		0,0655	.
FORET	abeilles	nls(abeRSa ~Flower*Vm/(K+Flower)	0,7653		0,0004	***
	araignées	lm(aracRSa ~ Area)	0,0780		0,0815	.
	vers de terre	--				
	plantes	--				
CNE	abeilles	glm(abeH ~ Flower + Flower^2, family=poisson)		0,2850	< 0,0001	***
	araignées	glm(aracRSa ~ insecti, family=poisson)		0,1357	0,0002	***
	vers de terre	lm(verH ~ herbi)	0,0780		0,0254	*
	plantes	lm(plaRSa ~ FoPest + FoSeed)	0,1986		0,0017	**
CE	abeilles	glm(abeH ~ herbi, family=poisson)		0,4386	0,0039	**
	araignées	glm(aracRSa ~ herbi + Flower, family=poisson)		0,1614	0,0008	***
	vers de terre	--				
	plantes	lm(plaRSa ~ Area)	0,1784		0,0407	*
PTEM	abeilles	nls(abeRSa ~Flower*Vm/(K+Flower)	0,8196		< 0,0001	***
	araignées	glm(aracRSa ~ Flower, family=poisson)		0,2452	0,0407	*
	vers de terre	lm(verRSa ~ FoSeed + Area)	0,5765		0,0055	**
	plantes	glm(plaRSa ~ residues + Area, family=poisson)		0,3212	0,0008	***
PNAT	abeilles	nls(abeH ~Flower*Vm/(K+Flower)	0,4650		< 0,0001	***
	araignées	lm(aracRSa ~ FoHarvest + Man)	0,1618		0,0013	**
	vers de terre	lm(verH ~ PH)	0,1296		0,0016	**
	plantes	lm(plaRSa ~ FoHarvest + residues + Flower)	0,2925		< 0,0001	***

significatif à : . 10% ; * 5% ; ** 1% ; *** 0,1%

en bleu : coefficients positifs
en rouge : coefficients négatifs

La diversité en vers de terre apparait comme la plus difficile à prédire, car seulement quatre modèles significativement différents du modèle nul, avec un risque α de 5%, ont pu être construits (Tab.7). La variance expliquée est de plus très faible dans les modèles des groupes CNE et PNAT (R^2 respectifs de 0,0780 et 0,1296). Dans les BROUS, il semble que, comme pour les abeilles et les araignées, l'intensité de gestion influence la diversité en vers de terre, mais l'humidité joue également un rôle et permet d'atteindre les 48,73% de variance expliquée (R^2 = 0,4873). Le meilleur modèle de prédiction pour les vers de terre est celui du groupe PTEM avec une variance expliquée de 57,65% (R^2 = 0,5765). Dans ces prairies temporaires, les variables dont dépend la diversité en vers de terre sont la préparation du lit de semence et la surface.

La diversité en plantes est également difficile à prédire : cinq modèles significativement différents du modèle nul avec un risque α de 5% ont pu être construits (Tab.7). Les modèles des groupes PTEM et PNAT ont les meilleures parts de variance/déviance expliquée (D^2 = 0,3212 pour PTEM et R^2 = 0,2925 pour PNAT). Dans ces habitats, la gestion de la végétation après fauche et les opérations de récolte sont les facteurs influençant le plus la diversité des plantes. La variance expliquée est plus faible pour les modèles des groupes HAIE (R^2 = 0,1942), CNE (R^2 = 0,1986) et CE (R^2 = 0,1784). Dans le groupe HAIE, la diversité en plantes est faiblement liée au pourcentage de fleurs, alors que dans le groupe CE, elle serait liée à la surface du plot. Dans les CNE, ce sont les variables « gestion » qui ressortent avec une influence des pesticides et de la préparation du lit de semence sur la diversité en plantes.

4 Discussion

4.1 Qualité des résultats obtenus

Les indicateurs BIOMOY et BIOMAX sont deux indicateurs de biodiversité qui n'apportent pas la même information. L'indicateur BIOMOY, qui indique la biodiversité moyenne de chaque groupe d'habitats dans une EA, a un rôle de « diagnostic » et peut permettre de juger de l'état de la biodiversité sur l'ensemble d'une EA. L'indicateur BIOMAX, qui indique la biodiversité maximale observée de chaque groupe d'habitats d'une EA, a plus une vocation pédagogique. En effet, si un plot possède une plus grande biodiversité que les autres plots du même groupe d'habitats dans la même EA, une réflexion peut être conduite avec l'agriculteur pour déterminer les facteurs environnementaux *lato sensu* qui favorisent la biodiversité de ce plot. On pourrait alors essayer de raisonner l'augmentation de la biodiversité dans les autres plots par une gestion au cas par cas. Si la biodiversité des plots les plus pauvres augmente, alors l'indicateur BIOMOY va lui aussi augmenter. Ces indicateurs BIOMAX et BIOMOY sont également intéressants pour évaluer l'absence d'un groupe d'habitats. La biodiversité étant proportionnelle au nombre d'habitats, il est logique que moins une EA possède d'habitats différents, plus son indicateur de biodiversité sera faible. Enfin, les résultats des indicateurs BIOMAX et BIOMOY, malgré l'information différente qu'ils apportent, permettent d'aboutir aux mêmes conclusions quant aux états de biodiversité dans les agroécosystèmes : les HCU des EA conventionnelles ont une plus faible biodiversité que les HSN des EA conventionnelles et que les HCU et HSN des EA biologiques. Cette comparaison globale doit cependant être détaillée car la biodiversité est représentée par quatre taxons indépendants. On a pu, en effet, voir dans cette étude que les EA biologiques favorisent la diversité en abeilles et en plantes, alors que les EA conventionnelles semblent favoriser la diversité en vers de terre. Pour la diversité des araignées, les résultats ne montrent pas de différence marquée. Ces résultats rejoignent ceux de Hole *et al.* (2005), qui ont conclu que l'agriculture biologique conduit à une plus grande abondance et richesse spécifique en plantes et en oiseaux que l'agriculture conventionnelle, mais que pour certains invertébrés comme les vers de terre, les papillons, les araignées et les scarabées, la différence n'est pas toujours nette. Le cas d'étude italien du projet BioBio, sur des exploitations viticoles, montre aussi qu'une gestion biologique est plus bénéfique à la diversité des plantes vasculaires qu'une gestion conventionnelle (Nascimbene *et al.,* 2012).

Qu'il s'agisse des modèles de prédiction de la biodiversité de l'ensemble des quatre taxons ou de chacun, la part de variance/déviance expliquée est faible. Seuls deux modèles dépassent les 60% de variance/déviance expliquée : la réponse de la richesse spécifique des abeilles au pourcentage de fleurs dans les prairies temporaires ($R^2 = 0,8196$) et dans les forêts de plus de 5m de hauteur ($R^2 = 0,7653$). Il est donc impossible de remplacer les indicateurs directs de biodiversité que sont les abeilles, araignées, vers de terre et plantes par les seules variables « gestion » et « habitat » utilisées dans cette étude. D'autres facteurs environnementaux tels que le type de travail du sol (labour, technique culturale simplifiée ou semis direct) et la texture du sol pourraient permettre d'améliorer ces modèles, notamment les modèles de prédiction de la diversité en vers de terre. Le travail du sol est une variable de gestion présente dans nos bases de données, malheureusement les seules EA étudiées qui utilisent le semis direct favorable à la biodiversité sont conventionnelles. Pour pouvoir étudier le lien entre biodiversité et travail du sol, il faudrait comparer des EA biologiques à des EA conventionnelles appliquant toutes le semis direct.

Le paysage entourant un habitat a également une influence sur la biodiversité de cet habitat. Les bandes enherbées, les haies et les prairies constituent des refuges pour les abeilles sauvages qui nichent directement dans le sol (Oertli *et al.*, 2005 ; Ockinger et Smith, 2007), ainsi plus il y a de haies, de bandes enherbées et de prairies autour d'une parcelle cultivée, plus on aura de chances d'avoir une biodiversité en abeilles importante dans cette parcelle. La zone explorée par les abeilles sauvages autour de leur nid ne dépassant que rarement les 500m (Zurbuchen *et al.*, 2010), il faudrait calculer le pourcentage de prairies, cultures, friches, haies et bandes enherbées dans un rayon (100, 200, 300, 400 et 500m) autour de chaque plot pour voir l'influence de ces variables spatiales sur nos modèles. Il faudrait également relever la présence de points d'eau et de fossés dans ces différents rayons, car la répartition des abeilles et des araignées dépend des bords de fossés (Sepp *et al.*, 2004 ; Öberg *et al.*, 2007), comme de nombreux autres taxons d'invertébrés, tels que les hémiptères, les coléoptères et les lepidotères, ainsi que des amphibiens vivant dans ces habitats aquatiques des agroécosystèmes (Herzon et Helenius, 2008).

4.2 Amélioration de l'indicateur de biodiversité proposé

La méthode de calcul des Ibios est une première tentative de mesure synthétique de la biodiversité. Pour les groupes d'habitats sans modèle prédictif basé sur l'ACP, le calcul des Ibios ne pose pas de problème majeur, si ce n'est que la richesse spécifique maximale utilisée est celle de nos données qui reposent sur un échantillon réduit. Il est possible qu'une richesse spécifique plus élevée soit observée dans certaines EA, ce qui requerra alors de mettre à jour notre calcul des Ibios. Les Ibios des groupes d'habitats pour lesquels un modèle prédictif basé sur l'ACP a pu être construit ont en revanche un problème de précision : les Ibios calculés sont des valeurs discrètes. Comme on a pu le constater pour le modèle Mce en *3.1.3*, il est nécessaire d'augmenter le nombre de plots pour améliorer la qualité de prédiction de ce modèle. Après une augmentation du nombre de plots estimée à cinquante, il faudrait pour chaque modèle transformer les valeurs des axes des ACP, non pas en valeurs discrètes de 0 à 5, mais en valeurs continues de 0 à 5.

Outre ce problème, le choix des axes représentant les gradients de biodiversité sur les ACP n'a pas été fait de manière optimale. Pour pouvoir définir l'axe de chaque ACP qui représente au mieux le gradient de biodiversité, il faudrait mettre au point un algorithme capable de déterminer l'axe permettant à la fois de faire la meilleure distinction possible entre les plots biologiques et conventionnels, et assurer une parfaite cohérence avec le gradient de biodiversité globale. Mais comme on a pu le constater pour le cas des prairies temporaires, il est impossible, pour certains groupes d'habitats, de définir un gradient de biodiversité de l'ensemble des quatre taxons permettant de distinguer gestions biologique et conventionnelle. Donc même avec un tel algorithme, il ne sera pas toujours possible de définir cet axe recherché.

4.3 Réduction des coûts de la méthode

L'objectif de remplacer les Ibios, indicateurs directs de biodiversité basés sur les quatre taxons abeilles, araignées, vers de terre et plantes vasculaires, par des indicateurs indirects que sont les indicateurs de gestion, et par des indicateurs directs moins onéreux que sont les indicateurs d'habitat, ne peut être atteint avec les indicateurs utilisés dans nos analyses. Une fois nos modèles de prédiction des Ibios complétés par des variables spatiales et d'autres facteurs environnementaux, il faudra comparer les Ibios prédits à ceux calculés dans cette étude pour pouvoir juger de la qualité de prédiction de la biodiversité par ces modèles. Si les résultats sont concluants, la mise au point d'un indicateur de biodiversité basé sur les indicateurs environnementaux pourra alors être envisagée. On peut également se questionner sur la pertinence de chacun des quatre taxons, ainsi que sur les protocoles de recueil des données sur ces communautés.

Les plantes vasculaires, de par les niches écologiques qu'elles fournissent aux autres communautés du vivant, et les abeilles, de par leur rôle de pollinisation, constituent des indicateurs des plus pertinents dans les agroécosystèmes. Les résultats obtenus pour ces deux taxons apparaissent également pertinents pour comparer les systèmes de production. Il faut donc conserver ces deux sous-indicateurs pour notre indicateur de biodiversité. La méthode de capture des abeilles est, en revanche, à remettre en question sur deux points : (i) la mauvaise reproductibilité de la méthode de capture par chasse à vue au filet à insecte et (ii) le coût temporel. Pour pallier ces deux points, la pose de pièges à abeilles dans les différents plots serait la meilleure piste à explorer.

Les araignées, représentantes du maillon haut de la chaîne alimentaire chez les arthropodes, donnent quant à elles des résultats discutables. Suivant l'habitat, leur richesse spécifique est plus élevée en agriculture biologique ou conventionnelle, mais qu'en est-il de leur diversité ? La diversité de Shannon des araignées n'a en effet pas été utilisée dans notre mise au point, car, à la différence des taxons abeilles et vers de terre, la richesse spécifique s'est révélée plus pertinente après étude des corrélations en *3.1.1*. Ce résultat s'explique par le fait que les courbes de saturation n'atteignent pas l'asymptote correspondant à la richesse spécifique maximale présente dans les plots échantillonnés (Annexe 3). A ce grand nombre d'espèces présentes correspond un faible nombre d'individus par espèce, ce qui influe certainement sur les calculs des indices de Shannon si le nombre total d'espèces présentes sur le plot est mal estimé. Cette importante richesse spécifique en araignées, recensée dans les Vallées et Coteaux de Gascogne, constitue de plus un véritable problème financier. En effet, le temps d'identification de ces espèces par un spécialiste est proportionnelle au nombre d'espèces capturées, et les honoraires de ce spécialiste sont proportionnelles au temps qu'il consacre à cette identification. L'utilisation des araignées en tant que sous-indicateur de biodiversité peut donc être remise en question.

Enfin les vers de terre donnent des résultats pertinents et constituent la principale communauté habilitée à régénérer le sol. Leur place de sous-indicateur dans notre indicateur de biodiversité dans les agroécosystèmes est donc confirmée. Afin de réduire les coûts d'identification, une option pourrait être de se limiter à la reconnaissance des morpho-espèces, associée à l'identification et à la mesure de biomasse des groupes fonctionnels (épigé, endogé et anécique) auxquels les vers de terre appartiennent (Olivier et Beattie, 1996). La contribution d'un spécialiste pour l'identification des vers de terre ne serait alors plus nécessaire.

4.4 Conclusion

A partir des seules variables de gestion agricole et d'état de l'habitat naturel décrites dans notre étude, il est impossible de remplacer notre indicateur de biodiversité basé sur la structure des communautés d'abeilles, d'araignées, de vers de terre et de plantes vasculaires. L'étape suivante est d'évaluer si l'ajout de variables spatiales, qui rendent compte de l'influence du paysage, peuvent améliorer les modèles prédictifs. Cette opportunité permettrait de diminuer les coûts financiers et temporels, mais avec une précision diminuée de nos estimations de la biodiversité.

Toutefois, dans la perspective de mettre en place un suivi de l'évolution de la biodiversité dans les agroécosystèmes, il est possible de conduire un suivi régulier à l'aide des variables de gestion, et de réaliser des mesures de biodiversité plus espacées dans le temps pour diminuer les coûts tout en ayant des résultats pertinents.

Bibliographie

Bockstaller, C., 2008. Les indicateurs de durabilité, du choix au tableau de bord. Colloque de restitution du projet OPTA, Lyon.

Dale, V.H., Beyeler, S.C., 2001. Challenges in the development and use of ecological indicators. Ecological Indicators 1, 3-10.

Duelli, P., Obrist, M.K., 2003. Biodiversity indicators: the choice of values and measures. Agriculture, Ecosystems and Environment 98, 87-98.

Feld, C.K., Martins da Silva, P., Sousa, J.P., de Bello, F., Bugter, R., Grandin, U., Hering, D., Lavorel, S., Mountford, O., Pardo, I., Pärtel, M., Römbke, J., Sandin, L., Jones, K.B., Harrison, P., 2009. Indicators of biodiversity and ecosystem services: a synthesis across ecosystems and spatial scales. Oikos 118, 1862-1871.

Feld, C.K., Sousa, J.P., Silva, P.M.d., Dawson, T.P., 2010. Indicators for biodiversity and ecosystem services: towards an improved framework for ecosystems assessment. Biodiversity and Conservation 19, 2895-2919.

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, C., Ramankutty, N., Snyder, P.K., 2005. Global Consequences of Land Use. Science 309, 570-574.

Geiger, F., Bengtsson, J., Berendse, F., Weisser, W.W., Emmerson, M., Morales, M.B., Ceryngier, P., Liira, J., Tscharntke, T., Winqvist, C., Eggers, S., Bommarco, R., Pärt, T., Bretagnolle, V., Plantegenest, M., Clement, L.W., Dennis, C., Palmer, C., Onate, J.J., Guerrero, I., Hawro, V., Aavik, T., Thies, C., Flohre, A., Hänke, S., Fischer, C., Goedhart, P.W., Inchausti, P., 2010. Persistent negative effects of pesticides on biodiversity and biological control potential on European farmland. Basic and Applied Ecology. doi: 10.1016/j.baae.2009.12.001.

Herzon, I., Helenius, J., 2008. Agricultural drainage ditches, their biological importance and functionning. Biological Conservation 141, 1171-1183.

Hole, D.G., Perkins, A.J., Wilson, J.D., Alexander, I.H., Grice, P.V., Evans, A.D., 2005. Does organic farming benefit biodiversity ? Biological Conservation 122, 113-130.

Nascimbene, J., Marini, L., Paoletti, M.G., 2012. Organic Farming Benefits Local Plant Diversity inVineyard Farms Located in Intensive Agricultural Landscapes. Environmental Management. doi: 10.1007/s00267-012-9834-5.

Öberg, S., Ekbom, B., Bommarco, R., 2007. Influence of habitat type and surrounding landscape on spider diversity in Swedish agroecosystems. Agriculture, Ecosystems and Environment 122, 211-219.

Ockinger, E., Smith, H.G., 2007. Semi-natural grasslands as population sources for pollinating insects in agricultural landscapes. J. Appl. Ecol. 44, 50-59.

Oertli, S., Müller, A., Dorn, S., 2005. Ecological and seasonal patterns in the diversity of a species-rich bee assemblage (Hymenoptera: Apoidea: Apiformes). Eur. J. Entomol. 102, 53-63.

Olivier, I., Beattie, A.J., 1996. Invertebrate Morphospecies as Surrogates for Species: A case Study. Conservation Biology 10, 99-109.

Perrings, C., Naeem, S., Ahrestani, F.S., Bunker, D.E., Burkill, P., Canziani, G., Elmqvist, T., Fuhrman, J.A., Jaksic, F.M., Kawabata, Z., Kinzig, A., Mace, G.M., Mooney, H., Prieur-Richard, A.H., Tschirhart, J., Weisser, W., 2011. Ecosystem services, targets, and indicators for the conservation and sustainable use of biodiversity. Frontiers in Ecology and the Environment 9, 512-520.

Qi, A., Perry, J.N., Pidgeon, J.D., Haylock, L.A., Brooks, D.R., 2008. Cost-efficacy in measuring farmland biodiversity – lessons from the Farm Scale Evaluations of genetically modified herbicide-tolerant crops. Annals of Applied Biology 152, 93-101.

Sepp, K., Mikk, M., Mänd, M., Truu, J., 2004. Bumblebee communities as an indicator for landscape monitoring in the agri-environmental programme. Landscape and Urban Planning 67, 173-183.

Targetti, S., Viaggi, D., Cuming, D., Sarthou, J.P., Choisis, J.P., 2011. Towards a cost-effectiveness analysis of the measurement of biodiversity indicators. 122nd Seminar of the European Association of Agricultural Economists. Ancona, Italy.

Turner, B.L., Lambin, E.F., Reenberg, A., 2007. The emergence of land change science for global environmental change and sustainability. PNAS (Proc. Natl. Acad. Sci.) 104, 20666-20671.

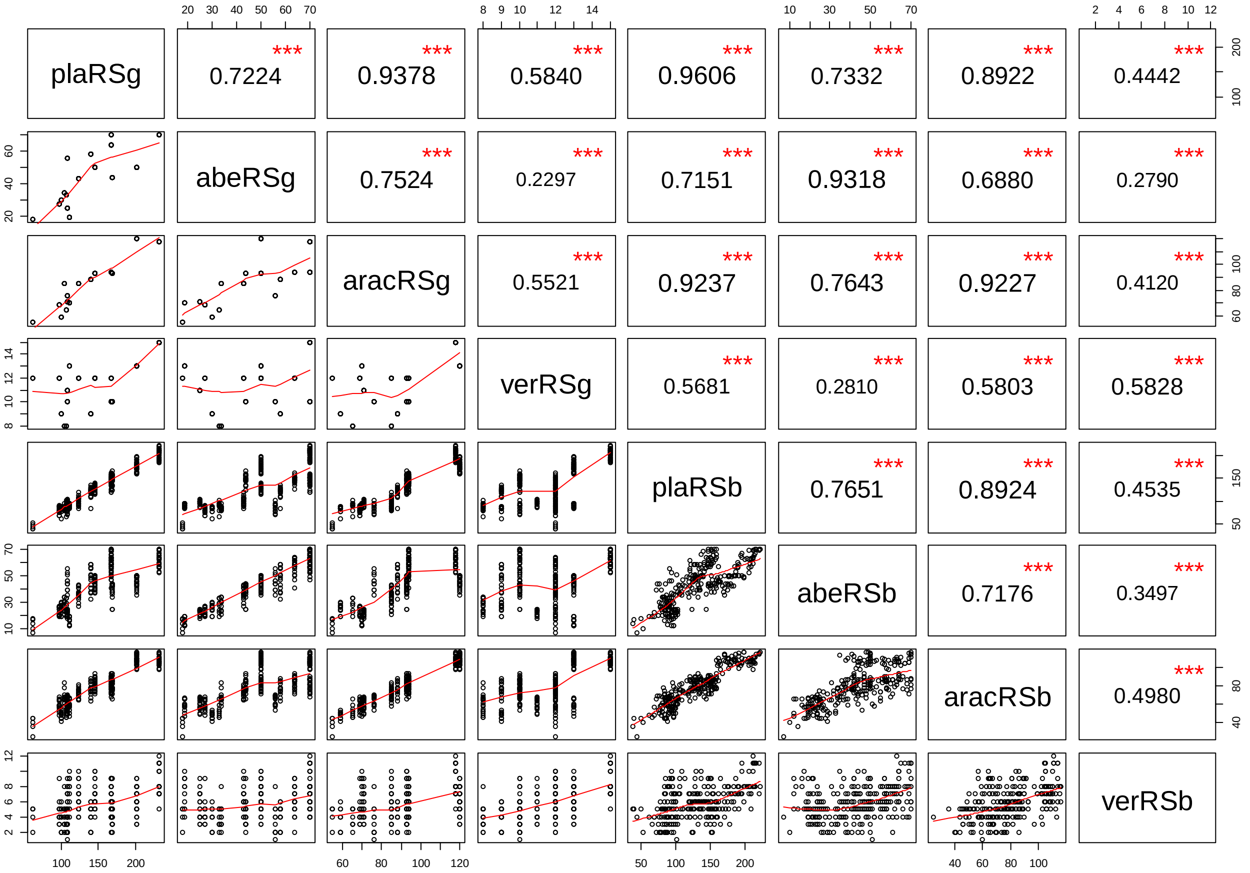
UNEP/CBD/COP7, 2003. Implementation of the strategical plan: evaluation of progress towards the 2010 biodiversity target: development of specific targets, indicators and a reporting framework. 7th Meet. Subsidiary Body on Scientific, Technical and Technological Advice to the Convention on Biological Diversity, Kuala Lumpur.

Winqvist, C., Bengtsson, J., Aavik, T., Berendse, F., Clement, L.W., Eggers, S., Fischer, C., Flohre, A., Geiger, F., Liira, J., Pärt, T., Thies, C., Tscharntke, T., Weisser, W.W., Bommarco, R., 2011. Mixed effects of organic farming and landscape complexity on farmland biodiversity and biological control potential across Europe. J. Appl. Ecol. 48, 570-579.

Zurbuchen, A., Cheesman, S., Klaiber, J., Müller, A., Hein, S., Dorn, S., 2010. Long foraging distances impose high costs on offspring production in solitary bees. Journal of Animal Ecology 79, 674-681.

Annexes

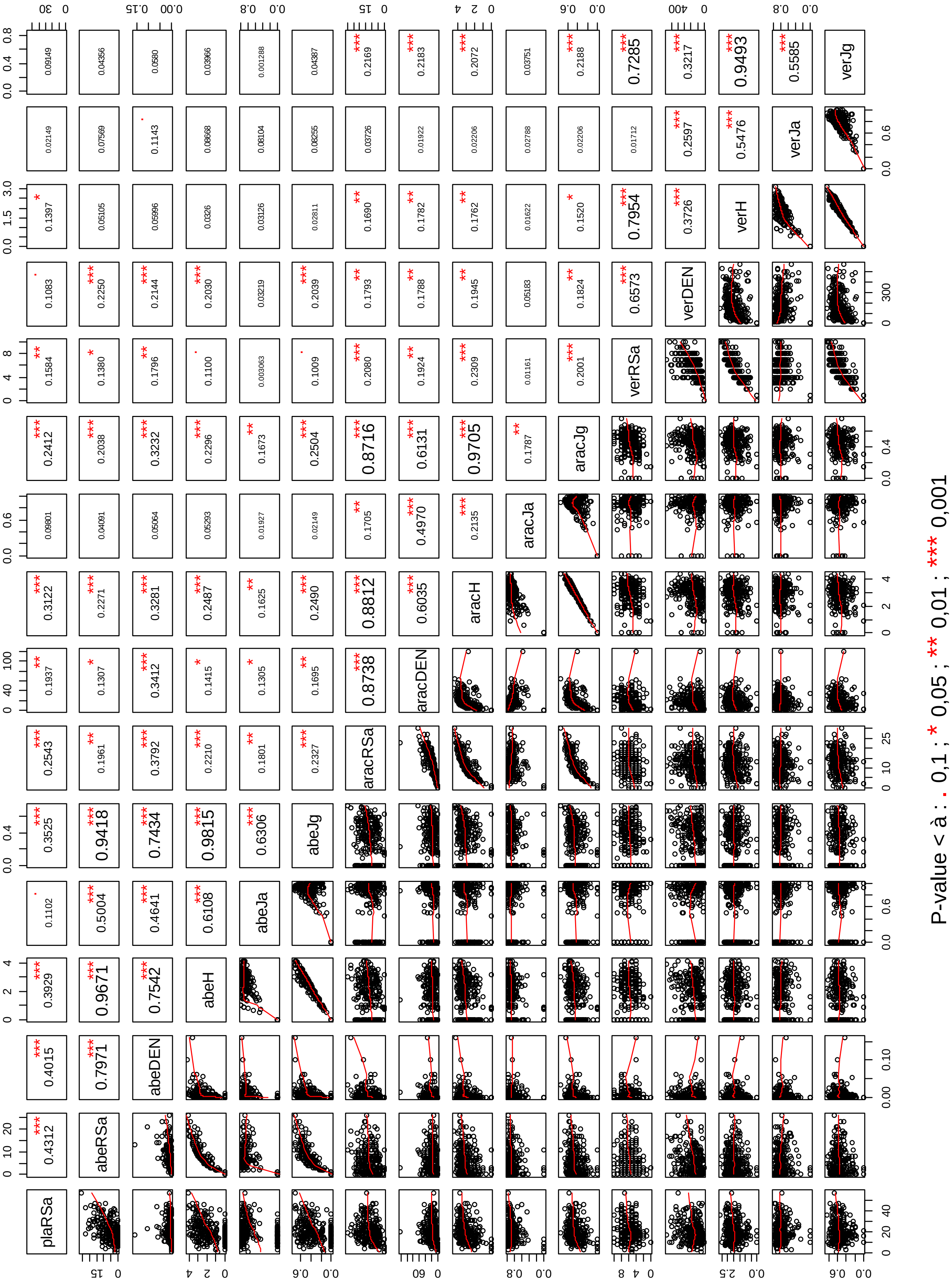
Annexe 1 : Corrélogramme des Richesses spécifiques gamma et beta



P-value < à : . 0,1 ; * 0,05 ; ** 0,01 ; *** 0,001

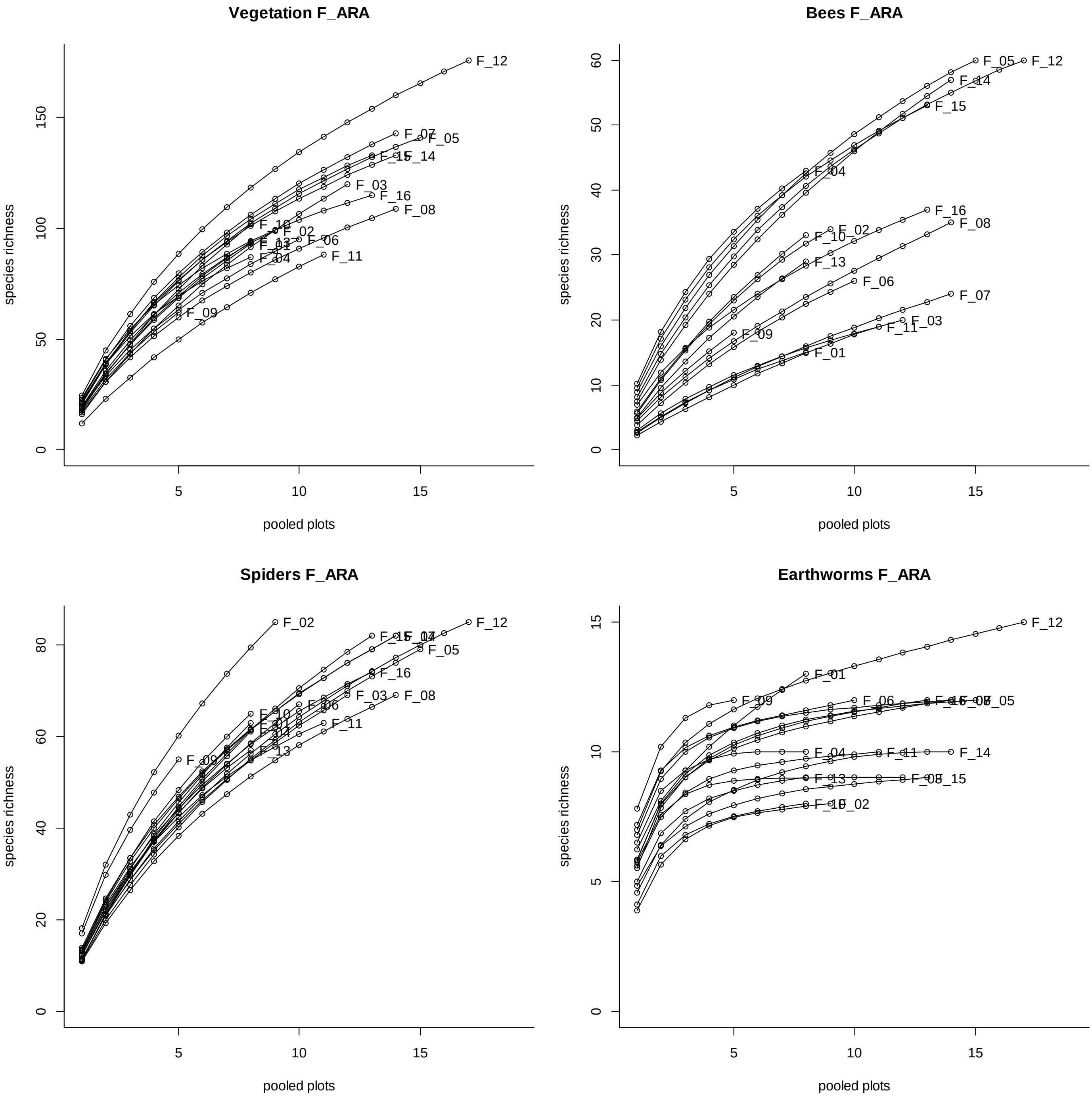
pla=plantes, abe=abeilles, arac=araignées, ver=vers de terre, RSg=richesse spécifique gamma, RSb=richesse spécifique beta. Les coefficients de corrélation ont été calculés selon la corrélation de Spearman.

Annexe 2 : Corrélogramme des variables de richesse spécifique alpha, densité, indice de shannon et indices de Piélou alpha et gamma.



pla=plantes, abe=abeilles, arac=araignées, ver=vers de terre, RSa=richesse spécifique alpha, DEN=densité (nb/m²), H=indice de Shannon, Ja=indice de Piélou alpha, Jg=indice de Piélou gamma. Les coefficients de corrélation ont été calculés selon la corrélation de Spearman.

Annexe 3 : Courbes d’accumulation des espèces d’araignées dans les seize EA du cas d’étude français du projet européen BioBio. Source : P. Jeanneret, BioBio Project 2012.

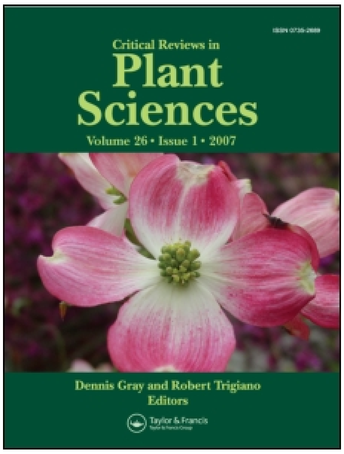


Résumé

Les changements d’utilisation du sol par l’Homme, notamment l’agriculture et l’urbanisme, ont conduit à une érosion de la biodiversité. Pour pouvoir évaluer et suivre les tendances d’évolution de la biodiversité dans les agroécosystèmes, le développement d’indicateurs fiables et pertinents s’avère nécessaire. Dans le cadre du projet européen BioBio, un indicateur de biodiversité a été mis au point pour rendre compte de l’état de la biodiversité dans les exploitations agricoles (EA) des Vallées et Coteaux de Gascogne. Composé de quatre sous-indicateurs taxonomiques, que sont des indices de richesse et de diversité spécifiques en plantes vasculaires, abeilles sauvages, araignées et vers de terre, cet indicateur a été élaboré en se basant sur la différence de biodiversité existante entre les EA biologiques et conventionnelles. Mais l’utilisation de sous-indicateurs taxonomiques a un coût, à la fois temporel et financier, qu’il faut minimiser pour promouvoir la mise en pratique d’un tel indicateur. A travers l’étude des liens entre espèces et environnement agricole, une éventuelle substitution des sous-indicateurs taxonomiques par des sous-indicateurs environnementaux, moins onéreux, a été explorée.

