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BIOBIO

Indicators for biodiversity in organic and low-input farming systems

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Conceptual foundations for biodiversity indicator selection for organic and low-input farming systems

Deliverable D2.1

Report on a) Conceptual foundation and on criteria for selecting scientifically sound biodiversity indicators for orgnabic/low – input farming systems, state of the art of existing indicator and monitoring programmes; b) The applicability of existing European databases for assessing the biodiversity in organic/low – input farming systems; c) Tentative list of indirect and direct biodiversity indicators to be reviewed by Stakeholder Advisory Board

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Conceptual foundations for biodiversity indicator selection for organic and low-input farming systems

Fourth draft - Final version of report

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Preface

This report was written in the context of the EU FP7 research project BIOBIO - *Indicators for biodiversity in organic and low-input farming systems*. At the start of this project, we set out to summarise the state of the art of indicator development for genetic, species, habitat and farm management indicators related to farming systems (FIG. 0.1). Based on this exercise, we submitted a list of candidate indicators to the Stakeholder Advisory Board which is associated with the project and – with support from the stakeholders – we selected 40 indicators to be tested in a field survey of 12 case studies across Europe, covering the major organic and low-input farming systems and the main European bio-geographical regions.

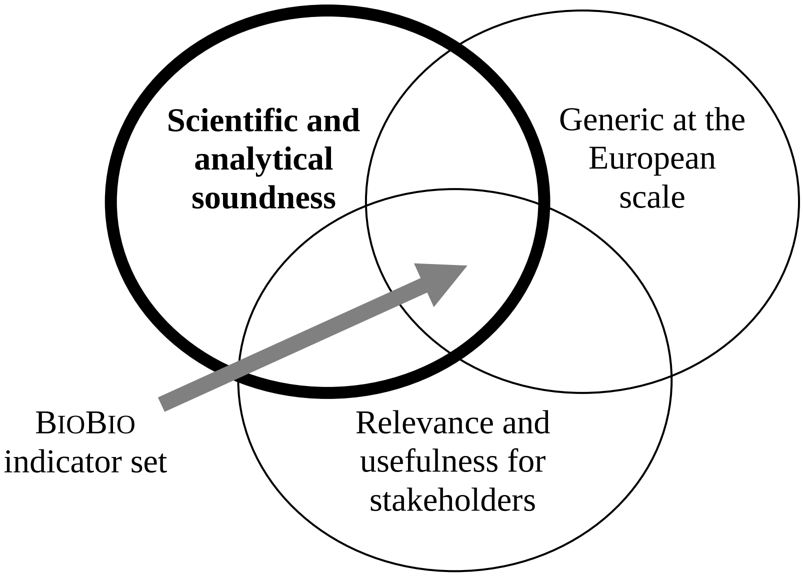

FIGURE 0.1. CONCEPTUAL FRAMEWORK OF REQUIREMENTS FOR BIODIVERSITY INDICATORS WHICH THE RESEARCH PROJECT BIOBIO WILL TEST AND PROPOSE.

CONTENTS

1. EXECUTIVE SUMMARY

1.1. This work characterised and identified indicators of biodiversity suitable for the evaluation of organic and low input farming systems.

1.2. A full review is provided of scientific properties of indicators and the current availability of indirect and direct biodiversity indicators applied in agricultural and other ecosystems.

1.3. Possible indirect indicators can be founded on existing farm accounts (FSS and FADN), farmer interviews and assessment of management intensity during farm visits.

1.4. Direct indicators are broad in scope and are available to assess the genetic diversity of cultivated plants and livestock breeds, the genetic characterisation of soil micro-organisms and the species diversity of plant and animal life, both domesticated and representing wildlife. The final group of indicators are measurements of habitats and landscape of farms, including linear features that are often refugia for much of the species diversity.

1.5. The four distinct lists of indicators: indirect, genetic, species and habitats/ landscape were evaluated by an expert group applying scientific selection criteria. This produced a priority list for each group for evaluation based on the application of criteria proposed by a stakeholder group and an assessment of cost of effort in the field, laboratory, for analysis and communication elements of the implementation of each candidate indicator.

1.6. Interactions between these indicator sets were next assessed and the most complementary combinations were selected to cover the necessary range of biological organisation and spatial scales.

1.7. The higher scoring list of indicators under the four headline groups were summarised in a series of Fact Sheets and scrutinised by the Stakeholder Advisory Board (SAB) prior to a workshop, applying the full list of 18 selection criteria. SAB recommendations were considered in the final choice of short list of candidate indicators for field testing in Work Package 3.

1.8. In total, the candidate list included 10 indicators for the genetic diversity, 5 for the species diversity, 13 for the habitat diversity and 12 indirect or farm management indicators/accounts records

1.9. Full methods for measuring parameters in the field or collecting farm records to derive indicators will be detailed in the Deliverable 2.2 Field manual for implementation and validation in Work Package 3.

2. TASK GROUPS AND CONTRIBUTORS

The objectives of WP 2 were achieved by the activity of eight Task Groups (FIG. 2.1). Each TG reviewed and selected key papers and reports, summarising the information into a bibliography. Lists of direct and indirect indicators of biodiversity are organised into tables justified with reference to supporting papers and reports. Candidate biodiversity indicators to take forward for validation in field trials (WP 3) are advocated under sections dealing with specific biodiversity indicator categories.

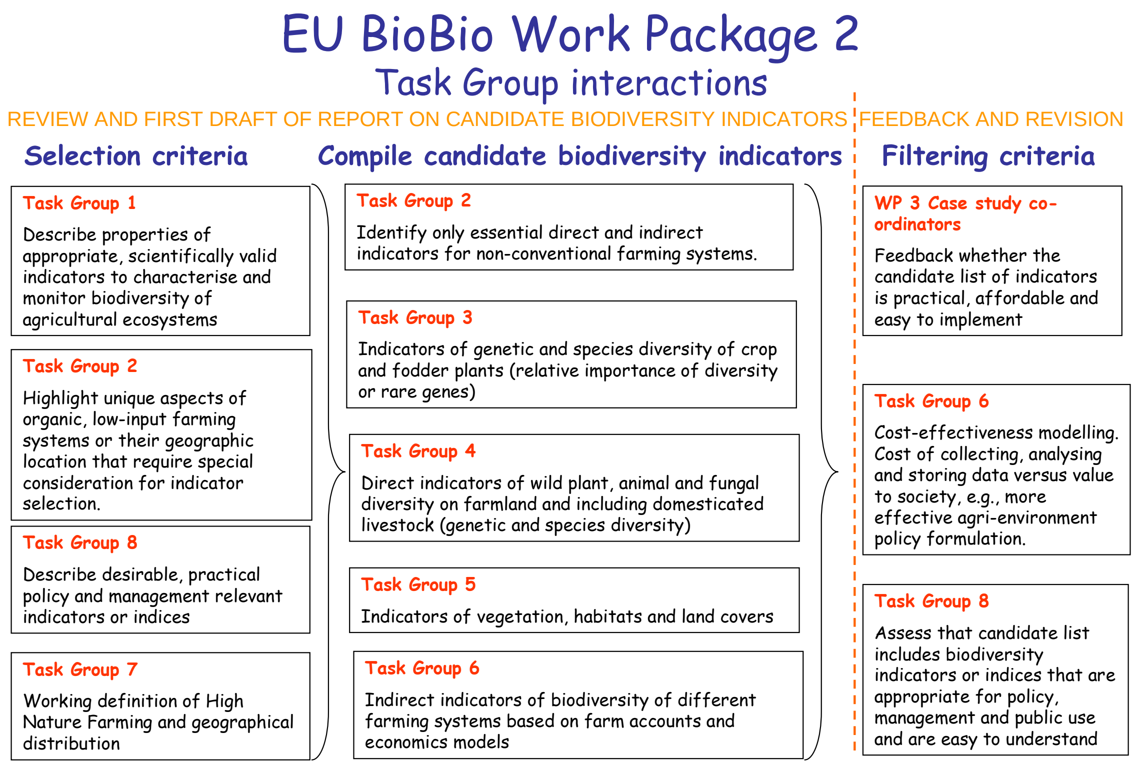

FIGURE 2.1. TASK GROUPS AND ACTIVITIES ORGANISED INTO THE SECTIONS FOR WP 2, DELIVERABLE 2.1 REPORT ON CANDIDATE INDICATORS

3. INTRODUCTION

3.1. INDICATOR THEORY, PROPERTIES AND SELECTION CRITERIA

Indicators have been defined in many ways. The definition by UNEP is suitable to the context of BioBio, i.e., the development and assessment of a scientifically-based set of indicators capable of detecting qualitative and quantitative linkages between different organic/low-input farming systems and biological diversity for Europe: "Indicators serve four basic functions: simplification, quantification, standardization and communication. They summarize complex and often disparate sets of data and thereby simplify information. They usually assess trends with respect to policy goals. They should provide a clear message that can be communicated to, and used by, decision makers and the general public" (Ad Hoc Expert Group on biodiversity indicators, NEP/CBD/SBSTTA/9/10).

Due to the complexity of all aspects of biodiversity, there is no doubt that biodiversity in the broadest sense of the Rio Convention cannot be measured as such and it is accepted that a single indicator for biodiversity cannot be devised (e.g., Büchs, 2003ab). Organisms are sensitive to the environmental conditions of the ecosystem in which they are living. Their occurrence and abundance may therefore vary according to the state of the ecosystem. A species or a taxon may be a good indicator for heavy metals in the environment without indicating biodiversity. This organism is considered as a bio-indicator of contamination but not as a biodiversity indicator (McGeoch, 1998). This

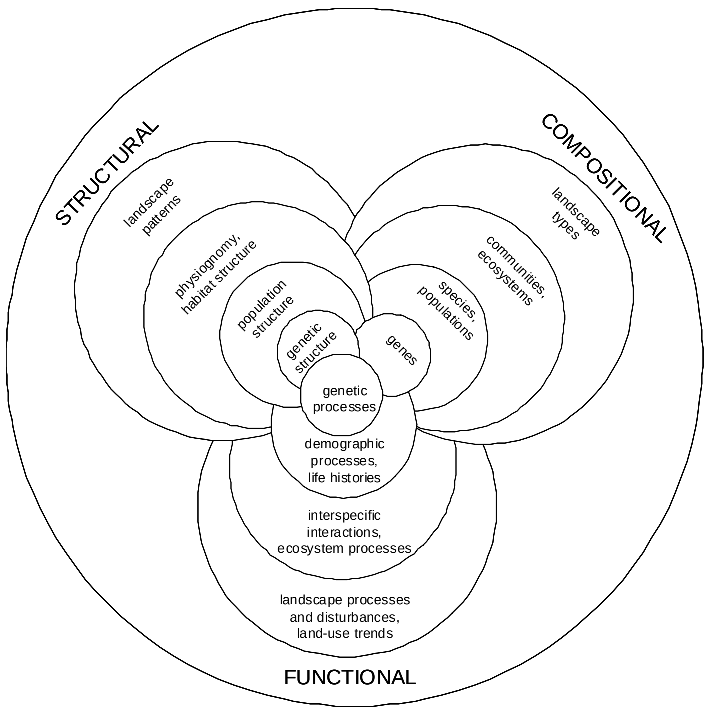

FIGURE 3.1. LEVELS OF INFORMATION THAT CAN BE CONSIDERED FOR BIODIVERSITY AND ECOSYSTEM STUDIES (NOSS 1990).

distinction is crucial to not consider every living organism as a biodiversity indicator. To this purpose, Duelli (2003) distinguished "indicator FOR or FROM biodiversity". Ideally indicators should be selected that express or represent both the biodiversity as a whole AND because they are sensitive to environmental conditions resulting from, in the case of organic and low-input farming, land use and agricultural management practices.