Abstract

Land use changes caused by Human, especially agriculture and urbanization, have led to a biodiversity loss. In order to evaluate and monitor evolution trends of biodiversity in agroecosystems, it is necessary to develop reliable and relevant indicators. Within the framework of the European BioBio Project, a biodiversity indicator has been developed to report on biodiversity status in farms located in Gascony Hills. This indicator is composed of four species sub-indicators, which are species richness and diversity of vascular plants, wild bees, spiders and earthworms, and it was constructed based on the biodiversity difference established between organic and conventional farms. However, the use of species sub-indicators entails temporal and economic costs that should be minimized to promote the practical implementation of such an indicator. Through the study of relationships between species and agricultural environment, the potential replacement of species sub-indicators by cheaper environmental sub-indicators has been explored.



Critical Reviews in Plant Sciences
Publication details, including instructions for authors and subscription information:
http://www.informaworld.com/smpp/title~content=t713400911

Environmental Impact of Different Agricultural Management Practices: Conventional vs. Organic Agriculture

Tiziano Gomiero[a]; David Pimentel[b]; Maurizio G. Paoletti[a]
[a] Laboratory of Agroecology and Ethnobiology, Department of Biology, Padova University, Padova, Italy [b] College of Agriculture and Life Sciences, Cornell University, Ithaca, New York, USA

Online publication date: 29 April 2011

To cite this Article Gomiero, Tiziano , Pimentel, David and Paoletti, Maurizio G.(2011) 'Environmental Impact of Different Agricultural Management Practices: Conventional vs. Organic Agriculture', Critical Reviews in Plant Sciences, 30: 1, 95 — 124

To link to this Article: DOI: 10.1080/07352689.2011.554355

URL: http://dx.doi.org/10.1080/07352689.2011.554355

Critical Reviews in Plant Sciences, 30:95–124, 2011

ISSN: 0735-2689 print / 1549-7836 online
DOI: 10.1080/07352689.2011.554355

Taylor & Francis
Taylor & Francis Group

Environmental Impact of Different Agricultural Management Practices: Conventional vs. Organic Agriculture

Tiziano Gomiero,[1] David Pimentel,[2] and Maurizio G. Paoletti[1]
[1]*Laboratory of Agroecology and Ethnobiology, Department of Biology, Padova University, Padova, 35121 Italy*
[2]*College of Agriculture and Life Sciences, Cornell University, Ithaca, New York 14853, USA*

Table of Contents

I. ORGANIC AGRICULTURE: AN INTRODUCTION 96
A. Organic Principles 96
B. Origins and Present Situation 97
C. Organic Standards 99

II. SOME ISSUES CONCERNING COMPARATIVE ANALYSIS 99

III. SOIL BIOPHYSICAL AND ECOLOGICAL CHARACTERISTICS 100
A. Soil Erosion and Soil Organic Matter 100
B. Soil Chemical Properties 101
C. Nitrogen Leaching 102
D. Water Use and Resistance to Drought 103
E. The Potential for Organically Managed Farming Systems to Operate as a Carbon Sink and Contribute to GHGs Reduction 104
F. Soil Ecology, Biodiversity, and Its Effects on Pest Control 104

IV. BIODIVERSITY 105
A. Organic Farming and Biodiversity 106
B. Biodiversity and Landscape 107
C. Biodiversity and Pest Control 108

V. ENERGY USE AND GHGs EMISSION 109
A. Energy Efficiency 109
B. GHGs Emission 110
C. Integrating Animal Husbandry 111

VI. CONSTRAINTS TO THE ADOPTION OF ORGANIC AGRICULTURE 113
A. Feasibility 113
B. Labor Productivity 113
C. Economic Performance 114
D. Environmental Services of Organic Agriculture 114
E. Organic Farming and Food Security 114
F. "Food Miles" Analysis 115

VII. CONCLUSIONS 115

Address correspondence to Tiziano Gomiero, Laboratory of Agroecology and Ethnobiology, Department of Biology, Padova University, Padova, 35121 Italy. E-mail: tiziano.gomiero@libero.it
Referee: Dr. Nadia El-Hage Scialabba, Natural Resources Management and Environment Department FAO, Rome.

ACKNOWLEDGMENTS 117

REFERENCES 117

Organic agriculture refers to a farming system that enhance soil fertility through maximizing the efficient use of local resources, while foregoing the use of agrochemicals, the use of Genetic Modified Organisms (GMO), as well as that of many synthetic compounds used as food additives. Organic agriculture relies on a number of farming practices based on ecological cycles, and aims at minimizing the environmental impact of the food industry, preserving the long term sustainability of soil and reducing to a minimum the use of non renewable resources. This paper carries out a comparative review of the environmental performances of organic agriculture versus conventional farming, and also discusses the difficulties inherent in this comparison process. The paper first provides an historical background on organic agriculture and briefly reports on some key socioeconomic issues concerning organic farming. It then focuses on how agricultural practices affect soil characteristics: under organic management soil loss is greatly reduced and soil organic matter (SOM) content increases. Soil biochemical and ecological characteristics appear also improved. Furthermore, organically managed soils have a much higher water holding capacity than conventionally managed soils, resulting in much larger yields compared to conventional farming, under conditions of water scarcity. Because of its higher ability to store carbon in the soil, organic agriculture could represent a means to improve CO_2 abatement if adopted on a large scale. Next, the impact on biodiversity is highlighted: organic farming systems generally harbor a larger floral and faunal biodiversity than conventional systems, although when properly managed also the latter can improve biodiversity. Importantly, the landscape surrounding farmed land also appears to have the potential to enhance biodiversity in agricultural areas. The paper then outlines energy use in different agricultural settings: organic agriculture has higher energy efficiency (input/output) but, on average, exhibits lower yields and hence reduced productivity. Nevertheless, overall, organic agriculture appears to perform better than conventional farming, and provides also other important environmental advantages, such as halting the use of harmful chemicals and their spread in the environment and along the trophic chain, and reducing water use. Looking at the future of organic farming, based on the findings presented in this review, there is clearly a need for more research and investment directed to exploring potential of organic farming for reducing the environmental impact of agricultural practices; however, the implications of reduced productivity for the socioeconomic system should also be considered and suitable agricultural policies should be developed.

Keywords organic agriculture, conventional agriculture, sustainability, energy use, GHGs emissions, soil organic matter, carbon sink, biodiversity

I. ORGANIC AGRICULTURE: AN INTRODUCTION

Organic agriculture refers to a farming system that bans the use of agrochemicals such as synthetic fertilizers and pesticides and the use of Genetically Modified Organisms (GMO), as well as many synthetic compounds used as food additives (e.g., preservatives, coloring) (IFOAM, 2008; 2010). Organic agriculture is regulated by international and national institutional bodies, which certify organic products from production to handling and processing (Codex Alimentarius, 2004; Courville, 2006; EC, 2007; USDA, 2007; IFOAM, 2008; 2010). Its origins can be traced back to the 1920–1930 period in North Europe (mostly Germany and UK) (Conford, 2001; Lotter, 2003; Lockeretz, 2007), and it is now widely spread all over the world.

In this paper we will briefly present the history of organic agriculture and introduce the key characteristics of organic practices and principles. The focus of the paper is, then, to review the main literature on the comparison between organic and conventional agriculture concerning their environmental performances. Some socioeconomic issues will also be addressed.

We are aware that conventional agriculture can adopt low input, environmentally friendly approaches to management (as in systems with reduced or no tillage, or integrated pest management farming). However, the very fact that organic agriculture is strictly regulated allows better comparison of the performances of farming systems with and without agrochemical inputs, and with or without the adoption of certain management practices. The main difficulty in comparisons is the blur definition of conventional practices, which range from traditional polycultures to highly industrial monocultures.

We wish to point out that in the review of the literature we found a number of studies published in gray literature (reports, conference proceedings, etc.) in local/national languages, which are then difficult to both reach and read. In this review we choose to reduce to a minimum the references to gray literature because of the difficulty for the reader to find and check the original works.

A. Organic Principles

The International Federation of Organic Agriculture Movements IFOAM, a grassroots international organization born in 1972, that today includes 750 member organizations belonging to 108 countries, for details see http://www.ifoam.org/index.html), states that: "Organic agriculture is a production system that sustains the health of soils, ecosystems and people. It relies on ecological processes, biodiversity and cycles adapted to local conditions, rather than the use of inputs with adverse effects. Organic agriculture combines tradition, innovation and science to benefit the shared environment and promote fair relationships and a good quality of life for all involved." (IFOAM, 2010).

The USDA National Organic Standards Board (NOSB) defines organic agriculture as follows: "Organic agriculture is an ecological production management system that promotes and enhances biodiversity, biological cycles and soil biological activity. It is based on minimal use of off-farm inputs and on management practices that restore, maintain and enhance ecological harmony." (Gold, 2007).

Organic agriculture relies on a number of farming practices that take full advantage of ecological cycles. In organic farming systems soil fertility is enhanced by crop rotation, intercropping, polyculture, covering crops and mulching. Pest control is achieved by using appropriate cropping techniques, biological control, and natural pesticides (mainly extracted from plants). Weed control, in many cases the main focal problem for organic farming, is managed by appropriate rotation, seeding timing, mechanic cultivation, mulching, transplanting, flaming, etc. (Howard, 1943; Altieri, 1987; Lampkin, 2002; Lotter, 2003; Altieri and Nichols, 2004; Koepf, 2006; Kristiansen *et al.*, 2006; Gliessman, 2007). As with any manipulation of a natural ecosystem, biological control must adopt a cautionary approach when introducing novel organisms to fight pests. Cases have been reported where introduced ally insects turned out to cause more harm than those they were supposed to fight (Simberloff and Stiling, 1996; Hamilton, 2000).

According to IFOAM, organic agriculture should be guided by four principles:

- *health*: organic agriculture should sustain and enhance the health of soil, plant, animal, human and planet as one and indivisible,
- *ecology*: organic agriculture should be based on living ecological systems and cycles, increased soil organic matter, work with them, emulate them and help sustain them,
- *fairness*: organic agriculture should build on relationships that ensure fairness with regard to the common environment and life opportunities,
- *care*: organic agriculture should be managed in a precautionary and responsible manner to protect the health and well-being of current and future generations and the environment.

IFOAM argues that organic agriculture is a holistic production management system which promotes and enhances agroecosystem health, including biodiversity, biological cycles, and soil biological activity. An organic production system is, then, designed to:

- enhance biological diversity within the whole system,
- increase soil biological activity,
- maintain long-term soil fertility,
- recycle plant and animal waste in order to return nutrients to the land, thus minimizing the use of nonrenewable resources,
- rely on renewable resources in locally organized agricultural systems,
- promote the healthy use of soil, water and air as well as minimize all forms of pollution that may result from agricultural practices,
- handle agricultural products with emphasis on careful processing methods in order to maintain the organic integrity and vital qualities of the product at all stages,
- become established on any existing farm through a period of conversion, the appropriate length of which is determined by site-specific factors such as the history of the land, and type of crops and livestock to be produced.

The organic philosophy aims at preserving the natural environment; concern towards local floras and fauna as goals for organic farming are often little understood by consumers and policy makers.

As stated by FAO (2004, p. iii): "Evidence suggests that organic agriculture and sustainable forest management not only produce commodities but build self-generating food systems and connectedness between protected areas. The widespread expansion of these approaches, along with their integration in landscape planning, would be a cost efficient policy option for biodiversity."

Concerning environmental performances, some authors warn that organic practices may not be applicable without considering the specific situation. Wu and Sardo (2010) list a number of examples in which the effects of agricultural techniques employed in organic agriculture could result in worse environmental impacts than conventional practices. The authors, for instance, argue that, on sloping land, environmental damages from erosion due to mechanical weed control can be more harmful than that from chemical origin, e.g., spraying with glyphosate [results from Teasdale *et al.* (2007), for organic farming on 15% slope, indicate that if properly managed and in proper condition, organic farming can still provide benefits for soil]. In addition, Wu and Sardo (2010) suggest that mulching with polyethylene sheets (permitted in organic farming) is more polluting than spraying glyphosate, and that flame weeders (permitted in organic farming) are more costly and energy demanding than glyphosate and much less efficient in the control of perennial weeds. It is to be noted that the evaluation of one practice ought to be contextualized, with the consideration of a range of factors that determine good or bad management of a landscape as a whole. For example, mechanical slope weeding on its own may be detrimental while if considered within the farm architecture, its local impact may be compensated with features such as hedges and perennials that ensure overall soil resilience.

Some authors (e.g., Guthman, 2004) argue that as organic farmers enter large distribution system they may be forced to shift once again into monoculture and industrial agriculture. That is because of the pressure from agrifood corporations that buy and distribute their organic products, and from the market itself.

B. Origins and Present Situation

In order to help the reader to better understand the foundation of organic farming, it may be useful to provide a brief sketch of the history of the organic agriculture movement. For details on this topic we will refer the reader to the extensive works of Conford (2001) and Lockeretz (2007) or, for a more concise summary, to Lotter (2003), Kristiansen (2006), Heckman (2006), and Gold and Gates (2007). Historical information can also be found at the website of the main organic associations such as the British "Soil Association" (http://www.soilassociation.org), or the international IFOAM (http://www.ifoam.org).

The first organized movement by alternative farmers, who wanted to adhere to the traditional way of production refusing the new chemical inputs, appeared in Germany at the end of 1920s. Some tens of farmers, agronomists, doctors and lay people grouped together after attending the lectures of the Austrian philosopher and scientist Rudolf Steiner (who developed also Anthroposophy), in 1924. The experimental circle of anthroposophical farmers immediately tested Steiner's indications in daily farming practice. Three years later a cooperative was formed to market biodynamic products forming the association Demeter (for details see Demeter web page at http://www.demeter.net). In 1928 the first standards for Demeter quality control were formulated. Biodynamic agriculture, as this method is named, is well grounded in the practical aspects of manuring the soil, which is the cornerstone of organic farming, but it also concerns lunar and astrological scheduling, communication with "nature spirits" and the use of special potencies or preparations, that are derived by what might be described as alchemical means (Koepf, 1976; 2006; Conford, 2001). These latter practices are not easily "measurable" in scientific terms, but performance can be assessed using usual agronomic indicators.

While Rudolf Steiner was establishing the roots for the growth of the biodynamic movement, Sir Albert Howard (1873–1947), a British agronomist based in India, was trying to develop a coherent and scientifically based system for preserving soil and crop health. Upon his return to the UK, he worked to promote his new approach (Howard, 1943; Conford, 2001). He was convinced that most agricultural problems were due to soil mismanagement, and that reliance on chemical fertilization could not solve problems such as loss of soil fertility and pest management. He maintained that the new agrochemical approach was misguided, and that it was a product of reductionism by "laboratory hermits" who paid no attention to how nature worked. In his milestone book, *An Agricultural Testament* (1943), Howard described a concept that was to become central to organic farming: "*the Law of Return*" (a concept expressed also by Steiner). The Law of Return states the importance of recycling all organic waste materials, including sewage sludge, back to farmland to maintain soil fertility and the land humus content (Howard 1943; Conford, 2001).

The first use of the word *organic* has been ascribed to Walter Northbourne, the author of *Look to the Land,* an influential book published in 1940 in the UK. Within it, he elaborates on the notion of a farm as an "organic whole," where farming has to be performed as a biologically complete process (Conford, 2001). The term "organic" then, in its original sense, describes a holistic approach to farming: fostering diversity, maintaining optimal plant and animal health, and recycling nutrients through complementary biological interactions.

In 1943 in the UK, Lady Eve Balfour (1899–1990) published the book *The Living Soil*, in which she described the direct connection between farming practice and plant, animal, human and environmental health. The book exerted a significant influence on public opinion, leading in 1946 to the foundation in the UK of "The Soil Association" by a group of farmers, scientists and nutritionists. In the following years, the organisation also developed organic standards and its own certification body. Eve Balfour, who was one of IFOAM's founders, claimed that: "The criteria for a sustainable agriculture can be summed up in one word—permanence, which means adopting techniques that maintain soil fertility indefinitely, that utilise, as far as possible, only renewable resources; to avoid those that grossly pollute the environment; and that foster biological activity throughout the cycles of all the involved food chains" (Balfur, 1977).

In 1940, in an article published in *Fact Digest*, Jerome I. Rodale introduced the term "organic agriculture" in the United States and techniques such as crop rotation and mulching, that have, since then, become accepted organic practices in the United States. Although, the idea of organic agriculture came mostly from the work of Albert Howard. However, Rodale expanded Howard's ideas in his book *Pay Dirt* (Rodale, 1945), adding a number of other "good farming practices."

Since 1990, with increased public concern for the environment and food quality, the organic farming movement has gained the attention of consumers and has undergone national and international institutional regulation (Willer and Yussefi, 2006). According to the recent data by IFAOM (Willer, 2011) there are 37.2 million hectares of organic agricultural land (including in-conversion areas). The regions with the largest areas of organic agricultural land are Oceania (12.2 million hectares—32.8%), Europe (9.3 million hectares—25%), and Latin America (8.6 million hectares—23.1%). The countries with the most organic agricultural land are Australia, Argentina, and the United States. It should be noted that it is difficult to compare figures coming from different countries: most of the area in Australia is pastoral land used for low intensity grazing, therefore one organic hectare in Australia is not directly equivalent (e.g., does not have the same productivity) to one organic hectare in a European country.

In the United States, in 2005, for the first time all 50 states had some certified organic farmland. In 2005, U.S. producers dedicated over 1.6 million ha of farmland to organic production systems: 690,000 ha of cropland and 910,000 ha of rangeland and pasture. California remains the leading State in certified organic cropland, with over 89,000 ha, mostly for fruit and vegetable production (Gold, 2007).

According to the data collected from Willer and Yussefi (2006), the main land uses in organic farming worldwide, as a percentage of the total global organic area, are as follows:

- 5% permanent crops: land cultivated with crops that do not need to be replanted after each harvest, such as cocoa, coffee; this category includes flowering shrubs, fruit trees, nut trees and vines, but excludes trees grown for wood or timber,
- 13% arable land: land used for temporary crops, temporary meadows for mowing or pasture, market and kitchen gardens and land temporarily fallow (less than five years).
- 30% permanent pasture: land used permanently (five years or more) for herbaceous forage crops, either cultivated or growing wild (wild prairie or grazing land),
- 52% certified land the use of which is not known but where wild products are harvested.

C. Organic Standards

Organic farming aims at providing farmers with an income while at the same time protecting soil fertility (e.g., by crops rotation, intercropping, polyculture, cover crops, mulching) and preserving biodiversity (even if tending the local flora and fauna as a goal for organic farming is often little understood by consumers and policy makers), the environment and human health. Broader ethical considerations regarding the above aims have also been made (Halberg *et al*., 2006; IFOAM, 2008).

In Europe, the first regulation on organic farming was drawn up in 1991 (Regulation EEC N° 2092/91 – EEC, 1991). Organic standards prohibit the use of synthetic pesticides and artificial fertilizers, the use of growth hormones and antibiotics in livestock production (a minimum usage of antibiotics is admitted in very specific cases and is strictly regulated). Genetically modified organisms (GMOs) and products derived from GMOs are explicitly excluded from organic production methods.

A revised EU regulation which came into force in 2007 (EC, 2007) added two main new criteria: firstly, food will only be able to carry an organic logo (certified as organic) if at least 95% of the ingredients are organic (nonorganic products will be entitled to indicate organic ingredients on the ingredients list only); secondly, although the use of GMOs will remain prohibited, a limit of 0.9 percent will be allowed as accidental presence of authorised GMOs.

In the United States, Congress passed the Organic Foods Production Act (OFPA) in 1990. The OFPA required the U.S. Department of Agriculture (USDA) to develop national standards for organically produced agricultural products, to assure consumers that agricultural products marketed as organic meet consistent, uniform standards. The OFPA and the National Organic Program (NOP) regulations require that agricultural products labelled as organic originate from farms or handling operations certified by a state or private entity that has been accredited by USDA (Gold, 2007).

Internationally, organic agriculture has been officially recognised by the Codex Alimentarius Commission (CAC).[1] In 1991, the CAC began elaborating guidelines for the production, processing, labelling and marketing of organically produced food, with the participation of observer organizations such as IFOAM and the EU. The CAC approved organic plant production in June 1999, followed by organic animal production in July 2001. The requirements in these CAC Guidelines are in line with IFOAM Basic Standards and the EU Regulation for Organic Food (EU Regulations 2092/91 and1804/99). There are, however, some differences with regard to the details and the areas, which are covered by the different standards.

In the Guidelines for the Production, Processing, Labelling and Marketing of Organically Produced Foods, CAC at point 5 states that: "Organic Agriculture is one among the broad spectrum of methodologies which are supportive of the environment. Organic production systems are based on specific and precise standards of production which aim at achieving optimal agroecosystems which are socially, ecologically and economically sustainable." (Codex Alimentarius, 2004, p. 4).

Some authors (e.g., Vogl *et al*., 2005; Courville, 2006) express concerns about the excessive bureaucratic control posed by standards on farmers, and warns that excessive bureaucratization of organic agriculture can result a serious burden to organic farmers because of the economic effort that it takes to accomplish with all the requirements.

II. SOME ISSUES CONCERNING COMPARATIVE ANALYSIS

Often, different approaches to farming system analysis are employed by different scholars, making comparison of findings difficult: this is especially true with regards to how the boundaries of the farming system are defined. For instance, in accounting for the energy in animal feed or agrochemicals, should we consider the energy spent for transportation? In a time of fast globalization where commodities travel from continent to continent such a question is not a negligible one.

Moreover, farming system may have different geographical, climatic and soil characteristics, different crops, different rotation systems (both in crop species and timing) and different sort of inputs.

Comparative studies tend to focus on specific crops, over a short period of time. Simplifying the focus of the farming system analysis, through single commodity versus whole farm productivity analysis, entails the risk of compromising the understanding of its complex reality and supplying incomplete

[1]The Codex Alimentarius Commission was created in 1963 by FAO and WHO to develop food standards, guidelines and related texts such as codes of practice under the Joint FAO/WHO Food Standards Program. The main purposes of this Program is protecting consumer health, ensuring fair trade practices in the food trade, and promoting coordination of all food standards work undertaken by international governmental and non-governmental organizations. (Codex Alimentarius web page at http://www.codexalimentarius.net/web/index_en.jsp)

information. Longer-term studies (e.g. a minimum of 10 years) should be encouraged to gather information—through comparable models—about the true sustainability of different farming systems.

Energy analysis in agriculture is a complex task (Fluck and Baird, 1980; Giampietro *et al*., 1992; Pimentel and Pimentel, 2008; Wood *et al*., 2006; Smil, 2008). Usually energy analysis focuses on fertilizers, pesticides, irrigation and machinery but fails to include important components such as insurance, financial services, repairs and maintenance, veterinary and other services (Fluck and Baird, 1980). Energy efficiency assessment presents many tricky issues (Giampietro *et al*., 1992; Giampierto, 2004; Smil, 2008), and the choice of the system boundary can account for differences as large as 50% on energy estimates among studies (Suh *et al*., 2004; Wood *et al*., 2006), and even higher when coming to the assessment of the whole agri-food system (Giampietro, 2004). Comparing organic and conventional systems is even more difficult (Dalgaard *et al.*, 2001; Haas *et al*., 2001; Pimentel *et al.*, 2005; Küstermann *et al.,* 2008; Thomassen *et al*., 2008; Wu and Sardo, 2010).

Wood *et al*. (2006), for instance, when studying a cohort of organic farmers in Australia, found that when direct energy use, energy related emissions, and greenhouse gas emissions are measured they are higher for the organic farming sample than for a comparable conventional farm sample. But when the whole Life-Cycle Assessment was considered, including the indirect contributions of all above-mentioned secondary factors, then conventional farming practices had a higher energy cost. The authors argue that indirect effects must be taken into account when considering the environmental consequences of farming, in particular with regards to energy use and greenhouse gas emissions. In a comprehensive Life-Cycle Assessment of milk production in The Netherlands, Thomassen *et al.* (2008) compared energy consumption (MJ kg^{-1} of milk) for conventional and organic milk (see also Table 5a and Table 5b). They found that when comparing direct energy consumption conventional performed much better (0.6 MJ kg^{-1} of milk) then organic (0.96 MJ kg^{-1} of milk). But when indirect costs were taken into account, the result was the opposite (conventional 4.47 MJ kg^{-1} of milk and organic 2.17 MJ kg^{-1} of milk). See also Küstermann *et al*. (2008) in section VB for another example concerning GHGs emissions.

Comparing efficiency may not be that simple also within the same experiment. For instance, Gelfand *et al.* (2010) report that an alfalfa growing organic system was half as efficient compared to a conventional system when employing tillage, and had one third of a conventional system efficiency when there was no tillage. But the fact that the authors accounted all the grain (included corn, and soybean) as used directly for human consumption, while alfalfa were not (of course) can be questioned. And, in fact, as the authors correctly argue (Gelfand *et al.,* 2010, p. 4009-4010): "This is because under the Food scenario alfalfa biomass can be used only as ruminant livestock feed and conversion efficiency of forage energy to weight gain by livestock is 9:1. Were we to assume that corn, soybean, and wheat were to be used for livestock production rather than direct human consumption, similar energy conversion efficiencies by livestock would apply. This would result in about 87% lower energy output from the grain systems, similar to Alfalfa energy yields."

This is an important consideration to keep in mind because in an organic farming system the value of a crop has to be understood within a whole cropping system that can span several years. On the contrary, conventional farming can be based on a simple system that alternates corn and soybean on a yearly basis.

To carry on extensive long-term trials for a number of crops in several different geographical areas would be of fundamental importance to understand the potential of organic farming as well as to improve farming techniques in general (Mäder *et al.*, 2002; Pimentel *et al*., 2005; Gomiero *et al*., 2008; Francis *et al*., this issue).

When comparing organic vs. conventional system "farm-to-fork" we should also be aware that a possible disadvantage of organic products is the fact that they account for less than 2% of global food retail: this smaller economic scale compared to conventional systems could contribute to lower energy efficiency of collection, preparation and distribution (El-Hage Scialabba and Müller-Lindenlauf, 2010).

III. SOIL BIOPHYSICAL AND ECOLOGICAL CHARACTERISTICS

In this section we will review the effects of organic agriculture on soil biophysical and ecological characteristics and how these effects relate to the long-term soil fertility. Attempts to develop a soil quality index can provide an effective framework for evaluating the overall effects of different production practices (organic, integrated, conventional etc.) on soil quality (Glover *et al*., 2000; Mäder *et al*., 2002a; Marinari *et al.*, 2006; Fließbach *et al*., 2007).

A. Soil Erosion and Soil Organic Matter

Soil erosion and loss of Soil Organic Matter (SOM) with the conversion of natural ecosystems to permanent agriculture are the most important and intensively studied and documented consequences of agriculture (Hillel, 1991; Pimentel *et al*., 1995; Lal, 2004, 2010; Montgomery, 2007a; 2007b; Quinton *et al.*, 2010). Intensive farming exacerbates these phenomena, which are threatening the future sustainability of crop production on a global scale, especially under extreme climatic events such as droughts (Reganold *et al*., 1987; Pimentel *et al*., 1995; Mäder *et al.*, 2002a; Sullivan, 2002; Lotter *et al*., 2003; Montgomery, 2007a; 2007b; Lal, 2010; NRC, 2010).

Clark *et al*. (1998) underlined that increases in SOM following the transition to organic management occur slowly, generally taking several years to detect. This is a very important point to be kept in mind when assessing the performances of farming systems under different management practices. Farmers, scientists and policy makers alike should take into consideration the evolving and complex nature of organic farming systems, a complex nature that contrasts with the extreme simplification and large dependency on external input that characterize conventional farming systems. When aiming at long-term sustainability, trade offs should also be considered between obtaining short-term high yields with the aid of agrochemicals, and maintaining soil health.

Given the crucial importance of soil health, the aim of organic agriculture is to augment ecological processes that foster plant nutrition yet conserve soil and water resources. Even if the soil characteristics are generally site-specific, to date many studies have proven organic farming to perform better in preserving or improving soil quality with regards to both biophysical (e.g., SOM) and biological (e.g., biodiversity) properties (e.g., Reganold *et al*., 1987; Reganold, 1995; Clark *et al*., 1998; Drinkwater *et al.*,1998; Siegrist *et al*., 1998; Fließbach *et al*., 2000; 2007; Glover *et al*., 2000; Stölze *et al*., 2000; Stockdale *et al*., 2001; Mäder *et al*., 2002a; Lotter *et al*., 2003; Delate and Cambardella, 2004; Pimentel *et al*., 2005; Kasperczyk and Knickel, 2006; Marriott and Wander, 2006; Briar e*t al*., 2007; Liu *et al*., 2007).

Although few in number, important long-term studies concerning SOM content and soil characteristics in organic and conventional soils have been carried out, both in the United States and Europe. In a long trial of nearly 40 years, Reganold *et al*. (1987) compared soils from organic and conventional farms in Washington, USA. They found that organic fields had surface horizon 3 cm thicker and topsoil 16 cm deeper than conventionally managed fields. Higher SOM matter content (along with other better biochemical performance indicators) resulted in much reduced soil erosion. In addition, soils under organic management showed $<75\%$ soil loss compared to the maximum tolerance value in the region (the maximum rate of soil erosion that can occur without compromising long-term crop productivity or environmental quality -11.2 t ha^{-1} yr^{-1}), while in conventional soil a rate of soil loss three times the maximum tolerance value was recorded.

As a result of the Rodale Institute Farming System Trial, Pimentel *et al*. (2005) reported that after 22 years the increase of SOM was significantly higher in both organic animal and organic legumes systems, where soil carbon increase by 27.9% and 15.1% respectively, when compared to the conventional system, where the increase was 8.6%. Moreover, soil Carbon (C) level was 2.5% in organic animal, 2.4% in organic legume and 2.0% in the conventional system.

In a 12-year trial in Maryland, Teasdale *et al*. (2007) found that organic farming can provide greater long-term soil benefits than conventional farming with no tillage, despite the use of tillage in organic management. A drawback of the organic system was the difficulty in controlling weeds, explained by the authors by a number of factors such as short crop rotation and remaining crop residues (Teasdale *et al*., 2007; Cavigelli *et al*., 2008). However, the authors argue that despite poor weed control, the organic systems improved soil productivity significantly as measured by corn yields in a uniformity trial conducted in the American Mid-Atlantic region. The same study also indicates that supplying adequate nitrogen (N) for corn and controlling weeds in both corn and soybean are the biggest challenges to achieving equivalent yields between organic and conventional cropping systems (Cavigelli *et al*., 2007). SOM increase for organic soil has been reported also by Marriott and Wander (2006) in a long-term U.S. trial.

In the longest trial so far (running for more than 150 years), and going on at the Rothamsted Experimental Station in the UK, SOM and soil total N levels have been reported to have increased by about 120% over 150 years in the organic manured plots, and only by about 20% in the plots employing NPK fertilizer. Yields for organic wheat have averaged 3.45 t ha^{-1} on organically manured plots, compared with 3.40 t ha^{-1} on plots receiving NPK (Tilman, 1998). Long-term trials in Poland (Stalenga and Kawalec, 2008) also report consistent increase of SOM under organic management.

Different findings have also been reported. In an 18-year-long study in Sweden, Kirchmann *et al*. (2007, did not find significant differences in soil carbon for organic systems compared to conventional systems. It is to be considered that some can be increased up to a certain level where it starts leveling-off.

B. Soil Chemical Properties

In an 8-year experiment in the California's Sacramento Valley, Clark *et al*. (1998) found that the transition from conventional to organic farming improved soil fertility by increasing soil organic C and the pools of stored nutrients. In Europe, a 21-year Swiss field study on loess soil analyzed the agronomic and ecological performance of biodynamic, organic, and conventional farming systems (Siegrist *et al*., 1998; Mäder *et al*., 2002a; Fließbach *et al*., 2007). The authors found that the aggregate and percolation stability of both bio-dynamic and organic plots were 10 to 60% higher than conventionally farmed plots. This also affected the water retention potential of these soils in a positive way and reduced their susceptibility to erosion. Soil aggregate stability was strongly correlated to earthworm and microbial biomass, important indicators of soil fertility (Mäder *et al*., 2002a). The long-term application of organic manure positively influenced soil fertility at the biological, chemical and physical level, whereas the repeated spraying of pesticides appeared to have negative effects. Compared to stockless conventional farming (mineral fertilizers, herbicides and pesticides), the aggregate stability in plots with livestock-based integrated production (mineral and organic fertilizers, herbicides and pesticides) was 29.4% higher, while in organic and bio-dynamic plots (organic fertilizers only) was 70% higher. The authors underline the importance of using manure, by means of organic agriculture, as a good practice for soil quality preservation (Fließbach *et al*., 2007). In addition, planting cover crops once the crop is harvested helps prevent soil erosion, as the soil is kept covered with vegetation all year long.

In North Carolina, Liu *et al*. (2007) found that soils from organic farms had improved soil chemical factors and higher levels of extractable C and N, higher microbial biomass carbon and nitrogen, and net mineralizable N. In Italy, Russo *et al*., (2010) comparing chemical and organic N uptake by crops, found that altogether more mineral N was released in soil and water from the organic fertilizer while more N was taken up by plants with the mineral fertilizer. While microbial population in the soil was unaffected by the type and amount of fertilizers, enzymatic activity responded positively to organic N and was depressed by the synthetic N form. According to Walden *et al*. (1998), organically managed soils may also use mineral nutrients in a more efficient manner and allow lower inputs.

C. Nitrogen Leaching

Nitrogen fertilizers are of key importance in intensive conventional agriculture. However, their use turns out to be a major cause of concern when coming to environmental pollution. The primary source of N pollution comes from N-based agricultural fertilizers, whose use is forecast to double or almost triple by 2050 (Tilman *et al*., 2001; Robertson and Vitousek, 2009; Vitousek *et al*., 2009).

A proportion of soluble N leaches deep into groundwater, ultimately affecting human health, whereas other soluble N volatilizes (e.g., NO_X) to increment GHGs. Considering that nitrous oxide is the most potent GHG and given the environmental problems associated with the production and use of synthetic fertilizer, there is a great need for researchers concerned with global climate change and nitrate pollution to evaluate reduction strategies (Tilman *et al.*, 2002; Millenium Ecosystem Assessment, 2005a; Robertson and Vitousek, 2009; Vitousek *et al*., 2009).

On average, agricultural system N balances (N input vs. N removed with crops) in the developed or rapidly developing worlds are positive (200–300 kg N yr^{-1}), implying substantial losses of N to the environment. A number of practices can be implemented in order to reduce N loss. In this regard, leguminacae can be used productively as cover crops, absorbing N through N_2 fixation and building SOM, and in some cases can also be used by intercropping. The development of crop varieties with higher efficiencies of N uptake could help capture more of the N added to annual cropping systems (e.g., Robertson and Vitousek, 2009; Vitousek *et al*., 2009). Techniques to reduce N loss and to increase the efficiency of N uptake are widely used in organic farming (Drinkwater *et al*.,1998; Lampkin, 2002; Kramer *et al.*, 2006), and many trials demonstrate the benefit of organic farming in reducing N leaching and increasing N uptake efficiency.

A 9-year trial has been conducted by Kramer *et al.* (2006) in commercial apple orchards in Washington State, USA. The study examined denitrification and leaching from organic, integrated, and conventional systems receiving the same amount of N inputs but in different forms. The authors found that annual nitrate leaching was 4.4–5.6 times higher in conventional plots than in organic plots, where microbial denitrifier activity is enhanced through C inputs as organic fertilizers, crop residues, or root exudates from cover crops. Integrated plots showed, intermediate leaching, somewhere between organic and conventional plots. This study demonstrates that organic and integrated fertilization practices support more active and efficient denitrifier bacterial communities and reduce environmentally damaging nitrate losses.

Drinkwater *et al*. (1998) reported better N uptake efficiency for organic systems, and argued that there are differences in the partitioning of nitrogen from organic versus mineral sources, with more legume-derived nitrogen than fertilizer-derived nitrogen immobilized in microbial biomass and SOM, so reducing leaching of NO_3^- of 60% compared to the conventional control. Küstermann *et al.* (2010) report a reduction of N loss in organic farming, compared with the conventional system. An 18-year field study in Swedenby Kirchmann *et al.* (2007) reports different results. The authors found that N leaching is not reduced in organic farming, even with use of cover crops. The authors argue that yield and soil fertility were superior in conventional cropping systems under cold-temperate conditions.

Possible drawbacks from organic fertilization have been reported by some authors (e.g., Tilman *et al*., 2002; Sieling and Kage, 2006; Kirchmann *et al*., 2007; Wu and Sardo, 2010): the 'slow release' of nutrients from organic compost or green manures can be difficult to control and harness and may fail to match crop demand, resulting in N losses through leaching and volatilization. Moreover, in organic systems, competition with weeds can greatly reduce N intake efficiency (Kirchmann *et al*., 2007).

Atmospheric nitrous oxide (N_2O) is a greenhouse gas nearly 300 times more effective at radiative warming than CO_2, and is produced mainly during the microbially mediated process of denitrification. There has been a marked increased in atmospheric N_2O over the past 150 years; about 80% of this source is associated with agriculture, largely (50%) with fertilized soils (Tilman *et al*., 2001; Robertson and Vitousek, 2009; Vitousek *et al*., 2009). Although N_2O contributed for only about 6% to of the global waring potential, it plays a substantial role in the agricultural contribution to climate change, and its emissions can offset efforts to use agricultural systems to mitigate climate change by sequestering CO_2 or providing alternative energy sources (Robertson and Vitousek, 2009)

Works by Mathieu *et al.* (2006) support the hypothesis that an increase in soil available organic carbon leads to N_2 emissions as the end product of denitrification, whilst Petersen *et al.* (2006), in a study concerning five European countries, found that N input is a significant determinant for N_2O emissions from agricultural soils, and that N_2O emissions from conventional crop rotations were higher than those from organic crop rotations (except in Austria), with significant differences between locations and crop categories. Stalenga and Kawalec (2008) found that N_2O emission for organic farming systems was about 66% lower than conventional systems and 50% lower than integrated systems.

In a long-term study in southern Germany, Flessa *et al.* (2002) also found reduced N_2O emission rates in organic agriculture, although yield-related emissions were not reduced. Contrasting result are reporter by Bos *et al.* (2006, in Niggli *et al.*, 2009) with a reduction of the GHGs on Dutch organic dairy farms and in organic pea production areas, and higher GHGs emissions for organic vegetable crops (e.g., leek and potato).

D. Water Use and Resistance to Drought

Water use efficiency is determined by the amount of crop yielded divided by the amount of water used (Stanhill, 1986; Morison *et al.*, 2008). Several ways to improve water use efficiency in organic agriculture have been proposed, including reducing evaporation through minimum tillage, mulching, using more water-efficient varieties and inducing microclimatic changes to reduce crop water requirements (Stanhill, 1986; Pretty *et al.,* 2006; Morison *et al.*, 2008). Sustainable agricultural practices can be effective in improving water use efficiency in particular in poor developing country affected by water scarcity (Pretty *et al.,* 2006). Organic farming proves to be effective both at enhancing soil water content and improve water use efficiency.

Long-term crop yield stability and the ability to buffer yields through climatic adversity will be critical factors in agriculture's capability to support society in the future. A number of studies have shown that, under drought conditions, crops in organically managed systems produce higher yields than comparable crops managed conventionally. This advantage can result in organic crops out-yielding conventional crops by 70–90% under severe drought conditions (Lockeretz *et al.*, 1981; Stanhill, 1990; Smolik *et al.*, 1995; Teasdale *et al.*, 2000; Lotter *et al.*, 2003; Pimentel *et al.*, 2005). According to Lotter *et al.* (2003), the primary mechanism for higher yields in organic crops is due to higher water-holding capacity of soils under organic management. Others studies have shown that organically managed crop systems have lower long-term yield variability and higher cropping system stability (Smolik *et al.*, 1995; Lotter *et al.*, 2003).

As part of the Rodale Institute Farming System Trial (from 1981 to 2002), Pimentel *et al.*, (2005) found that during 1999, a year of extreme drought, (with total rainfall between April and August of 224 mm, compared with an average of 500 mm) the organic animal system had significantly higher corn yield (1,511 kg per ha) than either organic legume (412 kg per ha) or the conventional (1,100 kg per ha) systems.

For soybean both organic systems performed much better than the conventional system (Table 1).

TABLE 1
The Rodale Institute Farming System Trial, crops performance under drought condition, data after Pimentel *et al.* (2005).

	Yield (kg ha^{-1})	
Farming system	Corn	Soybean
Organic animal	1, 511	1, 400
Organic legume	412	1, 800
Conventional	1, 100	900

Pimentel *et al*. (2005) estimated the amount of water held in the organic plots of the Rodale experiment in the upper 15 cm of soil at 816.000 liters per ha. In heavy loess soils in a temperate climate in Switzerland water holding capacity was reported being 20 to 40% higher in organically managed soils than in conventional ones (Mäder *et al.*, 2002a).

The primary reason for higher yield in organic crops is thought to be due to the higher water-holding capacity of the soils under organic management (Reganold *et al.*, 1987; Sullivan, 2002; Lotter *et al.*, 2003). Soils in the organic system capture more water and retain more of it, up to 100% higher in the crop root zone, when compared to conventional. Such characteristics make organic crop management techniques a valuable resource in this present period of climatic variability, providing a better buffer to environmental extremes, especially in developing countries.

A soil's texture (the proportions of sand, silt, and clay present in a given soil), and aggregation (how the sand, silt, and clay come together to form larger granules) determine air and water circulation, erosion resistance, looseness, ease of tillage, and root penetration. Texture is a given property of the native soil and does not change with agricultural activities. Aggregation, however, can be improved or weakened through the timing of farm practices. Among the practices that destroy or degrade soil aggregates are: excessive tillage, tilling when the soil is too wet or too dry, using anhydrous ammonia (because it speeds the decomposition of organic matter), using excessive nitrogen fertilization, or using salty irrigation water or sodium-containing fertilizers, which results in the excessive buildup of sodium (Sullivan, 2002). It has been estimated that for every 1% of SOM content, the soil can hold 10.000-11.000 liters of plant-available water per ha of soil down to about 30 cm (Sullivan, 2002).

However, it has to be pointed out that local specificity plays an important role in determining the performance of a farming system: what is sustainable for one region may not be for another region or area (Smolik *et al.*, 1995). So, more work has to be done to acquire knowledge about the comparative sustainability of different farming systems.

Adaptive measures to cope with climate change should treasure knowledge gained from organic farming. Extensive experimentation should be conducted to gain better understating of the complex interaction among farming practices, environmental characteristics and agroecosystem resilience.

E. The Potential for Organically Managed Farming Systems to Operate as a Carbon Sink and Contribute to GHGs Reduction

Annual fossil CO_2 emissions increased from an average of 6.4 Gt C (or 23.5 Gt CO_2) per year in the 1990s to 7.2 Gt C (or 26.4 $GtCO_2$) per year in 2000–2005. CO_2 emissions associated with land-use change are estimated to average 1.6 GtC (5.9 $GtCO_2$) per year over the 1990s, although these estimates have a large uncertainty (IPCC, 2007).

Agricultural activities (not including forest conversion) account for approximately 5% of anthropogenic emissions of CO_2 and the 10–12% of total global anthropogenic emissions of GHGs (5.1 to 6.1 Gt CO_2 eq. yr^{-1} in 2005), accounting for nearly all the anthropogenic methane and one to two thirds of all anthropogenic nitrous oxide emissions are due to agricultural activities (IPCC, 2000, 2007).

In 2008, in the United States, agricultural activities were responsible for about 7% of total U.S. GHGs emissions in 2008 (with livestock as major contributors) with an increase of 10% from 1998 to 2008 (U.S. EPA, 2010).

According to Smith *at al.* (2008) many agricultural practices can potentially mitigate GHG emissions, such as: improved cropland and grazing land management, restoration of degraded lands and cultivated organic soils; and point out that the current levels of GHG reduction are far below the technical potential of these agricultural practices. Smith *et al.* (2008) estimate that agriculture could offset, at full biophysical potential, about 20% of total global annual CO_2 emissions.

Some authors (Kern and Johnson, 1993; Schlesinger, 1999) report that converting large areas of U.S. cropland to conservation tillage (including no-till practices), could sequester all the CO_2 emitted from agricultural activities in the United States, and up to 1% of today's fossil fuel emissions in the United States. Similarly, alternative management of agricultural soils in Europe could potentially provide a sink for about 0.8% of the world's current CO_2 release from fossil fuel combustion.

Lal (2004) has estimated that the strategic management of agricultural soil that is moving from till to no-till farming (also known as *conservation tillage, zero tillage,* or *ridge tillage*) has the potential to reduce fossil-fuel emissions by 0.4 to 1.2 Gt C yr^{-1}. This equals to a reduction of 5% to 15% of global CO_2 emissions.

In a 10-year systems trial in American Midwest, Grandy and Robertson (2007) found that compared to conventional agriculture, increases in soil C concentrations from 0 to 5 cm occurred with no-till (43%), low input (17%) and organic (24%) management. Soil carbon fixation is possible for conventional agriculture ranging from 8.9 gC m^{-2} y^{-1} (0.89 t ha^{-1} y^{-1}) in row crops to 31.6 gC m^{-2} y^{-1} (3.16 t ha^{-1} y^{-1}) in the early successional forage crops. Reduction in land use intensity increases soil C accumulation in soil aggregates. The authors argue that soil tillage is of key importance to determine soil C accumulation and suggest that there is high potential for carbon sequestration and offsetting atmospheric CO_2 increases by effective management of agriculture land.

Evidence from numerous long-term agroecosystem experiments indicates that returning residues to soil, rather than removing them, converts many soils from "sources" to "sinks" for atmospheric CO_2 (Rasmussen *et al.*, 1998; Lal, 2004; Smith *et al.*, 2008).

Properly managed agriculture and SOM increase in cultivated soil play an important role in the storage of carbon, and this has been addressed by many authors (e.g., Janzen, 2004; Drinkwater *et al.*, 1998; Stockdale *et al.*, 2001; Pretty *et al.*, 2002; Holland, 2004; Lal, 2004; Pimentel *et al.*, 2005; IPCC, 2007; Smith *et al.*, 2008). This carbon can be stored in soil by SOM and by aboveground biomass through processes such as adopting rotations with cover crops and green manures to increase SOM, agroforestry, and conservation-tillage systems. According to a review carried out by Pretty *et al.* (2002), carbon accumulated under improved management increased by more than 10 times, from 0.3 up to 3.5 tC ha^{-1} yr^{-1}.

Organic agriculture practices play an important role in enhancing carbon storage in soil in the form of SOM. Results from a 15-year study in the United States, where three district maize/soybean, two legume-based and one conventional agroecosystems were compared, led Drinkwater *et al.* (1998) to estimate that the adoption of organic agriculture practices in the maize/soybean grown region in the U.S. would increase soil carbon sequestration by 0.13 to 0.30 10^{14} g yr^{-1}. This is equal to 1–2% of the estimated carbon released into the atmosphere from fossil fuel combustion in the USA (referring to 1994 figures of 1.4 10^{15} g yr^{-1}).

Both because there is a limit to how much carbon the soil can capture acting as a carbon sink and because fossil fuels are being used at a very rapid pace, conversion to organic agriculture only represents a temporary and partial solution to the problem of carbon dioxide emissions Foereid and Høgh-Jensen (2004) developed a computer model for organic agriculture acting as carbon sink, and simulations show a relatively fast increase in the first 50 years, by 10–40 g C m^{-2} y^{-1} on average; this increase would then level off, and after 100 years reach an almost stable level of sequestration.

Although organic agriculture may represents an important option to reduce CO_2, long-term solutions concerning CO_2 and GHGs emission abatement should rely on a more general change of our development path, for instance by reducing overall energy consumption.

F. Soil Ecology, Biodiversity, and Its Effects on Pest Control

One hectare of high-quality soil contains an average of 1,300 kg of earthworms, 1,000 kg of arthropods, 3,000 kg of bacteria, 4,000 kg of fungi, and many other plants and animals (Pimentel *et al.*, 1992; Lavelle and Spain, 2002). Transition to organic soil management can benefit soil biodiversity. In this context, it should also be noted that SOM play an essential role in increasing soil biodiversity (Pimentel *et al*., 2006).

Enhancement of soil microbes and soil microfauna by organic inputs has been demonstrated in alternative farming systems across different climatic and soil conditions (Paoletti *et al*., 1995, 1998; Gunapala and Scow, 1998; Fließbach and Mäder, 2000; Hansen *et al*., 2001; Mäder *et al*., 2002a; Marinari *et al.* 2006; Tu *et al*., 2006; Briar *et al*., 2007 Fließbach *et al*., 2007; Liu *et al.*, 2007; Birkhofer *et al*., 2008; Phelan, 2009).

Hansen *et al*. (2001), reviewing several studies on soil biology, found that organic farming is usually associated with a significantly higher level of biological activity, represented by bacteria, fungi, springtails, mites and earthworms, due to its versatile crop rotations, reduced applications of nutrients, and the ban on pesticides.

In a Swiss long-term experiment (Siegrist *et al*., 1998; Mäder *et al*., 2002a; Fließbach *et al*., 2007), soil ecological performance were greatly enhanced under biodynamic and organic management.

Microbial biomass and activity increased under organic management, root length colonized by mycorrhizae in organic farming systems was 40% higher than in conventional systems. Biomass and abundance of earthworms were from 30 to 320% higher in the organic plots as compared with conventional. Although the number of species of carabid beetles were not significantly higher in organic and biodynamic system compared to conventional (28–34 in biodynamic; 26–29 in organic and 22–26 in conventional), still some specialized and endangered species were reported to be present only in the two organic systems.

Concerning soil health, Briar *et al*. (2007) conclude that transition from conventional to organic farming can increase soil microbial biomass, N and populations of beneficial bacterivore nematodes while simultaneously reducing the populations of predominantly plant-parasitic nematodes. The authors also indicate that reducing tillage provides benefits for the development of a more mature soil food web.

In a seven-year experiment in Italy, Marinari *et al.* (2006) compared two adjacent farms, one organic and one conventional, and found that the fields under organic management showed significantly better soil nutritional and microbiological conditions; with an increased level of total nitrogen, nitrate and available phosphorus, and an increased microbial biomass content, and enzymatic activities.

Liu *et al*. (2007) report that in North Carolina microbial respiration in soils from organic farms was higher than that in low-input or conventional farms, indicating that microbial activity was greater in these soils, and that populations of fungi and thermophiles were significantly higher in soils from organic and low-input when compared to those of conventional fields.

Birkhofer *et al*. (2008) found that organic farming fosters microbial and faunal decomposers and this propagates into the aboveground system, sustaining a higher number of generalist predators, thereby increasing natural pest control. The authors, however, note that grain and straw yields were 23% higher in systems receiving mineral fertilizers and herbicides then the organic systems.

Soil management also seems to affect pest response. A number of studies report pest preferring plants which have been nurtured with synthetic fertilizer rather than those growing in organically managed soil (Phelan *et al*., 1995, 1996; Alyokhin *et al*., 2005; Hsu *et al*., 2009). This is explained by the "mineral balance hypothesis" (Phelan *et al*., 1996), which states that organic matter and microbial activity associated with organically managed soils allow to enhance nutrient balance in plants, which in turn can better respond to pest attack. Phelan and colleagues (Phelan *et al*., 1995; 1996; Phelan, 2009) report that under green house controlled experiments, females of European corn borer (*Ostrinia nubilalis*) were found to lay consistently fewer eggs in corn on organic soil than on conventional soil. Research on the effect of butterfly *Pieris rapae crucivora,* a cabbage pest, by Hsu *et al*. (2009) indicated that these butterflies preferred to lay eggs on foliage of synthetically fertilized plants (authors argue that proper organic fertilization can increase plant biomass production and may result lower pest incidence). Moreover, Alyokhin *et al*. (2005) reported that densities of Colorado potato beetle (*Leptinotarsa decemlineata*) were generally lower in plots receiving manure soil amendments in combination with reduced amounts of synthetic fertilizers compared to plots receiving full rates of synthetic fertilizers, but no manure.

A more complex relation between soil fertilization and crop pest has been found by Staley *et al*., (2010). The authors report that two aphid species showed different responses to fertilizers: the *Brassica* specialist *Brevicoryne brassicae* was more abundant on organically fertilized plants, while the generalist *Myzus persicae* had higher populations on synthetically fertilized plants. The diamondback moth *Plutella xylostella* (a crucifer specialist) was more abundant on synthetically fertilized plants and preferred to oviposit on these plants. The authors found also that glucosinolate concentrations were up to three times greater on plants grown in the organic treatments, while nitrogen content as maximized on plant foliage under higher or synthetic fertilizer treatments.

IV. BIODIVERSITY

Biodiversity refers to the number, variety and variability of living organisms in a given environment. It includes diversity within species, between species, and among ecosystems (Wilson, 1988; Gaston and Spicer, 2004; Koh *et al*., 2004; Chivian and Bernstein, 2008). The concept also covers how this diversity changes from one location to another and over time. Biodiversity assessment, such as the evaluation of the number of species in a given area, or the more affordable use of bioindicators, can help in monitoring certain aspects of biodiversity (Paoletti, 1999; Büchs, 2003; Duelli and Obrist, 2003; Paoletti *et al.*, 2007a), even if due attention should be paid to the comparison procedure (Gotelli and Colwell, 2001; Duelli and Obrist, 2003; Pocock and Jennings, 2007). Within the term biodiversity also fall the biodiversity of crops and reared animals and the management strategy of the farm itself (e.g., rotation pattern, intercropping) (Lampkin, 2002; Caporali *et al*., 2003; Noe *et al*., 2005; Norton *et al*., 2009)

The most dramatic ecological effect of agriculture expansion on biodiversity has been habitat destruction, which, along with soil erosion and the intensive use of agrochemicals (e.g., pesticides and fertilizers), has combined to threaten biodiversity (Paoletti and Pimentel, 1992; Pimentel *et al*., 1995; Krebs *et al*., 1999; Benton *et al*., 2003; Foley *et al*., 2005; Pimentel *et al*., 2006; Butler *et al*., 2007; Paoletti *et al*., 2007b). According to Czech *et al*. (2000), in the United States agriculture has contributed to endangering biodiversity more than any other cause except urbanization.

Organic farming can offer a possible solution to halt, or reduce, biodiversity loss by a number of means such as preservation of ecological elements of the landscape, reduction in the use of harmful chemicals and alleviation of stress caused on soil ecology.

A. Organic Farming and Biodiversity

Whether organic agriculture enhances biodiversity has been a matter of research and debate for the last decades (Paoletti and Pimentel, 1992; Moreby *et al*., 1994; Stockdale *et al*., 2001; Shepherd *et al*., 2003; Bengtsson *et al*., 2005; Fuller *et al*., 2005; Hole *et al*., 2005; Hyvönen, 2007; Norton *et al*., 2009).

Extensive analysis (e.g., Moreby *et al*., 1994; Pfiffner and Niggli, 1996; Mäder *et al*., 2002a; Caporali *et al*., 2003; Bengtsson *et al*., 2005; Fuller *et al*., 2005; Hole *et al*., 2005; Roschewitz *et al*.,2005; Gabriel *et al*., 2006, 2010; Clough *et al*., 2007a; Hyvönen, 2007; Hawesa *at al.,* 2010), suggest that organic farming is generally associated with higher levels of biodiversity with regards to both flora and fauna.

A wide meta-analysis by Bengtsson *et al*. (2005) indicated that organic farming often has positive effects on species richness and abundance: 53 of the 63 studies analyzed (84%) showed higher species richness in organic agriculture systems, but a range of effects considering different organism groups and landscapes. Bengtsson *et al*. (2005) suggest that positive effects of organic farming on species richness can be expected in intensively managed agricultural landscapes, but not in small-scale landscapes comprising many other biotopes as well as agricultural fields. A review of the literature carried out by Hole *et al*. (2005) confirms the positive effect of organic farming on biodiversity, but authors point out that such benefits may be achieved also by conventional agriculture when carefully managed (a finding that seems supported also by other authors, e.g., Gibson *et al*., 2007), and indicate the need for long term, system-level studies of the biodiversity response to organic farming.

Comparing local weed species diversity in organic and conventional agriculture in agricultural areas in Germany, Roschewitz *et al*. (2005) found that weed biodiversity was influenced by both landscape complexity and farming system. The authors reported that local management (organic vs. conventional) and complexity of the surrounding landscape had an influence on alpha, beta and gamma diversities of weeds in 24 winter wheat fields. Species diversity under organic farming systems was clearly higher in simple landscapes, but conventional vegetation reached similar diversity levels when the surrounding landscape was richer because of the presence of refugia for weed populations. Roschewitz *et al*. (2005) argue that agri-environment schemes designed to preserve and enhance biodiversity should not only consider the management of single fields but also that of the surrounding landscape. Along similar lines, in Finland, Hyvönen *et al*. (2003) studied diversity and species composition of weed communities during spring in cereal fields cultivated by organic, conventional cereal and conventional dairy cropping, and concluded that organic cropping tends to promote weed species diversity at an early phase of cropping history, in particular for species susceptible to herbicides. The authors, however, argue that a change in species composition would require a longer period of organic cropping. In Scotland, Hawesa *at al.* (2010) found significantly more weeds in the seedbank and emerged weed flora of organic farms compared to either integrated or conventional farms and concluded that organic systems tend to support a greater density, species number and diversity of weeds compared to conventional management.

It has been demonstrated that when farming management is turned from conventional to organic, the weed populations can be restored to a state comparable to that before application of intensive cropping measures (Hyvönen and Salonen, 2002; Hyvönen, 2007). However, the recovery of the weeds is reported to differ between species, with species with a more rapid recovery being nitrophilous species that suffered from the application of herbicides, or species that were tolerant against herbicides. Perennial species favored by grasslands showed the slowest recovery. The authors point out that application of diverse crop rotations in organic cropping is the focal factor affecting species composition of weed communities.

Pfiffner *et al.* (2001) conducted a review of 44 investigations worldwide concerning the effects of organic and conventional farming on fauna, and reported organic farming as performing much better on both organism abundance and species diversity.

In Swiss trials (Pfiffner and Niggli, 1996; Mäder *et al.*, 2002a; Pfiffner and Luka, 2003), earthworms, carabids, epigeal spiders and other epigeal arthropods have been reported to be more abundant and with higher biodiversity in organic/biodynamic fields compared to conventional fields. They suggest the higher abundance might depend upon low-input and organic fertilization, more favorable plant biota protection management (especially weed management) and possibly upon closer interaction with semi-natural habitats.

Ekroos *et al*. (2010), comparing both weed and carabid beetles biodiversity, find that, in the case of weds, organic farming increased both insect-pollinated as well as overall weed species richness, whereas the proportion of insect-pollinated weed species within the total species richness was unaffected by farming practices; on the other hand, in the case of carabid beetles a positive correlation with organic farming was less evident. Pfiffner and Niggli (1996) reports higher diversity and abundance of carabid beetles (90% greater) and other epigeic arthropods on organic plots of winter wheat than in conventional plots. Research carried out in North Eastern Italy in different types of orchards and vineyards found that arthropods, carabid species and earthworms were more abundant in organic than in conventional agroecosystems (Paoletti *et al*., 1995, 1998). Greater abundance of earthworms (up to more than 100%) and insects for organic farms has been reported also for Swiss farming system (Pfiffner and Mäder, 1997; Pfiffner and Luka, 2007).

In the largest and most comprehensive study of organic farming in the UK to date, Fuller *et al*. (2005) shows that organic farms provide greater benefits for a range of wildlife (including wild flowers, beetles, spiders, birds and bats) than their conventional counterparts. Fuller *et al*., (2005) found that organic fields were estimated to hold 68–105% more plant species and 74–153% greater abundance of weeds (measured as cover) than nonorganic fields support, 5–48% more spiders in preharvest crops, 16–62% more birds in the first winter and 6–75% more bats (see also Wickramasinghe *et al.,* 2004, who have found that organic farming is beneficial to bats, both through provision of more structured habitats and higher abundance of insect prey). These studies indicate that organic farming systems provide greater potential for biodiversity than their conventional counterparts, as a result of greater variability in habitats and more wildlife-friendly management practices, which results in real biodiversity benefits, particularly for plants. Plants indeed showed far more consistent and pronounced responses to the use of organic systems when compared to other taxa, as reported also by Bengtsson *et al*. (2005).

In the case of other taxa, Fuller *et al*. (2005) report that even where significant differences were detected, the results showed high variability and wide confidence intervals. Compared to the review by Bengtsson *et al*. (2005), Fuller *et al*. (2005) in their meta-analysis find that predatory invertebrates showed a significant response to agricultural practices only infrequently.

Results from Swedish research on butterfly species diversity in organic and conventional farms (Rundlöf and Smith, 2006; Rundlöf *et al*., 2008) indicate that both organic farming and landscape heterogeneity significantly increased butterfly species richness and abundance. Authors report also that there was a significant interaction between farming practice and landscape heterogeneity, and organic farming significantly increased butterfly species richness and abundance only in homogeneous rather than heterogeneous landscapes.

A previous Swedish study (Weibull *et al*., 2003) did not find differences when comparing the biodiversity and abundance of plants, butterflies, rove beetles and spiders in organic and conventional farms, while carabids richness was higher in conventional farms. The authors argued that species richness was higher on farms with a heterogeneous landscape, while farming practice was of relatively less importance in relation to landscape features for species richness.

A review of literature on carabid beetles in organic and conventional farming system in Germany and Switzerland by Döring and Kromp (2003) found that in most cases species richness was higher in the organically than in the conventionally managed fields.

No difference for carabids biodiversity were instead reported by the USDA Farming Systems Project in Maryland, by Clark *et al*. (2006) in organic, no-till, and chisel-till cropping systems.

According to van Elsen (2000), economic pressure leads to an improvement in mechanical weed control and undersowing, so that supporting and developing a diverse arable field flora cannot be done automatically just by converting to organic farming. Rather, an integration with the guiding vision of organic agriculture is needed, and measures to support the richness of species of arable field plants in organic fields have to be developed.

B. Biodiversity and Landscape

An increasing body of evidence indicates that landscape heterogeneity is a key factor in promoting biodiversity in the agricultural landscape (Benton *et al*., 2003; Purtauf *et al*., 2005; Schmidt *et al*., 2005; Tscharntke *et al*., 2005; Gabriel *et al.,* 2006, 2010; Rundlöf and Smith, 2006; Clough *et al*., 2007b; Norton *et al*., 2009). A mosaic landscape may support a larger number of species in a given area, simply because the landscape contains a larger number of habitats. Organic farming system produced greater field and farm complexity than farms employing a nonorganic system (Gabriel *et al.,* 2006, 2010; Clough *et al*., 2007b; Norton *et al*., 2009). In Germany, Gabriel *et al.* (2006, 2010) found that plant species in wheat organic farming made the greatest contribution to total species richness at the meso (among fields) and macro (among regions) scale due to environmental heterogeneity. Rundlöf and Smith (2006) argue that organic farming, with its exclusion of pesticides and longer crop rotation, may, on a landscape scale, increase habitat heterogeneity and biodiversity.

Some scholars argue that because many organic farms are often isolated units, embedded in nonorganic farmland managed with conventional levels of pesticide and fertilizer inputs, offering a relatively low levels of habitat heterogeneity, this may reduce the benefits offered by organic farming as well as by species colonization. In these cases, organic farming probably offer insufficient resources to affect population sizes of species with large spatial needs, such as birds (Bosshard *et al*., 2009; Brittain *et al*., 2010).

Concerning invertebrates, agricultural landscapes with organic crops have overall been reported to support higher biodiversity for pollinator (Holzschuh *et al*., 2008), butterfly (Rundlöf and Smith, 2006), carabid beetle (Purtauf *et al*., 2005), spiders (Fuller *et al*., 2005; Schmidt *et al*., 2005), and a number of invertebrates taxa (Benton *et al*., 2003; Bengtsson *et al*., 2005; Clough *et al*., 2007). It has to be pointed out that the extent of non-crop habitat in the vicinity of organic farms (usually larger than for conventional farms) is likely to be beneficial for biodiversity (Holzschuh *et al*., 2007; Norton *et al*., 2009). Holzschuh *et al.* (2007), for instance, found that landscape heterogeneity and the availability of semi-natural nesting habitats resulted in higher bee diversity on farmland.

It would appear that the extension of organic farming is a potential means of reestablishing heterogeneity of farmland habitats, and thereby enhancing farmland biodiversity. However, the total area of organic farmland relative to nonorganic is generally small (a few points percentage of the total agricultural area per country). Strategies aimed at increasing both the total extent of organic farming and the size and contiguity of individual organic farms could help to restore biodiversity in agricultural landscapes (Fuller *et al*., 2005; Tscharntke *et al*., 2005; Bosshard *et al.*, 2009). This strategy is supported also by other authors. Benton *et al.* (2003) for instance, argue that, rather than concentrating on particular farming practices, promoting heterogeneity widely across agricultural systems should be a universal management objective.

Given the body of evidence accumulated so far, it is clear that measures to preserve and enhance biodiversity in agroecosystems should be both landscape and farm specific (e.g., Paoletti, 1999; Thies and Tscharntke, 1999; Hole *et al*., 2005; Pimentel *et al*., 2005; Roschewitz *et al*., 2005; Tscharntke *et al*., 2005; Gabriel *et al.,* 2006, 2010; Rundlöf and Smith, 2006; Holzschuh *et al*., 2008; Norton *et al*., 2009). Unfortunately, it is difficult to provide reliable recommendations concerning agricultural land management in order to enhance biodiversity and ecosystem services, because there is still little knowledge about the relation among agricultural land management, both at farm and at landscape level, and ecosystem services. (Tscharntke *et al*., 2005; Gabriel *et al.,* 2006, 2010).

C. Biodiversity and Pest Control

One key feature of agricultural intensification has been the increasing specialization in the production process, resulting in reduction in the number of crop and livestock species, leading to monoculture and intensive farming (Zhu *et al.*, 2000; Matson *et al*., 1997; Tscharntke *et al*., 2005). On the other hand, it has been demonstrated that increasing crop genetic diversity can play an important role in pest management and in controlling crop disease, as well as enhance pollination services and soil processes (Zhu *et al.,* 2000; Barberi, 2002; Hajjar *et al*., 2008). Zhu *et al.* (2000), for instance, demonstrated that crop heterogeneity is a possible way to solve the problem of vulnerability of monoculture crops to disease. Barberi (2002) argues that weed management should be tackled on a long time frame and needs deep integration with the other cultural practices, so as to optimize whole system control.

Agriculture intensification results also in a dramatic simplification of landscape composition and in a sharp decline of biodiversity. This also affected the functioning of natural pest control, as natural habitats provide shelter for a broad spectrum of natural species that operate as pest control for all crops (Pimentel *et al*., 1992; 1997; Kruess and Tscharntke, 1994; Pimentel, 1997; Thies and Tscharntke, 1999; Barbosa, 2003; Altieri and Nicholls, 2004; Perfecto *et al*., 2004; Bianchi *et al*., 2006; Crowder *et al*., 2010).

Preserving landscape-ecological structures (e.g., hedgerows, herbaceous strips, woodlot) means also preserving their function as a haven for beneficial organisms that can provide useful services to agriculture. On the contrary, reducing ecological structures and causing habitat fragmentation results in a significant reduction in local biodiversity and its impact in the biological control of pests (Kruess and Tscharntke, 1994; Sommaggio *et al*., 1995; Paoletti *et al.*, 1997; Thies and Tscharntke, 1999; Letourneau and Goldstein, 2001; Thies *et al.*, 2003, 2005; Bianchi *et al*., 2006; Gardiner *et al*., 2009).

Letourneau and Bothwell (2008) argue that few studies have measured biodiversity effects on pest control and yield on organic farms compared to conventional farms, while relevant studies suggest that an increase in the diversity of insect predators and parasitoids can have both positive and negative effects on prey consumption rates. As mentioned earlier in this paper, Briar *et al*. (2007) reported the positive role of the transition from conventional to organic farming in increasing populations of beneficial bacterivore nematodes while reducing plant-parasitic nematodes.

Perfecto *et al*. (2004) found that in coffee farms in Chiapas, Mexico, birds could potentially reduce pest outbreak in farms with higher floristic diversity, thus providing partial evidence in support of the "insurance hypothesis." In organic cereal fields in Germany, Westerman *et al*. (2003) found that seed predation by birds contributes substantially to the containment of weed population growth.

Other experiments proved the role of vegetation and bird presence in reducing pest outbreaks. Mols and Visser (2002, 2007), for instance, found that big tit (*Parus major* L.), a European cavity-nesting bird, reduces the abundance of harmful caterpillars in apple orchards by as much as 50 to 99%. In the Netherlands, the foraging of *P. major* increased apple yields by 4.7 to 7.8 kg per tree.