In a large sense, Noss (1990) has shown that it is possible to develop a hierarchy of indicators from gene to landscape level based on the distinction between structure, composition, and function (FIG. 3.1). Examples of structural indicators in the context of a cultivated field are cultivated plant architecture and openness of the cultivation. The second group comprises compositional indicators. These can be functionally important species that are sensitive to and thus indicate management practices, isolation of the habitat, but also habitat structure indicators. The third group comprises functional indicators. These are indicators of the abiotic and biotic disturbance factors and management regimes that are present, e.g., razing impact, cutting regimes.

3.1.1. MOTIVATIONS FOR USING BIODIVERSITY INDICATORS

Indicators may be categorized according to three important motivations to preserve and enhance biodiversity in the agricultural context (Duelli, 2003), i.e., (i) indicators reflecting nature protection purposes (species conservation with focus on rare and endangered species), (ii) indicators reflecting ecological resilience (focus on genetic and species diversity) and (iii) indicators reflecting plant protection purposes (biological control of potential pest organisms with focus on predatory and parasitoid arthropods). This last category may be extended to additional issues with respect to important ecosystem services in agriculture, e.g., indicators of soil health and fertility (markers for soil microbial and fungal diversity and macro-invertebrates), indicators including beneficial organisms (in addition to predatory and parasitoid arthropods) providing biological control of pests, and pollinators. The BioBio approach seems to be promising for the purpose of developing appropriate indicators for the linkage between organic/low-input farming systems and biodiversity because it considers nature conservation goals (species conservation), genetic resources and other components of biodiversity (ecological resilience) and economic aspects (crop production). According to Clergue *et al. (*2005), the three parts may be extended to three main functions, respectively, i.e., patrimonial, ecological and agronomical functions.

BioBio is not intended to develop new and previously untested biodiversity indicators, motivations of the project are:

- a concise and stringent evaluation of existing indicator systems according to clear criteria relevant for organic and low-input farming systems at the European level;
- the maximisation of synergies with already existing European indicator systems, be they landscape, biodiversity or farm economics oriented, for application in the context of organic and low-input farming systems;
- the development of indicators that combine measurements at a fine spatial resolution (farm/landscape) with requirements for reporting for large geographical areas;
- a practical test of biodiversity indicators across all major organic and low-input farming systems in Europe;

- a practical test of biodiversity indicators in selected ICPC countries to assess the adaptability of the indicators and their wider relevance for organic/low-input farming systems globally;
- the assessment of private and public economic benefits, and non-monetary value of biodiversity promoted by organic and low-input farming;
- a systematic integration of European and local stakeholders throughout the research project, furthering mutual understanding between researchers and stakeholders;
- production of standardised protocols and recommendations that will enable establishment of biodiversity monitoring across different farming systems and countries, thus laying the foundations for increasing understanding of the links between farming practices and biodiversity at the European scale and beyond.

3.1.2. CRITERIA FOR SELECTING BIODIVERSITY INDICATORS

Several authors and institutions, such as UNEP (2003), have proposed lists of criteria that should be met by indicators (TABLE 3.1).

TABLE 3.1. QUALITY CRITERIA OF BIODIVERSITY INDICATORS (UNEP, 2003)

Criterion	Description
For individual indicators:	
1. Policy relevant and meaningful.	Indicators should send a clear message and provide information at a level appropriate for policy and management decision making by assessing changes in the status of biodiversity (or pressures, responses, use or capacity), related to baselines and agreed policy targets if possible.
2. Biodiversity relevant.	Indicators should address key properties of biodiversity or related issues as state, pressures, responses, use or capacity.
3. Scientifically sound.	Indicators must be based on clearly defined, verifiable and scientifically acceptable data, which are collected using standard methods with known accuracy and precision, or based on traditional knowledge that has been validated in an appropriate way.
4. Broad acceptance.	The power of an indicator depends on its broad acceptance. Involvement of the policymakers, and major stakeholders and experts in the development of an indicator is crucial.
5. Affordable monitoring.	Indicators should be measurable in an accurate and affordable way and part of a sustainable monitoring system, using determinable baselines and targets for the assessment of improvements and declines.
6. Affordable modelling.	Information on cause-effect relationships should be achievable and quantifiable, in order to link pressures, state and response indicators. These relation models enable scenario analyses and are the basis of the ecosystem approach.
7. Sensitive.	Indicators should be sensitive to show trends and, where

	possible, permit distinction between human-induced and natural changes. Indicators should thus be able to detect changes in systems in time frames and on the scales that are relevant to the decisions, but also be robust so that measuring errors do not affect the interpretation. It is important to detect changes before it is too late to correct the problems being detected.
For sets of indicators: 8. Representative.	 The set of indicators provides a representative picture of the pressures, biodiversity state, responses, uses and capacity (coverage).
9. Small number.	The smaller the total number of indicators, the more communicable they are to policy makers and the public and the lower the cost.
10. Aggregation and flexibility.	Indicators should be designed in a manner that facilitates aggregation at a range of scales for different purposes. Aggregation of indicators at the level of ecosystem types (thematic areas) or the national or international levels requires the use of coherent indicators sets (see criteria 8) and consistent baselines. This also applies for pressure, response, use and capacity indicators.

3.1.3. FRAMEWORK FOR STRUCTURING INDICATOR SYSTEMS

We will use the DPSIR framework (Klotz, 2007; EEA, 2005; IRENA operation) to structure the indicators according to the different components of the system (FIG. 3.2). Coarse processes of land use/land cover, farming practice categories etc. drive the actual pressures and benefits, i.e., the concrete farm operations, which in turn act on farmland biodiversity (state/impact indicators, direct indicators). If pressures have a positive or negative impact on the state of an ecosystem or species (negative or positive trends), then they will stimulate managers or policy makers to act and give a response, through management decisions, quality requirements or through technical renewal (new farm practices) to improve the situation of farmland biodiversity.

Several classifications of biodiversity indicators have been proposed (Levrel, 2007; Waldhardt, 2003) which can be explained by the diversity of criteria used to characterize the indicators. In the context of agriculture, a review of the main methods to evaluate environmental impacts of management and cultural systems has recently been conducted by Bockstaller *et al.* (2008). Methods for monitoring state, drivers and responses based on indicators are very diverse, although analysis is restricted to indicators related to relationships between agriculture and environment: kind of objectives (evaluation, decision making, scientific, political, economic issues, etc.), kind of targets (institutions, social groups, etc.), relevant scale, etc.

In BioBio we propose to divide the biodiversity indicators into direct and indirect indicators as suggested in Le Roux *et al.* (2008). Both types of indicators may report on biodiversity itself but may also provide information on associated functions (Clergue, 2005).

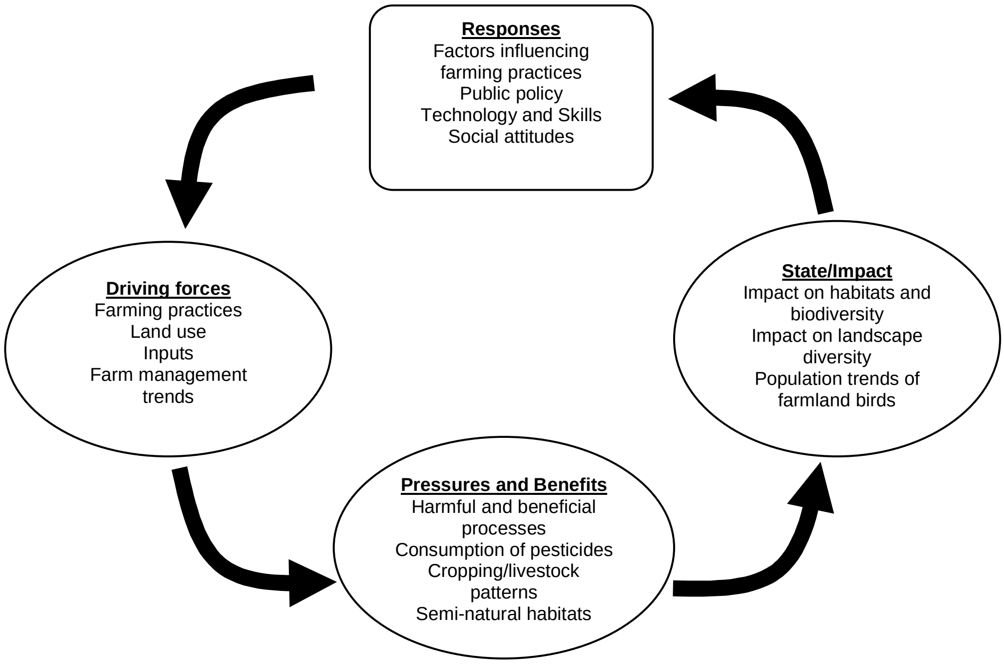

FIGURE 3.2. DPSIR FRAMEWORK FOR DEVELOPING FARMING AND BIODIVERSITY INDICATORS (EXAMPLES SHOW RELEVANT ISSUES FOR AGRICULTURE).

3.2. USEFULNESS AND APPLICABILITY OF EXISTING INDICATOR SYSTEMS

There has been rapid development of environmental indicators to fulfil demands for international environmental monitoring programmes since the UNEP Environmental data report (1987). The increasing need to assess the ecological effects of pollution and climate change (WCED, 1987; EEA, 2004b; ALTER-Net, 2008) drove a demand for biological indicators. Indicator development at a European level has focused on regional and national scale monitoring (EEA, 1999; Delbaere 2002; EEA 2004a) to assess national progress towards national biodiversity targets since the Convention on Biodiversity, Rio 1992 (CBD, 2004) and renewed commitments to halt the loss of biodiversity by 2010 (CEC, 2006; EU Council, 2004; EP, 2004; EEA, 2006; EEA, 2007). Current efforts are directed towards developing harmonised and integrated monitoring programmes across Europe using common biological indicators (ALTER-Net, 2008; SEBI2010, 2007). Examples include the European land cover map (CORINE), common bird survey (Gregory *et al.,* 2005; PECBM, 2007) and butterfly survey (Roy *et al.,* 2007). Indicators have been designed for Pan-European use across all ecosystems either in dedicated Long-Term Ecological Research sites (ETC/NPB, 2003; ALTER-Net, 2008) or in the wider countryside (Bredemeier *et al.,* 2007).

Much of the wider countryside in the European context is under agricultural land use. Indicators of environmental effects of agricultural policy have been developed at the regional and national scale (Dramstad *et al.,* 2002; EEA, 2005; EEA, 2006; Gaillard *et al.,* 2003; OECD, 2001; OECD, 2003, ter Brink, 2000). These have been increasingly adapted to assess the effects of particular farming systems or agri-environment schemes on biodiversity (De Roeck, 2005; Wascher, 2000). However, most of the indicators have not been tested with real data. Although some major studies of biodiversity have been carried out at the farm scale, notably the evaluation of genetically modified crops in the UK (Firbank *et al.,* 2006), biological indicators have not been developed for specific farming systems. Reports mentioned above identify and provide lists of indicators which result from political and scientific compromises, and are useful for policymakers. These indicators have been developed for global scale assessment (regional, national and international) and are therefore not primarily adapted to evaluate the agricultural management at the plot or farm level.

When choosing between very similar indicators of equal quality, those that are already in use by EEA/ OECD (TABLE 3.2) should be given priority. In such cases, the same definitions should be used in order to avoid confusion.

TABLE 3.2. INDICATORS USED IN EXISTING INTERNATIONAL AND SELECTED NATIONAL INDICATOR SYSTEMS FOR AGRICULTURAL BIODIVERSITY.

Biodiversity level	Indicator(s)	Indicator system
Genetic diversity	Diversity of cattle, pig, sheepand goat, poultry breeds	IRENA 25 OECD Swiss AEI
	Livestock breeds registered and certified for marketing for the main livestock categories (i.e., cattle, pigs, poultry, sheep and goats)	OECD
	Three dominant livestock breeds in total livestock numbers for the main livestock categories (i.e., cattle, pigs, poultry, sheep and goats)	OECD
	Endangered risk status of major livestock breeds (cattle, pig, sheep, goat, poultry)	IRENA 25
	Livestock (i.e., cattle, pigs, poultry and sheep) in endangered and critical risk status categories and under conservation programmes	OECD Swiss AEI
	Plant varieties registered and certified for marketing for the main crop categories (i.e., cereals, oilcrops, pulses and beans, root crops, fruit, vegetables and forage)	OECD Swiss AEI
	Five dominant crop varieties in total marketed production for selected crops (i.e., wheat, barley, maize, oats, rapeseed, field peas and soyabeans)	OECD
	Status of plant and livestock genetic resources under in situ and ex situ national conservation programmes	OECD Swiss AEI
Species diversity	Bird diversity	IRENA 28 OECD Swiss AEI Norwegian 3Q
	Butterfly diversity	IRENA 33.2 Swiss AEI
	Plant diversity	Swiss AEI
	Land snail diversity	Swiss AEI
	Potential effect of farming practices on 11 species groups (LCA approach)	Swiss AEI
	Threatened species	IRENA

Habitat diversity	Area of major habitat types	IRENA Swiss AEI Norwegian 3Q UK Countryside Survey Swedish NILS
	Quality and evolution of semi-natural habitats	Swiss AEI Swedish NILS

3.3. DISTINCTION OF ORGANIC FARMING FROM CONVENTIONAL AND LOW INPUT FARMING SYSTEMS

3.3.1. DEFINITION OF ORGANIC: CERTIFICATION AS ORGANIC TO MINIMUM EU STANDARDS (EC 834/2007 AND EC889/2008)

Certain principles govern organic farming and these are implemented in practice via regulations for organic production. The EU standards EC 834/2007 (outlining the principles of organic production) and EC 889/2008 (giving the implementing rules for organic production) are the minimum requirement for any producer wishing to certify as organic within the EU. Being certified organic is expected to have positive impacts on biodiversity because of the following practices required under the regulation EC 889/2008:

- Restriction on external imports and reliance on internal cycling and natural processes to maintain soil fertility and plant production
- Use of tillage and cultivation to increase soil organic matter, stability and biodiversity
- Fertility and biological activity of soil maintained and increased by multiannual crop rotations (grassland, legumes, forage crops, arable crops, root crops, etc) and application of animal manure
- Fertilisers and soil conditioners restricted – no mineral nitrogen fertilisers
- Restricted use of herbicides and pesticides
- Restricted use of animal medicines
- Restrictions on livestock numbers and balance of livestock types (e.g., sheep to cattle) to help control parasites

The farming practices required under organic regulations have many positive impacts on biodiversity and these impacts are described in Anon (2005), Flade *et al.* (2003), Fowler *et al.* (2004), Gardner and Brown (1998), Hole *et al.* (2005), Norton *et al.* (2009) and Siebrecht and Hulsbergen (2009).

3.3.2. DEFINITION OF LOW INPUT FARMING SYSTEMS

In the context of the IRENA 15 indicator, the European Environmental Agency classifies the farms in Europe according to the funds spent on farm inputs:

- Low-input farms spend < 80 Euro per ha per year on fertilisers, crop protection and concentrated feedstuff,
- Medium-input farms spend between 80 Euro and 250 Euro per ha per year on these inputs
- High-input farms spend > 250 Euro per ha per year on such inputs.

3.3.3. WORKING DEFINITION OF HIGH NATURE VALUE AREAS UNDER AGRICULTURE

In BioBio, HNV needs to be defined in terms that can be translated into field measurements. The proposed HNV definition to be used in BioBio has been synthesized from the available literature and the field measures are summarized in the TABLE 3.2. The published definitions are conceptual and are not suitable for direct application in the field hence TABLE 3.3 given below. Whilst there is a core of agreement between these definitions some factors e.g., cultivated land are only included in one definition.

HNV farms will usually consist of land with a high proportion of semi-natural vegetation of high biodiversity and quality, but including cultivated crops. However, most farms in NW Europe will only contain fields of low plant biodiversity. HNV farms may also consist of mosaics of habitats of different structures. These may be in patches or consist of networks of linear features or combinations. HNV farms may also contain European and regionally important habitats as well as rare species and those of more general conservation interest. A given farm may only satisfy a few of these criteria.

These farmland types may be present within one farm. Whilst some farms may be entirely composed of semi-natural vegetation many others will contain mixtures of various types. In many cases the actual farmland may be of low biodiversity in terms of the vegetation but may support important species from other groups e.g., intensively managed grassland on Islay (Scotland) and in the Netherlands supports geese. Also the lowland part of the farm may be of low diversity but the upland area might be rich, as in parts of Romania.

Farmland is defined in the present project using the system developed for the Seamless project (Anderson *et al.,* 2006). This procedure describes eight classes, five of which define farmland and exclude extensive forest, urban land and roads. These classes have been tested.

TABLE 3.3. HIGH NATURE VALUE FARMING AMENABLE TO ASSESSMENT WITH BIOBIO INDICATORS

HNV farmland types	**Measures in BioBio**
Farmland with a high proportion of semi-natural vegetation.	1. Percentage area of farm with semi-natural habitats
High diversity of vegetation	2. Number of species in vegetation plots
High functional diversity of vegetation	3. Application of statistical programs to record functionality of the vegetation, e.g., number of stress tolerant species
Farmland with a mosaic of agricultural elements, including crops grasslands and heathland.	4. Fragstats, e.g., patch size (see McGarrigle and Marks, 1995)
Farmland with an extensive length of field margins, hedgerows, stonewalls,	5. Lengths measured in the field, e.g. length of hedgerow.

wood edges, grass strips, walls/terrace walls and lines of trees.	
Low intensity farmland types	6. Average herbage yield based yield classes estimated from habitat characteristics.
Extent of European important habitats	7. Area of Annex 1 habitats of the Habitat Directive.
Extent of regionally important habitats	8. Area of regionally important habitats as identified by local consultants.
Farmland supporting rare species	9. A. Number of rare species recorded in the vegetation plot B. Number of rare plant species noted by the field recorders C. Number of rare plant species obtained via consultation with local or international experts.
Presence of rare livestock breeds	10. Field observation
Presence of local/national rare crop varieties	11. Consultation with farmer

3.4. STAKEHOLDER PERSPECTIVE: DESIRABLE PRACTICAL POLICY AND MANAGEMENT INDICATORS

The Stakeholder Advisory Board operates under Work Package 7 and a detailed report of SAB recommendations for the selection of biodiversity indicators has been produced therein (Pointereau, P. 2009. Deliverable 7.1 Report on Stakeholder requirements for biodiversity indicators for organic and low input farming systems). The Executive summary and table of 18 recommendations of the report (TABLE 3.4) are reproduced here but please refer to the report for the full justification and background.

The stakeholder advisory board (SAB) consists of 20 experts from major interest groups: NGO Nature protection and environment (5), NGO consumers' association (1), farmer organisations (3), territorial and national administration (3), farmer adviser and agrarian institutes (2) and European administration (6).

The SAB accompanies the project from the start (conceptual phase) to the end (dissemination), will support the BioBio research and development approach and will formulate their main expectations and criteria for relevant and useful biodiversity indicators for organic and low input farming systems. This process was launched during the kick-off meeting of the project in Zurich and the SAB workshop I (March 25-27, 2009).

The SAB considers that it is important and necessary to precisely state the objectives concerning the different uses of biodiversity indicators. Six objectives have been listed during the first meeting:

- Training and awareness of several types of stakeholders
- Advice and consultancy for farmers (including environmental reporting and monitoring at farm level)
- Management plans for protected areas where agriculture plays an important role
- Assessment and justification of public subsidies for agriculture
- Calculation of agro-environmental premium
- Certification of agricultural products or production methods

It is important also to define the biodiversity "level" (or quality) which is expected or to be achieved.

The SAB has proposed 18 recommendations which can constitute a grid to analyse the selected bio-indicators (TABLE 3.4). These recommendations should be weighted and linked to the objectives. They concerned the type of indicators (direct, indirect) but also how indicators are presented / communicated (i.e., score, trend, list of species, etc.). The biodiversity indicators should be easy to develop, to record, to use, be comprehensive and flexible, low cost, integrate emblematic species, be appropriate for use by farmers, consumers and administration. They should assess the farmer progress, management plans and agricultural policies. They should contribute to evaluate all types of farming systems and if possible be common to all Europe. They should take into account existing indicators and tools to assess biodiversity in agriculture and observatories. The biodiversity indicators should be available at different scales, take into account functional biodiversity and also provide information on other environmental issues.

TABLE 3.4. RECOMMENDATIONS OF THE BIOBIO STAKEHOLDER ADVISORY BOARD REGARDING THE SELECTION OF BIODIVERSITY INDICATORS FOR ORGANIC AND LOW-INPUT FARMING SYSTEMS

No.	Requests	Remarks
1	Easy to develop: indirect Indicators	Improve indirect indicators; strengthen the relation between direct and indirect indicators. Indicators based on diversity
2	Easy to use, not too expensive to apply	Advisers do not have much time to assess one farm (less than one day). Constraints concerning the season
3	Comprehensive and flexible	The methodology must be explained (abundance, rare species, specialist species, number of species, trend, indicative species, set of species, trophic indicator)
4	Integrate emblematic species	Use flagship species, umbrella species? Are they good indicators? If not explain why
5	Appropriate for farmers, consumers and administrators	Observable and understandable. Common species which people can recognise. Easy interpretation
6	Low cost indicators	Applicable by advisers and not only by specialists

7	Assess the farmer progress and be appropriated	Sensitive to the practices implemented. Adapted to the duration of the contract (AEM) – 5/7 years-. Linked to agricultural practices. Status indicators probably not practical
8	Assess projects managed by the stakeholders	Effectiveness of the project. Compare our experience with others
9	Assess the agricultural policy (OF and AEM but not only) and national action plans on biodiversity	Make the difference between the application and the effectiveness of the measures. Not only measure the level of implementation but also results. Contribute to assess the contribution of agriculture to biodiversity action plans
10	Contribute to evaluate all types of farming	Be used also in "conventional" or "intensive" agriculture. Measure the environmental efficiency
11	Develop common indicators in Europe	Indicators should be recognised by member states and the European Commission. Adapted to the local situation
12	Available for different scales: Farm and region	Indicators must be available for the different scales (same indicators or different indicators)
13	Take into account functional biodiversity	Functional biodiversity explains how the farming system works and shows the contribution which stems from biodiversity
14	Provide information on other environmental issues	Contribute to the assessment of other environmental issues (carbon storage, water quality, preserving natural resources, farming sustainability)
15	Take into account the environmental impact of the production of farm inputs	Think about system boundaries. E.g.,integrate the environmental impacts of soya production in America which feed part of our animal
16	Define the targets and the objective	What level of biodiversity do we want to achieve (high, medium?). What is the target?
17	Take into account the existing indicators and observatories	Realize a benchmarking and a state of the art of existing biodiversity indicators and tools or of biodiversity surveys. Give priority to existing indicators and improve them
18	Explain how to use the indicators	It is important to describe how the indicators have to be used

4. INDIRECT/ MANAGEMENT INDICATORS

"Indirect indicators" are factors acting on biodiversity and represent risk or opportunity for biodiversity, or are consequences of biodiversity state. These indicators are primarily oriented toward decision making and the evaluation of measures that favour biodiversity (e.g., change of agricultural practices, success of agri-environmental measures). Broad criteria as proposed by UNEP (2003) can be applied to the selection of indirect indicators. Nevertheless, indirect indicators are not part of biodiversity. The relationship between the candidate indirect indicator and any direct indicator has therefore to be confirmed as requisite criteria (indicator of indicators, see Table 4.1).