Although some studies do not find a correlation between landscape complexity and parasitoid diversity (e.g., Menalled *et al*., 1999), most of them do confirm the importance of ecological structures for harbouring beneficial organisms. Research in Italy found that hedgerows in organic farming can improve consistently the number and abundance of invertebrates and can host important key species of predators and parasitoids that can provide a natural pest control for crops (Paoletti and Lorenzoni, 1989; Sommaggio *et al*., 1995; Paoletti *et al*., 1997). In an extensive experiment to assess the effectiveness of natural pest control provided to soybean by natural pest predators, 26 replicate fields were set across Michigan, Wisconsin, Iowa, and Minnesota over two years (2005–2006) (Gardiner *et al*., 2009). The authors found that the abundance of Coccinellidae was related to landscape composition, with beetles being more abundant in landscapes with an abundance of forest and grassland compared with landscapes dominated by agricultural crops. Landscape diversity and composition at a scale of 1.5 km surrounding the focal field explained the greatest proportion of variation in biological control service index (based on relative suppression of aphid populations and on Coccinellidae abundance). The authors conclude that management aimed at maintaining or enhancing landscape diversity has the potential to stabilize or increase biocontrol services.

Bianchi *et al*. (2006) reach the same conclusions. They find that enhanced natural enemy activity showed correlation with presence of herbaceous habitats such fallows and field margins (80% of cases), and also with presence of wooded habitats (71%), and of landscape patchiness (70%). The authors conclude that all these landscape characteristics are equally important in enhancing natural enemy populations, and claim that diversified landscapes hold most potential for the conservation of biodiversity and perform a pest control function.

It is often assumed that if the reduction in agrochemicals on organic farms allows the conservation of biodiversity, it on the other hand must have some cost in terms of increased pest damage. In an experiment in tomato farms in California, Letourneau and Goldstein (2001) tested such a claim. The authors found no evidence of increased crop loss when synthetic insecticides are withdrawn. The authors stress the importance of large-scale on-farm comparisons for testing hypotheses about the sustainability of agroecosystem management schemes and their effects on crop productivity and associated biodiversity.

Recently, Crowder *et al*. (2010) showed that such insecticides disrupt the communities of pest natural enemies, reducing the effectiveness of pest control. Authors claim that organic farming methods can mitigate this ecological damage by promoting evenness among natural enemies, implying that ecosystem functional rejuvenation requires restoration of species evenness, rather than just richness, and that organic farming can offer a means of reestablishing functional evenness to ecosystems. Bahlai *et al.* (2011), however, point out that organic pesticides may not represent always the best solution to mitigate environmental risk.

It has to be pointed out that biodiversity conservation, by retaining local food web complexity can also represent an effective management strategy against the spread of invasive species that often act as pests in new environments (Kennedy *et al.*, 2002). This may help to avoid the drawback from using exotic natural enemies to fight novel invasive species, as species introduced for biocontrol can act as invasive species in their own right (Thomas and Reid, 2007).

V. ENERGY USE AND GHGs EMISSION

A. Energy Efficiency

Organic farming has been reported to provide a better ratio of energy input/output (Table 2). (For further figures see also the review by Lynch *et al.*, 2011)

The main reasons for higher efficiency in the case of organic farming are: (1) lack of input of synthetic N-fertilizers (which require high energy consumption for production and transport and can account for more than 50% of the total energy input), (2) low input of other mineral fertilizers (e.g., P, K), lower use of highly energy-consumptive foodstuffs (concentrates), and (3) the ban on synthetic pesticides and herbicides (Lockeretz *et al*., 1981; Pimentel *et al*., 1983; 2005; Refsgaard *et al*., 1998; Cormack, 2000; Stockdale *et al*., 2001, Haas *et al*., 2001; FAO, 2002; Lampkin, 2002; Hoeppner *et al*., 2006; Kasperczyk and Knickel, 2006; Küstermann *et al.,* 2008; Lynch *et al.*, 2011). According to estimates carried out in a study conducted by the Danish government (Hansen *et al*., 2001), upon 100% conversion to organic agriculture a 9–51% reduction in total energy use would ensue (the rate of reduction depending on the level of imported feeds and the numbers of animals reared).

However, when calculating energy input in terms of physical output units, a reduced advantage in employing organic systems was observed (Cormack, 2000; Stockdale *et al*., 2001). On average, yield from arable crops is reported to be 20% to 40% lower in organic systems compared to conventional systems, whereas the yield for horticultural crops could be as low as 50% that of conventional; grass and forage production is reported between 0 and 30% lower for organic systems (Cormack, 2000; Stockdale *et al*., 2001; Mäder *et al.*, 2002a, 2002b; Cavigelli *et al*., 2007; Kirchmann *et al*., 2007; Küstermann *et al.*, 2008).

Dalgaard *et al*. (2001) argue that the energy efficiency, calculated as the yield divided by the energy use (MJ ha^{-1}), was generally higher in the organic system than in the conventional system, but the yields were also lower. This meant that conventional crop production had the highest net energy production, whereas organic crop production had the highest energy efficiency.

In industrial societies, energy efficiency *per se* may not be the goal. Increasing productivity per hour of labor is in fact what modern society aims at, and this may lead us in the opposite direction (decreasing overall energy efficiency) (Giampietro, 2004). This inverse relation between total productivity and efficiency is typical for traditional agriculture and intensive agriculture. When comparing corn production in intensive U.S. farming systems and a Mexican traditional farming system the former had an efficiency (output/input) of 3.5:1 while the latter of 11:1 (using only manpower). However, when coming to total net energy production, intensive farming system accounted for 17.5 million kcal $ha^{-1}yr^{-1}$(24.5 in output and 7 in input), while traditional just 6.3 million kcal $ha^{-1}yr^{-1}$ (7 million in output and 0.6 million in input) (Pimentel, 1989).

On the other hand, some studies have found organic production comparable to that of conventional systems (Clark *et al*., 1999; Pimentel *et al*., 2005). Clark *et al*. (1999) argue that organic and low-input tomato systems can produce yields similar to those of conventional systems but that factors limiting yield may be more difficult to manage: N availability in the case of organic systems and water availability in that of conventionally managed systems. In the Rodale long-term study (Pimentel *et al*., 2005) organic performance is comparable to conventional

TABLE 2
Comparison of energy efficiency (input/output) per unit of production of organic as percent of conventional farming systems.

Farming System	Reference	Energy Efficiency organic as % of conventional
Analysis for crops under organic and conventional management		
Wheat in USA	Pimentel *et al.* (1983)	+29/+70
Wheat in Germany (various studies)	Stölze *et al.* (2000)	+21/+43
Wheat in Italy	FAO (2002)	+25
Corn in USA	Pimentel *et al.* (1983)	+35/+47
Apples in USA	Pimentel *et al.* (1983)	−95
Potatoes in Germany (3 studies)	Stölze *et al.* (2000)	+7/+29
Potatoes USA	Pimentel *et al.* (1983)	−13/−20
Rotations of different crop systems in Iran	Zarea *et al.* (2000) (in FAO, 2002)	+81
Rotations of different crop systems in Poland	Kus and Stalenga (2000) (in FAO, 2002)	+35
Danish organic farming	Jørgensen *et al.* (2005)	+10
Whole system analysis (Midwest – USA) with comparable output	Smolik *et al.* (1995)	+60/+70
Crop rotations (wheat-pea-wheat-flax and wheat-alfalfa-alfalfa-flax) in Canada	Hoeppner *et al.* (2006)	+20
Apricot in Turkey	Gündomuþ (2006)	+53
Olive in Spain	Guzmán and Alonso (2008)	+50
Crop rotations	Küstermann *et al.* (2008)	+9
Results from Long-Term Agroecosystem Experiments		
Apples in USA	Reganol *et al.* (2001)	+7
Various crop systems	Mäder *et al.* (2002)	+20/+56%
Organic and animals	Pimentel *et al.* (2005)	+28
Organic and legumes	Pimentel *et al.* (2005)	+32
Organic vs. conv. with tillage	Gelfand *et al.* (2010)	+10
Organic vs. conv. no tillage	Gelfand *et al.* (2010)	−30

performance with respect to key agronomic indicators (Table 3).

As previously mentioned, it has to be pointed out that under drought conditions organic systems produce higher yields than comparable crops managed conventionally, up to 70–90% (Lockeretz *et al*., 1981; Stanhill, 1990; Smolik *et al*., 1995; Lotter *et al*., 2003; Pimentel *et al*., 2005).

It appears that the energetic performances of different farming systems depend on the crops cultured and specific farm characteristics (e.g., soil, climate). Pimentel *et al*. (1983), who reported lower energy efficiency in organic potatoes, ascribed it to reduced yield due to insect and disease attacks that could not be controlled in the organic system. In the case of apples there is a striking difference between data reported by Pimentel *et al*. (1983) and Reganold *et al*. (2001). This can be explained by different management techniques and their improvement in the last 20 years.

TABLE 3
A comparison of the rate of return in calories per fossil fuel invested in production for major crops - average of two organic systems over 20 years in Pennsylvania (based on Pimentel, 2006, modified).

Crop	Technology	Yield (t ha^{-1})	Labor (hrs ha^{-1})	Energy (kcal x 10^6)	output/ input
Corn	*Organic*[1]	*7.7*	*14*	*3.6*	*7.7*
Corn	Conventional[2]	7.4	12	5.2	5.1
Corn	Conventional[3]	8.7	11.4	8.1	4.0
Soybean	*Organic*[4]	*2.4*	*14*	*2.3*	*3.8*
Soybean	Conventional[5]	2.7	12	2.1	4.6
Soybean	Conventional[6]	2.7	7.1	3.7	3.2

[1] Average of two organic systems over 20 years in Pennsylvania.
[2] Average of conventional corn system over 20 years in Pennsylvania.
[3] Average U.S. corn.
[4] Average of two organic systems over 20 years in Pennsylvania.
[5] Average conventional soybean system over 20 years in Pennsylvania.
[6] Average of U.S. soybean system.

B. GHGs Emission

Agricultural contributions to CO_2 emissions come from consumption of energy in the form of oil and natural gas, both directly (e.g., field work, machinery) and indirectly (e.g., production and transport of fertilizers and pesticides). Changes in soil ecology can also result in carbon release into the atmosphere. Deforestation is an important contributor to CO_2 emissions, occurring when forest land is removed to provide more land to plant crops. NH_4 emissions come from livestock, mainly from enteric fermentation but also from manure and rice fields. N_2O comes mainly from the soil (denitrification) and to a lesser extent from animal manure (IPCC, 2007). On the other hand, it is possible to reduce direct and indirect carbon emissions by reducing the use of agrochemicals, pumped irrigation and mechanical power, which account for most of the energy input in agriculture. It has also been suggested that organic farms can develop biogas digesters to produce methane for home and commercial use (Pretty *et al*., 2002; Hansson *et al*., 2007). This technology is, however, not limited to organic management.

Stölze *et al*. (2000), in their review of European farming systems, saw trends toward lower CO_2 emissions in organic agriculture but were not able to conclude that overall CO_2 emissions are lower per unit of product in organic systems compared to the conventional ones. The authors reported that the 30% higher yields in conventional intensive farming in Europe can compensate for the lower CO_2 emissions per unit of products in organic agriculture.

Haas *et al*. (2001) conducted a Life Cycle Assessment of the environmental impacts of 18 grassland farms in three different farming intensities (intensive, extensified, and organic) in southern Germany. They found that extensified and organic farms reduce energy consumption and Global Warming Potential (GWP). The authors found that the area-related GWP decreases for intensive (9.4 t CO_2 eq. ha^{-1}), extensified (7.0 t CO_2 eq. ha^{-1}) and organic farms (6.3 t CO_2 ha^{-1}), accordingly. With regards to product-related energy use, extensified farms (1.0 t CO_2 eq. ha^{-1}) cause the lowest GWP, whereas intensified and organic farms (1.3 t CO_2 eq. ha^{-1}) produce the same emissions. Lower CO_2^- and N_2O^-emissions of organic farms are compensated by a higher emission of CH_4 per unit of produced milk, because of lower milk yields.

Comparing the performances of single crops can produce very different results from those obtained when comparing the whole cropping system within which that specific crop is found. Küstermann *et al*. (2008), for instance, report that GHGs per ha for winter wheat are comparable between organic and conventional system. On a harvested biomass basis, lower yields in organic farming involved higher emissions (496 kg CO_2 eq. Mg^{-1} for the organic system and 355 kg CO_2 eq. Mg^{-1} for the conventional), when all products relating to the whole crop rotation are considered, organic management is shown to result in lower emission (263 kg CO_2 eq. Mg^{-1}, for the organic system against 376 kg CO_2 eq. Mg^{-1} for the conventional system) (Table 4).

Modeling of a transition to organic production in Canada, Pelletier *et al*. (2008) found that a total transition of Canadian canola, corn, soybean and wheat production to organic management may reduce the overall national energy consumption by 0.8%, GHGs emissions by 0.6%, and acidifying emissions (from N and S compounds) by 1%. The authors argue that although organic farming systems have a slightly higher fuel-related energy consumption, still their average total energy demand has been estimated at about 40% that of conventional management, mainly due to the use of synthetic fertilizer and pesticide (quite costly in terms of energy demand) in conventional systems. Such calculations, however, do not account for organic compost shipments over long distance.

Wood *et al.* (2006) carried out a comprehensive environmental impacts analysis of Australian agriculture, and argue that organic production has smaller indirect impacts than conventional production, and that a transition to organic farming could be a viable way of reducing energy use and GHG emissions, while maintaining employment and economic benefits. In their review, Lynch *et al*. (2011) fond that organic systems has gnerally lower GHGs emission per ha but the results are variable on a per unit of product basis.

C. Integrating Animal Husbandry

In organic farming, animal husbandry is carried out taking into account ethical concerns regarding the well being of the animals, and therefore, amongst other practices, it promotes natural behavior of cows by having them spend most of the grazing period outdoors, it limits the use of drugs and endorses the use of feed coming from crops where the use of synthetic fertilizers and pesticides is forbidden (Lund, 2006). This translates to better consumer health, having meat without an extra supply of (synthetic) hormones and traces of antibiotics.

According to some authors (Subak, 1999; Cederberg and Stadig, 2003; Koneswaran and Nierenberg, 2008a, 2008b)

TABLE 4
CO_2 emissions for some productions (data from Küstermann *et al.,* 2008).

	GHGs emission per ha (kg CO_2 eq. ha^{-1})			GHGs emission per production unit (kg CO_2 eq. t^{-1})		
Study	Conv.	Organic	Org. as % of conv.	Conv.	Organic	Org. as % of conv.
Winter wheat	2,333	1,669	71	355	496	140
Similar crop rotation	2,717	887	32	376	263	70

organic animal husbandry has the potential to reduce GHG emissions and sequester carbon through better pasture management. Raising cattle for beef organically on grass, in contrast to fattening confined cattle on concentrated feed, may emit 40% less GHGs and consume 85% less energy than conventionally produced beef. According to Williams *et al*. (2006), most organic animal production reduces primary energy use by 15% to 40%, with the exception of organic poultry meat and egg production, which increase energy use by 30% and 15% respectively.

How to develop appropriate analytical methods to assess the sustainability of organic meat and milk production is, however, still work in progress and a matter of debate (e.g., De Boer, 2003; Avery and Avery, 2008; Koneswaran and Nierenberg, 2008a; 2008b; Müller-Lindenlauf *et al*., 2010).

A study of German dairies by Haas *et al*. (2001) reports an energy use per unit of milk for organic agriculture that is less than half of that of conventional farming, and less than one-third per unit of land. For instance, De Boer (2003), argued that at present we cannot directly compare results of different LCA studies. The author noted that, for example, absolute GWP differs largely among studies because of differences in allocation or normative values used with respect to CH_4 and N_2O emission. Lacking a standardized protocol for LCA, De Boer (2003, p. 76) stated that "conventional and organic production systems can be compared only within a case study." Avery and Avery (2008) of the Huston Institute (a think tank based in Washington D.C.), challenged the data by Koneswaran and Nierenberg (2008a), whose figures indicated organic animal production systems performing better than conventional, claiming that the authors were comparing highly different environmental and cultural contexts (Sweden and Japan), and citing different studies to support different conclusion. Koneswaran and Nierenberg (2008a; 2008b), on the other hand, replied that the LCA cited by Avery and Avery (2008) are still misleading and, in some cases, wrongly quoted. Further to the LCA issue, De Boer (2003), argued also that experimental farms, from which comparison between organic and conventional animal production are made, do not necessarily represent corresponding production systems. Müller-Lindenlauf *et al*. (2010), called for the adoption of a more complex approach, arguing that focussing only on the classical environmental impact categories (e.g. energy efficiency, GWP) may lead to different results than a system approach that includes a broader range of relevant impacts and ecological benefits. However, there were slightly higher methane emissions per unit of organically produced milk, and the authors estimated that the final GWP of the two farming systems was similar (Tables 5a and 5b). Most LCA undertaken thus far report that organic management results in a bit less or equal footprints as compared to conventional. While outcomes rate organic management positively on a per hectare basis, performance per unit of production is less positive as organic management tends to yield less than conventional.

A German study based on a multicriterial assessment of milk production of organic and conventional farms (Müller-Lindenlauf *et al.,* 2010), concludes that organic farming tends to have less negative environmental effects than conventional farming. Results are, however, not neat. The authors found that intensive farm types tend to be advantageous in global categories such as climate impact and land demand. On the other hand, low-input farm types have significant advantages with regards to ammonia emissions, animal welfare and milk quality. The authors argue that carrying on an environmental impact assessment analyzing only a few indicators, e.g., GHGs emission and energy consumption, leads to different conclusions than an overall analysis taking into account a large number of regional and local factors. When considering land demand Müller-Lindenlauf *et al.* (2010) report that arable land demand (ha/1000 kg milk) was 0.07 for organic grasslands vs. 0.1 for conventional grasslands, and 0.03 for organic mix farm vs. 0.1 for conventional mix farm. That means that organic milk production was 3 to 10 times less dependent on arable land. Even if organic management resulted slightly higher on the overall land

TABLE 5a
Energy use and carbon emission in milk production in organic and conventional systems.

	Energy Consumption (GJ ha^{-1})			Energy Consumption (GJ t^{-1})		
Study	Conv.	Organic	Org. as % of conv.	Conv.	Organic	Org. as % of conv.
Cederberg and Mattsson (1998)	22.2	17.2	77	2.85	2.41	85
Refsgaard *et al.* (1998)	–	–	–	3.34	2.16/2.88	75/87
Cederberg and Mattsson (1998) in Haas *et al.* (2001)	–	–	–	2.85	2.4	92
Haas *et al.* (1995) in Haas *et al.* (2001)	19.4	6.8	35	–	–	–
Haas *et al.* (2001)	19.1	5.9	31	2.7	1.2	46
Thomassen *et al.* (2008)*				4.4	2.17	51
Müller-Lindenlauf *et al.* (2010) – Grassland	–	–	–	1.52	1.2	79
Müller-Lindenlauf *et al.* (2010) – Mix farm	–	–	–	1.17	1.32	113

(*) including indirect costs.

TABLE 5b
Energy use and carbon emission in milk production in organic and conventional systems.

	CO_2 Emission (kg CO_2 ha^{-1})			CO_2 Emission per Production Unit (kg CO_2 t^{-1})		
Study	Conv.	Organic	Org. as % of conv.	Conv.	Organic	Org. as % of conv.
Haas *et al.* (2001)	9,400	6,300	67	1,280[a]	428[a]	33
Haas *et al.* (2001)	–	–	–	1,300[b]	1,300[b]	0
Thomassen *et al.* (2008)*	–	–	–	1,400	1,500	107
Müller-Lindenlauf *et al.* (2010)–Grassland	–	–	–	1,036	1,172	113
Müller-Lindenlauf *et al.* (2010)–Mix farm	–	–	–	917	1,082	118

[a]considering only CO_2 emission; [b]summing up CH_4 and N_2O emissions as CO_2 equivalents, the CH_4 and N_2O emissions are comparably low, but due to the high Global Warming Potential (GWP) of these trace gases their climate relevance is much higher.
(*) including indirect costs.

demand (0.31 and 0.28 for organic vs. 0.27 and 0.22 for conventional), still the impact of organic farming on soil (e.g., soil loss, SOM, biodiversity) can be considered lower than that of conventional farming. Again, neither chemical residues in milk nor pesticide use in crops production were taken into consideration as sustainability indicators (and in some contexts pesticide use is indeed a cause of concern). The points raised should not be taken as criticism, as the work just described can be considered a nice and welcomed attempt to adopt a multicriterial approach in order to account for key indicators in a comprehensive farming system analysis. Our aim is to illustrate the complex nature of farming system analysis when attempting a comparison between different systems and the assessment of what is "the best."

In a review comparing milk production performance of organic and conventional systems, De Boer (2003) claims that few exact figures are available, especially on the amount of NO_2 and CH_4 emitted from dairy cattle production, and concludes that, firstly, the potential environmental impact of conventional and organic milk production is based largely on comparison of experimental farms, which do not necessarily represent the corresponding production systems Secondly, he suggests that different indicators provide different levels of performance; for instance, CH_4 emission appears higher in organic systems, while eutrophication potential per tonne of milk and per ha appears lower for organic milk production than for conventional. Thirdly the author argues that organic milk production potentially reduces leaching of NO_3^- and PO_4^-, due to lower fertilizer application rates.

VI. CONSTRAINTS TO THE ADOPTION OF ORGANIC AGRICULTURE

A. Feasibility

The benefits associated with the adoption of organic farming practices have been questioned by many authors to different degrees. Some authors claim that organic farming is an ideology rather than a scientific approach to agriculture (e.g., Kirchmann and Thorvaldsson, 2000; Rigby and Càceres, 2001; Trewavas, 2001, 2004; Edwards-Jones and Howells, 2001; De Gregori, 2003). Others express a milder form of criticism based on the concern that not all organic agriculture strategies can be applied globally and without many local adjustments, and because of this lack of coherence, they suggest that this approach may actually lead to a worsening of agricultural problems (e.g., Tilman *et al.*, 2002; Elliot and Mumford, 2002; Wu and Sardo, 2010).

Some authors (e.g., Elliot and Mumford, 2002) suggest the adoption of integrated farming, rather than upholding solely organic practices, which they find more harmful than conventional farming, for instance in the case of pest control technologies.

B. Labor Productivity

When assessing the socioeconomic sustainability of farming enterprises, labor productivity is a key indicator. Organic farms, although performing better in terms of energy efficiency, generally require more labor than conventional ones, ranging from about 10% up to 90% (in general about 20%), with lower values for organic arable and mixed farms and higher labor inputs for horticultural farms (Lockeretz *et al.*, 1981; Pimentel *et al.*, 1983; 2005; FAO, 2002; Foster *et al.*, 2006).

Case studies in Europe for organic dairy farms report a comparable high labor input (FAO, 2002). Little data exists for pig and poultry farms. In some long term trials, productivity per hectare and hour of work for organic and conventional crops (corn and soybean) were comparable (Pimentel *et al.*, 2005; Pimentel, 2006).

In order to gain insight into the sustainability of a farming system, different perspectives such as land use, working time and energy use should be employed at the same time (Giampietro, 2004; Gomiero *et al.*, 2006). Data on energy efficiency cannot be detached from the "metabolism" of the social system where agriculture is performed. High energy efficiency may imply low total energy output that, for a large society with limited land, may not be a sustainable option, menacing food supply for urban populations. With the current emphasis on promoting a green economy and paying farmers for environmental services, organic agriculture offers great potential to generate green jobs and revitalize rural areas. We warn, however, about looking at organic agriculture as a mean to produce biofuels (Giampietro and Ulgiati, 2005; Pimentel and Patzek, 2005; Giampietro and Mayumi, 2009; Gomiero *et al*., 2010).

C. Economic Performance

Comparing organic and conventional system is still not an easy task because authors often adopt quite different methodologies, and different geographical areas (e.g., developed and developing countries) have distinctive characteristics that should be properly taken into consideration (Nemes, 2009). Although yields in organic systems tend to be lower, input costs are usually lower. A number of studies report no major revenue difference for organic farming compared to conventional (e.g., Drinkwater *et al*., 1998; Delate *et al*. 2003; Pacini *et al.,* 2003; Mahoney *et al*., 2004; Pimentel *et al.,* 2005, for a comprehensive review of the topic see Nemes, 2009).

According to the U.S. Department of Agriculture (USDA, 2010a; Bowman, 2010), data from the organic farming census reveal that the 14,540 organic farms included in the census had an estimated average net income (total sales less expenses) of $20,249 per farm per year, a figure higher than the figure in which all farm types were included.

This has been reported also in some broad research conducted in developing countries. For instance, Eyhorn *et al*. (2007) found that in India the average gross margins from organic cotton fields were 30–40% higher than in conventional fields, due to 10–20% lower total production costs and a 20% organic price premium. Authors argue that although the crops grown in rotation with cotton were sold without premium, organic farms achieved all the same 10–20% higher incomes from agricultural activity.

Other studies, however, indicate that the impact of organic price premiums is large, and sometimes needed to match conventionally generated income and compensate for lower yields (e.g., Reganold *et al*., 2001; Pacini *et al.,* 2003; Chavas *et al*., 2009; Nemes, 2009). Recent analysis for southern Wisconsin (USA) by Chavas *et al.* (2009) shows that, under the market scenarios that prevailed between 1993 and 2006, intensive rotational grazing and organic grain and forage systems were the most profitable systems. On, highly productive land organically grown corn resulted more profitable than continuous corn cropping. Once the premium was taken into account, organic farming resulted more profitable in all systems. Results for Low External Input (LEI) agriculture in the United States (Liebman *et al*., 2008) shows that corn and soybean yields in LEI systems can be sustained at levels that match or exceed levels obtained from conventional systems. Scenario analysis by Lohr and Park (2007) indicates that economic gains will be realized as farm size increases, creating pressure on organic farmers to expand operations. Protecting small organic farms is likely to become a policy issue in the near future.

D. Environmental Services of Organic Agriculture

Economic benefits from agriculture management cannot be limited to yield or commodities production, or account only for farm investment and revenue. For instance, issues such as energy efficiency and GHGs emissions, preserving water supply, biodiversity and landscape preservation and reduction in the use of agrochemicals are usually not assessed when conducting farming cost-benefit analyses. Still they play a key role for the long term sustainability of our support system and our environment, even if they have to be addressed on a broader spatial and temporal scale (Paoletti and Pimentel, 1992; Pimentel *et al.*, 1997; Tilman *et al*., 2001, 2002; Pretty *et al.*, 2003; FAO, 2004; Foley *et al.*, 2005; Millennium Ecosystem Assessment, 2005a; 2005b; Molden, 2007; Bosshard *et al*., 2009; Vitousek, *et al*., 2009).

It should be noted that organic agriculture provides many beneficial "by-products" both for the environment (e.g., conservation of soil fertility, CO_2 storage, fossil fuel reduction, preserving biodiversity) and for people (e.g., eliminating the use of agrochemicals such as synthetic fertilizers and pesticides, preserving landscape). We wish to stress that preserving or increasing soil organic matter content has to do not only with a farm long-term sustainability (and benefit), but, and maybe most importantly, with preserving a country's long term food security, guaranteeing that it can overcome and recover from possible future climate extremes.

In this sense it is important to get a deeper understanding of the nature of agroecosystems: they are embedded in complex ecological networks, characterized by nonlinearity and stochasticity. Theoretical and empirical research reveals that ecological systems persist and generate ecosystem services as a result of complex interacting components (Ehrlich and Ehrlich, 1981; Paoletti and Pimentel, 1992; Cliff, 1997; Pimentel *et al*., 1997; 2006; Loreau *et al*., 2002; Luck *et al*., 2009; Vandermeer *et al*., 2010). Benefits from insect services in the United States, for instance, are valued at $57 billion per year (Losey and Vaughan, 2006). But insect do not live in a vacuum, they are constrained by the environment-landscape characteristics. Eventually, benefits provided by insects depend on how we decide to manage the environment in which they may find their living from which they depend on. So, in order to fully benefit from ecosystems environmental services, we should manage our environmental at a broader scale than that of the single farm.

At the same time, economic analysis should take full account ("internalization") of the economic impact of conventional agriculture, addressing the issue of its long term sustainability (Pimentel *at al.*, 1995, 1997; Pretty *et al*., 2000, 2003; Buttel, 2003).

E. Organic Farming and Food Security

According to some authors organic agriculture can be a promising approach to sustain food security while decreasing the environmental impact of agriculture, especially in some developing countries (Pretty and Hine, 2001; Altieri, 2002; FAO, 2002, 2008; Pretty, 2002; van Veluw, 2006; Niggli *et al*., 2007, 2008; El-Hage Scialabba, 2007; Badgley *et al*., 2008; El-Hage Scialabba and Müller-Lindenlauf, 2010). In low input systems, and especially in arid and semi-arid areas where most of the food-insecure people live, organic systems are reported to greatly improve yields (Pretty and Hine, 2001; Pretty, 2002). Although for perennial cropping, such as coffee or banana, significant yield reductions are reported, under appropriate agroforestry system, the lower yields for the main crop are compensated by producing other foodstuff and goods (El-Hage Scialabba and Müller-Lindenlauf, 2010).

Some authors (e.g., Pretty and Hine, 2001; FAO, 2002, 2008; Halberg *et al*., 2006; Badgley and Perfecto, 2007; Badgley *et al*., 2007; El-Hage Scialabba, 2007; Niggli *et al.,* 2007, 2008) argue that organic agriculture could benefit developing countries because organic practices contribute considerably to increasing soil stability and resilience, an important factor in food supply stability, and also save water, another critical resource in many areas. The authors claim that the productivity of organic compared to conventional farming depends strongly on soil and climate conditions as well as on choice of crops being compared, and under less favorable soil conditions, organically managed crop yields equal those from conventional agriculture. Recent models of a hypothetical global food supply grown organically (Badgley, *et al*., 2007; Halberg, *et al*., 2006) indicates that organic agriculture could produce enough food on a global *per capita* basis for the current world population.

In their review, Badgley *et al*. (2007) compared yields of organic versus conventional or low-intensive food production for a global dataset of 293 examples and estimated the average yield ratio (organic vs. nonorganic) of different food categories for the developed and the developing world, and found that for most food categories the average yield ratio was slightly <1.0 for studies in the developed world and >1.0 for studies in the developing world. The authors found also that in developed countries average yield losses under organic management ranged from 0 to 20% (Badgley *et al*., 2007). Pretty and Hine (2001) surveyed 208 projects in developing tropical countries in which contemporary organic practices were introduced, and found that average yield increased by 5–10% in irrigated crops, and by 50–100% in rain-fed crops. Data from Pretty and Hine (2001) have been challenged by some authors (e.g., McDonald *et al*., 2005; Cassman, 2007; Hudson Institute, 2007; Hendrix, 2007), who dispute the correctness of both the accounting (they hold that, in some of the cases reported, pesticides may have been used) and comparative methods employed. Cassman (2007) criticizes both the findings and the approach to the problem of food security adopted by the supporters of organic farming, and argues that what is needed to produce 60% more food by 2050 to meet demand from growth in both population and income is ecological intensification of crop production systems rather than relying on the organic farming approach.

F. "Food Miles" Analysis

Most energy in the food system is post-production. Food processing, distribution, wholesale and retail can amount to two thirds of total energy expenditure (Pimentel and Pimentel, 2008; Smil, 2008). It has been estimated that in the United States, on-farm production amounts to approximately 20% of the total food system energy, with about 40% of this amount going into making chemical fertilizers and pesticides (Keoleian and Keoleian, 2000).

National and international trade results in increasing "food miles" (the distance that food travels from the field to the grocery store), which may lead to increasing the overall energy consumption and CO_2 emissions associated with a given product (Pimentel *et al*., 1973; Steinhart and Steinhart, 1974; DEFRA, 2005; Pretty *et al*., 2005; Schlich and Fleissner, 2005; Foster *et al*., 2006, Pimentel and Pimentel, 2008). To avoid such a problem, environmental groups and organic associations are advising consumers to consume locally produced food as part of environmentally friendly eating habits. This, however, may limit export of organic products from developing countries to western markets, reducing the income for poor farmers and the adoption of sustainable farming practices.

Some authors challenge such a claim as too simplistic a view, and make the point that agricultural products imported from far away may cause lower environmental impact than locally produced products, for example when the latter have to be kept stored in fridges for several months (e.g., fruits) (Wells, 2001; Saunders *et al*., 2006; Williams, 2007; El-Hage Scialabba and Müller-Lindenlauf, 2010). Saunders *et al*. (2006), for instance, report that in the case of dairy and sheep meat production, New Zealand is by far more energy efficient than the UK even including transport costs, twice as efficient in the case of dairy, and four times as efficient in case of sheep meat. Wells (2001) found that New Zealand dairy production was on average less energy intensive than in North America or Europe even though on-farm primary energy input had doubled in 20 years and energy ratio (outputs vs. inputs) had increased by 10%. Williams (2007) reports that Dutch CO_2 emissions for rose cultivation were about 6 times larger than producing them in Kenya and delivering the product to Europe.

VII. CONCLUSIONS

In the last century, intensive farming has successfully achieved high crop yields. On the other hand this came with a cost on the environmental side because of the high intensity of energy use (agrochemical, machinery, water pumping etc) and GHGs emissions, water consumption and the large use of agrochemicals, which, other than being costly in energy terms, have also detrimental effects on the health of organisms, humans included.

When comparing the performances of organic and conventional agricultural practices it has been shown that organic generally performs better or much better than conventional for a wide range of key indicators (Table 1). Such improved performances have been summarised in previous reviews such as Stölze *et al*. (2000), Stockdale *et al*. (2001); FAO (2002), Lotter *et al*. (2003), Shepherd *et al*. (2003), Kasperczyk and Knickel (2006), Niggli *et al*. (2007), Gomiero *et al*. (2008), as well as proven in long term monitoring trials (e.g., Reganold *et al*., 1987; Matson *et al*., 1997; Paoletti *et al*., 1998, Drinkwater *et al*., 1998; Mäder *et al.,* 2002a, 2002b; Pacini *et al.,* 2003; Pimentel *et al*., 2005; Badgley *et al*., 2007). However, it has to be pointed out that in some cases performance can vary according to specific crop species and crop patterns and in relation to the environmental context where agricultural activity is performed.