Turner and Doolittle (1978) and Shriar (2000) argue that "Output per unit area is likely to be the ideal measure of intensity because it makes no presumptions about the effect of inputs on productivity (…)". However, in large scale studies which will cover many different farming systems of organic and low-input farming, there will be no single agricultural commodity which will be common to all systems. Assessing their monetary value would make these outputs comparable; however, farm gate prices vary considerably both temporally and between countries (Shriar, 2000).

Alternatively, therefore, agricultural land-use intensity can be assessed by quantifying agricultural inputs that aim to increase productivity. Labour, skills and capital, which materialise through for example, mechanisation, fertilizer and pesticide inputs, can be used as surrogates for intensity (Turner and Doolittle, 1978; Lambin *et al.,* 2000; Shriar, 2000; Kerr and Cihlar, 2003). It is hypothesised that these inputs will increase the agricultural output.

Methods have been developed to evaluate environmental impacts of farming systems based on standard agricultural statistics as indirect measures of biodiversity (Huelsbergen, 2005; McRae, 2000) but either for single case study farms (Kuestermann, 2007; Noe, 2005) or low resolution across broad geographic areas (Buchs, 2003; Hoffman, 2001). Such methods are based on indirect indicators derived from management practices. Indirect indicators for biodiversity have been implemented in the evaluation of environmental impacts of agriculture, e.g., in life cycle assessment (LCA) method (e.g., Swiss Agricultural Life Cycle Assessment, SALCA), and in agro-environmental diagnosis of farms (INDIGO and DIALECTE in France, KUL/USL and REPRO in Germany).

In SALCA, impacts of agricultural practices on biodiversity are estimated at field and farm level by fuzzy-coding of published experimental or observational investigations and of expert knowledge by means of 11 species groups (e.g., birds, small mammals, spiders)(Jeanneret, 2006). In the farm-based system REPRO (Hülsbergen, 2003) the complex relationships between farm management and biodiversity are divided in 1) structural parameters describing the area, the land use and the cropping structure, 2) fertilizer and pesticide inputs, and 3) specific indicators of process design and management features. These indicators are finally aggregated to the "Biodiversity Development Potential".

Whilst those methods have been developed primarily for national applications, large datasets like the Farm Accounts Data Network (FADN) and Farm Statistics Service (FSS) may be helpful in providing indirect indicators relating to input use and land-use diversity (number of crop and livestock enterprises per holding) for organic/low-input holdings at the European level. Low-input holdings can be determined in relation to the value of crop fertiliser and pesticides used and livestock feed inputs and stocking rates (the latter also applies to FSS). Some of these approaches have been applied in the IRENA framework, though not separating out organic/low-input farms specifically. For both datasets, it would be possible to differentiate the analysis by farm type and region as part of an EU wide assessment, though the spatial resolution will be limited by the number of FADN samples per sub-region. It is envisaged that a greater level of detail will be collected with respect to organic farms for the pan EU agricultural census in 2010, but the results of the census are not expected to be available in the life time of the project and reliance will need to be placed on data collected in earlier years (FADN is collected annually, the last FSS survey was conducted in 2007). However, it will be important that selected indicators take account of European Commission (in particular DG Agri, DG Enviro and Eurostat) plans for agricultural, rural and agri-environmental development from 2010, to increase the chance of the biodiversity indicators being developed in this project being adopted. Nevertheless, indirect indicators have to be discussed and chosen with caution. As argued by Wascher *et al. (*2000), because of the huge number of species and the complexity of ecological processes within agricultural habitats, many potentially influencing factors may be unrecognised and not monitored. The intensity of agricultural management varies considerably across Europe (Herzog *et al.*, 2006) and the environmental heterogeneity of the European continent reduces the certainty with which predictions about the link between agricultural management on biodiversity can be made (Dormann *et al.*, 2008). Moreover, impacts of agricultural practices are often poorly understood so that the most relevant parameters that can be practically monitored are unclear. Therefore, indicators of the actual state of biodiversity are essential.

Recent EU research projects have investigated the relationship between indirect/ management indicators and selected farmland species diversity indicators. They did not rely on existing national or European datasets (FADN, FSS) but both indirect and direct indicators were measured on farms distributed across Europe. Kleijn *et al. (*2009) related N-input to plant species richness in a pan-European dataset (130 grasslands and 141 arable fields across 6 countries). Confounding factors (latitude, altitude, precipitation, temperature, landscape diversity) were removed, thereafter the relation was statistically significant. There was, however, a high variability and the statistical analysis was strongly affected by sites with very low nitrogen inputs. Liira *et al. (*2008) also related plant diversity to nitrogen input (25 agricultural landscapes, 7 temperate European countires) but found not significant correlation. Other indicators they tried include: (i) share of intensively fertilised land (> 150 kg N $ha^{-1}y^{-1}$), (ii) no. of crops in the rotation, (iii) livestock units, (iv) no of pesticide applications. Of these, only the no. of crops in the rotation could be related to plant species richness (of growth form classes, of nature value groups, of life span classes). On the same dataset but with different statistical methods, Billeter *et al. (*2008) found a statistically significant (negative) relation between the share of intensively fertilised land (>150 kg N ha^{-1} y^{-1}) and the richness of vascular plants.

Regarding arthropods, Schweiger *et al. (*2005) related intensity and landscape diversity to arthropod (Apidae, Araneae, Carabidae, Heteroptera, Syrphidae) diversity in 24 landscapes of 7 temperate European countries (170,000 individuals of 628 species). Land-use intensity (LUI) was assessed as “stress” by means of the indicators (i) nitrogen input and (ii) number of pesticide applications as well as the “spatiotemporal” pattern indicators, (iii) crop diversity (no. of crops in the rotation) and (iv) share of intensively fertilised land (> 150 kg N ha^{-1} y^{-1}). Intensity explained around 10% of the variability of the arthropod data, the "spatial" indicators having the stronger effect for all groups except Syrphidae. This can be due to the indicator "crop diversity", which correlated with arthropod diversity. With the same arthropod dataset, Hendrickx *et al.* (2007) tested the relation to a composite intensity index (LUI index) derived from (i) Nitrogen input, (ii) n° of pesticide applications and (iii) livestock density. They found a significant main effect of LUI on overall arthropod species richness (Alpha diversity). For Gamma and Beta Diversity, no significant main effect was found, but taxon-specific effects of LUI on Apoidea and Araneae (Gamma only). Dormann *et al.* (2007), using the same data, found (negative) relations between the number of pesticide applications and bees and syrphid diversity. In a smaller scale study, Bailey *et al.* (submitted) investigated the effect of the amount and fragmentation of 30 traditional orchards on the diversity of vascular plants, spiders, land snails, beetles, bugs and birds. Although they tried to standardise for farm management and remove its effect as a confounding variable, spider and bug diversity was negatively affected by increasing intensity of management. The latter was assessed by means of 11 indicators relating to the management of the understory grassland (fertilisation, cutting frequency, grazing) and of the trees (pesticide applications, pruning, fertilisation). The indicators were aggregated into an intensity index by means of correspondence analysis.

With respect to farmland birds, several investigations yielded correlations between indirect indicators and bird diversity:

- In the greater Paris region, Filippi-Condaccioni (2008) investigated bird communities on 58 farms. She compared conventional farming to organic farming and to farming with conservation tillage. Intensity was assessed as (i) the number of pesticide applications, (ii) length of the crop rotation (similar to crop diversity), (iii) yield of wheat and (iv) nitrogen input. All four intensity indicators differentiated between conventional and organic farming (but not all between conventional farming and conservation tillage). All four indicators correlated strongly and the number of pesticide applications was selected to represent them. This indicator was then related to a bird specialisation index.
- Britschgi *et al.* (2006) found that in alpine meadows, the time of first cut and the frequency of grassland cutting affected the survival of the offspring of a typical meadow bird species.
- Billeter *et al.* (2008, 24 landscapes in 7 European countries) found a negative relation between nitrogen input and bird richness.

Whilst these analyses were based on field work (for both, bird diversity and farm management indicators), at a larger scale, other authors tried to link statistical data of both components:

- Chamberlain *et al.* (2000) in the UK analysed time series of bird census data and of agricultural census data over 3 decades. They worked with 31 agricultural variables representing crop area, livestock, fertilizer, grass production, pesticide use. Management / intensity indicators were strongly intercorrelated. They were aggregated by means of detrended correspondence analysis (categorical variables) and principal components analysis (continuous variables) into a composite index. The population changes of farmland birds could be related to changes in farming practices; there was a timelag between those changes and the shift in bird populations.
- Donald *et al.* (2001) conducted a statistical analysis of the evolution of farm management indicators and bird populations over 3 decades for 30 or so European countries. Farm management data were obtained from FAOSTAT: (i) agriculture

population density, (ii) milk yield, (iii) density of cattle, (iv) fertilizer use, (v) cereal yield, (vi) number of tractors per worker, (vii) number of harvesters per worker. Cereal yield explained over 30% of the variation in bird population trends

- In Canada, Kerr and Cihlar (2003, 2004) compiled agricultural census data and remote sensing information. The farm management indicators (i) amount of pesticides purchased, (ii) amount of fertilisers purchased, (iii) manure production, (iv) livestock density were aggregated into an index by means of principal component analysis. They found a positive relation between increasing intensity and the density of endangered species (not only birds).

Amongst the above mentioned farm management indicators, the diversity of crops – which is directly related to the length of the rotation/ the number of crops which succeed in a rotation – could be related to plants and almost all arthropod groups which have been investigated (TABLE 4.1). Nitrogen applications seem to directly affect plant and bird diversity whereas pesticides could be related to some arthropod groups and to birds. Specific grassland management indicators have been shown to affect meadow birds. Composite indices of farm management (intensity) reflected significant effects on some arthropod groups and on birds. The level of crop yield is not actually a farm management indicator but results from the level of intensity of the management. It could be related to bird diversity in two large scale studies (TABLE 4.1).

TABLE 4.1. STUDIES WHICH DETECTED STATISTICALLY SIGNIFICANT RELATIONS BETWEEN FARM MANAGEMENT INDICATORS AND THE DIVERSITY OF TAXONOMIC GROUPS. LARGE SCALE STUDIES ARE UNDERLINED.