In the following section we provide some more detailed comments on the performances of organic agriculture on some key environmental issues. We will deal in particular with soil, biodiversity, energy and GHG emission.

Table 6 is an attempt to further develop the qualitative review efforts made by Stölze *et al*. (2000) and Lotter, (2003). Assessments are only indicative and no claim is made to provide weighted qualitative values of farming performance.

As pointed out by Pacini *et al.* (2003), the fact that in most cases organic farming systems perform better environmentally than conventional or integrated farming system, does not directly imply that they are sustainable when compared to the intrinsic carrying capacity and resilience of a given ecosystem. Comparison between organic and conventional (or other) farming systems is much needed, but to assess sustainability in the long term, proper comparisons have to be made taking into account the local (and global) carrying capacity of the agroecosystem.

To date, many studies prove organic farming to perform better in improving soil quality with respect to both biophysical and ecological properties. Organic farming prevents soil erosion, increases SOM (promoting soil biodiversity and soil health) and can reduce N leaching. Increases in SOM following the transition to organic management occur slowly. This has to be of concern when assessing the performances of farming systems under different management practices. Soil under organic management greatly increases their water holding capability and under drought conditions crops in organically managed systems produce higher yields than comparable crops managed conventionally. Adaptive measures to cope with climate change should treasure knowledge gained from organic farming. Local characteristics deserve attention, as agricultural practices should not be adopted blindly, but with much concern for specific local features. What may fit a given area may not be practicable with the same results in another (e.g., plain vs. sloping land).

Agriculture intensification results also in a dramatic simplification of landscape composition and in a sharp decline of biodiversity. This affects the functioning of natural pest control, as natural habitats provide shelter for a broad spectrum of natural species that operate as pest-control in agriculture crops. Organic farming tends to rely on a higher number of crops, compared to conventional, because of the very nature of the management system, involving rotation, cover crops, intercropping and set

TABLE 6

Overall qualitative assessment of organic farming systems relative to conventional farming*. (Organic farming performs: ++ much better, + better, 0 the same, − worse, – much worse).

Indicator − Performance	Qualitative Assessment				
	++	+	0	−	−−
Agronomic					
Productivity as yield per ha		+	0	−	−−
Productivity as yield per hr				−	−−
Biodiversity					
Crop diversity	++	+	0		
Floral diversity	++	+			
Aboveground faunal diversity (invertebrate and vertebrate)	++	+			
Habitat diversity	++	+	0		
Effect on pest control and pollinators	++	+			
Soil biophysical characteristics					
Organic matter	++	+	0		
Structure	++	+	0		
Soil biology	++	+	0	−	
Microbial biomass	++	+			
Microbial activity	++	+			
Mycorrhizae	++				
Biodiversity	++	+			
Effect on pest control	++	+	0		
Ground and surface water					
Nitrate leaching	++	+	0	−	
Pesticides	++				
Greenhouse emissions (including CO_2, CH_4, N_2O, NH_3)					
GHGs per ha	++	+			
GHGs per ton biomass		+	0	−	
Farm input and output					
Nutrient use		+			
Water use		+	0		
Energy use per ha	++	+			
Energy use per ton biomass		+	0	−	
Animal welfare and health					
Husbandry		+			
Health	++	+			
Quality of product food					
Pesticides residues	++	+			
Nitrate		+	0	−	
Mycotoxins		+	0	−	
Heavy metals		+	0	−	
Antibiotics	++				

(*): the list of indicators has been expanded from Stölze *et al*. (2000) and Lotter (2003), and quality assessment modified according to the data found by the present review.

aside. A more complex crop pattern offers more chances for "wild biodiversity" to thrive.

According to the studies reviewed, organic farming provides greater potential for biodiversity than its conventional counterpart, as a result of greater habitat variability and more wildlife-friendly management practices, and, to a lesser extent, due to the exclusion of pesticides. This greater potential is more readily observed primarily for wild plants, but also for their hosts. Indeed, an increasing body of evidence indicates that landscape heterogeneity is a key factor in promoting biodiversity in the agricultural landscape.

The effect of organic agriculture on promoting biodiversity may also vary according to the specific taxa and the surrounding conditions where a farm operates. Research indicates the need for long term, system-level studies of the biodiversity response to organic farming. It is noted that such benefits may be achieved also by conventional agriculture when carefully managed.

Promoting heterogeneity widely across agricultural systems should be a universal management objective. Large areas converted to organic management may generate positive feedbacks on biodiversity because of scale effect (the larger the areas the greater the benefits), suggesting that measures to preserve and enhance biodiversity in agroecosystems should be both landscape- and farm-specific.

Energy efficiency and GHG emission reduction are certainly important indicators of farming system performances. Organic farming has been shown to providing a better of energy input/output ratio. The main reasons for higher efficiency are lack of input of synthetic agrochemical (e.g., fertilizers, pesticides) and lower use of highly energy-consumptive foodstuffs (concentrates). However, due to the general lower yield of crops under organic farming, when calculating energy input in terms of unit of physical output, the advantage to organic systems was generally not as significant. Organic agriculture may represent a means for reducing GHG emission, both because of its lower energy consumption and of its soil management practices that help to reduce GHG emission and absorb carbon in soil. Conversion to organic agriculture, however, only represents a temporary solution to the problem of carbon abatement because the possibility to stock carbon in the soil has limits. Long-term solutions concerning CO_2 and GHG emission abatement should rely on a more general change of our development path, for instance in containing energy consumption in general. Other beneficial "by-products" provided by organic farming both for the environment (e.g., reducing pollution, fostering biodiversity) and for human health (e.g., exposure to harmful chemicals), also should be properly accounted for.

Carrying out extensive long-term trials for diverse crops in diverse areas would be of fundamental importance in order to understand the potential of organic farming as well as to improve farming techniques in general. Investing in organic farming research will help to gain knowledge and experience about best practices for agroecosystem management.

According to Niggli *et al*. (2008), there are three strategic research priorities for agricultural and food research:

- Viable concepts for the empowerment of rural economies in a regional and global context
- Securing food and ecosystems by means of eco-functional agricultural intensification
- High quality foods—a basis for healthy diets and a key for improving our quality of life and health.

Researching organic food and farming systems can contribute greatly towards the overall sustainability of agriculture and food production by providing a holistic analysis of system factor interactions and trade-offs in order to meet new challenges.

We would like to conclude by reminding each of us that we all depend inescapably on agriculture for our life. We feel that maybe there has been too much focus on agriculture as a mere economic activity, forgetting that, differently from all other economic activities, this is the only one that we cannot afford to dismiss or allow ourselves to lose.

ACKNOWLEDGMENTS

We wish to thank Nadia El-Hage Scialabba at Natural Resources Management and Environment Department at FAO, Rome, for her valuable comments, which helped to improve the manuscript. We also wish to thank Dr. Lucio Marcello, Glasgow Biomedical Research Centre, University of Glasgow, for helping to edit this manuscript. Part of this work has been carried out within the EU FP7 research project "Indicators for biodiversity in organic and low-input farming systems" (BIO-BIO KBBE-2008-1-2-01). The European Union or the European Union Commission cannot be held responsible for results and opinions quoted in the text.

REFERENCES

Alyokhin, A., Porter, G., Groden, E., and Drummond, F. 2005. Colorado potato beetle response to soil amendments: A case in support of the mineral balance hypothesis? *Agric. Ecosys. & Environ.* **109:** 234–244.

Altieri, M. 1987. *Agroecology: The Science of Sustainable Agriculture*. Westview Press, Boulder, NY, USA.

Altieri, M. 2002. Agroecology: The science of natural resource management for poor farmers in marginal environments. *Agric., Ecosys. and Environ.* **93:** 1–24.

Altieri, M., and Nicholls, C. I. 2004. *Biodiversity and Pest Management in Agroecosystems.* (2nd ed.) CRC Press, Boca Raton, FL, USA.

Anderson, K., Cockburn, J., and Martin, W. 2010. *Agricultural Price Distortions, Inequality, and Poverty*. World Bank Publications, Washington D.C.

Avery, A., and Avery, D. 2008. Beef production and greenhouse gas emissions. *Environmental Health Perspectives* **116:** 373–374.

Badgley, C., and Perfecto, I. 2007.Can organic agriculture feed the world? *Renewable Agriculture and Food Systems* **22:** 80–85.

Badgley, C., Moghtader, J., Quintero, E., Zakem, E., Chappell, J. M., Avilés-Vázquez, K., Samulon, A., and Perfecto, I. 2007. Organic agriculture and the global food supply. *Renewable Agriculture and Food Systems* **22:** 86–108.

Bahlai, C. A., Xue, Y., McCreary, C. M., Schaafsma, A. W., and Hallett, R. H. 2011. Choosing Organic Pesticides over Synthetic Pesticides May Not Effectively Mitigate Environmental Risk in Soybeans. *PlosOne* **5**(6): e11250.

Balfour, E. 1977. Towards a Sustainable Agriculture: The Living Soil. IFOAM conference, Switzerland, 1977. http://www.soilandhealth.org/01aglibrary/010116balfourspeech.html Accessed on 20 November 20009.

Barberi, P. 2002. Weed management in organic agriculture: Are we addressing the right issues? *Weed Research* **42:** 177–193.

Barbosa, P. 2003. *Conservation Biological Control*. Academic Press, San Diego, USA.

Bengtsson, J., Ahnstrom, J., and Weibull, A-C. 2005. The effects of organic agriculture on biodiversity and abundance: A meta-analysis. *J. Appl. Ecol.* **42:** 261–269.

Benton, T. G., Vickery, J. A., and Wilson, J. D. 2003. Farmland biodiversity: Is habitat heterogeneity the key? *Trends Ecol. Evol.* **18**:182–188.

Bianchi, F.J.J.A., Booij, C. J. H., and Tscharntke, T. 2006. Sustainable pest regulation in agricultural landscapes: A review on landscape composition, biodiversity and natural pest control. *Proc. R. Soc. B* **273:** 1715–1727.

Birkhofer, K., Bezemer, T. M., et al. 2008. Long-term organic farming fosters below and aboveground biota: Implications for soil quality, biological control and productivity. *Soil Biology & Biochemistry* **40:** 2297–2308.

Bosshard, A., Reinhard, B. R., and Taylor, S. (Eds.). 2009. *IFOAM Guide to Biodiversity and Landscape Quality in Organic Agriculture.* IFOAM http://www.ifoam.org/growing_organic/7_training/training_pdf/Biodiversity Book–2009-07-15-AZ-Teaser.pdf Accessed on 20 August 2009.

Bowman, G. 2010. USDA census shows profitability of organic farming. Rodale Institute. http://www.rodaleinstitute.org/20100319/nf_USDA-census-shows-profitability-of-organic-farming Accessed on 20 July 2010.

Briar, S. S., Grewal, P. S., Somasekhar, N., Stinner, D., and Miller, S. A. 2007. Soil nematode community, organic matter, microbial biomass and nitrogen dynamics in field plots transitioning from conventional to organic management. *Appl. Soil. Ecol.* **37:** 256–266.

Brittain, C., Bommarco, R., Vighi, M., Settele, J., and Potts, S. G. 2010. Organic farming in isolated landscapes does not benefit flower-visiting insects and pollination. *Biological Conservation* **143:** 1860–1867.

Büchs, W. (Ed.) 2003. Biotic indicators for biodiversity and sustainable agriculture. *Agric. Ecosys. & Environ*. **98,** Special Issue. Elsevier Amsterdam.

Butler, S. J., Vickery, J. A., and Norris, K. 2007. Farmland biodiversity and the footprint of agriculture. *Science* **315:** 381–384.

Buttel, F. H. 2003. Internalizing the societal costs of agricultural production. *Plant Physiol.* **133:** 1656–1665.

Caporali, F., Mancinelli, R., and Campiglia, E. 2003. Indicators of cropping system diversity in organic and conventional farms in Central Italy. *Int. J. Agric. Sust*. **1:** 67–72.

Cassman, K. 2007. Editorial response by Kenneth Cassman: Can organic agriculture feed the world-science to the rescue? *Renew. Agric. and Food Syst.* **22:** 83–84.

Cavigelli, M. A., Teasdale, J. R., and Conklin, A. E. 2007. Long-term agronomic performance of organic and conventional field crops in the Mid-Atlantic Region. *Agronomy J.* **100:** 785–794.

Cederberg, C., and Stadig, M. 2003. System expansion and allocation in life cycle assessment of milk and beef production. *Int. J. Life Cycle Assess*. **8:** 350–356.

Chandler, S., and Dunwell, J. M. 2008. Gene flow, risk assessment and the environmental release of transgenic plants. *Critical Reviews in Plant Sciences* **27:** 25–49.

Chavas, J-P, Posner, J. L., and Hedtck, J. L. 2009.Organic and conventional production systems in the wisconsin integrated cropping systems trial: II. Economic and risk analysis 1993–2006. *Agron. J*. **101**:288–295.

Chivian, E., and Bernstein, A. (Eds). 2008. *Sustaining Life: How Human Health Depends on Biodiversity.* Oxford University Press, New York.

Clark, M.S., Horwath, W. R., Shennan, C., and Scow, K. M. 1998. Changes in soil chemical properties resulting from organic and low-input farming practices. *Agronomy J.* **90:** 662–671.

Clark, M. S., Horwath,W. R., Shennan,C., Scow, K. M., Lantni, W.T., and Ferris, H. 1999. Nitrogen, weeds and water as yield-limiting factors in conventional, low-input, and organic tomato systems. *Agric. Ecosys & Environ.* **73:** 257–270.

Clark, S., Szlavecz, K., Cavigelli, M. A., and Purringto, F. 2006. Ground Beetle (Coleoptera: Carabidae) assemblages in organic, no-till, and chisel-till cropping systems in Maryland. *Environmental Entomology* **35:** 1304–1312.

Cliff., B. 1997. Economic and environmental benefits of biodiversity. *BioScience* **47:** 747–757.

Clough, Y., Kruess, A., and Tscharntke, T. 2007a. Local and landscape factors in differently managed arable fields affect the insect herbivore community of a non-crop plant species. *J. Appl. Ecol.* **44:** 22–28.

Clough, Y., Holzschuh, A., Gabriel, D., Purtauf, T., Kleijn, D., Kruess, A., Steffan-Dewenter, I., and Tscharntke, T. 2007b. Alpha and beta diversity of arthropods and plants in organically and conventionally managed wheat fields. *J. Appl. Ecol.* **44:** 804–812.

Codex Alimentarius. 2004. G*uidelines for the Production, Processing, Labelling and Marketing of Organically Produced Foods* (GL 32 – 1999, Rev. 1 – 2001). Internet http://www.codexalimentarius.net/web/standard_list.do?lang=en. Accessed on November 24, 2007.

Conford, P. 2001. *The Origins of the Organic Movement*. Floris Books, Glasgow, UK.

Cormack, W. F. 2000. *Energy Use in Organic Farming Systems*. Final report for project OF0182 for the Department for Environment, Food and Rural Affairs. http://orgprints.org/8169/01/OF0182 181 FRP.pdf. Accessed on December 12, 2008 from

Courville, S. 2006. Organic standards and certification. **In:** *Organic Agriculture. A Global Perspective.* pp. 201–220. Kristiansen, P., Taji, A., and Reganold, J., Eds., CSIRO Publishing, Collingwood, Australia.

Crowder, D. W., Northfield, T. D., Strand, M. R., and Snyder, W. E. 2010. Organic agriculture promotes evenness and natural pest control. *Nature* **466:** 109–112.

Czech, B., Krausman, P. R., and Devers, P. K. 2000. Economic associations among causes of species endangerment in the United States. *BioScience* **50:** 593–601.

Dalgaard, T., Halberg, N., and Porter, J. R. 2001. A model for fossil energy use in Danish agriculture used to compare organic and conventional farming. *Agric., Ecosys. & Environ.* **87:** 51–65.

De Boer, I.J.M. 2003. Environmental impact assessment of conventional and organic milk production. *Livestock Production Science* **80:** 69–77.

DEFRA (Department for Environment Food and Rural Affairs – UK). 2005. The validity of food miles as an indicator of sustainable development. Report number ED50254. Retrieved September 16 2007 from http://statistics.defra.gov.uk/esg/reports/foodmiles/default.asp.

De Gregori, T. R. 2003. *Origins of the Organic Agriculture Debate*. Wiley-Blackwell, Oxford.

Delate, K., and Cambardella, C. A. 2004. Agroecosystem performance during transition to certified organic grain production. *Agron. J.* **96:** 1288–1298.

Delate, K., Duffy, M., Chase, C., Holste, A., Friedrich, H., and Wantate, N. 2003. An economic comparison of organic and conventional grain crops in a long-term agroecological research (LTAR) site in Iowa. *J. of Altern. Agric.***18:** 59–69.

Döring, T. F., and Kromp, B. 2003.Which carabid species benefit from organic agriculture? A review of comparative studies in winter cereals from Germany and Switzerland. *Agric. Ecosys. & Environ*. **98:** 153–161.

Drinkwater, L. E., Wagoner, P., and Sarrantonio, M. 1998. Legume-based cropping systems have reduced carbon and nitrogen losses. *Nature* **396:** 262–265.

Duelli, P., and Obrist, M. K. 2003. Biodiversity indicators: The choice of values and measures. *Agric. Ecosys. & Environ.* **98:** 87–98.

EC (European Commission) 2007. Council Regulation (EC) No 834/2007, of 28 June 2007 on organic production and labelling of organic products and repealing Regulation (EEC) No 2092. Internet http://eurlex.europa.eu/LexUriServ/site/en/oj/2007/l_189/l_18920070720en 00010023.pdf. Accessed on 25 November 2009.

Edwards-Jones, G., and Howells, O. 2001. The origin and hazard of inputs to crop protection in organic farming systems: Are they sustainable? *Agricultural Sys.* **67:** 31–47.

Ehrlich, P., and Ehrlich, A. 1981. *Extinction: The Causes and Consequences of the Disappearance of Species*. Random House, New York.

Ekroos, J., Hyvönen, T., Tiainen, J., and Tiira, M. 2010. Responses in plant and carabid communities to farming practises in boreal landscapes. *Agric. Ecosys. & Environ*. **135:** 288–293.

El-Hage Scialabba, N. 2007. Organic agriculture and food security. International Conference on Agriculture and Food Security, FAO, Rome. ftp://ftp.fao.org/paia/organicag/ofs/OFS-2007-5.pdf. Accessed on 20 May 2010.

El-Hage Scialabba, N., and Müller-Lindenlauf, M. 2010. Organic agriculture and climate change. *Renewable Agriculture and Food Systems* **25:** 158–169.

Elliot, S.L., and Mumford, J.D. 2002. Organic, integrated and conventional apple production: Why not consider the middle ground? *Crop Protection* **21:** 427–429.

Enserink, M. 2010. Tough lesson from golden rice. *Science* **320:** 468–471.

Eyhorn, F., Ramakrishnan, M., and Mäder, P. 2007. The viability of cotton-based organic agriculture systems in India. *Int. J. Agric. Sust.* **5:** 25–38.

FAO. 2002. *Organic Agriculture, Environment and Food Security.* Environment and Natural Resources Service Sustainable Development Department. http://www.fao.org/DOCREP/005/Y4137E/y4137e00.htm#TopOfPage. Accessed on 12 January 2009.

FAO. 2004. *The Scope of Organic Aagriculture, Sustainable Forest Management and Ecoforestry in Protected Area Management*. FAO, Rome. http://www.fao.org/docrep/007/y5558e/y5558e00.htm#toc. Accessed on November 15, 2009.

FAO. 2006. *Livestock's Long Shadow*. FAO, Rome. ftp://ftp.fao.org/docrep/fao/010/A0701E/A0701E00.pdf Accessed on 20 November 2009.

FAO. 2008. The state of food insecurity in the developing world. FAO, Rome. http://www.fao.org/docrep/012/i0876e/i0876e00.HTM Accessed on January 20, 2010.

Flessa, H., Ruser, R., Dörsch, P., Kamp, T., Jimenez, M. A., Munch, J. C., and Beese, F. 2002. Integrated evaluation of greenhouse gas emissions (CO2, CH4, N2O) from two farming systems in southern Germany. *Agric. Ecosyst. & Environ.* **91:** 175–189.

Fließbach, A., and Mäder, P. 2000. Microbial biomass and size-density fractions differ between soils of organic and conventional agricultural systems. *Soil Biol. Biochem*. **32:** 757–768.

Fließbach, A, Mäder, P., and Niggli, U. 2000. Mineralization and microbial assimilation of ^{14}C-labeled straw in soils of organic and conventional agricultural systems. *Soil Biology & Biochem.* **32:** 1131–1139.

Fließbach, A., Oberholzer, H. R., Gunst, L., and Mäder, P. 2007. Soil organic matter and biological soil quality indicators after 21 years of organic and conventional farming. *Agric. Ecosyst. & Environ.* **118:** 273–284.

Fluck, R. C. and Baird, C. D. 1980. *Agricultural Energetics*. Avi Publishing Company Inc., Westport, CT, USA.

Foereid, B., and Høgh-Jensen, H. 2004. Carbon sequestration potential of organic agriculture in northern Europe – a modelling approach. *Nutr. Cycl. in Agroecosyst.* **68:** 13–24.

Foley, J. A., DeFries, R., Asner, G. P., Barford. C., and fifteen other. 2005. Global consequences of land use. *Science* **309:** 570–574.

Foster, C., Green, K., Bleda, M., Evans, B., Flynn, A., and Myland, J. 2006. *Environmental Impact of Food Production and Consumption*. A report to the Department of Environment, Food and Rural Affair (DEFRA). Manchester Business School, DEFRA, London. http://www.defra.gov.uk/science/project_data/DocumentLibrary/EV02007/EV02007_4601_FRP.pdf Accessed on 20 June 2010.

Fuller R. J., Norton L. R., Feber R. E., Johnson P. J., Chamberlain D. E., Joys A. C., Mathews F., Stuart R. C., Townsend M. C., Manley W. J., Wolfe M. S., Macdonald D. W., and Firbank L. G. 2005. Benefits of organic farming to biodiversity vary among taxa. *Biol. Lett.* **1:** 431–434.

Gabriel, D., Roschewitz, I., Tscharntke, T., and Thies, C. 2006. Beta diversity at different spatial scales: Plant communities in organic and conventional agriculture. *Ecol. Appl.* **16:** 2011–2021.

Gabriel, D., Sait, S. M., Hodgson, J. A., Schmutz, U., Kunin, W. E., and Benton, T. G. 2010. Scale matters: The impact of organic farming on biodiversity at different spatial scales. *Ecology Letters* **13:** 858–869.

Gardiner, M. M., Landis, D. A., Gratton, C., DiFonzo, C. D., O'Neal, M., Chacon, J. M., Wayo, M. T., Schmidt, N. P., Mueller, E. E., Heimpel, G. E. 2009. Landscape diversity enhances biological control of an introduced crop pest in the north-central USA. *Ecological Applications* **19:** 143–154.

Gaston, K. J., and Spicer, J. I. 2004. *Biodiversity. An Introduction*. Blackwell Publishing, Oxford, UK.

Gelfand, I., Snapp, S., and Robertson, G. P. 2010. Energy efficiency of conventional, organic, and alternative cropping systems for food and fuel at a site in the U.S. Midwest. *Environmental Science & Technology* **44:** 4006–4011.

Giampietro, M. 2004. *Multi-Scale Integrated Analysis of Agroecosystems*. CRC Press, Boca Raton, London.

Giampietro, M. and Mayumi, K. 2009. *The Biofuel Delusion: The Fallacy of Large Scale Agro-biofuels Production*. Earthscan, London.

Giampietro, M., Pimentel, D., and Cerretelli, G. 1992. Energy analysis of agricultural ecosystem management: Human return and sustainability. *Agric. Ecosyst. & Environ.* **38:** 219–244.

Giampietro, M. and Ulgiati, S. 2005. An integrated assessment of large-scale biofuel production. *Critical Review in Plant Sciences* **24:** 1–20.

Gibson, R. H., Pearce, S., Morris, R. J., Symondson, W. O. C., and Menmott, J. 2007. Plant diversity and land use under organic and conventional agriculture: A whole-farm approach. *J. Appl. Ecol.* **44:** 792–803.

Gliessman, S. R. 2007. *Agroecology: Ecological Processes in Sustainable Agriculture*. (2nd ed.) Lewis Publisher, Boca Raton, New York.

Global Footprint Network (2009). World Footprint: Do we fit on the planet? http://www.footprintnetwork.org/en/index.php/GFN/page/world_footprint/ Accessed on the 23 April 2010.

Glover, J. D., Reganold, J. P., and Andrews, P. K. 2000. Systematic method for rating soil quality of conventional, organic,and integrated apple orchards in Washington State. *Agric. Ecosyst. Environ.* **80:** 29–45.

Gold, M. V., and Gates J. P. 2007. Tracing the evolution of organic/sustainable agriculture: A selected and annotated bibliography. http://www.nal.usda.gov/afsic/pubs/tracing/tracing.shtml Accessed on 20 January 2010.

Gomiero, T., Giampietro, M., and Mayumi, K. 2006. Facing complexity on agro-ecosystems: a new approach to farming system analysis. *Int. J. of Agric. Res. Gov. and Ecol.* **5:** 116–144.

Gomiero, T., Paoletti, M. G., Pimentel, D. 2008. Energy and environmental issues in organic and conventional agriculture. *Crit. Rev. Plant Sci.* **27:** 239–254.

Gomiero, T., Paoletti, M.G. and Pimentel, D. 2010. Biofuels: Ethics and concern for the limits of human appropriation of ecosystem services. *Journal of Agriculture and Environmental Ethics* **23:** 403–434.

Gold, M. V. 2007. Organic production. U.S. Department of Agriculture http://www.ers.usda.gov/data/organic/ Accessed on 20 June 2010.

Gotelli, N. J., and Colwell, R. K. 2001. Quantifying biodiversity: Procedures and pitfalls in the measurement and comparison of species richness. *Ecol. Lett.* **4:** 379–391.

Grandy, S. A., and Robertson, G. P. 2007. Land-use intensity effects on soil. *Ecosystems* **10:** 58–73.

Green, R. E., Cornell, S. J., Scharlemann, J. P. W., and Balmford, A. 2004. Farming and the fate of wild nature. *Science* **307:** 550–555.

Gunapala, N., and Scow, K. M. 1998. Dynamics of soil microbial biomass and activity in conventional and organic farming systems. *Soil Biol. Biochem.* **30:** 805–816.

Gündomuþ, E. 2006. Energy use on organic farming: A comparative analysis on organic versus conventional apricot production on small holdings in Turkey. *Energy Conservation and Management* **47:** 3351–3359.

Guthman, J. 2004. *Agrarian Dreams: The Paradox of Organic Farming in California*. University of California Press, Los Angeles.

Guzmán, G. I., and Alonso, A. M. 2008. A comparison of energy use in conventional and organic olive oil production in Spain. *Agricultural Systems* **98:** 167–176.

Haas, G., Wetterich, F., and Kopke, U. 2001. Comparing intensive, extensified and organic grassland farming in southern Germany by process life cycle assessment. *Agric., Ecosys. and Environ.* **83:** 43–53.

Hajjar, R., Jarvis, D. I., and Gemmill-Herren, B. 2008. The utility of crop genetic diversity in maintaining ecosystem services. *Agr. Ecosyst. & Environ.* **123:** 261–270.

Halberg, N., Alroe, H. F., Knudsen, M. T., and Kristensen, E. S. (Eds.) 2006. *Global Development of Organic Agriculture: Challenges and Prospects*. CABI Publishing.

Hamilton, G. 2000. When good bugs turn bad. *New Scientist* **15 January:** 30–33.

Hansen, B., Alrøe, H. F., and Kristensen, E. S. 2001. Approaches to assess the environmental impact of organic farming with particular regard to Denmark. *Agric. Ecosyst. & Environ.* **83:** 11–26.

Hawesa, C., Squirea, G. R., Hallett, P. D., Watsonb, C. A., and Young, M. 2010. Arable plant communities as indicators of farming practice. *Agric. Ecosyst. & Environ.* **138:** 17–26.

Heckman, J. 2006. A history of organic farming transitions from Sir Albert Howard's war in the soil to the USDA National Organic Program. *Renewable Agriculture and Food Systems* **21:** 143–150.

Hendrix, J. 2007. Editorial response by Jim Hendrix. *Renew. Agric. and Food Syst.* **22:** 84–85.

Hillel, D. 1991. *Out of the Earth. Civilization and the Life of the Soil.* California Univ. Press, CA.

Hoeppner, J. W., Entz, M. H., McConkey, B. G., Zentner, R. P., and Nagy, C. N. 2005. Energy use and efficiency in two Canadian organic and conventional crop production systems. *Renewable Agriculture and Food Systems* **21:** 60–67.

Hole, D. G., Perkins A. J., Wilson J. D., Alexander I. H., Grice P. V., and Evans A. D. 2005. Does organic farming benefit biodiversity? *Biol. Conserv*. **122**:113–130.

Holland, J. M. 2004. The environmental consequences of adopting conservation tillage in Europe: reviewing the evidence. *Agric., Ecosys. and Environ.* **103:** 1–25.

Holzschuh, A., Steffan-Derwenter, I., Kleijn, D., amd Tscharntke, T. 2007. Diversity of flower-visiting bees in cereal fields: Effects of farming system, landscape composition and regional context. *J. Appl. Ecol.* **44:** 41–49.

Holzschuh, A., Steffan-Dewenter, I., and Tscharntke, T. 2008. Agricultural landscapes with organic crops support higher pollinator diversity. *Oikos* **117:** 354–361.

Howard, A. 1943. *An Agricultural Testament*. Oxford University Press, London. Available on internet at URL: http://journeytoforever.org/farm_library/howardAT/ATtoc.html

Hsu, Y.-T., Shen, T.-C., and Hwang S.-Y. 2009. Soil fertility management and pest responses: A comparison of organic and synthetic fertilization. *J. Econ. Entomol.* **102:** 160–169.

Hyvönen, T. 2007. Can conversion to organic farming restore the species composition of arable weed communities? *Biological Conservation* **137:** 328–390.

Hyvönen, T., Ketoj, E., Salonen, J., Jalli, H., and Tiainen, J. 2003. Weed species diversity and community composition in organic and conventional cropping of spring cereals. *Agric. Ecosys. & Environ*. **97:** 131–149.

Hyvönen, T., and Salonen, J. 2002. Weed species diversity and community composition in cropping practices at two intensity levels – a six-year experiment. *Plant Ecology* **154:** 73–81.

Hudson Institute 2007. *"Organic Abundance" Report: Fatally Flawed*. http://www.cgfi.org/cgficommentary/organic-abundance-report-fatallyflawed/. Accessed on 2 May 2010.

IFOAM (International Movement of Organic Agriculture Movements), 2008. *The World of Organic Agriculture - Statistics and Emerging Trends 2008*. International Federation of Organic Agriculture Movements -IFOAM, Bonn, Germany.

IFOAM (International Movement of Organic Agriculture Movements), 2010. Definition of Organic Agriculture. IFOAM http://www.ifoam.org/growing_organic/definitions/doa/index.html, and http://www.ifoam.org Accessed on 10 March 2010.

IPCC (Intergovernmental Panel on Climate Change). 2000. *Emissions Scenarios*. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. Nakicenovic, N., and Swart, R., Eds. Available on internet at URL http://www.ipcc.ch/ipccreports/sres/emission/index.php?idp=0 Accessed on 20 May 2010.

IPCC (Intergovernmental Panel on Climate Change). 2007. *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change.* Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (Eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. Available on internet at URL http://www.ipcc.ch/. Accessed on 20 May 2010.

Janzen, H. H. 2004. Carbon cycling in earth systems—a soil science perspective. *Agric., Ecosys. & Environ.* **104:** 399–417.

Kasperczyk, N., and Knickel, K. 2006. Environmental impact of organic agriculture. **In:** *Organic Agriculture. A Global Perspective*. pp. 259–294. Kristiansen, P., Taji, A., Reganold, J., Eds., CSIRO Publishing, Collingwood, Australia.

Kennedy, T. A., Naeem, S., Howe, K. M., Knops, J. M. H., Tilman, D., and Reich, P. 2002. Biodiversity as a barrier to ecological invasion. *Nature* **417:** 636–638.

Keoleian, H. and Keoleian, G. A. 2000. *Life Cycle-Based Sustainability Indicators for Assessment of the U.S. Food System*. The University of Michigan, Center for Sustainable Systems, Ann Arbor, MI. http://css.snre.umich.edu/css_doc/CSS00-04.pdf Accessed on 20 June 2010.

Kern, J. S., and Johnson, M. G. 1993. Conservation tillage impacts on national soil and atmospheric carbon levels. *Soil Sci. Soc. Am.* **57:** 200–210.

Kirchmann, H., Bergström, L., Kätterer, T., Mattsson, L., and Gesslein, S. 2007. Comparison of long-term organic and conventional crop–livestock systems on a previously nutrient-depleted soil in Sweden. *Agron. J.* **99:** 960–972.

Kirchmann, H., and Thorvaldsson, D. 2000. Challenging targets for future agriculture. *European Journal of Agronomy* **12:** 145–161.

Koepf, H. H. 1976. *Biodynamic Agriculture: An Introduction.* The Anthroposofic Press, Spring Valley, New York. Available at URL: http://continuousharvest.com/Documents/Bio-Dynamic%20Agriculture%20book.pdf

Koepf, H. H. 2006. *The Biodynamic Farm*. Steiner Books, Dulles, VA.

Koh, L. P., Dunn, R. R., Sodhi, N. S., Colwell, R. K., Proctor, H. C., and Smith, V. S. 2004. Species coextinctions and the biodiversity crisis. *Science* **305:** 1632–1634.

Koneswaran, G. and Nierenberg, D. 2008a. Global farm animal production and global warming: impacting and mitigating climate change. *Environ. Health Perspect*. **116:** 578–582.

Koneswaran, G. and Nierenberg, D. 2008b. Beef production: Koneswaran and Nierenberg respond. *Environ. Health Perspect*. **116:** 365–376.

Kramer, S. B., Reganold, J. P., Glover, J. D., Bohannan, B.J.M., and Mooney, H. A. 2006. Reduced nitrate leaching and enhanced denitrifier activity and efficiency in organically fertilized soils. *PNAS* **103:** 4522–4527.

Krebs, J. R., Wilson, J. D., Bradbury, R. B., and Siriwardena, G. M. 1999. The second Silent Spring? *Nature* **400:** 611–612.