	Plants	**Spiders**	**Bugs**	**Bees**	**Carabid beetles**	**Hoverflies**	**Birds**
Nitrogen application	<u>Billeter *et al.* (2008), Kleijn *et al.* (2009)</u>						<u>Billeter *et al.* (2008)</u>
Pesticide application				<u>Dormann *et al.* (2007)</u>		<u>Dormann *et al.* (2007)</u>	Filippi-Condaccioni (2008)
Crop diversity	<u>Liira *et al.* (2008)</u>	<u>Schweiger *et al.* (2005)</u>	<u>Schweiger *et al.* (2005)</u>	<u>Schweiger *et al.* (2005)</u>	<u>Schweiger *et al.* (2005)</u>		
Meadow cutting regime							Britschgi *et al.* (2006)
Composite index		<u>Hendrickx *et al.* (2007), Bailey *et al.* (submitted)</u>	Bailey *et al.* (submitted)	<u>Hendrickx *et al.* (2007)</u>			Chamberlain *et al.* (2000)
Crop yield							<u>Donald *et al.* (2001), Kerr and Cihlar (2003, 2004)</u>

A full list of available indirect indicators has been summarised under major categories (TABLE 4.2).

TABLE 4.2. PROPOSED INDIRECT BIODIVERSITY INDICATORS

Indicator Category	Sub category	Specific indicators	Comments and references	Other relevant indicators	Reflected by Candidate Indicator[$]
1. Regulatory and certification		1.1 Definition of organic and low input systems	Anon, 2007		---
		1.2 Area of land under organic management			CertOrg Certified as Organic (D6)
		1.3 Percentage of ag. area without application of pesticides	Includes all organic land (Fowler *et al*., 2004) Other references (Siebrecht and Hülsbergen, 2009)	1.2	PestUse Area without/with reduced use of pesticide (D10)
		1.4 Length of time converted to organic	Newly converted fields found to be just as likely to have high species richness as long term organic fields (Anon, 2005)		---
2. Farm type and structure	2.1 Farm type	2. 11 Farm Type	Farm type (lowland, upland, hill) has big impact on differences in biodiversity between organic and conventional. In the hills and uplands differences not great The biggest differences occurred where farm also in another agri-environment scheme (in addition to organic). Keatinge (2005) Farm type (arable, lowland, upland, hill etc) an important indicator (Fowler *et al*., 2004)		---
	2.2 Field size	2.21Average field size at individual farm and regional level	Smaller fields, more hedges per unit area, diverse rotations, more grassland (Anon, 2005); ß-, and γ-diversity were higher in organic than conventional fields and higher at the field edge than in the field centre at all spatial scales (Gabriel *et al*., 2006) Length of field margins (Siebrecht and Hulsbergen, 2008)	2.11,	---

Indicator Category	Sub category	Specific indicators	Comments and references	Other relevant indicators	Reflected by Candidate Indicator[$]
3. Enterprises	3.1 Degree of diversity in farming systems	3.11Number and relative land area of crops and enterprises at individual farm and regional level	Organic farmers tend to have greater diversity of crop structure (Unwin and Smith, 1995) Organic has positive effect on species richness and abundance – contrast greatest on intensive conventional vs organic. Differences unlikely to be great on smaller scale landscapes with range of biotypes. Bengtsson *et al.*, 2005 Cereals on organic farms are more likely to part of a mixed systems than cereals on conventional farms (Norton *et al.*, 2009) Proportion of arable land is an appropriate indicator for biodiversity in agriculturally dominated landscapes of eastern Austria (Sauberer *et al.* (2008)) Percentage area of arable land higher on conventional than organic farms, area of grass significantly greater on organic farms (Bates and Harris, 2009) Mixed cropping , diverse rotations, more grassland; the use of clover and herbs in grass leys, cover crops and undersowing, green manures (Anon, 2005; Fowler *et al.*, 2004; Gardner and Brown, 1998) Preservation of mixed farming, which is largely intrinsic (but not exclusive) to organic farming is one of the factors that benefits farmland wildlife (Hole *et al.*, 2005) Organic farmers incorporated grass-clover leys into their rotations and planted a wider variety of cereal types which were frequently under-sown with a ley (Norton *et al.*, 2009; Altieri and Letourneau, 1982; Siebrecht and Hülsbergen, 2009) Set-aside, crop rotations with grass leys, spring sowing, permanent pasture, green manuring and intercropping all observed to influence biodiversity (Gardner and Brown, 1998)	2.11	DivEnt Diversity of Enterprises (D1) Undersowing (D---) Manure & Green Manure (D---) Grass-clover/ legumes (D---) CropRot Crops in Rotation (C4)

Indicator Category	Sub category	Specific indicators	Comments and references	Other relevant indicators	Reflected by Candidate Indicator[$]
		3.12 Share of permanent grassland	Will tend to be higher in extensively managed farms. Different types of biodiversity in grassland than in crop fields.		Grass-clover/ legumes (D---) GrassArea Permanent Grassland (C6)
	3.2 Stocking rates and enterprise balance	3.21 Average stocking rates (LSU/ Ha) – 3.22 Relative proportions (in terms of LSU?) of different livestock species on farms	Stocking rates are lower on organic (restricted by regulations) – results in reduced defoliation and treading, impacts of pasture species mix and productivity, positive impacts on bird habitats and ground dwelling mammals (Fowler *et al*. 2004). Other references (Flade *et al*. (2003) Mixed grazing with cattle and sheep results in structural diversity in grazed grasslands. Impacts on parasite control (cattle:sheep in range of 40: 60 to 60: 40 ideal). Cow dung source of food for carabids (Fowler *et al*., 2004)		AvStock Average Stocking Rate (D2)
	3.3 Indigenous breeds and strains of livestock (also plant species) adapted to local environment	3.31 Number and proportion of indigenous breeds in flock/ herds	Less external, chemical inputs required to support. Less intensive level of production. Utilise rough grazing. Genetic diversity (Fowler *et al*., 2004)	2.11	Breeds Proportion of breeds (A1)
4. Specific management practices	4.1 Prohibited/ restricted use of mineral fertilisers and	4.11Area of land over which these inputs are not used	Feature of organic systems (Fowler *et al*., 2009; Anon, 2007; Hole *et al*., 2005)	1.1, 1.2, 1.3	MinFert No use mineral fertiliser (D3)

Indicator Category	Sub category	Specific indicators	Comments and references	Other relevant indicators	Reflected by Candidate Indicator$
	pesticides				PestUse Without/reduced use pesticide (D10)
	4.2 Soil management and fertility	4.21Fertilisation intensity (frequency and application rate of synthetic fertilisers?)	Siebrecht and Hüsbergen, 2009 Use of artificial sources of P and K prohibited – slow release rock forms allowed. Better conditions for less competitive plant species. Fowler *et al*., 2004		Percentage grass-clover or legumes (D---)
		4.22 Area and proportion of fertility building legumes used	Legumes main source of fertility (Fowler *et al*., 2004)	3.11	NitroIn N input or balance (D5)
		4.33 N and Organic matter balances	Balances for N, organic matter and energy reflect most agricultural activities and give direct or indirect hints on the environmental effects. They meet the demands for useful agricultural-environment indicators. (Flade *et al*., 2003)	3.11	Manure or Green Manure (D----)
		4.23 pH	Increased pH has been one of the many agronomically and environmentally desirable changes to biological, physical and chemical properties in cultivated organically managed soil identified. Lime encourages soil biological activity. (Fowler *et al*., 2004)		
		4.24 Frequency and application rate of FYM and composts)	Use of FYM and compost, - encouraged in organic farming – promotes soil flora and fauna biodiversity		
	4.3 Structure and diversity of crop cultivations	4.31 Number, frequency and type of cultivation	(Flade *et al*., 2003). Mechanical weed control in organic. Sensitive and timely cultivations. (Fowler *et al*., 2004)	2.1, 3.11,	FieldOp Field Operations (D11)

Indicator Category	Sub category	Specific indicators	Comments and references	Other relevant indicators	Reflected by Candidate Indicator[$]
	4.4 Sowing & Harvesting (crops)	4.41 Dates of sowing on low input and organic farms vs conventional farrms	Sowing date. Organic farmers always sowed crops later than their conventional counterparts (Norton *et al*., 2009)		-----
		4.42 Dates of harvest and harvesting techniques in organic systems vs conventions	Diversity of harvesting techniques (Flade M., Plachter H., Henne E., Anders K. (2003). Lower in extensively managed systems and proven correlation to biodiversity (Donald *et al*. (2001), Kerr and Cihlar (2003; 2004)		
	4.5 Plant protection	4.51 Number of pesticide applications	Affects namely arthropod diversity (Dormann *et al*., 2007), strongly restricted in OF		TFI Treatment Frequency Index (D9)
	4.6 Grassland management	4.61 Date of first cut or pasturing	Affects biodiversity and is generally later in more extensive farming systems (Britschgi *et al*., 2006) same		GrazInt Grazing Intensity (D12) FieldOp Field Operations (D11) Mowing (D----)
		4.62 Frequency of cuts and/or grazing	Different aspects of biodiversity in meadows than in pastures		
		4.63 Dominating use: mowing or grazing			
	4.7 Energy use	4.71 Direct and indirect energy use (per Ha, per unit produce	(Flade *et al*., 2003).		EnerIn Energy Input (D5)
	4.8 Animal health	4.8.1 Frequency and type of medication used to	Treatments restricted in organic. Some chemical harmful to biodiversity e.g., Synthetic pyrethroids		------

Indicator Category	Sub category	Specific indicators	Comments and references	Other relevant indicators	Reflected by Candidate Indicator[$]
		control external parasites	harmful to aquatic biodiversity, Avermectin family of drenches persist in soil and have negative impact on soil and soil fauna/ dung beetle biodiversity. (Fowler *et al.*, 2004)		
5 Over all diversity indicators	General	5.1 Area and types of habitat on individual farms 5.2 Uncultivated biotype area 5.3 NSCP (number of shape characterising points, Moser *et al.* 2002) 5.4 Degree of habitat isolation, sizes of the study sites	Differences in biodiversity between organic and conventional thought to be related to the greater quantity of habitat available in organic.(Anon 2005) Sympathetic management of un-cropped areas increases biodiversity (Hole *et al.*, 2005) Unspecific measure of wildlife habitat (Noe *et al.* 2005) Appropriate indicator for biodiversity in agriculturally dominated landscapes of eastern Austria (Sauberer *et al.*, 2008) Species number (butterflies and burnet moths) higher in large (>10 ha) than in small remnants in 1972 and 2001, smaller in highly isolated habitats than in less isolated ones (Wenzel, M *et al*, 2006))		HabDensity Habitat Density (C1) HabRich Habitat Richness (C2) HabDiv Habitat Diversity (C3)
Utilisation frequency		Define more clearly	Siebrecht and Hülsbergen 2009		FieldOp Field Operations (D11)
Travelling frequency		Define more clearly	Siebrecht and Hülsbergen 2009		FieldOp Field Operations (D11)

4.1. GENERAL FARM CATEGORIES

4.1.1. DIVERSITY OF ENTERPRISES – NUMBER AND RELATIVE LAND AREA OF CROPS AND ENTERPRISES AT INDIVIDUAL FARM AND REGIONAL LEVEL

Organic farms have been found to have a more diverse enterprise mix than conventional farms resulting in positive impacts on biodiversity (Bates and Harris, 2009; Norton *et al.,* 2009; Siebrecht and Hülsbergen, 2009; Holzschuh *et al.,* 2007; Anon, 2005; Bengtsson *et al.,* 2005; Hole *et al.,* 2005; Fowler *et al.,* 2004; Gardner and Brown, 1998; Unwin and Smith, 1995). The diversity of enterprises can be measured directly at the farm level by counting the number of enterprises or can be identified at a regional/national level through FADN data using organic farm and enterprise codes.

4.1.2. FARM TYPE (LESS FAVOURED AREA (LFA) VS NON-LFA)

This indirect indicator is not recommended but is worthy of further discussion. There is evidence of the beneficial effects of organic farming practices on biodiversity and environment (Bengtsson *et al.,* 2005; Hole *et al.,* 2005). However, to date research on biodiversity impacts of organic farming has concentrated on lowland arable and stock farming (non-LFA) (Fuller *et al.,* 2005; Hole *et al.,* 2005). It is unclear how this work can be applied to the situation in the hills and uplands, and whether the environmental benefits of organic farming are as great in this context (Keatinge, 2005; Fowler *et al.,* 2004). A combination of altitude, soil characteristics and climatic conditions restricts most farms within the hills and uplands to sheep and cattle production, and many such livestock farms have relatively low fertiliser and chemical input when compared to lowland equivalents.

However, Fowler *et al. (*2004) stated that policy influences prior to the 2003 CAP reform resulted in intensification of farming systems in the hills and uplands in the form of:

- High stocking levels, and large scale hill land improvement to increase grassland production
- An emphasis on sheep-only systems, or sheep and beef systems where beef are the minority stock on the holding
- Reliance on annual use of N, P and K fertilisers to maintain *Lolium perenne* swards for grazing and forage conservation
- Purchase of supplementary livestock feed (including forage, as hay or big bale silage, and concentrates) to provide winter keep for the high stock numbers
- Regular use of prophylactic allopathic medicines, to maintain stock health, particularly where large numbers of a single species led to high incidences of intestinal parasites. This in turn allowed a more set-stocking approach as opposed to rotational grazing within enclosed land.
- Increase in size of fields and degradation of traditional field boundaries and of dry stone walls and hedgerows as traditional grazing management practices disappeared to be replaced by extensive ranching type practices.

The effect of these developments was detrimental to biodiversity in the hills and uplands. The removal of headage payments and the introduction of the Single Farm Payment post CAP reform have reduced reliance on high stocking rates but conventional hill and upland farms tend to still rely on sheep only systems supported by allopathic medicines (particularly for internal parasite control) – which can have a negative impact on biodiversity.

The problem with this as an indicator is that the definition of LFA land types varies from country to country – in the European Union, less-favoured area (LFA) is a term used to describe an area with natural handicaps (lack of water, climate, short crop season and tendencies of depopulation), or that is mountainous or hilly, as defined by its altitude and slope. More specific definitions are not common across the EU.

LFA or non-LFA (or its informal sub-classifications used in the UK (lowland, hill and upland)) is not defined clearly enough for this to be a suitable indicator of biodiversity across the EU. Also, the differences between organic and conventional farming practices in terms of their impacts on biodiversity in LFA areas (hills and uplands in the UK) are not known. Farm systems also vary considerably in terms of inputs, there may be conventional LFA farmers who are farming very similarly to organic practices and there may be others that are farming in a more intensive way.

4.2. LIVESTOCK PRODUCTION

4.2.1. AVERAGE STOCKING RATES (LIVESTOCK UNITS/HA)

Stocking rates tend to be lower on organic farms (maximum limits set under the organic regulations EC 834/2007 and EC 889/2008) due to restrictions on inputs such as fertilisers and pesticides to boost feed production and also due to restrictions on animal medicines which are often used to support artificially high stocking levels and which have a detrimental effect on biodiversity. Lower stocking rates result in decreased defoliation and treading, positive impacts on pasture species mix and productivity and positive impacts on bird habitats and ground dwelling mammals (Fowler *et al.,* 2004; Flade *et al.,* 2003).

4.2.2. RELATIVE PROPORTIONS OF LIVESTOCK SPECIES ON FARMS

Mixed grazing with cattle and sheep results in structural diversity in grasslands and having an appropriate ratio of cattle to sheep (in the range 40: 60 to 60: 40) has positive impacts on parasite control in sheep. This results in a reduction in the use of anthelmintics which can have a negative impact on microbe diversity in the soil (especially the avermectin family of drenches). Cow dung also provides a source of food for carabids (Fowler *et al.,* 2004).

4.3. CULTIVATION PRACTICES AND CROPPING SYSTEMS

4.3.1.STRUCTURE OF CROP CULTIVATION AND CROP ROTATION

4.3.1.1. Presence or percentage of grass-clover and legumes in the rotation

Organic farms often have a lower percentage area of arable land and a higher percentage area of grass (Bates and Harris, 2009) and they use more rotational practices including grass. Generally, the share of arable land on a farm and the total biodiversity is related negatively (Sauberer *et al.,* 2008). Organic farmers incorporated grass-clover leys into their rotations and planted a wider variety of cereal types which were frequently undersown with a ley. This contributed to greater field and farm complexity (Norton *et al.,* 2009). Mainly fodder legumes in the rotation can be regarded as fertility building crops because they contribute positively to soil organic matter balance (Kolbe, 2005). Grass-clover in the rotation was found to enhance populations of non-pest butterfly species (Feber *et al.,* 1997).

Grass-clovers and Lucerne are known to be beneficial for field breeding birds like Skylark, Grey Partridge, Quail and also for Brown Hare (e.g., Jenny, 1990; Fuchs and Saacke, 2006; Kragten *et al.,* 2008). To maintain a sufficient breeding success for nesting birds it is crucial to establish a minimum time-window without cutting (see 4.3.2.3. Utilisation frequency and timing).

4.3.1.2. Diversity of crop structure

Flade *et al. (*2003) recommend the structure of cultivation / relation of different crops, among other indices, as an indicator for farm management. Organic farms produced greater field and farm complexity than non-organic farms in the study of Norton *et al. (*2009). Organic farmers tend to have greater diversity of crop structure (e.g., Kragten and de Snoo, 2007, 2008; Filippi-Codaccioni *et al.,* 2009; Unwin and Smith, 1995). The diversity of crop structure may benefit a variety of species that require a structurally diverse crop/ habitat mosaic (see Hole *et al.,* 2005). For clarifying the effects of crop structure on direct biodiversity indicators, crop species, cultivar and both sowing and harvesting date should be stated.

4.3.1.3. Intercropping and undersowing

Organic farmers tend to have more under-sown crops (Altieri and Letourneau, 1982). Intercropping can be used to suppress weeds; it increases vegetation structures and heterogeneity, enhances invertebrate populations, e.g., sawflies, carabids, and spiders; provides a greater abundance of invertebrate food resources for birds and mammals (see Hole *et al.,* 2005) and enhances bird populations (Jones and Sieving, 2006).

4.3.2. SOIL CULTIVATION AND TILLAGE

4.3.2.1. Soil cultivation/ minimum tillage

Tillage is a major factor in arable farming. It is influencing the position of residues and food for soil organisms, the position and emergence of crops and wild plants, the habitats of soil organisms, the competition and phytopathogenic situation of crops and ultimately, crop yield.

There are many papers which show the influence of tillage measures and tillage systems (ploughing vs. reduced tillage) on
soil organisms:

- Soil microrganisms **(**Kandeler *et al*., 1993; Emmerling, 2003),
- Collembolla (Bassemir, 2002),
- Enchytraeides (Brockmann, 1987) or

- Earthworms (Kainz, 2003; Claupein, 1992; Edwards and Lofty, 1975; Wenz, 2003; Friebe, 1993).

Minimum tillage, also known as reduced tillage or non-inversion tillage, avoids detrimental effects of inversion ploughing on invertebrate populations (Weber and Emmerling, 2005; Hole *et al*., 2005). The number of tillage treatments and nest density in organic fields are negatively correlated (Loekmoen and Beiser, 1997) and in general minimum tillage is thought to have beneficial effects for the farmland bird food chain (Cunningham *et al*., 2004). Because cultivation implements differ in their effects on weeds, the type of implements used should be stated.

The wild plants within an arable field will be buried or uprooted with tillage measures. Some cultivation selects certain plant species (e.g., Gruber *et al*., 2000). Reduced tillage often increases permanent weeds and reduces annual weeds (Albrecht and Sprenger, 2008; Albrecht, 2004; Sprenger, 2004) and weed diversity (Albrecht and Sprenger, 2008). Summarized, tillage has a distinct effect on segetal flora (also: Schmidt and Leithold, 2002; Bilalis *et al*., 2001; Derksen *et al*., 1998; Feldmann *et al*., 1997; Pallutt, 1999).

The crop yields and, therefore, the effectivity of systems are directly influenced by tillage due to changing the competition between plants and changing the conditions of growth. Pathogens, the air and water conductivity in soils, the turn over of residues and the release of nutrients will be greatly influenced by soil tillage. Therefore, tillage is one of the most important pressures in arable systems.

4.3.2.2. Mechanical weeding: frequency and timing

Mechanical weeding is often less efficient than using herbicides and contributes to a greater abundance of non-crop flora in arable fields, indirectly supporting higher densities of arthropods (see Hole *et al.,* 2005). On the other hand, higher mechanisation rates and mechanical weeding on organic farms may cause high mortality amongst eggs and young of ground-nesting bird species (Hansen *et al.,* 2001; Kragten and de Snoo, 2007) unless timed carefully (Hole *et al.,* 2005).

4.3.2.3. Utilisation frequency and timing

Utilisation frequency of grassland or grass-clover leys describes effects on organisms as a result of management intensity and disturbance (Siebrecht and Hülsbergen, 2009).
The interval between two cuts in the breeding season is crucial for nesting success of ground nesting birds (e.g., Jenny, 1990; Fuchs and Saacke, 2006; Kragten and de Snoo, 2007). For successful Skylark broods a mowing interval of at least seven weeks has been proposed (Fuchs and Saacke, 2006) and shifting cutting dates of grass-lucerne leys accordingly had no significant detrimental effects on productivity (Pietsch *et al.,* 2009).

4.3.3. SOIL FERTILITY MANAGEMENT

4.3.3.1. Area of land without use of mineral-based fertilisers

The non-use of mineral-based nitrogen fertilisers is a principle of organic farming, e.g., in the organic regulations EC 834/2007 and EC 889/2008.
Direct effects: This avoids detrimental direct impacts on biodiversity resulting from high levels of inorganic fertiliser application, e.g., increased crop structural density that alters microclimate at soil level with potentially negative consequences for invertebrate fauna and limits foraging and nesting opportunities for bird species (see Hole *et al.,* 2005).
Indirect effects: Due to the prohibition of mineral N fertiliser, organic farms rely on legumes as the main source of nitrogen. This has a wide range of effects on soil fertility management/ fertilisation practice, crop rotation structure and weed control.

The input of fertiliser was reduced in organic systems compared to the conventional system by 34 to 53 % in the study of Mäder *et al. (*2002). It should be stated which mineral-based nutrients (N, P, K, S) are applied/ not applied because different nutrients affect crop and weed growth in different way.

4.3.3.2. N balances

Balances for N, besides organic matter and energy balances, reflect most agricultural activities and give direct or indirect hints on the environmental effects. They meet the demands for useful agricultural-environment indicators like availability of data, deducibility from farm data, reproducibility (Flade *et al.,* 2003). The N balance surplus is reduced in organic farms compared to conventional farms in most cases (Kratochvil, 2002).

4.3.3.3. Manuring, green manuring

Application of farmyard manure/ compost/ green manure generally supports a greater abundance of invertebrates that rely on un-degraded plant matter as a food source, e.g., earthworms, carabids, and more diverse microbial communities (see Hole *et al.,* 2005). Slurry, however, may differ in its effects. Therefore the kind of manuring, the application rate and frequency should be stated.