Kristiansen, P., Taji, A., and Reganold, J. (Eds.) 2006. *Organic Agriculture. A Global Perspective*. CSIRO Publishing, Collingwood, Australia.

Kruess, A., and Tscharntke, T. 1994 Habitat fragmentation, species loss, and biological control. *Science* **264:** 1581–1584.

Küstermann, B., Kainz, M., and Hülsbergen, K.-J. 2008. Modeling carbon cycles and estimation of greenhouse gas emissions from organic and conventional farming systems. *Renewable Agriculture and Food Systems* **23:** 38–52.

Küstermann, B., Christen, O., and Hülsbergen, K-J. 2010. Modelling nitrogen cycles of farming systems as basis of site- and farm-specific nitrogen management. *Agric. Ecosys. & Environ.* **135:** 70–80.

Lal, R. 2004. Soil carbon sequestration impact on global climate and food security. *Science* **304:** 1623–1627.

Lal, R. 2010. Enhancing eco-efficiency in agro-ecosystems through soil carbon sequestration. *Crop Sci.* **50:** 120–131.

Lamine, C., and Bellon, S. 2009. Conversion to organic farming: A multidimensional research object at the crossroads of agricultural and social sciences. A review. *Agron. Sustain. Dev.***29:** 97–112.

Lampkin, N. 2002. *Organic Farming.* (Revised edition). Old Pond Publishing, Suffolk, UK.

Lavelle, P., and Spain, A. V. 2002. *Soil Ecology*. Kluwer Academic Publishers, Amsterdam.

Letourneau, D. K. and Goldstein, B. 2001. Pest damage and arthropod community structure in organic vs. conventional tomato production in California. *J. Appl. Ecol.* **38:** 557–570.

Letourneau, D. K., and Bothwell, S. G. 2008. Comparison of organic and conventional farms: challenging ecologists to make biodiversity functional. *Frontiers in Ecology and the Environment* **6:** 430–438.

Liebman, M., Gibson, L. R., et al. 2008. Agronomic and economic performance characteristics of conventional and low-external-input cropping systems in the Central Corn Belt. *Agron. J.* **100:** 600–610.

Liu, B. Tu, C., Hu, S., Gumpertz, M., and Ristaino, J. B. 2007. Effect of organic, sustainable, and conventional management strategies in grower fields on soil physical, chemical, and biological factors and the incidence of Southern blight. *Appl. Soil Ecol.* **37:** 202–214.

Lockeretz, W. (Ed.) 2007. *Organic Farming: An International History*. CABI, Cambridge, MA.

Lockeretz, W., Shearer, G., and Kohl, D. H. 1981. Organic farming in the Corn Belt. *Science* **211:** 540–546.

Lohr, L., and Park, T. A.,2007. Efficiency analysis for organic agricultural producers: The role of soil-improving inputs. *J. Environ. Manag.* **83:** 25–33.

Loreau, M., Naeem, S., and Inchausti, P. (Eds). 2002. *Biodiversity and Ecosystem Functioning: Synthesis and Perspectives*. Oxford University Press, Oxford.

Losey, J. E., and Vaughan, M. 2006. The economic value of ecological services provided by insects. *BioScience* **56:** 311–323.

Lotter, D. W. 2003. Organic agriculture. *J. of Sust. Agric.* **21:** 59–128.

Lotter, D. W., Seidel, R., and Liebhart, W. 2003. The performance of organic and conventional cropping systems in an extreme climate year. *Am. J. Alternative Agriculture* **18:** 146–154.

Luck, G. W., Harrington, R., et al. 2009. Quantifying the contribution of organisms to the provision of ecosystem services. *BioScience* **59:** 223–235.

Lund, V. 2006. Natural living—a precondition for animal welfare in organic farming. *Livestock Science* **100:** 71– 83.

Lynch, D. H., MacRae, R., and Marti, R. C. 2011. The Carbon and global warming potential impacts of organic farming: Does it have a significant role in an energy constrained world? *Sustainability* **3:** 322–362.

Mäder, P., Fließbach, A., Dubois, D., Gunst, L., Fried, P., and Niggli, U. 2002a. Soil fertility and biodiversity in organic farming. *Science* **296:** 1694–1697.

Mäder, P., Fließbach, A., Dubois, D., Gunst, L., Fried, P., and Niggli, U. 2002b. The ins and outs of organic farming. *Science* **298:** 1889–1890.

Mahoney P. R., Olson, K. D., Porter, P. M., Huggins, D. R., Perillo, C. A., and Crookston, C. K. 2004. Profitability of organic cropping systems in Southwestern Minnesota. *Renewable Agriculture and Food Systems* **19:** 35–46.

Marinari, S., Mancinelli, R., Campiglia, E., and Grego, S. 2006. Chemical and biological indicators of soil quality in organic and conventional farming systems in Central Italy. *Ecological Indicators* **6:** 701–711.

Marriott, E. E., and Wander, M. 2006. Qualitative and quantitative differences in particulate organic matter fractions in organic and conventional farming systems. *Soil Biol. & Biochem.* **38:** 1527–1536.

Mathieu, O., Lévêque, J., Hénault, C., Milloux, M.-J., Bizouard, F., and Andreux, F. 2006. Emissions and spatial variability of N2O, N2 and nitrous oxide mole fraction at the field scale, revealed with 15N isotopic techniques. *Soil Biol. & Biochem.* **38:** 941–951.

Matson, P. A., Naylor, R. L., and Ortiz-Monasterio, I. 1998. Integration of environmental, agronomic, and economic aspect of fertilizer management. *Science* **280**:112–115.

Matson, P. A., Parton, W. J., Power, A. G., and Swift, M.J. 1997. Agricultural intensification and ecosystem properties. *Science* **277:** 504–509.

McCandless, M., Venema, H. D., Barg, S., and Oborne, B. 2008. *Full Cost Accounting for Agriculture*. International Institute for Sustainable Development. http://www.iisd.org/pdf/2008/measure_fca_2008.pdf Accessed on 20 January 2010.

McDonald, A. J., Hobbs, P. R., and Riha, S. J. 2005. Does the system of rice intensification outperform conventional best management? A synopsis of the empirical record. *Field Crops Res.* **96:** 31–36.

Millennium Ecosystem Assessment. 2005a. *Millennium Ecosystem Assessment: Vol. 2, Scenario assessment*. http://www.millenniumassessment.org/en/Scenarios.aspx Accessed 25 November 2009.

Millennium Ecosystem Assessment. 2005b. *Ecosystems and Human Well-Being: Synthesis*. Island Press, Washington, DC. http://www.millenniumassessment.org/en/Synthesis.aspx Accessed 25 November 2009.

Molden, D. (Ed.). 2007. *Water for Food, Water for Life. A Comprehensive Assessment of Water Management in Agriculture*. Earthscan, London. The publication is available on internet at URL: http://www.iwmi.cgiar.org/assessment/Publications/books.htm

Mols, C. M. M., and Visser, M. E. 2002. Great Tits can reduce caterpillar damage in apple orchards. *J. Appl. Ecol.* **39:** 888–899.

Mols, C. M. M., and Visser, M. E. 2007. Great Tits (*Parus major*) reduce caterpillar damage in commercial apple orchards. *PLoS ONE* **2:** e202.

Montgomery, D. R. 2007a. Soil erosion and agricultural sustainability. *PNAS* **104**:13268–13272.

Montgomery, D. R. 2007b. *Dirt: The Erosion of Civilization*. University of California Press, Berkeley.

Moreby, S. J., Aebischer, N. J., Southway, S. E., and Sotherton, N. W. 1994. A comparison of the flora and arthropod fauna of organically and conventionally grown winterwheat in southern England. *Annals Applied Biol.* **125:** 13–27.

Morison, J. I. L., Baker, N. R., Mullineaux, P. M., and Davies, W. J. 2008. Improving water use in crop production. *Phil. Trans. B* **363:** 639–658.

Müller-Lindenlauf, M., Deittert, C., and Köpke, U. 2010. Assessment of environmental effects, animal welfare and milk quality among organic dairy farms. *Livestock Science* **128:** 140–148.

Murphy, J. D. 2007. *People, Plant & Genes*. Oxford University Press, New York.

Nemes, N. 2009. Comparative analysis of organic and non-organic farming systems: A critical assessment of farm profitability. Food and Agriculture Organization of the United Nations, Rome, June 2009. ftp://ftp.fao.org/docrep/fao/011/ak355e/ak355e00.pdf

Niggli, U., Earley, J., and Ogorzalek, K. 2007. Organic agriculture and environmental stability of the food supply. International Conference on Agriculture and Food Security, FAO Rome. ftp://ftp.fao.org/docrep/fao/meeting/012/ah950e.pdf Accessed 24 October 2009.

Niggli U., Slabe A., Schmid O., Halberg N., and Schlüter M. 2008. Vision for an organic food and farming research agenda to 2025 organic knowledge for the future. IFOAM http://www.tporganics.eu/upload/TPOrganics_VisionResearchAgenda.pdf Accessed on 20 August 2009.

Niggli, U., Fließbach, A., Hepperly, P. and Scialabba, N. 2009. Low greenhouse gas agriculture: mitigation and adaptation potential of sustainable farming systems. FAO, Rome. April 2009, Rev. 2 – 2009. ftp://ftp.fao.org/docrep/fao/010/ai781e/ai781e00.pdf Accessed on 20 May 2010.

Noe, E., Halberg, N, and Reddersen, J. 2005. Indicators of biodiversity and conservational wildlife quality on Danish organic farms for use in farm management: A multidisciplinary approach to indicator development and testing. *J. Agric. and Environ. Ethics* **18:** 383–414.

NRC (National Research Council). 2010. *Toward Sustainable Agricultural Systems in the 21st Century.* National Academies Press http://www.nap.edu/catalog/12832.html Accessed on 20 March 2010.

Norton, L., Johnson, P., Joys, A., Stuart, R., et al. 2009. Consequences of organic and non-organic farming practices for field, farm and landscape complexity. *Agric. Ecosys & Environ.* **129:** 221–227.

Organic World, 2008. http://www.organic-world.net/. Accessed on March 16, 2008.

Pacini, C., Wossink, A., Giesen, G., Vazzana, C., and Huirne, R. 2003. Evaluation of sustainability of organic, integrated and conventional farming systems: a farm and field-scale analysis. *Agric. Ecosys & Environ* **95:** 273–288.

Paoletti, M. G. 1999. Using bioindicators based on biodiversity to assess landscape sustainability. *Agric. Ecosys & Environ.* **74:** 1–18.

Paoletti, M. G., and Lorenzoni, G. G. 1989. Agroecology patterns in Northeastern Italy. *Agric. Ecosys. & Environ.* **27:** 139–154.

Paoletti, G. M., and Pimentel, D. (Eds). 1992. *Biotic Diversity in Agroecosystems*. Elsevier, Amsterdam.

Paoletti, M. G., Schweigl, U., and Favretto, M. R.1995. Soil macroinvertebrates, heavy metals and organochlorines in low and high input apple orchards and a coppiced woodland. *Pedobiologia* **39:** 20–33.

Paoletti, M. G., Boscolo, P., and Sommaggio, D. 1997. Predators-parasitoids and beneficial insects in fields sorrounded by hedgerows in North Eastern Italy. *Biological Agriculture and Horticulture* **15:** 311–323.

Paoletti, M. G., Thomson, L. J., and Hoffmann, A. (Eds.) 2007a. Using invertebrate bioindicators to assess agricultural sustainability in Australia: Proposals and current practices. *Experimental Journal of Australian Agriculture* **47**(special issue): 379–478.

Paoletti M. G., Tsitsilas A., Thomson L. J., Taiti S., Umina, P. A. 2007b. The flood bug, *Australiodillo bifrons* (Isopoda: Armadillidae): A potential pest of cereals in Australia? *Appl. Soil Ecol.* **39**:76–83.

Paoletti, M. G., Sommaggio, D., Favretto, M. R., Petruzzelli, G., Pezzarossa, B., and Barbafieri, M. 1998. Earthworms as useful bioindicators of agroecosystem sustainability in different input orchards. *Appl. Soil Ecol.* **10:** 137–150.

Pelletier, N., Arsenault, N., and Tyedmers, P. 2008. Scenario modeling potential eco-efficiency gains from a transition to organic agriculture: Life cycle perspectives on Canadian canola, corn, soy, and wheat production. *Environ. Manage.* **42:** 989–1001.

Perfecto, I., Vandermeer, J. H., Bautista, G. L., Nunez, G. I., Greenberg, R., Bichier, P., and Langridge, S. 2004. Greater predation in shaded coffee farms: The role of resident neotropical birds. *Ecology* **85:** 2677–2681.

Petersen, S. O., Regina, K., Pöllinger, A., Rigler, E., Valli, L., Yamulki, S., Esala, M., Fabbri, C., Syväsalo, E., and Vinther, F. P. 2006. Nitrous oxide emissions from organic and conventional crop rotations in five European countries. *Agric. Ecosys. & Environ.* **112:** 200–206.

Pfiffner, L., and Niggli, U. 1996. Effects of bio-dynamic, organic and conventional farming on ground beetles (Col. Carabidae) and other Epigaeic arthropods in winter wheat. *Biological Agriculture and Horticulture* **12:** 353–364.

Pfiffner, L. and Mäder, P. 1997. Effects of biodynamic, organic and conventional production systems on earthworm populations. Entomological research in organic agriculture. *Biological Agriculture and Horticulture* **15:** 3–10.

Pfiffner, L., and Luka, H. 2003. Effects of low-input farming systems on carabids and epigeal spiders—a paired farm approach. *Basic Appl. Ecol.* **4:** 117–127.

Pfiffner, L., and Luka, H. 2007. Earthworm populations in two low-input cereal farming systems. *Appl. Soil Ecol.* **37:** 184–191.

Pfiffner, L., Häring, A., Dabbert, S., Stolze, M., and Piorr, A. 2001. Contributions of organic farming to a sustainable environment. http://orgprints.org/2943/1/pfiffner-2001-proceedings-copenhagen.pdf Accessed on 20 May 2010.

Phelan, P. L. 2009. Ecology-based agriculture and the next green revolution. **In:** *Sustainable Agroecosystem Management,* pp. 97–135. Bohlen, P. J., and House, G., Eds, CRC Press, Boca Raton, FL.

Phelan, P. L., Mason, J. F., and Stinner, B.R. 1995. Soil-fertility management and host preference by European corn borer, *Ostrinia nubilalis* (Hübner), on *Zea mays* L.: A comparison of organic and conventional chemical farming. *Agric. Ecosys. & Environ.* **56:** 1–8.

Phelan, P. L., Norris, K. H., and Mason, J. F. 1996. Soil-management history and host preference by *Ostrinia nuhilalis:* Evidence for plant mineral balance mediating insect-plant interactions. *Environ. Entomol*. **25**:1329–1336.

Pimentel, D. 1989. Energy flow in food system. **In:** *Food and Natural Resources*. pp. 1–24. Pimentel, D., and Hall, C. W. Eds. Academic Press, New York.

Pimentel, D. 1997. *Technique for Reducing Pesticides Use. Economic and Environmental Benefits*. J.Wiley & Sons, New York.

Pimentel, D. 2006. *Impacts of Organic Farming on the Efficiency of Energy Use in Agriculture: An Organic Center State of Science Review*. (The Organic Center). Internet http://www.organic-center.org/reportfiles/ENERGY_SSR.pdf. Accessed on 12 January 2008.

Pimentel, D. and Patzek, T. 2005. Ethanol production using corn, switchgrass, and wood: biodiesel production using soybean and sunflower. *Natural Resources Research* **14:** 65–76.

Pimentel, D., and Pimentel, M. 2008. *Food, Energy, and Society. 3^{d}* ed. CRC Press, Boca Raton, FL.

Pimentel, D., Berardi, G., and Fast, S. 1983. Energy efficiency of farming systems: Organic and conventional agriculture. *Agric. Ecosys. & Environ.* **9:** 359–372.

Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M, Crist, S., Sphpritz, L., Fitton, L., Saffouri, R., and Blair, R. 1995. Environmental and economic costs of soil erosion and conservation benefits. *Science* **267:** 1117–1123.

Pimentel, D., Hepperly, P., Hanson, J., Douds, D., and Seidel, R. 2005. Environmental, energetic, and economic comparisons of organic and conventional farming systems. *BioScience* **55:** 573–582.

Pimentel, D., Hurd, E., Bellotti, A. C., Forster, M. J., Oka, I. N., Sholes, O.D., and Whitman, R. J. 1973. Food production and the energy crisis. *Science* **182:** 443–449.

Pimentel, D., Petrova, T., Riley, M., Jacquet, J., Ng, V., Honigman, J., and Valero, E. 2006. Conservation of biological diversity in agricultural, forestry, and marine systems. **In:** *Focus on Ecology Research*. pp. 151–173, Burk, A. R., Ed., Nova Science Publishers, New York. Also, 2007. **In:** *Focus on Biodiversity Research*. pp. 1–25. Schwartz, J., Ed., Nova Science Publishers, New York.

Pimentel, D., Stachow, U., Takacs, D. A., Brubaker, H. W., Dumas, A. R., Meaney, J. J., O'Neill, J. A. S., Onsi, D. E. and Corzelius, D. B. 1992 Conserving biological diversity in agricultural/forestry systems. *Bioscience* **432:** 354–362.

Pimentel, D., Wilson, C., McCullum, C., Huang, R., Dwen, P., Flack, J., Tran, Q., Saltman, T., and Cliff, B., 1997. Economic and environmental benefits of biodiversity. *BioScience* **47:** 747–757.

Pocock, M.J. O., and Jennings, N. 2007. Testing biotic indicator taxa: The sensitivity of insectivorous mammals and their prey to the intensification of lowland agriculture. *J. Appl. Ecol.***45:** 151–160.

Pretty, J. 2002. Lessons from certified and non certified organic projects in developing countries. **In:** *Organic Agriculture, Environment and Food Security*. pp. 139–162. El-Hage Scialabba, N., and Hattam, C., Eds., FAO, Rome, Italy.

Pretty, J., and Hine, R. 2001. *Reducing Food Poverty with Sustainable Agriculture: A Summary of New Evidence*. Final report from the 'SAFE World' Research Project, University of Essex. Internet http://www.essex.ac.uk/ces/esu/occasionalpapers/SAFE%20FINAL%20-%20Pages1-22.pdf. Accessed on 12 February 2010.

Pretty, J. N., Ball, A. S., Lang, T., and Morison, J. I. L. 2005. Farm costs and food miles: An assessment of the full cost of the UK weekly food basket. *Food Policy* **30:** 1–19.

Pretty, J. N., Ball, A. S., Xiaoyun, L., and Ravindranath, N. H. 2002. The role of sustainable agriculture and renewable-resource management in reducing greenhouse-gas emissions and increasing sinks in China and India. *Phil. Trans. A* **360:** 1741–1761.

Pretty, J. N., Brett, C., Gee, D., Hine, R. E., Mason, C. F., Morison, J. I. L., Raven, H., Rayment, M. D., and van der Bijl, G. 2000. An assessment of the total cost of external costs of UK agriculture. *Agricultural Systems* **65:** 113–136. Pretty, J. N., Mason, C. F., Nedwell, D. B., Hine, R. E., Leaf, S. and Dils, R. 2003. Environmental costs of the eutrophication of the fresh waters in England and Wales. *Environ. Sci. & Technol.* **37:** 201–208.

Pretty, J. N., Noble, A. D., Bossio, D., Dixon. J., Hine, R. E., Penning derVires, F. V. T., and Morrison J. I. L. 2006. Resource-conserving agriculture increases yields in developing countries. *Environ. Sci. & Technol.* **40:** 1114–1119.

Purtauf, T., Roschewitz, I., Dauber, J., Thies, C., Tscharntke, T. and Wolters, V. 2005. Landscape context of organic and conventional farms: Influences on carabid beetle diversity. *Agric. Ecosys. & Environ.* **108:** 165–174.

Quinton, J. N., Govers, G., Van Oost, C., and Bardgett, R. D. 2010. The impact of agricultural soil erosion on biogeochemical cycling. *Nature Geoscience* **3:** 311–314.

Rasmussen, P. E., Goulding, K.W. T., Brown, J. R., Grace, P. R., Janzen, H. H., and Körschens, M. 1998. Long-term agroecosystem experiments: Assessing agricultural sustainability and global change. *Science* **282:** 893–896.

Refsgaard, K., Halberg, N., and Kristensen, E. S. 1998. Energy utilization in crop and dairy production in organic and conventional livestock production systems. *Agricultural Systems* **57:** 599–630.

Reganold, J. P. 1995. Soil quality and profitability of biodynamic and conventional farming systems: A review. *Am. J. Alternative Agric.* **10:** 36–46.

Reganold, J., Elliott, L., and Unger, Y. 1987. Long-term effects of organic and conventional farming on soil erosion. *Nature* **330:** 370–372.

Reganold, J., Glover, J., Andrews, P., and Hinman, H. 2001. Sustainability of three apple production systems. *Nature* **410:** 926–929.

Rigby, D., and Càceres, D. 2001. Organic farming and the sustainability of agricultural systems. *Agricultural Systems* 68: 21–40.

Robertson, G. P., and Vitousek, P. M., 2009. Nitrogen in agriculture: Balancing the cost of an essential resource. *Annu. Rev. Environ. Res.* **34:** 97–125.

Rodale, J. I. 1945. *Pay Dirt: Farming & Gardening With Composts*. Devin-Adair Co., New York.

Roschewitz, I., Gabriel, D., Tscharntke T., and Thies, T., 2005. The effects of landscape complexity on arable weed species diversity in organic and conventional farming. *J. Appl. Ecol.* 42: 873–882.

Rundlöf, M., Bengtsson, J., and Smith, H. G. 2008. Local and landscape effects of organic farming on butterfly species richness and abundance. *J. Appl. Ecol.* **45:** 813–820.

Rundlöf, M., and Smith, H. G. 2006. The effect of organic farming on butterfly diversity depends on landscape context. *J. Appl. Ecol.* **43:** 1121–1127.

Russo, M. A., Belligno, A., Wu, J. Y., and Sardo, V. 2010. Comparing mineral and organic nitrogen fertilizer impact on the soil-plant-water system in a succession of three crops. *Recent Research in Science and Technology* **2:** 14–22.

Saunders, C., Barber, A., and Taylor, G. 2006. *Food Miles – Comparative Energy/Emissions Performance of New Zealand's Agriculture Industry*. Research Report No. 285 Agribusiness & Economics Research Unit, Lincoln University, New Zealand. http://www.jborganics.co.nz/saunders_report.pdf Accessed on 20 July 2010.

Schlesinger, W. H. 1999. Carbon and agriculture: Carbon sequestration in soils. *Science* **284:** 2095.

Schlich, E., and Fleissner, U. 2005. The ecology of scale: Assessment of regional energy turnover and comparison with global food. *The International Journal of Life Cycle Assessment* **10:** 213–223.

Schmidt, M. H., Roschewitz, I., Thies, C., and Tscharntke, T. 2005. Differential effects of landscape and management on diversity and density of ground-dwelling farmland spiders. *J. Appl. Ecol.* **42:** 281–287.

Shepherd, M., Pearce, B., Cormack, B., Philipps, L., Cuttle, S., Bhogal, A., Costigan, P., Unwin, R. 2003. *An Assessment of the Environmental Impacts of Organic Farming*. Department for Environment, Food and Rural Affairs UK. http://orgprints.org/6784/2/OF0405_909_TRP.pdf Accessed on 15 March 2010.

Siegrist, S., Staub, D.; Pfiffner, L. and Mäder, P. 1998. Does organic agriculture reduce soil erodibility? The results of a long-term field study on loess in Switzerland. *Agric. Ecosyst. & Environ.* **69:** 253–264.

Sieling, K., and Kage, H. 2006. N balance as an indicator of N leaching in an oilseed rape-winter wheatwinter barley rotation. *Agric. Ecosys. & Environ.* **115:** 261–269.

Simberloff, D., and Stiling, P. 1996. How risky is biological control. *Ecology* **77:** 1965.

Smil, V. 2008. *Energy in nature and society*. MIT Press, Cambridge, MA.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., et al. 2008. Greenhouse gas mitigation in agriculture. *Phil. Trans. B* **363:** 789–813.

Smolik, J. D., Dobbs, T. L., and Rickerl, D. H. 1995. The relative sustainability of alternative, conventional and reduced-till farming system. *Am. J.Alternative Agric.* **10:** 25–35.

Sommaggio, D., Paoletti, M. G. and Ragusa, S. 1995. Effects of microhabitat conditions, nutrients and predators on the abundance of herbivores on stinging nettles (*Urtica dioica* L.). *Acta Oecologica* **16:** 671–686.

Stalenga, J., and Kawalec, A. 2008. Emission of greenhouse gases and soil organic matter balance in different farming systems. *Int. Agrophysics.* **22:** 287–290.

Stanhill, G. 1986. Water use efficiency. *Adv. Agron.* **39:** 53–85.

Stanhill, G. 1990. The comparative productivity of organic agriculture. *Agric. Ecosyst. & Environ.* **30:** 1–26.

Staley, J. T., Stewart-Jones, A., Pope, T. W., Wright, D. J., Leather, R. S., Hadley, P., Rossiter, J. T., van Emden, H. F., and Poppy, G. M. 2010. Varying responses of insect herbivores to altered plant chemistry under organic and conventional treatments. *Proceedings of the Royal Society B* **277:** 779–786.

Steinhart, J. S., and Steinhart, C. E. 1974. Energy use in the U.S. food system. *Science* **184:** 307–316.

Stockdale, E. A., Lampkin, N. H., Hovi, M., Keatinge, R., Lennartsson, E. K. M., Macdonald, D. W., Padel, S., Tattersall, F. H., Wolfe, M. S., and Watson, C. A. 2001. Agronomic and environmental implications for organic farming systems. *Advances in Agronomy* **70:** 261–327.

Stölze, M., Piorr, A., Häring, A., and Dabbert, S. 2000. The environmental impact of organic farming in Europe. **In:** *Organic Farming in Europe: Economics and Policy*. University of Hohenheim: Hohenheim, Germany. http://orgprints.org/2366/02/Volume6.pdf Accessed 23 November 2009.

Subak, S. 1999. Global environmental costs of beef production. *Ecol. Econ.* **30:** 79–91.

Suh, S., Lenzen, M., et al. 2004. System boundary selection in life-cycle inventories using hybrid approaches. *Environ. Sci. Technol.* **38:** 657–664.

Sullivan, P. 2002. *Drought Resistant Soil*. ATTRA, National Center for Appropriate Technology USDA. http://attra.ncat.org/attra-pub/PDF/drought.pdf Accessed 24 November 2009.

Teasdale, J. R., Coffman, C. B., and Mangum, R. W. 2007. Potential long–term benefits of no–tillage and organic cropping systems for grain production and soil improvement. *Agron. J.* **99:** 1297–1305.

Teasdale, J. R., Rosecrance, R. C., Coffman, C. B., Starr, J. L., Paltineanu, I. C., Lu, Y. C., and Watkins, B. K. 2000. Performance of reduced-tillage cropping systems for sustainable grain production in Maryland. *Am. J. Alternative Agric.* **15**:79–87.

Thies, C., Roschewitz, I., and Tscharntke, T. 2005. Landscape context of cereal aphid-parasitoid interactions. *Proc. R. Soc. Lond. B* **272:** 203–210.

Thies, C., Steffan-Dewenter, I., and Tscharntke, T. 2003. Effects of landscape context on herbivory and parasitism at different spatial scales. *Oikos* **101:** 18–25.

Thies, C., and Tscharntke, T. 1999. Landscape structure and biological control in agroecosystems. *Science* **285:** 893–895.

Thomas, M. B., and Reid, A. M. 2007. Are exotic natural enemies an effective way of controlling invasive plants? *Trends in Ecology and Evolution* **22:** 447–453.

Thomassen, M. A., van Calker, K. J., Smits, M.C.J., Iepema G. L., and de Boer, I.J.M. 2008. Life cycle assessment of conventional and organic milk production in the Netherlands. *Agric. Syst.* **96:** 95–107.

Tilman, D. 1998. The greening of the green revolution. *Nature* **396:** 211–212.

Tilman, D., Cassman, K. G., Matson, P. A., Naylor, R., and Polasky, S. 2002. Agricultural sustainability and intensive production practices. *Nature* **418:** 671–677.

Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, A., Howarth, R., Schindler, D., Schlesinger, W. H., Simberloff, D., and Swackhamer, D. 2001. Forecasting agriculturally driven global environmental change. *Science* **292:** 281–284.

Trewavas, A. 2001. Urban myths of organic farming. *Nature* **410:** 409–410.

Trewavas, A. 2004. A critical assessment of organic farming-and food assertions with particular respect to the UK and the potential environmental benefits of no-till agriculture. *Crop Prot*. **23:** 757–781.

Tscharntke, T., Klein, A. M., Kruess, A., Steffan-Dewenter, I., and Thies, C., 2005. Landscape perspectives on agricultural intensification and biodiversity—ecosystem service management. *Ecology Letters* **8:** 857–874.

Tu, T., Louws, F. J., Creamer, N. G., Mueller, J. P., Brownie, C., Fager, K., Bell, M., and Hu, S. 2006. Responses of soil microbial biomass and N availability to transition strategies from conventional to organic farming systems. *Agric. Ecosyst. & Environ.* **113:** 206–215.

Vandermeer, J., Perfecto, I., and Philpott, S. 2010. Ecological complexity and pest control in organic coffee production: Uncovering an autonomous ecosystem service. *BioScience* **60:** 527–537.

van Elsen, T. 2000. Species diversity as a task for organic agriculture in Europe. *Agric. Ecosyst. & Environ.* **77:** 101–109.

van Veluw, K. 2006. Voice from the other side: A Ghanaian view on organic. **In:** *Organic Agriculture. A Global Perspective*, pp. 351–359. Kristiansen, P., Taji, A., and Reganold, J. Eds., CSIRO Publishing, Collingwood, Australia.

Vitousek, P. M., Naylor, R., Crews, T., David, M. B., Drinkwater, L. E., Holland, E., Johnes, P. J., et al. 2009. Nutrient imbalances in agricultural development. *Science* **324:** 1519–1520.

Vogl, C. R., Kilcher, L., and Schmidt, H. 2005. Are standards and regulations of organic farming moving a way from small farmers' knowledge? *J. Sustain. Agric.* **26:** 5–25.

USDA (United States Department of Agriculture). 2007. Agricultural Statistics. U.S. Department of Agriculture, Washington, DC.

USDA (United States Department of Agriculture). 2010a. Census of agriculture. Organic Production Survey (2008). United States Department of Agriculture. http://www.agcensus.usda.gov/Publications/2007/Online_Highlights/Organics/ORGANICS.pdf Accessed on 20 July 2010.

USDA (United States Department of Agriculture). 2010b. Energy use in the U.S. food system. http://www.ers.usda.gov/Publications/ERR94/ERR94_ReportSummary.pdf Accessed on 20 May 2010.

U.S. EPA (United States Environmental Protection Agency). 2010. Inventory of U.S. greenhouse gas emissions and sinks: 1990-2008. April 2010, U.S. EPA # 430-R-10-006. http://epa.gov/climatechange/emissions/downloads10/US-GHG-Inventory-2010_ExecutiveSummary.pdf Accessed on 14 May 2010.

Walden, H., Gliessman, S., and Buchananb, M. 1998. Agroecosystem responses to organic and conventional management practices. *Agric. Syst.* **51:** 65–75.

Weibòò, A. C., Östamn, Ö., and Granqvist, Å. 2003. Species richness in agroecosystems: The effect of landscape, habitat and farm management. *Biodiversity and Conservation* **12:** 1335–1355.

Wells, C. 2001. Total energy indicators of agricultural sustainability: Dairy farming case study. Final report. Prepared for Ministry of Agriculture and Forestry by University of Otago, Dunedin. http://www.maf.govt.nz/mafnet/publications/techpapers/techpaper0103-dairy-farming-case-study.pdf Accessed on 20 July 2010.

Westerman, P. S., Wes, J. S., Kropff, M. J., and van der Werf, W. 2003. Annual losses of weed seeds due to predation in organic cereal fields. *J. Applied Ecol.* **40:** 824–836.

Wickramasinghe, L., Harris S., Jones G., and Vaughan, N. 2004. Abundance and species richness of nocturnal insects on organic and conventional farms: Effects of agricultural intensification on bat foraging. *Conserv. Biol*. **18**:1–10.

Willer, H. 2011. The World of Organic Agriculture 2011: Summary. **In:** *The World of Organic Agriculture. Statistics and Emerging Trends 2011*. pp. 26–32. Willer, H. and Kilcher, L. (Eds.) IFOAM, Bonn, & FiBL, Frick. Internet http://www.organic-world.net/fileadmin/documents/yearbook/2011/world-of-organic-agriculture-2011-page-1-34.pdf Accessed on 23 February 2011.

Willer, H., and Yussefi, M. (Eds.). 2006. The World of organic agriculture: Statistics and emerging trends. International Federation of Organic Agriculture Movements (IFOAM), Bonn Germany & Research Institute of Organic Agriculture FiBL, Frick, SwitzerlandSOEL-Survey 2006. Internet http://www.soel.de/inhalte/publikationen/s/s_74_08.pdf. Accessed on February 24, 2008.

Williams, A. 2007. Comparative study of cut roses for the British market produced in Kenya and the Netherlands. http://www.fairflowers.de/fileadmin/flp.de/Redaktion/Dokumente/Studien/Comparative_Study_of_Cut_Roses_Feb_2007.pdf Accessed on 20 July 2010.

Williams, A. G., Audsley, A., and Sandars, D. L. 2006. Energy and environmental burdens of organic and non-organic agriculture and horticulture. *Aspects of Applied Biology* **79:** 19–23. http://orgprints.org/10162/1/Energy_and_environmental_burdens_of_organic_and_non-organic_agriculture_and_horticulture.pdf Accessed 20 May 2010.

Wilson, E.O. (Ed.). 1988. *Biodiversity.* National Academy Press. Available on online at http://www.nap.edu/catalog.php?record_id=989

Wojtkowski, P. 2006. *Introduction to Agroecology: Principles and Practices*. Island Press, Washington D.C.

Wood, R., Lenzen, M., Dey, C., and Lundie, S. 2006. A comparative study of some environmental impacts of conventional and organic farming in Australia. *Agric. Syst.* **89:** 324–348.