4.3.4. PLANT PROTECTION

4.3.4.1. Area of land without/ with reduced use of chemical pesticides

The use of chemical pesticides is significantly restricted in organic farming according to the organic regulations EC 834/2007 and EC 889/2008. This restriction results in a reduced input of pesticides in organic systems compared to conventional systems, e.g., a 97 % reduction found by Mäder *et al. (*2002). Organic systems rely on a variety of practices (e.g., crop rotation, biological control, mechanical weed control) to manage weeds and invertebrate pests instead (Lampkin, 2002). This avoids direct and indirect pesticide effects, as follows.
Direct effects: Herbicides are a significant factor in the declines of many common arable flowers in Europe (Andreasen *et al.,* 1996). Insecticides cause a major negative influence on invertebrates (see Hole *et al.,* 2005).
Indirect effects: Weed communities were found to have a higher diversity on organic farms than on conventional ones (Tyser *et al.,* 2008). Chemical pesticides lead to a reduction in plant food resources and invertebrate abundance (Dubois *et al.,* 2003). This is a factor in the declines of a range of farmland bird species (see Hole *et al.,* 2005).

4.3.5. POTENTIAL ADDITIONAL INDICATORS (LIMITED UTILITY).

4.3.5.1. Spring-sown cereals, sowing date of spring-sowing

Organic and conventional farms may differ in percentage of spring sown cereals and sowing time. Whereas the maximum percentage of winter and spring sown cereals in conventional farming is around 70 % and 50 %, respectively (Baeumer, 1992), these percentages may be lower in organic farming due to preventive plant protection and avoidance of specific weeds. Spring-sown cereals play an important role for the field breeding Skylark (e.g., Wilson *et al.,* 1997; Chamberlain *et al.,* 2000; Donald and Vickery, 2000; Kragten *et al.,* 2008).

Spring sowing frequently results in stubble fields being left over part of the winter. This affects weed community and provides a crucial winter food source for seed eating birds (Siriwardena *et al.,* 2007; 2008; Perkins *et al.,* 2008; see also Hole *et al.,* 2005). Organic farmers in the study of Norton *et al. (*2009) always sowed crops later than their conventional counterparts. The occurrence of spring-sown cereals is already included in the indicator "Structure of crop cultivation and crop rotation" which is a more general indicator. The date of spring sowing probably has a minor effect compared to the structure of the crop rotation and the diversity of the crop structure.

4.3.5.2. Green manuring (Gardner and Brown, 1998)

Effects are similar to farmyard manure or compost application therefore those are combined into one topic.

4.3.5.3. Farm structure: farm type/ mixed farming, field size

Important indicators that are considered under Section 5.5. Habitat assessment and monitoring in the wider countryside; see Hole *et al.,* (2005).

4.3.5.4. Travelling frequency

Travelling frequency is one indicator reflecting disturbances (Siebrecht and Hülsbergen, 2009). However, once utilisation frequency and mechanical weeding frequency are regarded as indicators (see above), travelling frequency gives little extra information.

4.3.5.5. Diversity of harvesting techniques

This indicator, suggested by Siebrecht and Hülsbergen (2009), assesses effects as a result of physical contact to organisms or disturbance. It gives additional information but cannot substitute the indicator "utilisation frequency and timing" that is regarded as a better indicator for management intensity.

4.3.5.6. Fertilisation intensity

As an indirect (pressure) indicator for material load, Siebrecht and Hülsbergen (2009) recommend among others "fertilisation intensity", reflecting the potential effect of eutrophication. However, they do not define this term operationally. Therefore, N balance that is also reflecting nutrient input, is preferred over fertilisation intensity.

4.3.5.7. Organic matter balances

Balances for N, organic matter and energy reflect most agricultural activities and give direct or indirect hints on the environmental effects. They meet the demands for useful agricultural-environment indicators (Flade *et al.,* 2003). But there is no common method yet for site-adapted soil organic matter balances. The accuracy of existing organic matter balances differs clearly depending on site, cultivation method and organic matter factors used (Kolbe, 2005).

4.3.5.8. Soil pH value

Soil pH value may be lower in conventional than in organic farming due to the use of acidifying mineral-based fertilisers. But often differences between farming systems are small (Diez and Weigelt, 1986; Mäder *et al.,* 2002) and sometimes inconsistent (Diez and Weigelt, 1986; Fowler *et al.,* 2004).

4.4. THE ENERGY BALANCE

Agricultural systems all over Europe vary for different aspects. Beside the general system classification (mixed farming systems, grassland systems, arable systems) the intensity and the efficiency are very important objects. The description of intensity can be done by different means like the livestock units per area, the amount of used fertilizer, the share of irrigated crops, etc. In this context it is known that the intensification of agriculture of the last decades raises the energy use. Almost each measure on a farm is connected with an input of energy. Especially the production of modern crops is characterized by high inputs of fossil energy. But the consumption of energy differs to a large extent. In some low-input arable farming systems the energy input on arable land is lower than 1 GJ ha^{-1}, whereas in some modern high-input farming systems in Western Europe, it can exceed 30 GJ ha^{-1} (Faidley, 1992). The indirect energy consumption is often higher than the direct energy consumption.

The energy efficiency relates the energy consumption to the output and shows if the energy is well used. A large variability in French dairy farms has been observed going from 50 Litre Equivalent of Fuel (LEF) for 100 litres of milk to 200 LEF and no simple link exists between intensity and efficiency (Pointereau, 2008). 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 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 agroecosystem. With this the potential for environmental effects increases and the farming system has a stronger impact on biodiversity.

4.4.1. METHODS AND DATA

The most common method for characterizing energy use is the energy balance. This indicator is based on an input/output-calculation. In this respect it is comparable e.g., to the indicators of nitrogen balance. The recommended method is an integrated indicator of the Model REPRO (System for Environmental Impact Assessment; Hülsbergen, 2003) or the Model DIALECTE, PLANETE and PLANETE-GES (System for Energy and GES assessment; Risoud, 2002; Bochu, 2004). In these approaches input values are described by the consumption of "direct energy" (fuel, gas and electricity) and "indirect energy" (energy expended beyond the farm for the manufacture of fertilizers, plant protection agents, feedstuffs, machines, etc.) (Hülsbergen *et al.,* 2001; 2003; Ridoud, 2002). Human labour and solar energy are not considered as inputs.

To express the indirect input of energy associated with the manufacture of production means in terms of primary energy input "energy equivalents" are used. Special emphasis must be put on the energy equivalents of fertilizers, because the rate of fertilizer application has a particularly strong effect on the fixation of energy (Greef *et al.,* 1993). Energy outputs are defined as the calorific value of the harvested biomass (main products and by-products). It is computed by multiplying the DM yield by the calorific value of the plant material. By converting the yields into grain equivalents (GE, Woermann, 1944), the yields of crops that differ in chemical composition and, thus, energetic value for humans can be aggregated. This enables researchers to compare yields of crop rotations and farming systems.

These methods provide different indices for the evaluation of energy use. The detailed de-scription of the method REPRO is given Hülsbergen (2001) and the detailed description of the method PLANETE is presented by Risoud (2002).

TABLE 4.3 shows the parameters and indices for the REPRO method. For the use in the BioBio project we suggest to use energy intensity (4). This index represents directly the intensity of energy use. The output/ input ratio (5) in contrast includes the yields which vary much more (weather conditions, crops, competence of farmer) and a similar value of the output/ input ratio can appear at different intensity levels.

TABLE 4.3. DEFINITION OF ENERGETIC PARAMETERS FOR ENERGY BALANCING

No	Energetic parameter	Definition	Required data	Unit
1.	Energy input (E)	$E = E_d + E_i$	Calculation	GJ ha^{-1}
1.1	Direct energy input (E_d)	Input of diesel / fuel	Field measures + used machines	GJ ha^{-1}
1.2	Indirect energy input (E_i)	$E_i = E_S + E_{MD} + E_{OD} + E_{PSM} + E_M$	Calculation	GJ ha^{-1}
1.2.1	Energy seeds (E_S)	Energy for production of seeds	Seed amount + energy equivalents	GJ ha^{-1}
1.2.2	Energy mineral fertilizer (E_{MD})	Energy for production of mineral fertilizer	Fertilizer amount + energy equivalents	GJ ha^{-1}
1.2.3	Energy organic fertilizer (E_{OD})	Energy for production of organic fertilizer	Fertilizer amount + energy equivalents	GJ ha^{-1}
1.2.4	Energy pesticides (E_{PSM})	Energy for production of pesticides	Pesticide amount + energy equivalents	GJ ha^{-1}
1.2.5	Energy machines (E_M)	Energy for production	Used machines +	GJ ha^{-1}

		of machines	energy equivalents	
2.	Energy output (EO)	Energy in the harvested biomass (main products + by products) – energy in the seed	Yields of crops + energy equivalents	GJ ha^{-1}
3.	Net energy output (NEO)	NEO = EO – E	Calculation	GJ ha^{-1}
4.	Energy intensity (EI)	EI = E / GE or E / ha	Calculation	MJ per GE or ha
5.	Output/input ratio (OI)	OI = EO / E	Calculation	-

The described approach of the model REPRO is typically applied on the level of a single field. For each field the information of the grown crop, the used inputs (fertilizer, pesticides etc.) and machines and the yield of the crop are inquired. The necessary energy equivalents are used from an integrated database to calculate the indirect inputs. With this data all parameters and indices can be determined. For an overall assessment on the level of the farm, the field based energy balances are aggregated (weighted average).

The PLANETE methodology which is simplified in the agro-environmental diagnosis DIALECTE, is used also for mixed farms with animals and includes as direct energy electricity (irrigation, milking, greenhouse heating) and as indirect energy, the energy feedstuff bought and the energy for the farm building (see FIG. 4.1). The output also includes meat and milk.