Wu, J. Y., and Sardo, V. 2010. Sustainable versus organic agriculture. **In:** *Sociology, Organic Farming, Climate Change and Soil Science.* pp. 41–76. Lichtfouse, E., Ed., Springer, Dordrecht.

Yunus, M. 2009. *Creating a World Without Poverty: Social Business and the Future of Capitalism*. PublicAffairs Cambridge, MA.

Zhu, Y., Chen, H., Fan, J., Wang, Y., et al. 2000. Genetic diversity and disease control in rice. *Nature* **406:** 718–722.

Bio Bio

Biodiversität von Regenwürmern im Marchfeld, Ostösterreich

Friedel, J.K.[1], Heiner, B.[2], Frank, T.[2], Papaja-Hülsbergen, S.[3], Arndorfer, M.[1]

[1] Universität für Bodenkultur, Institut für Ökologischen Landbau, Gregor Mendel-Str. 33, A-1180 Wien, juergen.friedel@boku.ac.at, www.boku.ac.at
[2] Universität für Bodenkultur, Institut für Zoologie, Gregor Mendel-Str. 33, A-1180 Wien
[3] Wissenschaftszentrum Weihenstephan für Ernährung, Landnutzung und Umwelt der Technischen Universität München, Lehrstuhl für Ökologischen Landbau und Pflanzenbausysteme, Alte Akademie 12, D-85354 Freising

BOKU

Universität für Bodenkultur Wien
Department für Nachhaltige Agrarsysteme

Hintergrund

In bisherigen Untersuchungen wurden auf Acker-Standorten im Marchfeld zwar endogäische, aber keine tiefgrabenden, anözischen Regenwurm-Arten angetroffen. Diese wurden nur lokal in Hecken, Krautstreifen und Ruderalflächen gefunden. Dies ist auf standörtliche Besonderheiten zurückzuführen: geringe Niederschläge (durchschnittlich 530 mm /Jahr), strukturschwache, schluffreiche Lehmböden.
Anözische Regenwürmer haben positive Auswirkungen auf Wasserinfiltration und Bodenstruktur.

Ziele

- Das Vorkommen von Regenwürmern in Ackerflächen und halb-natürlichen Begleithabitaten an einer größeren Anzahl von Flächen erheben
- Feststellen, ob das Auftreten von anözischen Arten durch das Bewirtschaftungssystem, die Bodenbearbeitung oder durch die Ausstattung mit halbnatürlichen Begleithabitaten bestimmt wird

Material und Methoden

Abb. 1: Austreiben der Regenwürmer: Herstellen der AITC-Lösung und Aufbringen auf den Boden

- Auswahl von 8 ökologisch und 8 konventionell bewirtschafteten Ackerbaubetrieben
- Ermitteln der Habitattypen auf den Betrieben nach der BioHab-Methode (Bunce et al. 2008)
- Auswahl jeweils einer Fläche je festgestelltem Habitattyp zur Beprobung
- Beprobung von jeweils drei Plots (30 cm * 30 cm) je ausgewählter Fläche nach Entfernen der Vegetation mit einer Kombination von Austreibung und Handauslese (Pelosi et al. 2009)
- Austreibung durch 0,1 g/L Allyl-Isothiocyanat (**AITC**)-Lösung (Abb. 1)
- Sammeln der Regenwürmer für 10 min
- Handauslese bis 20 cm Tiefe für 20 min
- Bestimmung auf Artniveau soweit möglich
- Einfluss des Bewirtschaftungssystems: einfaktorielle ANOVA
- Einfluss der Bodenbearbeitung: Flächenanteil mit Pflugeinsatz als Kovariable

Ergebnisse

- Insgesamt beprobt: 130 Habitate; 390 Plots; 64 flächige Habitate, davon 58 Ackerflächen; 66 lineare Habitate
- Mittelwert je Betrieb: 8,1 beprobte Habitate
- Abundanzen (Tab. 1)
  5200 gesammelte Regenwürmer, 3955 Juvenile oder Fragmente (= 74%); im Mittel 48,9 Individuen pro m^2, etwa ¾ davon Juvenile oder Fragmente; bestimmte Arten: 87 % endogäische, 12 % anözische, 1 % epigäische
- gesamte Artenzahl: 11; Artenzahl auf Ackerflächen: 7 (Tab. 1)
- Häufigste Arten (Anzahl Exemplare)
  Aporrectodea rosea (542), *Aporrectodea caliginosa* (412), *Lumbricus terrestris* (145), *Allolobophora chlorotica* (70), *Octolasion tyrtaeum* (52)
- Betriebssystem (ökologisch – konventionell): kein Einfluss auf Habitatanzahl und Abundanzen; mehr ausdauernde Kulturen auf ökologischen Betrieben (öko: 7 von 8, konv: 4 von 8)
- Bodenbearbeitung (Pflug – Grubber): kein Einfluss auf Abundanzen
- Vorkommen anözischer Arten:
  - in halbnatürlichen Begleithabitaten: Hecken, Krautstreifen, Grasstreifen, Gebüsche, …
  - in Ackerflächen nur in mehrjährigen Kulturen mit Bodenruhe: Luzerne, Kleegras, *Miscanthus* (vereinzelte Funde in Getreide und Mais)

Tabelle 1: Abundanzen (Indiv. m^{-2})§ und Artenzahlen der Regenwürmer auf den beprobten Flächen nach Lebensformtypen und Art der Flächen

	gesamt	juvenile*	endogäische	anözische	epigäische
Alle Flächen (N = 130)	48,9 (11,2)	37,3 (8,2)	10,0 (3,9)	1,4 (0,9)	0,1 (0,2)
Ackerflächen (N = 58)	45,1 (17,2)	34,6 (13,3)	10,0 (6,4)	0,4 (0,6)	0,1 (0,1)
Arten ges.	11	-	5	3	3
Arten Acker	7	-	4	1	2

§ Mittelwert (N Flächen) und Standardabweichung (16 Betriebe) in Klammern;
* juvenile inkl. Fragmente; Art nicht bestimmbar, Lebensformtyp nicht feststellbar

Fazit

- Regenwurm-Abundanz mit unter 50 Indiv. pro m^2 im Marchfeld vergleichsweise gering
- Endogäische Arten dominieren
- Anözische Arten kommen mit sehr wenigen Ausnahmen nur auf Flächen ohne Bodenbearbeitung vor
- Kein Unterschied zwischen ökologisch und konventionell bewirtschafteten Betrieben hinsichtlich Habitatausstattung und Regenwurmvorkommen feststellbar
- Kein Zusammenhang zwischen reduzierter Bodenbearbeitung (Grubber statt Pflug) und Regenwurmvorkommen feststellbar

Literatur

Bunce, R. G. H., et al. (2008) A standardized procedure for surveillance and monitoring European habitats and provision of spatial data. Landscape Ecology 23, 11-25.
Pelosi, C., M. Bertrand, Y. Capowiez, H. Boizard, and J. Roger-Estrade (2009) Earthworm collection from agricultural fields: Comparisons of selected expellants in presence/absence of hand-sorting. European Journal of Soil Biology 45, 176-183.

University of Natural Resources and Applied Life Sciences, Vienna
Department of Sustainable Agricultural Systems

Diese Untersuchung wurde im Rahmen des EU FP7-Projekts „BioBio – Indicators for Biodiversity in Organic and Low-input farming" durchgeführt

Environmental Management
DOI 10.1007/s00267-012-9834-5

Organic Farming Benefits Local Plant Diversity in Vineyard Farms Located in Intensive Agricultural Landscapes

Juri Nascimbene · Lorenzo Marini · Maurizio G. Paoletti

Received: 7 November 2011 / Accepted: 18 February 2012

Abstract The majority of research on organic farming has considered arable and grassland farming systems in Central and Northern Europe, whilst only a few studies have been carried out in Mediterranean agro-systems, such as vineyards, despite their economic importance. The main aim of the study was to test whether organic farming enhances local plant species richness in both crop and non-crop areas of vineyard farms located in intensive conventional landscapes. Nine conventional and nine organic farms were selected in an intensively cultivated region (i.e. no gradient in landscape composition) in northern Italy. In each farm, vascular plants were sampled in one vineyard and in two non-crop linear habitats, grass strips and hedgerows, adjacent to vineyards and therefore potentially influenced by farming. We used linear mixed models to test the effect of farming, and species longevity (annual vs. perennial) separately for the three habitat types. In our intensive agricultural landscapes organic farming promoted local plant species richness in vineyard fields, and grassland strips while we found no effect for linear hedgerows. Differences in species richness were not associated to differences in species composition, indicating that similar plant communities were hosted in vineyard farms independently of the management type. This negative effect of conventional farming was probably due to the use of herbicides, while mechanical operations and mowing regime did not differ between organic and conventional farms. In grassland strips, and only marginally in vineyards, we found that the positive effect of organic farming was more pronounced for perennial than annual species.

J. Nascimbene (✉) · M. G. Paoletti
Department of Biology, University of Padova,
via U. Bassi 58/B, 35121 Padova, Italy
e-mail: junasc@libero.it

L. Marini
DAFNAE, University of Padova, viale dell'Università
16, 35020 Legnaro, Padova, Italy

Keywords Conventional farming · Disturbance · Grassland strip · Hedgerow · Herbicide · Semi-natural habitats

Introduction

Comparing biodiversity of organic and conventional farming has been a subject for research and debate for the last decades (Bengtsson and others 2005; Fuller and others 2005; Gomiero and others 2011; Hadjicharalampous and others 2002; Hole and others 2005; Hyvönen 2007; Paoletti and Pimentel 1992; Moreby and others 1994; Norton and others 2009; Stockdale and others 2001). Although in some cases the effectiveness of organic farming was not clear, large scale synthesizing studies suggest that organic farming is generally associated with higher levels of biodiversity of both plant and animal taxa (Bengtsson and others 2005; Caporali and others 2003; Clough and others 2007; Fuller and others 2005; Gabriel and others 2006, 2010; Hawesa and others 2010; Hole and others 2005; Hyvönen 2007; Mäder and others 2002; Moreby and others 1994; Roschewitz and others 2005). For instance the meta-analysis of Bengtsson and others (2005) showed that organic farming has positive effects on species richness and abundance in the 84% of the studies included in their review. These studies indicate that organic farming systems provide greater potential for biodiversity than their conventional counterparts, mainly because of reduced soil disturbance and chemical applications associated with management practices.

Although, at the local scale (i.e. within a field), biodiversity is often higher in organic farms, the benefits of organic farming also depends on the composition of the surrounding landscape (Ekroos and others 2010; Rundlöf and Smith 2006; Rundlöf and others 2008, 2010; Weibull and others 2003). This landscape dependence is particularly relevant for highly mobile organisms such as flying insects and birds, while plants usually exhibit lower dependence on landscape processes (Gabriel and others 2010; Marini and others 2008a, b; but see Rundlöf and others 2010). Plant species richness under organic farming is expected to be higher than under conventional farming in simple landscapes, while conventional farms might reach similar diversity levels to those of organic farms when the surrounding landscape is complex (Batáry and others 2011; Roschewitz and others 2005; Tscharntke and others 2005).

A large body of research has considered arable and grassland farming systems mostly in Central and Northern Europe, while only a few studies have been carried out in other economically important agro-ecosystems of the Mediterranean regions such as vineyards, citrus and olive orchards (Cárdenas and others 2006; Gago and others 2007). Despite the fact that in several countries vineyards are economically relevant cultivations, only a very few studies have evaluated the effects of management practices on biodiversity (Brittain and others 2010; Brugisser and others 2010; Paoletti and others 2007).

Hence, the main aim of this study was to test whether organic farming enhances local plant species richness in vineyard farms located in a homogenous intensive conventional landscape. Specifically, this study had two main objectives: (i) to compare plant species richness and composition in three habitat types (vineyard, grassland strip, and hedgerow) between conventional vs. organic vineyard farms located in intensively managed conventional landscape, and (ii) to explore whether species richness responses to farm management differ between annual and perennial species. Perennial species are expected to be more sensitive to mechanical disturbance (McIntyre and others 1999) than annual species. Frequent disturbance destroys in general both annuals and perennials established, but it creates more favourable conditions for new germinations of annuals (Gago and others 2007).

Materials and Methods

Study Area

The study was carried out in the administrative region of Veneto (Fig. 1). The region is one of the most important areas of north Italy for grape production with c. 71,360 ha cultivated as vineyards. In Veneto, vineyard farms are mainly managed with conventional practices, and only a small percentage is managed under organic farming (~2%). Because of the dominance of conventional farming, the few organic farms are composed of scattered patches within a matrix of an intensively farmed landscape. In these areas, semi-natural habitats such as deciduous forests and semi-natural grasslands have been progressively converted to farmland, resulting in a relatively simplified landscape. Due to the dominance of conventional farming, it was not possible to include a gradient in landscape composition along with local management. We, therefore, kept constant landscape composition and tested only local farming practices. This was done by sampling farms in landscapes with more or less the same cover of semi-natural habitats (see below).

Sampling

Nine conventional and nine organic farms were selected to keep landscape composition constant and to test the effect of local management (Fig. 1). The farms were selected using a regional database to include the main productive districts of the region. Organic and conventional farms had 10.6% (SD = 15.3%) and 6.8% (SD = 7.5%) of cover of semi-natural habitats in the surroundings (quantified within a 1-km radius), respectively. There was no significant difference between the conventional and organic farms in the proportion of surrounding semi-natural habitats (unpaired t test, $n = 18$, $p = 0.514$). The landscape was characterized by a matrix of intensive conventional vineyards or arable lands where vineyard fields are usually located in a continuous matrix of vineyards and are sometimes bordered by grasslands strips and hedgerows.

Management information was obtained through farmer interviews. Sampled organic farms were converted to organic farming at least 10 years before sampling, except in the case of one farm which was converted in 2005. In organic farms, weeds are controlled only by mowing (3–4 times per year) whereas in conventional farms they are controlled both mechanically and chemically using herbicides (mainly glyphosate) 2–4 times per year, both on the rows and between them (Table 1). Mown vegetation is generally removed from the fields. Tillage on the rows is the main mechanical operation used in both organic and conventional farms. The number of fungicide and insecticide treatments is similar between organic and conventional farms, obviously differing in the type of product. Organic farms use only substances admitted by the management guidelines for organic farming (e.g., copper hydroxide, *Bacillus thuringiensis*, and pyrethroids), while conventional farms use various agrochemical (e.g., Fenamidone, Mancosin, Tebufenozide).

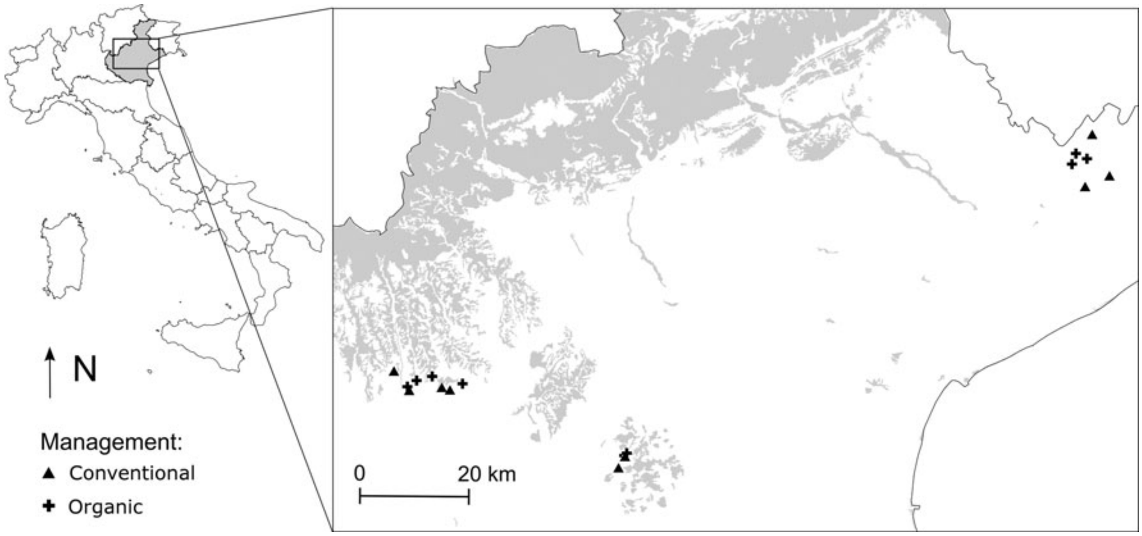


Fig. 1 Location of the study area and of the 9 organic and 9 conventional farms. The *grey* area in the panel on the right represents the cover of semi-natural habitats in the region

Table 1 Average ± SD (min–max) values of the main management treatments applied in organic versus conventional farms

	Organic ($n = 9$)	Conventional ($n = 9$)
Mean surface (ha)	22 ± 10 (12–39)	36 ± 35 (8–115)
Mean length of grass strips	55 ± 22 (35–90)	60 ± 35 (40–120)
Mean length hedgerows	45 ± 18 (20–90)	35 ± 10 (18–56)
Years of organic farming	16 ± 7 (5–25)	–
N herbicide treatments	0	2 ± 1.4 (1–4)
N fungicide treatments	15.3 ± 4.5 (9–21)	12.8 ± 4.2 (10–20)
N insecticide treatments	2 ± 2.4 (0–7)	1.4 ± 1.1 (0–3)
Use of mineral fertilizer	No	Yes
Use of organic fertilizer	Yes	Yes
N total (kg ha^{-1})	88 ± 36 (50–130)	82 ± 50 (36–160)
Tillage on the rows	Yes	Yes
Mowing frequency	3–4	3–4

In each farm, vascular plants (both herbaceous and woody species) were sampled in one randomly selected vineyard field and in the two linear habitats closest to the vineyard field: (i) grassland strips (∼4 to 6 m width) dominated by herbaceous vegetation, used for tractors crossing and usually managed with the same criteria as for vineyard, being actually part of the "vineyard system", and (ii) single-row hedgerows with trees and shrubs, usually coppiced (∼5 to 8 m width; ∼6 to 8 m far from the vineyard). For the hedgerows, we placed the plot directly next to the hedgerows. In one farm and three farms there was no grassland strip and no linear hedgerow respectively. In each habitat, plant sampling was carried out once between April 15th and May 10th, 2010 before any management interventions. In each vineyard, plants were sampled within a single 10 × 10 m^2 plot placed in the centre of the cultivated area. For each species, the abundance was visually estimated using 5% cover classes. Plants in linear habitats (one grassland strip and one hedgerow in each farm) were recorded within 1 × 1 m^2 plots placed in the centre of the linear element. The abundance of each species was estimated using the same approach applied within vineyard fields.

Data Analysis

Plant species were classified into two groups on the basis of their longevity: (i) annual species, and (ii) perennial species (including biennial species). For each group, species richness was computed as the total number of species recorded in each plot. Linear mixed models were used to test the main effects on species richness of management (conventional vs. organic) and species longevity (annual, and perennial) and their interaction separately for the three habitats (vineyard, grassland strip, and hedgerow). We included farm and plot as random factors in the model. The plot as random factor was included to account for the fact that the numbers of species in each trait group were quantified at the same sites. A significant interaction between management and longevity would imply a differential response to management of annual versus perennial species. We used linear mixed model assuming normal error distribution using the lmer function implemented in the "nlme" package using a restricted maximum likelihood estimation procedure (Pinheiro and others 2009) in R (R Development Core Team 2008, version 2.8.0). We verified assumptions of linear mixed models by inspecting diagnostic plots of model residuals.

Differences in species composition between organic and conventional farms were evaluated for each habitat separately both by Multi-Response Permutation Procedures (MRPP) and Indicator Species Analysis (ISA; Dufrêne and Legendre 1997) as implemented in PC-ORD (McCune and Mefford 1999). MRPP was used to test the significance of differences in plant species composition between the two management types, using a Monte Carlo p-value which describes the likelihood of an equal or smaller effect size "A" than that measured by the procedure (McCune and Grace 2002). The effect size is the value 1–0 (within group heterogeneity/randomly expected heterogeneity). When $A = 1$, there is perfect agreement within the group, and when $A = 0$ the agreement within the group is equal to random probability. A significant effect size of 0.1 is commonly observed in community data (McCune and Grace 2002). The Sørensen distance measure and rank transformation of the distance matrices was used. ISA was used to determine how strongly each species was associated with each type of management. For each species, the Indicator Value (INDVAL) ranges from 0 (no indication) to 100 (maximum indication). Statistical significance of INDVAL was tested by means of a Monte Carlo test, based on 10,000 randomizations.

Results

A total of 211 species were found (50 annuals, and 161 perennials), 162 in conventional and 171 in organic farms. Regional species pool of vineyards and hedgerows in organic farms was richer than their conventional counterparts (Table 2).

Management type and longevity had different effects depending on the habitat (Table 3). Within the vineyard fields, we found a positive effect of organic farming on species richness. There was a tendency for a stronger effect of organic farming on perennial than annual species. Species richness did not differ between annual and perennial species (Fig. 2a). Within the grassland strips, we found a significant interaction between longevity and management, i.e. the number of perennial species was higher in the grassland strips under organic than conventional farming while annual species richness was not affected by management. Annual species richness was smaller than perennial species richness (Fig. 2b). Within the hedgerows, we did not find any effect of management. Plant assemblages of were mainly composed of perennial species both in conventional and organic farms (Fig. 2c).

The multi-response permutation procedure found no difference in overall species composition between farming types (MRPP: effect size $A < 0.003$, and $p > 0.3$ for all the habitats). Also no indicator species were detected by ISA as being linked to either organic or conventional farms in any of the three habitats.

Table 2 Total plant species richness in the three habitats in conventional and organic farms

Habitat	Management		Total
	Conventional	Organic	
Vineyards	71 (49.3% annual)	98 (38.8% annual)	106
Grassland strips	94 (38.3% annual)	89 (30.3% annual)	125
Hedgerows	78 (14.1% annual)	92 (17.4% annual)	128

The percentage of annual species is included in parenthesis for each habitat × management combination

Table 3 Results of the linear mixed model testing the main effects of management (organic vs. conventional), and longevity and their interaction on plant species richness within the three habitats in the 18 farms

	d.f.	*F*	*P*
Vineyard			
Management	1, 16	6.00	<0.001
Longevity	1, 16	1.54	0.232
Management × Longevity	1, 16	2.33	0.146
Grassland			
Management	1, 15	3.88	0.067
Longevity	1, 15	13.32	0.002
Management × Longevity	1, 15	4.52	0.050
Hedgerow			
Management	1, 13	2.17	0.167
Longevity	1, 13	93.55	<0.001
Management x Longevity	1, 13	0.23	0.634

d.f. are different due to missing values, i.e. in some farms no semi-natural habitat was present

Discussion

Our study evaluated the effect of organic farming on plant species richness in both crop and non-crop areas of vineyard farms. In our intensive agricultural landscapes organic farming promoted local plant species richness in vineyard fields, and grassland strips while we found no effect for linear hedgerows. Differences in species richness were not associated with differences in species composition, indicating that similar plant communities grew in vineyard farms independently of the management type. However, the plant species pool was poorer in conventional than in organic farms.

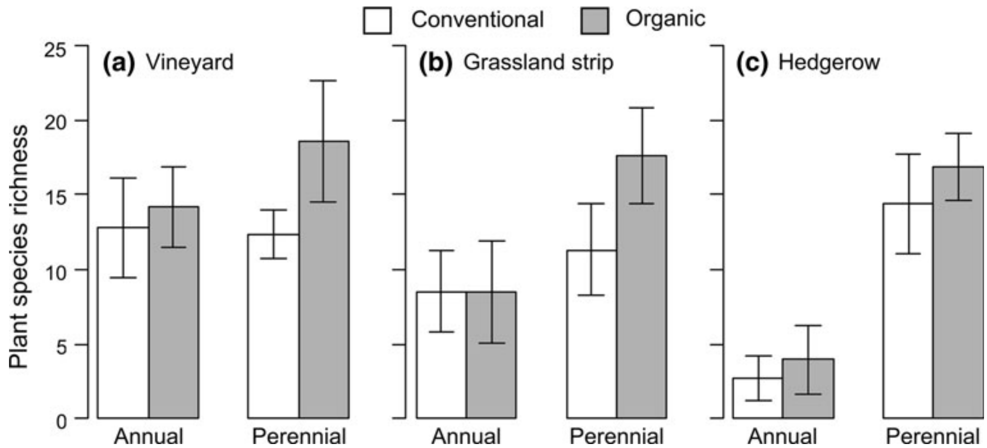


Fig. 2 Mean annual and perennial plant species richness in conventional versus organic farms in **a** vineyards, **b** grassland strips, and **c** hedgerows. The *bars* indicate the 95% intervals of confidence

In vineyard fields, the main difference between the two farming regimes in terms of management practices potentially influencing plants was the use of herbicides under conventional farming, while mechanical operations and mowing regime did not differ between the two management regimes. The use of herbicides was probably the main cause of the reduced establishment of large species richness under conventional farming (Hald 1999; Rundlöf and others 2010).

Research on the evaluation of the effects on biodiversity of organic versus conventional farming has sometimes led to contrasting results even within the same agro-ecosystem, depending on the taxonomic groups and spatial scales considered. Our results contrast with those of Brugisser and others (2010) who found that in vineyards plant species richness was not enhanced by organic farming compared to conventional farming. They conclude that biodiversity in vineyards may not benefit to the same degree from organic farming as in annual arable systems. They interpreted their results in the light of the intermediate disturbance regime hypothesis applied to vineyards compared to the more intensive management regime applied to annual crops. In their study, the positive effect of the intermediate disturbance regime on species richness overrode the benefit provided by organic farming, allowing conventional vineyards to reach comparable richness levels to their organic counterparts. In our study, both conventional and organic farms were scattered in an intensive-dominated conventional landscape sometimes extending for thousands of hectares. Intensively farmed and homogenous landscapes dominated by few crops have been considered as those where organic farming should produce the best benefits to biodiversity (Roschewitz and others 2005; Rundlöf and others 2010; Tscharntke and others 2005). Consistently, in our study, the intensive landscape management at the regional level could explain the positive effect of organic farming on local plant species richness. However, this conclusion does not necessarily apply to more mobile organisms such as insects or birds which often depend on suitable habitats and resources at larger spatial scale (Brittain and others 2010).

We further investigated the effect of organic farming on plant species richness in two common semi-natural habitats associated with vineyard cropping systems finding a beneficial effect for grassland strips and no effect for linear hedgerows. Grassland strips are expected to be more directly influenced by crop management than hedgerows due to their position and use within the farm. While grassland strips are usually managed with the same criteria as for vineyard, being actually part of the "vineyard system", linear hedgerows are not expected to be managed differently in organic versus conventional farms. They are usually coppiced for wood fuel production and/or traditionally used to mark field boundaries. Moreover, hedgerows are located 6–10 m far from the vineyard reducing the potential negative effect of agro-chemical drifts from crop areas.

In grassland strips, and only marginally in vineyards, we found that the positive effect of organic farming was more pronounced for perennial than annual species. The use of herbicides was probably the main cause of the reduced establishment of perennial species under conventional farming. Accordingly, Gago and others (2007) found that weed control, tillage, and mowing allow annual species to complete their life cycle. On the contrary, the development of perennial species is effectively hindered by herbicides, whilst tillage and mowing have a lower impact. Boer and Stafford Smith (2003) also showed that annual species richness was enhanced by habitat degradation and disturbance removing perennial and biennial species (McIntyre and others 1999).

In both organic and conventional vineyard farms, our three habitats are distributed along a gradient of disturbance, being higher in vineyard fields, intermediate in grassland strips and lower in hedgerows. While in vineyards there is no difference between the number of annual and perennial species, in grassland strips and hedgerows perennial species richness was higher than annual species richness irrespectively of the management type. In the case of hedgerows this is an obvious result related to the features of the vegetation which is largely composed by shrubs and tree species. In the case of grassland strips this situation could be indicative of less disturbed and more stable conditions allowing this habitat to host a higher number of more ecologically sensitive species.

In conclusions, our results demonstrate that in vineyard farms located in intensive conventional landscapes, local plant species richness benefits from organic farming within vineyard fields, grassland strips but not within linear hedgerows. As studies on the effects of organic farming on biodiversity within vineyards are still limited to a very few single-region studies, larger scale synthesizing studies are urgently needed to clarify the response of other organisms occupying different trophic levels, and to monitor in the long term the effects of the conversion from conventional to organic farming.

Acknowledgments Part of this project was funded by the FP7 EU project "Indicators for biodiversity in organic and low-input farming systems" (GA 227161). We thank Péter Batáry and two anonymous reviewers for providing insightful comments on the manuscript. Bob Bunce and Tiziano Gomiero provided stimulating comments on the manuscript, which improved its clarity. Tommaso Zanetti is thanked for his precious support in the field work. Diego Ivan, Valeria Covre, and Giulia Moresco collaborated for the field work. Giovanni Caniglia and Mariano Brentan helped in the identification of some critical specimens. We are grateful to the owners of the farms who kindly allowed us to work in their vineyards and provided information on farm management. The experiment fulfils the current laws of the country in which it was performed.

References

Batáry P, Báldi A, Kleijn D, Tscharntke T (2011) Landscape-moderated biodiversity effects of agri-environmental management—a meta-analysis. Proceedings of Royal Society B 278: 1894–1902

Bengtsson J, Ahnstrom J, Weibull A-C (2005) The effects of organic agriculture on biodiversity and abundance: a meta-analysis. Journal of Applied Ecology 42:261–269

Boer M, Stafford Smith M (2003) A plant functional approach to the prediction of changes in Australian rangeland vegetation under grazing and fire. Journal of Vegetation Science 14:333–344

Brittain C, Bommarco R, Vighi M, Settele J, Potts SG (2010) Organic farming in isolated landscapes does not benefit flower-visiting insects and pollination. Biological Conservation 143:1860–1867

Brugisser OT, Schmidt-Entling MH, Bacher S (2010) Effects of vineyards management on biodiversity at three trophic levels. Biological Conservation 143:1521–1528

Caporali F, Mancinelli R, Campiglia E (2003) Indicators of cropping system diversity in organic and conventional farms in Central Italy. International Journal of Agricultural Sustainability 1:67–72

Cárdenas M, Ruano F, García P, Pascual F, Campos M (2006) Impact of agricultural management on spider populations in canopy of olive trees. Biological control 38:188–195

Clough Y, Kruess A, Tscharntke T (2007) Local and landscape factors in differently managed arable fields affect the insect herbivore community of a non-crop plant species. Journal of Applied Ecology 44:22–28

Dufrêne M, Legendre P (1997) Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecological Monographs 67:345–366

Ekroos J, Hyvönen T, Tiainen J, Tiira M (2010) Responses in plant and carabid communities to farming practises in boreal landscapes. Agriculture, Ecosystems & Environment 135:288–293

Fuller RJ, Norton LR, Feber RE, Johnson PJ, Chamberlain DE, Joys AC, Mathews F, Stuart RC, Townsend MC, Manley WJ, Wolfe MS, Macdonald DW, Firbank LG (2005) Benefits of organic farming to biodiversity vary among taxa. Biology Letters 1:431–434

Gabriel D, Roschewitz I, Tscharntke T, Thies C (2006) Beta diversity at different spatial scales: plant communities in organic and conventional agriculture. Ecological Applications 16:2011–2021

Gabriel D, Sait SM, Hodgson JA, Schmutz U, Kunin WE, Benton TG (2010) Scale matters: the impact of organic farming on biodiversity at different spatial scales. Ecology Letters 13:858–869

Gago P, Cabaleiro C, García J (2007) Preliminary study of the effect of soil management system on the adventitious flora of a vineyard in northwestern Spain. Crop Protection 26:584–591

Gomiero T, Pimentel D, Paoletti MG (2011) Environmental impact of different agricultural management practices: conventional vs. organic agriculture. Critical Reviews in Plant Sciences 30:95–124

Hadjicharalampous E, Kalburtji KL, Mamolos AP (2002) Soil arthropods (Coleoptera, Isopoda) in organic and conventional agroecosystems. Journal of Environmental Management 29:683–690

Hald AB (1999) Weed vegetation (wild flora) of long established organic versus conventional cereal fields in Denmark. Annals of Applied Biology 134:307–314

Hawesa C, Squirea GR, Hallett PD, Watsonb CA, Young M (2010) Arable plant communities as indicators of farming practice. Agriculture, Ecosystems & Environment 138:17–26

Hole DG, Perkins AJ, Wilson JD, Alexander IH, Grice PV, Evans AD (2005) Does organic farming benefit biodiversity? Biological Conservation 122:113–130

Hyvönen T (2007) Can conversion to organic farming restore the species composition of arable weed communities? Biological Conservation 137:328–390

Mäder P, Fließbach A, Dubois D, Gunst L, Fried P, Niggli U (2002) Soil fertility and biodiversity in organic farming. Science 296: 1694–1697

Marini L, Scotton M, Klimek S, Pecile A (2008a) Patterns of plant species richness in Alpine hay meadows: local vs landscape controls. Basic and Applied Ecology 9:365–372

Marini L, Fontana P, Scotton M, Klimek S (2008b) Vascular plant and orthoptera diversity in relation to grassland management and landscape composition in the European alps. Journal of Applied Ecology 45:361–370

McCune B, Grace JB (2002) Analysis of ecological communities. MjM Software, Gleneden Beach

McCune B, Mefford MJ (1999) Multivariate analysis of ecological data. Version 4.25. MjM Software, Gleneden Beach

McIntyre S, Lavorel S, Landsberg J, Forbes TDA (1999) Disturbance response in vegetation—towards a global perspective on functional traits. Journal of Vegetation Science 10:621–630

Moreby SJ, Aebischer NJ, Southway SE, Sotherton NW (1994) A comparison of the flora and arthropod fauna of organically and conventionally grown winterwheat in southern England. Ann Appl Biol 125:13–27

Norton L, Johnson P, Joys A, Stuart R, Chamberlain D, Feber R, Firbank L, Manley W, Wolfe M, Hart B, Mathews F, Macdonald D, Fuller RJ (2009) Consequences of organic and non-organic farming practices for field, farm and landscape complexity. Agriculture, Ecosystems & Environment 129:221–227

Paoletti MG, Pimentel D (1992) Biotic diversity in agroecosystems. Elsevier, Amsterdam

Paoletti MG, Thomson LJ, Hoffmann AA (2007) Using invertebrate bioindicators to assess agricultural sustainability in Australia: proposals and current practices. Australian Journal of Experimental Agriculture 47:379–478

Pinheiro J, Bates D, DebRoy S, Sarkar, D, the R Core team, 2009. nlme: Linear and Nonlinear Mixed Effects Models. R package version 3. pp 1–92

Roschewitz I, Gabriel D, Tscharntke T, Thies T (2005) The effects of landscape complexity on arable weed species diversity in organic and conventional farming. Journal of Applied Ecology 42:873–882

Rundlöf M, Smith HG (2006) The effect of organic farming on butterfly diversity depends on landscape context. Journal of Applied Ecology 43:1121–1127

Rundlöf M, Nilsson H, Smith HG (2008) Interacting effects of farming practice and landscape context on bumble bees. Biological Conservation 141:417–426

Rundlöf M, Edlund M, Smith HG (2010) Organic farming at local and landscape scales benefits plant diversity. Ecography 33:514–522

Stockdale EA, Lampkin NH, Hovi M, Keatinge R, Lennartsson EKM, Macdonald DW, Padel S, Tattersall FH, Wolfe MS, Watson CA (2001) Agronomic and environmental implications for organic farming systems. Advances in Agronomy 70:261–327

R Development Core Team (2008) R: a language and environment for statistical computing. R foundation for statistical computing, Vienna. ISBN 3-900051-07-0. http://www.R-project.org

Tscharntke T, Klein AM, Kruess A, Steffan-Dewenter I, Thies C (2005) Landscape perspectives on agricultural intensification and biodiversity—ecosystem service management. Ecology Letters 8:857–874

Weibull AC, Östman Ö, Granqvist Å (2003) Species richness in agroecosystems: the effect of landscape, habitat and farm management. Biodiversity and Conservation 12:1335–1355

ИНТЕНЗИВНОСТ НА ФЕРМИ С НИСКИ ВЛОЖЕНИЯ И СЪЗДАВАНЕ НА ИНДИКАТОРИ ЗА БИОРАЗНООБРАЗИЕ
INTENSITY IN LOW-INPUT FARMS AND DEVELOPMENT OF INDICATORS FOR BIODIVERSITY

Сийка Стоянова*[1], Феликс Херцог[2], Яна Гутева[1], Сийка Ангелова[1], Цонка Оджакова[3], Мария Събева[1]
Siyka Stoyanova*[1], Felix Herzog[2], Yana Guteva[1], Siyka Angelova[1], Zonka Odjakova[3], Maria Sabeva[1]

[1]Институт по растителни генетични ресурси – Садово
[2]Опитна станция АРТ – Агроскопе - Рекенхолц – Теников, Федерален отдел по икономика и селско стопанство Цюрих, Швейцария
[3]Опитна станция по земеделие и животновъдство – Смолян, България

[1]Institute of Plant Genetic Resources, Sadovo
[2]FDEA-ART - Agroscope Reckenholz-Tänikon Research Station ART, Zurich, Switzerland
[3]Agricultural and Stockbreeding Experimental Station, Smolian

***E-mail: s_stoyanova@gbg.bg**

Резюме

Съхранението на биоразнообразието е неотложна задача за нашето общество с особено значение за пасищните територии. Европейският проект БиоБио по Седма рамкова програма – *Индикатори за биоразнообразие във фермерски системи за биологично и с ниски вложения земеделие*, има за цел да разработи научно обосновани, приложими и атрактивни индикатори за биоразнообразие в Европа във ферми за биологично и с ниски вложения земеделие. Районът за изследване в България е представен с фермерство с ниски вложения в пасища в Родопите. Проведено е предварително селекциониране на 16 ферми с ниски вложения за детайлно изследване на индикаторите за биоразнообразие. За да се намери подходяща скала, която да обхване фермите, са очертани социално-икономически компоненти, фактори на средата и отчасти интензивност на използване. Използвани са три основни групи от фактори и условия, за да се създаде подходящ модел и определяне на скала: общи социално-икономически фактори, социално-икономически условия, независещи от собственика, зависещи от собственика социално-икономически условия. Сумарното влияние на оценените фактори и условия е представено с индекс за интензивност на фермата. Разработеният модел позволява предварително разделяне на фермите в три групи по интензивност. Научно се потвърждава разпределението на фермите в подходяща скала преди определянето на индикаторите за биоразнообразие в пасища с ниски вложения.

Abstract

Biodiversity conservation is an urgent task for our society with special attention for grassland areas. The EU FP7 research project BIOBIO - *INDICATORS FOR BIODIVERSITY IN ORGANIC AND LOW-INPUT FARMING SYSTEMS* aims at developing scientifically sound, useful and attractive biodiversity indicators for organic and low-input farming in Europe. The Bulgarian case study region in the project is presented by low-input farming in grasslands in the Rhodopi mountains. Preliminary selection of 16 low-input farms in Bulgaria for a detailed assessment of biodiversity indicators is carried out. The socio-economic components, environmental factors and partially management intensity have been outlined in order to find farms covering a reliable scale of low-input farming. Three main groups of factors and conditions are implemented for developing an appropriate model for scaling: general socio-economic factors, non-enterprise specific socio-economic conditions, enterprise specific socio-economic conditions. The summarized effect of evaluated factors and conditions is presented by the farm intensity index. The developed model allows preliminary differentiation of farms in three main groups of intensity. A reliable scaling of farms prior to detailed assessment of indicators for biodiversity in low-input grasslands has been confirmed scientifically.

Ключови думи: биоразнообразие, пасища, ниски вложения, ферми, интензивност.
Key words: biodiversity, grasslands, low-input, farms, intensity.

INTRODUCTION

Biodiversity conservation – preserving species and their genetic variability, ecological communities, and landscape variety – is an urgent task for our society. Compared to other community types, European grasslands have a rich flora and they may develop a very high small-scale species density (Pärtel et al., 2005).

High grassland biodiversity is generally associated with low-input livestock systems that support less than 1LU (livestock unit) per ha (Duru and Hubert, 2003). Today biodiverse grasslands only survive where economic drivers towards intensification cannot operate (e.g. due to climatic or topographic constraints) or where there is adequate compensation against intensification via agro environment subsidies (Hodson et al., 2005). Biodiversity rich areas are often characterized by marked differences in management between fields that reflect topographic and environmental differences. Although grassland biodiversity may provide numerous potential utilization functions (Swift et al., 2004) at an ecosystem or landscape level, to livestock farmers the essential function is to feed herbivores.

The EU FP7 research project BIOBIO - *Indicators for biodiversity in organic and low-input farming systems* aims at developing scientifically sound, useful and attractive biodiversity indicators for organic and low-input farming in Europe. In the first project phase, possible indicators for genetic, species and habitat diversity as well as for farm management practices were screened for their scientific soundness and ranked by stakeholders for their usefulness and attractiveness in WP2 (BIOBIO Deliverable 2.1, Dennis et al., 2009). Developing biodiversity indicators for organic and low-input farming systems is the overall research objective for BioBio. Low-input farming systems (LIFS) are found all across Europe. "LIFS can be defined as a way to optimise the management and use of internal production inputs (i.e., on-farm resources) ... and to minimise the use of production inputs (i.e., off-farm resources), such as purchased fertilisers and pesticides, wherever and whenever feasible and practicable, to lower production costs, to avoid pollution of surface and groundwater, to reduce pesticide residues in food, to reduce a farmer's overall risk, and to increase both short- and long- term farm profitability" (Parr et al. 1990). LIFS are often located in marginal areas or in areas which are at risk of marginalisation due to unfavourable natural conditions for agriculture. Frequently LIFS in Europe are grazing systems. The second report (BIOBIO Deliverable D3.1, Arndorfer et al., 2010) summarises the characteristics of the case study regions and documents the process of selecting BioBio case study farms in each region. Eventually 10 to 20 farms per region were identified for a detailed assessment of biodiversity indicators in 2010.

The Bulgarian case study region in the BioBio project consists of low-input farming in grasslands of the Rhodope mountains. The most valuable ecosystems in Bulgaria are part of the agricultural landscape. About 350,000 ha of natural and semi-natural grazing habitats in the country are important for protection of biodiversity (Meshinev et al., 2005). They include diverse types of meadows and pastures that cover about 30% of the agricultural land of Bulgaria (Meshinev et al., 2005). The grazing habitats are considered being of high natural value (HNV) because of rich biodiversity that includes 51.5% of the Bulgarian flora. The agricultural land with HNV can be classified into 17 habitat types habitats included in Directive 92/43 (EC). Large parts of mentioned ecosystems are maintained predominantly by extensive agriculture – grazing for domestic animals and haymaking.

As part of this project the preliminary selection of 16 low-input farms in Bulgaria for a detailed assessment of biodiversity indicators is presented. This paper outlines the socio-economic components, environmental factors and partially management intensity for seeking farms covering a gradient of farming intensity. To this end an intensity index I is proposed.

MATERIALS AND METHODS

Preconditions for selecting farms were discussed during BioBio Workshop in Vienna in 2009 (Arndorfer et al., 2010). To ensure a sufficiently homogenous sample, two sets of potentially confounding factors can be identified: 1) Environmental conditions: biogeographical region, geomorphologic and soil features, landscape situation, altitude; 2) Farm characteristics: type of farm production.

In Bulgaria the Smolyan region was selected as 'Case Study Region' on the basis of the above mentioned BioBio selection criteria and public information on geographic and socio-economic conditions (Anonymous, 2009). The landscape is mountainous with grasslands and woodlands (predominantly coniferous) prevailing. Variation in altitude: 900 m to 1400 m; Soil: predominantly brown forest soils; Climate: Contingent on the relief and altitude. The average annual temperature in the region varies between 5°C and 10°C. The average annual rainfall is between 750 mm and 1100 mm. The average duration of snow cover is between 3 and 6 months.

Tourism is the priority branch in the regional economy which influences production area, ecology, sociology and cultural traditions of the region. Cattle-breeding and sheep-breeding are the basis for production of various original dairy products. The farm type could be described as „low-input farming system because of the limited (or non existing) use of fertilizers and pesticides in grasslands. About 30% of the Smolyan region is included in NATURA 2000 – bird protected areas.

Preliminary information on farms was collected by direct contact with farmers during three missions carried out in July-October 2009. This information was implemented for selection of case study farms. Because of confidentiality requirements in EC research projects, the place of farms and name of farmers are not presented. Farms are designated by numbers 1-16.

RESULTS AND DISCUSSION

Preliminary information on farms includes: name of the farmer, address and contacts, area of pastures and meadows (own and rented); species and number of livestock; other economic activities that could support the farmer's household - eco-tourism, rest house keeping, food production, bee keeping, etc. (Table 1). Forty two farms were involved in total, of which 34 confirmed their interest in the project BIOBIO. The number of case study farms was reduced to 32 because 2 of the interviewed farmers subsequently stopped farming. Among all 32 farms of the preliminary farm screening, 16 were selected randomly. The geographical distribution of selected farms in the frame of Smolian region is illustrated in Figure 1.

The socio-economic and ecological principles play an important role in grassland management and land-use practices (Mayer and Wytrzens, 1998; Duru and Hubert, 2003). On the basis of similar arguments we suggest that socio-economic and ecological factors (and the complex interactions between the two) are responsible for the differences in grassland management intensities. Recently it was reported that farm size and farming intensity are not related (Herzog et al., 2006). However we used farm size indirectly for calculation of livestock units per area. Creation of models is widely used for detecting differences in farm intensity (Mayer and Wytrzens, 1998; Trisorio et al., 2008; 2010; Reidisma et al., 2007). In this study we implement modeling approach for pre-selecting farms to ensure the needed pattern of variance for further evaluation of biodiversity indicators in farms. For scaling low-input farms a model design was developed using three main groups of factors and conditions: (1) general farm characteristics, (2) socio-economic framework conditions, (3) socio-economic farm characteristics (Table 2).

The general farm characteristics (1.1-1.3) remain constant across an enterprise or a group of enterprises including:

- LU per farm area – presented like *'conventional animal unit per ha' (CLU)*; Earlier this unit is described as 0.5 cattle per ha or 5 sheep per ha for mountainous grasslands (Bondev, 1991; Georgiev and Christov, 1944; Yancheva et al., 2002; Cheshmedzhiev, 1980; Christov, 1961). According to the last regulation of the Ministry of agriculture (RD-09-116/21.02.2011) is recommended the minimum grassland area per livestock unit (MGA): 0,6 ha for cattle, 0,1ha for sheep/goat, 0,6 ha for horse. This recommendation is used for calculation of conventional LU/ha in the present study;
- Road accessibility to the farm: track - 1; country road - 2; asphalt road - 3;
- Ownership: rented land for one up to three years - 1; governmental (long-term use) - 2; own – 3. The assumption that the rented land (1-3 years)

Table 1. Preliminary information for pre-selected low-input farms

	Farm characteristic	Number of farms involved	Mean value	Minimum value	Maximum value
1.	Utilized agricultural area, ha	32	43,96	4,8	120
2.	Type and number of livestocks				
	Sheep	28	321	5	800
	Cattle	5	35	2	120
	Goats	4	39	4	120
	Horses	6	45	1	200
	Ancien breed *Rhodope short horn cattle*	1	13	13	13
3.	Other economic acivities				
	Bee keeping (number of bee hives)	4	-	1	25
	Dog breeding *Bulgarian shepherd dog* (number of dogs for work and for breeding)	32	-	2	11
	Dairy products on farm production	10	-	-	-
	Rest house keeping	2	-	-	-

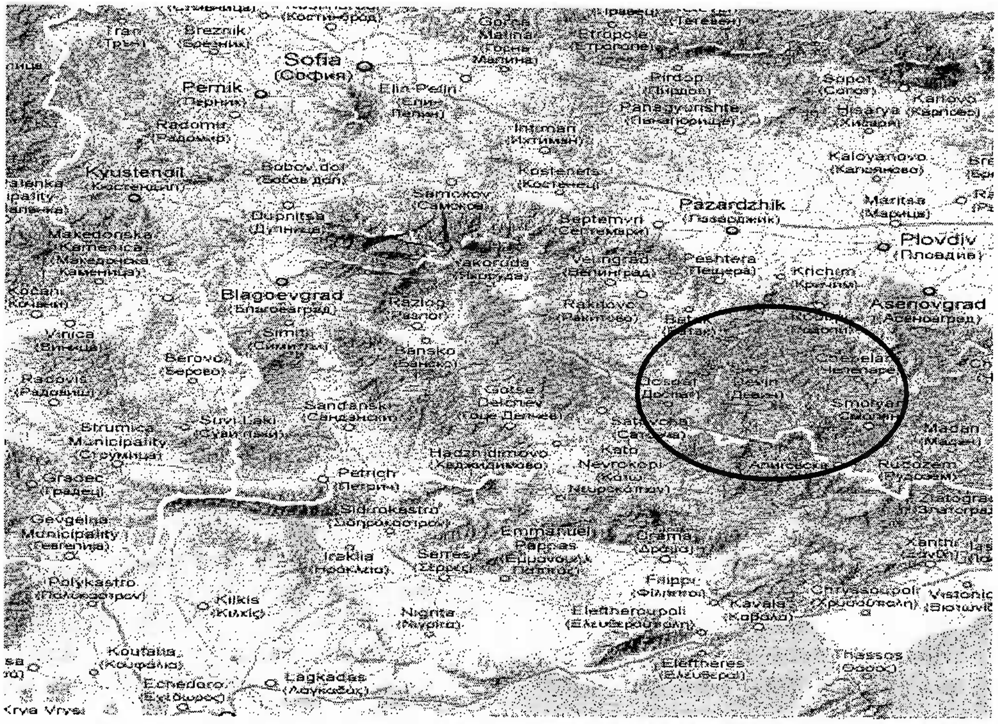


Fig. 1. Place of case study farms in Smolian region shown with red shade in the market area

is not so intensively used comparing to long-term rented is based on the findings for grassland maintenance. Supporting practices for cleaning, pruning or uproot of harmful plants in short-term rented lands are implemented seldom than in long-term rented or own lands.

The socio-economic framework conditions refer to potential supporting factors like: market for dairy products processed on farm, grants and subsidies and other incomes outside agriculture (2.1-2.3). These factors could be associated more convincing with farm sustainability than with farm intensity (Olale, 2011). However in our case should be pointed that incomes outside agriculture in visited farms more often are used for farm intensification including: enlargement of flocks, more resources for concentrated fodder instead of hay locally produced and investments for mechanization. As presented in other studies the regional agricultural policy for farming is based on standard incomes in farms where direct payment of subsidies relates to more intensive non-HNV farming (Trisorio et al., 2008; 2010). As opposite the same authors suggest that the economic support measures to prevent abandonment and payments to prevent intensification or land conversion could be the proper tool targeted on less intensive HNV farming. In our study agricultural subsidies include mostly direct payment per agricultural area but not for low-intensive farming and could be suggested as conductive to farm intensity. This group of conditions depends more often on administrative criteria and limits in the region for farming. Their evaluation in the scale yes/no is described respectively by 1/0.

The socio-economic farm conditions in a farm are specific and vary considerably (3.1.-3.6). The overall conditions together with local socio-economic conditions prevailing in individual farms lead to a large gradient with respect to: farming method; farm employment status; production factors; equipment and mechanization; food processing on farm and guest-house maintenance; education of the farmer. These conditions are described in a scale of 1 to 3 (Table 2). The relation of farming method to farm intensity is widely recognized (Kleijn et al., 2009). The grading of this factor is described on the basis of information for inputs use – fertilizers and pesticides. There should be pointed that the acting of other factors described above (3.2 – 3.6) on the level of intensity is suggested indirectly on the basis of other studies (Reidisma et al., 2007; Trisorio et al., 2008; 2010). Farm employment or 'work units' is described as important variable in economic analyses and modeling of European farms (Reidisma et al, 2007). In the present study the scaling of this factor is described with 1 to 3 depending to the number of employed. Factors 3.3, 3.4 and 3.5 illustrate the technical conditions for farming and their scaling is created on cumulative principle as shown in Table 2. The factor 'education of farmer' affects technology adoption and concerns farmers' characteristics for detailed surveys (Reidisma et al., 2007). The conditions in this group are very specific. However they represent the real situation at enterprise level and illustrate the differences of pre-selected farms.

Using the described modeling approach the pattern of socio-economic conditions within 16 low-input farms in

Table 2. Model design describing the scale in grassland management of low-input farms

	Factors and conditions	Description	Scale
1.General Socio-Economic Factors	1.1. LU density per area and Conventional Animal Unit per ha	CAU=LU X MGA Where MGA = 0,6 ha for cattle; 0,1ha for sheep/goat; 0,6 ha for horse	
	1.2. Accessibility	track country road asphalt road	1 2 3
	1.3. Ownership	rented land 1-3 years governmental (long-term use) own	1 2 3
2. Not Enterprise Specific Socio-Economic Conditions	2.1. Market for dairy products on farm processed	yes; no	1; 0
	2.2. Grants and subsidies	yes; no	1; 0
	2.3. Potential sources of incomes outside agriculture	yes; no	1; 0
3.Enterprise Specific Socio-Economic Conditions	3.1. Farming method	-low-inputs (no fertilizers or pesticides used) -reduced input use (organic fertilizers used on some parts of grasslands) - artificial fertilizers used, no pesticides)	1 2 3
	3.2. Farm employment status	- farmer's family (1-2 persons) - farmer's family plus 1-3 employed - farmer's family plus more than 3 employed	1 2 3
	3.3. Availability of production factors	- own land - own land + workers - own land + workers + technical equipment	1 2 3
	3.4. Availability of farming equipment	- stables/pens - stables/pens plus fodder storage capacity - stables/pens plus fodder storage capacity inside mechanization (milker, sterilizer, milk cooling bath, storage cold room)	1 2 3
	3.5. Food processing and farmhouse holydays	yes; no	1; 0
	3.6. Education of farmer	- secondary school - specialized/college education; - university education	1 2 3

Rhodope mountains is observed (Table 3). Three indices are described with respect to above mentioned groups: I_1 – using the general farm characteristics (1.1-1.3); I_2 – based to the socio-economic framework conditions (2.1-2.3); I_3 – as result of the specific socio-economic farm characteristics (3.1.-3.6). The summarized effect of all described factors and conditions is presented by the farm intensity index:

$$I = I_1 + I_2 + I_3$$

The intensity index for preliminary scaling of farms is used to detect more scientifically sound arguments for farms differentiation. This modeling approach is illustrated in the diagram (Fig. 2). Three groups of low-input farms with respect to intensity index (I) are clear distinguished:
- Low-intensive with I < 15 (Farm 3; Farm 4; Farm 8; Farm 9; Farm10; Farm 13)
- Medium intensive where 15 < I < 20 (Farm 2; Farm 5; Farm 12; Farm 15)
- Relatively intensive where I > 20 (Farm 1; Farm 6; Farm 7; Farm 11; Farm 14; Farm 16).

The suggested scaling of farm intensity allows a differentiation of the farms before the comprehensive study of biodiversity indicators. It describes that, although the

Table 3. Pattern of socio-economic conditions within 16 low-input farms in Rhodope mountains

FACTORS/CONDITIONS	FARM NUMBER															
	B01	*B02*	*B03*	*B04*	*B05*	*B06*	*B07*	*B08*	*B09*	*B10*	*B11*	*B12*	*B13*	*B14*	*B15*	*B16*
1. Socio-Economic Factors																
Farm size, ha	30	4	6	9	42	36	37	7	18	9	49	7	9	34	32	60
Livestock Units:																
Sheep/Goat (0,1)	260		170	240	260	250	680	160	410	150	300	250		540	180	700
Cattle (0,6)		14				99							14			
Horse (0,8)					8									210		
1.1. Conventional animal unit per ha	0,9	2,1	2,8	2,6	0,8	2,3	1,8	2,3	2,3	1,7	0,6	3,5	0,9	6,5	0,6	1,2
1.2. Accessibility	3	1	1	2	2	3	2	2	1	2	3	1	1	2	1	3
1.3. Ownership	3	3	1	3	3	2	3	1	1	1	3	3	3	1	3	1
Index 1	6,9	6,1	4,8	7,6	5,8	7,3	6,8	5,3	4,3	4,7	6,6	7,5	4,9	9,5	4,6	5,2
2. Not Enterprise Specific Socio-Economic Conditions																
2.1. Market for dairy products on farm processed	1	1	1	0	1	1	0	0	0	0	1	0	0	0	1	0
2.2. Grants and subsidies	1	1	1	0	1	1	1	1	1	1	1	1	1	1	1	1
2.3. Potential sources of incomes outside agriculture	1	0	0	0	0	1	1	0	0	0	0	1	0	1	0	1
Index 2	3	2	2	0	2	3	2	1	1	1	2	2	1	2	2	2
3. Enterprise Specific Socio-Economic Conditions																
3.1. Farming method	2	2	1	1	1	3	2	1	1	1	1	1	1	2	1	2
3.2. Farm employment status	2	1	2	2	1	3	3	1	1	1	2	2	2	3	2	3
3.3. Availability of production factors	3	2	1	2	2	3	3	1	1	1	3	2	1	3	2	3
3.4. Availability of farming equipment	3	2	1	1	2	3	3	1	1	1	2	1	1	3	1	3
3.5. Food processing and farmhouse holydays	1	1	0	0	1	1	1	0	0	0	1	0	0	1	1	1
3.6. Education of farmer	1	3	1	1	1	3	1	3	2	1	3	2	3	2	2	2
Index 3	12	11	6	7	8	16	13	7	6	5	12	8	8	14	9	14

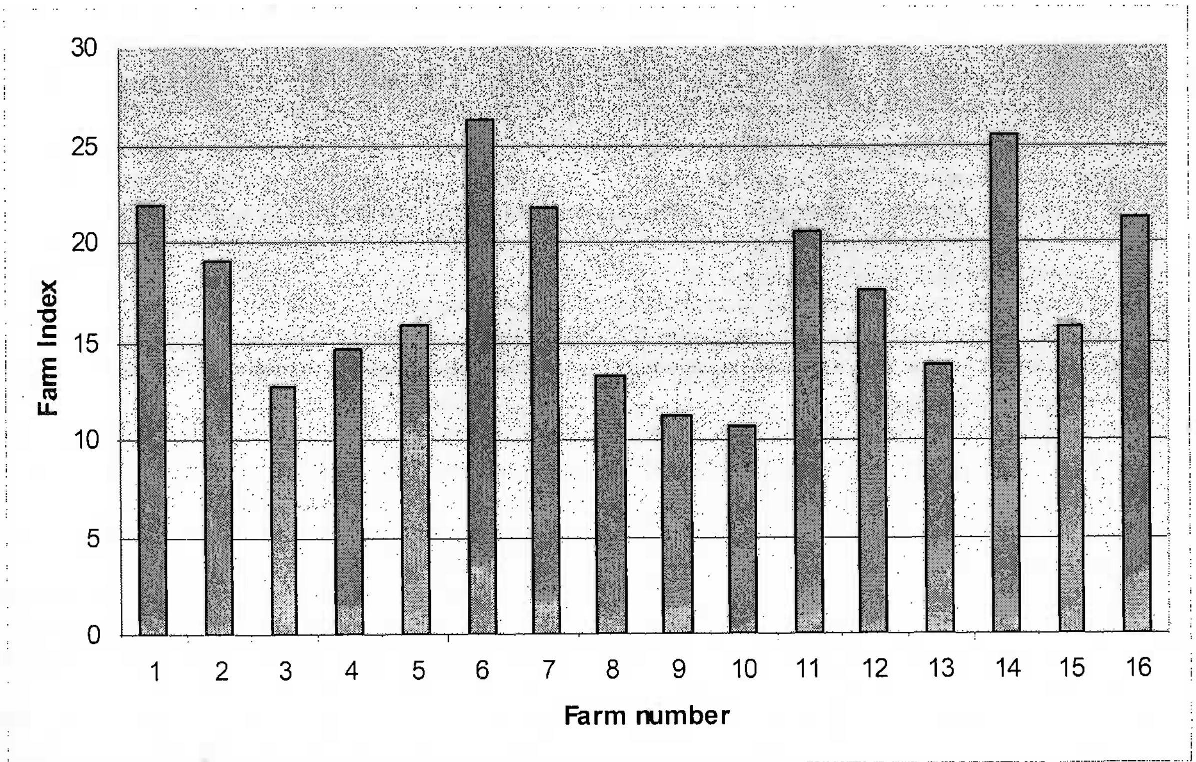


***Fig. 2**. The Intensity Index Scale for graduation of 16 farms for low-input farming in Rhodope mountains: $I = I_1 + I_2 + I_3$*
Low-intensive (I < 15): Farm 3; Farm 4; Farm 8; Farm 9; Farm10; Farm 13;
Medium intensive (15 < I < 20): Farm 2; Farm 5; Farm 12; Farm 15;
Relatively intensive (I > 20): Farm 1; Farm 6; Farm 7; Farm 11; Farm 14; Farm

farms were randomly selected, they span a gradient of intensity based on socio-economic factors and conditions.

CONCLUSIONS

The socio-economic components, environmental factors and partially farm management could be used to create a scale of low-input farms by their intensity. Three main groups of factors and conditions are implemented for development of appropriate model for scaling: (1) general socio-economic factors, (2) specific socio-economic conditions non-enterprise related, (3) enterprise specific socio-economic conditions. The total effect of evaluated factors and conditions is presented by the farm intensity index. This index allows preliminary scaling of farms for detection of more scientifically sound arguments for farms differentiation in three groups of intensity: low-intensive, medium intensive and relatively intensive.

The presented model confirms a reliable scale for farm differentiation before detailed assessment of indicators for biodiversity in low-input grasslands.

REFERENCES

Anonymous, 2009. National Strategy for regional development of Republic Bulgaria during 2005-2015 http://www.europe.bg/htmls/page.php?category=264&id=2231

Arndorfer, M., Angelova, S., Balázs, K., Bogers, M. I. , Centeri, C., Choisis, J.-P., Choisis, N. , Dennis, P., Eiter, S., Falusi, E., Fjellstad, W., Friedel, J., Geijzendorffer, I., Gomiero, T., Griffioen, A. J., Guteva, Y., Jongman, R., Juarez, E., Kainz, M., Kelemen, E., Lüscher G., Mayr, J., Moreno, G., Paoletti, M. G., Podmaniczky, L., Sarthou, J.-P., Skutai, J., Stoyanova, S., Schneider M., Siebrecht, N., Wolfrum, S., Wilkes J., Zanetti, T., 2010. Delimination of BIOBIO case study regions and the selection of case study farms. Deliverable 3.1 of the EU FP7 Project BioBio. www.biobio-indicator.org

Bondev, I., 1991. The vegetation of Bulgaria. "Sv. Kliment Ochridski" University publishers, p.183. (bulg.).

Cheshmedzhiev, I., 1980. Forage characteristics of natural pastures and meadows in Bulgaria. Zemizdat publishers, Sofia, p. 211. (bulg.).

Christov, M. A., 1961. The questions of forage production. Zemizdat publishers, Sofia, 215-225.

Dennis, P., Arndorfer, M., Balázs, K., Bailey, D., Boller, B., Bunce, R.G.H., Centeri, Cs., Corporaal, A., Cuming, D., Deconchat, M., Dramstad, W., Elyakime, B., Falusi, E., Fjellstad, W., Fraser, M.D., Freyer, B., Friedel, J.K., *Geijzendorffer, I., Jongman, R., Kainz, M., Marcos, G.M., Gomiero, T., Grausgruber-Gröger, S., Herzog, F., Hofer, G., Jeanneret, P., Kelemen, E., Kölliker, R., Moakes, S.R., Nicholas, P., Paoletti, M.G., Podmaniczky, L., Pointereau, P., Sarthou, J.-P., Siebrecht, N., Sommaggio, D., Stoyanova, S.D., Teufelbauer, N., Viaggi, D., Vialatte, A., Walter, T., Widmer, F., Wolfrum, S.*, 2009. Conceptual foundations for biodiversity indicator selection for organic and low-input farming systems. Aberystwyth, Deliverable 2.1 of the EU FP7 Project BioBio. ISBN 978-3-905733-16-7. http://www.biobio-indicator.wur.nl/UK/Publications+and+deliverables/

Duru, M., Hubert, B., 2003. Management of grazing systems: from decision and biophysical models to priciples for actio. – *Agronomie*, 23, 689-703.

Georgiev, T., M. Christov, 1944. Pasture types in Bulgaria, Ann. Report, Sofia University, 1; 181-217.

Herzog, F., R., Dirksen, J., Dormann, C.F., De Filippi, R., Frossard, E., Liira, J., Schmidt, T., R. Stöckli, R.,Thenail, C., van Wingerden, W., Bugter, R., 2006. Assessing the intensity of temperate European agriculture at the landscape scale. – *Europ. J. Agronomy* 24, 165-181.

Hodson, J.G., Montserrat-Martı, G., Tallowin, J., Thompson, K., Dıaz, S., Cabido, M., Grime, J.P., Wilson, P.J., Band, S.R., Bogard, A., Cabido, R., Caceres, D., Castro-Dıez, P., Ferrer, C., Maestro-Martınez, M., Perez-Rontome, M.C., Charles, M., Cornelissen, J.H.C., Dabbert, S., Perez-Harguindeguy, N., Krimly, T., Sijtsma, F.J., Strijker, D., Vendramini, F., J. Guerrero-Campo, J., Hynd, A. , Jones, G., Romo-Dıez, A., de Torres Espuny, L., Villar-Salvador, P., Zak, M.R., 2005. How much will it cost to save grassland diversity? *Biological Conservation*, 122, 263–273.

Kleijn, D., Kohler, F., Baldi, A., Batary, P., E. D. Concepcion, E.D., Clough, Y., Diaz, M., Gabriel, D., Holzschuh, A., Knop, E., Kovacs, A., Marshall, E. J. P., Tscharntke, T., Verhulst, J,. 2009. On the relationship between farmland biodiversity and land-use intensity in Europe. – In: Proceedings of the Royal Society B, 276, 903–909.

Mayer, C. and Wytrzens, H. K., 1998. A multidisciplinary framework for optimising the intensity of grassland management in mountain regions of Austria. 2nd LSIRD conference on livestock production in the European LFAs, Bray, Ireland. December, 1998.

Meshinev, T., Apostolova, I., Georgiev, V., Dimitrov, V., Petrova, A., Veen, P., 2005. Grasslands of Bulgaria, Final report on the National Grasslands Inventory Project – Bulgaria, 2001-2004 (PINMATRA / 2001/020), p. 104.

Olale, E., 2011. The influence of market barriers and farm income risk on non-farm income diversification. – Journal of Development and Agricultural Economics, 3(8), 379-386.

Parr, J. F. et al., 1990. Sustainable Agriculture in the United States. – In: Clive A. Edwards, R. Lal, P. Madden, R. Miller, G. House (eds.) Sustainable Agricultural Systems. Ankeny IA: Soil and Water Conservation Society, p. 52.

Pärtel, M., Bruun, H. H., Sammul M., 2005. Biodiversity in temperate European grasslands: origin and conservation. In: 13th Symposium on "Integrating Efficient Grassland Farming and Biodiversity" Grassland Science in Europe, 10, p. 666.

Reidsma, P., Ewert, F., & Alfons Oude Lansink, A., 2007. Analysis of farm performance in Europe under different climatic and management conditions to improveunderstanding of adaptive capacity. Climatic Change, 84, 403-422.

Swift, M. J. et al., 2004. Biodiversity and ecosystem services in agricultural landscapes – are we asking the right questions? – Agriculture, Ecosystem & Environment, 104, 113-134.

Yancheva, Ch., Angelova, S., Koeva, R., 2002. Management of steppe habitats in pastures of the region of Kaliakra reserve in Bulgaria; Grassland Science in Europe, XIX-th. General meeting of EGFMultifunction grasslands; La Roñhelle, France: 860-862.

Trisorio, A., Povellato, A., Bortolozzo, D., 2008. High Nature Value Farming Systems in Italy: an Economic Perspective . – In: Proceedings of the Conference Using Evaluation to Enhance the Rural Development Value of Agro-Environmental Measures", Parnu, June 17-19.

Trisorio, A., Povellato, A., Bortolozzo, D., 2010. High Nature Value Farming Systems in Italy: a Policy Perspective. OECD Workshop: Agri - environmental Indicators: Lessons Learned and Future Directions, 23-26 March, 2010, Leysin, Switzerland.

Acknowledgement: This study was supported by the EU FP7 Project BIOBIO: *Indicators for biodiversity in organic and low-input farming systems*, KBBE-2008-1-2-01, Contract № 227161.

Статията е приета на 7.10.2011 г.
Рецензент – доц. д-р Владислав Попов
E-mail: vpopov_bg@yahoo.com