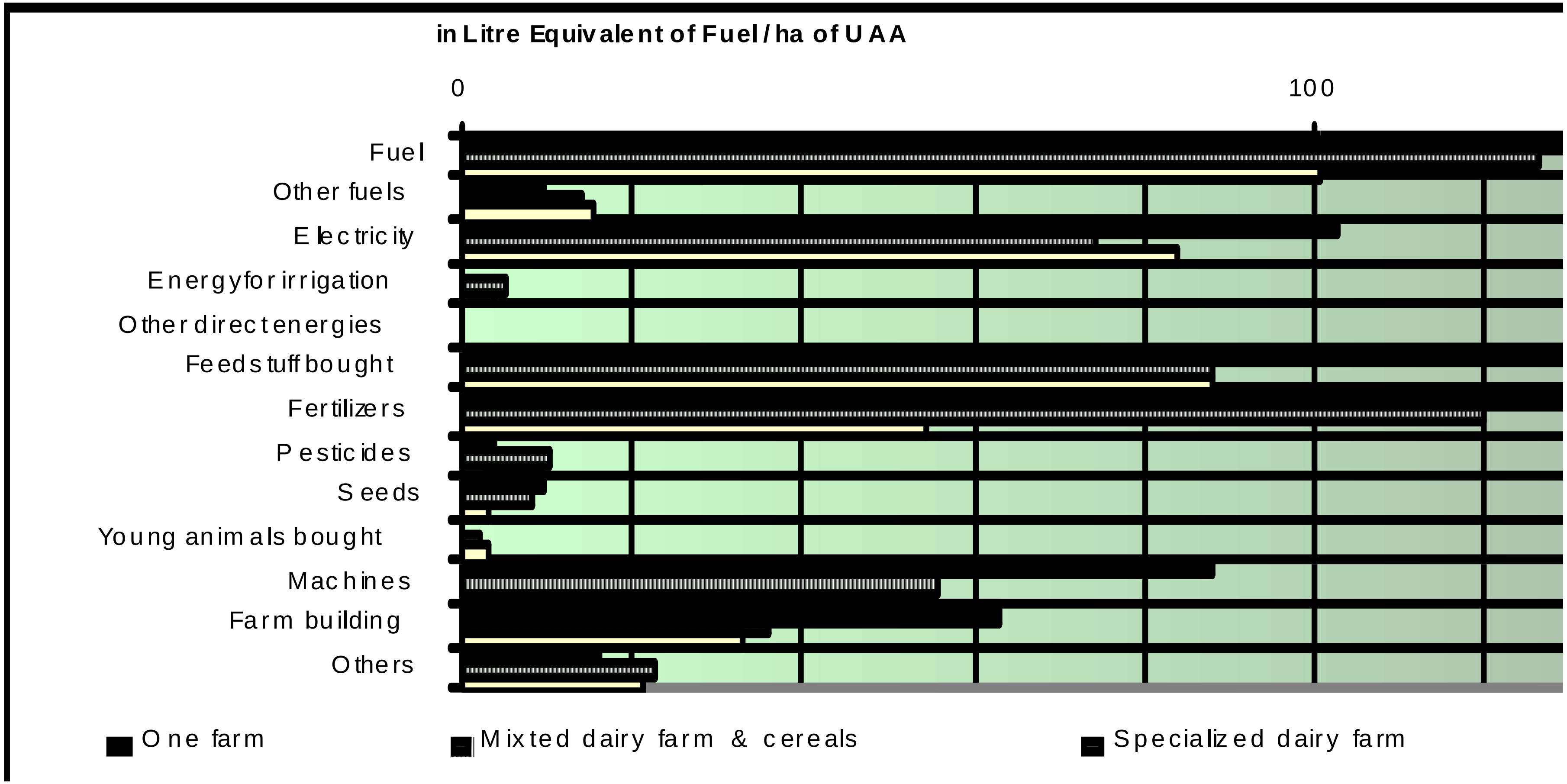

FIGURE 4.1. OUTPUTS OF DAIRY FARMS – EXAMPLE OF THE PLANETE RESULTS

If not all necessary input data are available we can try to use a "gradual realization". Farming systems with all information can apply the full energy balance. In farming systems without these data we need to operationalize it differently. While in conventional farming systems energy of mineral fertilizer is the most important input, organic or low-input systems don´t use it. Therefore we have to choose a different parameter like the direct energy input. In these systems the parameter should describe the intensity level of energy. For mixed farms it is necessary to get the quantity of feedstuff bought which is often an important input.

Similar to all other discussed indicators we have to define the method and scale for the assessment of the direct energy input, e.g., it would be possible to use a questionnaire and ask for the amount of used fuel in the year, use machine hours and so on.

4.5. INDIRECT INDICATORS OF BIODIVERSITY OF DIFFERENT FARMING SYSTEMS BASED ON FARM ACCOUNTS AND ECONOMICS MODELS

4.5.1. TOWARDS INDICATORS

The literature review assessed the relevant areas of economic literature related to the economic characterisation of organic and low input systems (structural data, income, etc.), as well as those found in section 6.2. This includes statistical information and studies about comparative characteristics of organic, low input and conventional systems. There are a number of studies concerning performances of organic systems. It led to the selection of candidate indicators (TABLE 4.4). These indicators have been selected because they can potentially be related to direct measurments of biodiversity (mainly species diversity, see TABLE 4.1) and/or because they are in use by existing European statistical surveys. They will be complemented by a comprehensive set of farm management parameters required to characterise farm management types and intensity.

TABLE 4.4. SUMMARY TABLE OF CANDIDATE INDICATORS PLUS REFERENCES

Candidate indicators name	**Candidate indicators details (e.g., measures, scale of applicability etc.)**	**Comments**	**Bibliographic references**
IRENA indicator 7/CSI 026: FADN/FSS organically managed percentage of farm holding	Percentage of UAA that is managed organically, and can be applied across EU-25/27 FADN/FSS dataset	Easy to calculate with access to the full or partial dataset. Covers around 60,000 holdings across all EU countries and can be assessed at various levels	Petersen, J. E. (2004)
IRENA indicator 2: Area under agri-environment	Data available from FSS or national	Easy to calculate with access to the	EEA (2005)

support	statistics	full dataset. Covers around 60,000 holdings across all EU countries	
Number of different crops grown per holding	FADN indicates number of crops grown	More crops will generally indicate more biodiversity	Williams Collins, W. Qualset, C.O. (1999) Biodiversity and Pest Management in Agroecosystems
Mixed farming methods	FADN indicates livestock and cropping data	A mixture of livestock and crops will generally provide greater biodiversity	Hole *et al. (*2005)
Livestock stocking rate (LU/ha)	FADN provides LU and UAA for calculation	Lower LU/ha will indicate less intensive farming methods	EEA (2006)
IRENA indicator 15: Intensification/extensification	FADN measure of inputs per ha UAA. Low-input farms spend less than 80 Euro per ha per year on fertilisers, crop protection and concentrated feedstuff. Medium-input farms spend between 80 and 250 Euro per ha per year and high-input farms more than 250 Euro per ha per year on these inputs.	Simple measure of farming intensity	EEA (2005)
IRENA indicator 5.2: Organic farm incomes	FADN/national data farm income data. Probably assessed as farm net value added per agricultural working unit (FNVA/AWU) and the family farm income per family working unit (FFI/FWU)	Provides indication of profitability of farms with enhanced environmental benefits. Are agri-environment schemes attractive to farmers? Are consumers willing to pay for biodiversity benefits?	EEA (2005)
FADN/FSS Farm typology	Farm type will indicate likely level of intensity, particularly when	E.g.,an organic arable LFA farm is likely to have higher biodiversity	Andersen (2007)

	combined with organic status, LFA status.	than an conventional upland livestock farm	
Crop protection use per ha	FADN crop protection costs data per ha.	Intensity vales vary by farm type	Andersen (2007) SEAMLESS 6th Framework EU program

4.5.2. TEXT EXPLANATION OF INDICATOR SELECTION CRITERIA AND CHOICES

The work of the EEA in developing the IRENA indicators is a good foundation for the selection of indicators for BioBio. Their indicator selection was based upon data availability and data quality for the EU-15 countries. As the EU statistics improve for EU-25 and now EU-27 data such as FADN and FSS can potentially be used to provide an indication of biodiversity across the whole EU at regional level.

IRENA indicator 7/EEA core set of indicators CSI 026; FADN/FSS organic land proportion (i.e., the land area for each holding that is organic or in-conversion, as a percentage of total Utilizable Agricultural Area, UAA) provides a simple indication of farming methods that seek to minimise damage to the environment. The measure can be applied to the FADN EU-25/27 dataset to indicate regional, national and EU levels of "more environmentally friendly" farming methods across Europe.

IRENA indicator 2: Area under agri-environment support will indicate the amount of land per EU region or country that is being paid additional funding for protection or enhancement of the environment. The level of funding will vary, and protection given will vary by country and scheme, but as a general indicator identifies reduced farming intensity and therefore probably higher biodiversity. Number of different crops grown per holding will tend to show an increasing level of biodiversity, the more different crops grown, the higher the biodiversity. Organic farming, particularly in lowland areas encourages this practise, and in particular a mixture of grasslands, cereals and legumes or horticulture is likely to provide the highest levels of biodiversity.

Mixed farming can be identified from FADN, and it is a measure complementary with the number of crops. Mixed livestock and arable farming, with mixtures of crops and livestock species will give the highest potential biodiversity due to variances in plant growth patterns, soil cover, as well as livestock grazing patterns and behaviour. Livestock stocking rate (GLU ha^{-1}) gives an indication of intensity of livestock production. Intensively farmed livestock areas will tend to have higher run-off and potential for other negative effects such as poaching of ground, and are likely to rely on higher input levels to maintain production.

IRENA indicator 15: Intensification/extensification is a simple measure of farming intensity, calculated as total input costs per ha. EEA has developed criteria, summarised in Section 3.3.2 for categorizing farms as low, medium or high intensity.

5. DIRECT INDICATORS OF BIODIVERSITY

In a more restrictive sense, "direct indicators" are based on the observation, counts or estimates of the occurrence and/or the abundance of varieties, races, genes, species, taxa and ecosystems and aim to directly evaluate biodiversity. These indicators may be simple (e.g., a species) or complex (e.g., a guild or community) or composite (e.g., a diversity index such as the Shannon index). Within the framework given in section 3.1 the choice of indicators for biodiversity depends primarily on the objects of the study (e.g., Noss, 1990; Huston, 1994) and secondly on whether knowledge is available of the properties of the proposed species and habitats. In the context of organic and low input farming, a criteria matrix can be prepared that takes into account both the objects of the study, for example the linking of the species to agricultural activity, their occurrence and their significance in the agricultural landscapes, and general criteria such as their distribution, their habitats and their place in the food chain (Pearson, 1995; Stork, 1995). The indicators must make it possible to estimate the impact of agricultural activity at plot and farm level, for biodiversity in the agricultural landscape is influenced by local (e.g., crop management method) and landscape factors (e.g., number of semi-natural habitats, (Burel, 1995; Duelli, 1997; Jeanneret, 2003).

5.1. GENETIC DIVERSITY OF CROP AND FORAGE PLANT SPECIES

Potential indicators for the genetic diversity of crops (including trees and vines) and forage plant species are listed in TABLE 5.1.

TABLE 5.1. POTENTIAL DIRECT BIODIVERSITY INDICATORS – GENETIC DIVERSITY OF CROP AND FORAGE SPECIES

Indicator type	Indicator	Variation detected	Sample throughput	Reproducibility / Reliability	Labour required	Technology required	Comment	Recommended for BioBio
On-farm survey	Number of cultivars / landraces grown	Low	High	Low	Low	Low	Strongly dependent on reliability of information obtained. No direct indication of genetic diversity of a species.	Tentative
On-farm survey	Origin of cultivars / landraces	Low	High	Medium	Low	Low	Cultivars out of the same breeding program may be genetically similar	Yes
On-farm survey	Area of cultivars / landraces cultivated	Low	High	Medium	Low	Low		Yes
On-farm survey	Phenotypic characteristics of cultivars / landraces	Low	High	Low	Low	Low	Characteristics of an accession as perceived by the farmer	Yes
Pedigree Analysis	Coefficient of parentage	Medium	Medium	High	Medium	Low	Information only available for cultivars of selected species	Yes
Phenotypic markers	Genetic variance, mean value	Low	High	Medium	High	Low	Environmental influence requires replicated field experiments for reliable detection	No
Isozymes	Protein-Marker-	Medium	Medium	Medium	Medium	Medium	Number of loci available is limited and	Tentative

Indicator type	Indicator	Variation detected	Sample throughput	Reproducibility / Reliability	Labour required	Technology required	Comment	Recommended for BioBio
	Diversity (e.g., Euclidean distance)						consequently the ability to distinguish closely related populations or individuals is limited. Comparability of results across laboratories may be difficult	
RFLP[1]	DNA-Marker-Diversity (e.g., genetic distance)	Medium to high	Low	High	High	High	Very specific, highly informative markers, technically demanding, low sample throughput	No
RAPD[2]	DNA-Marker-Diversity (e.g., genetic distance)	Medium to high	Medium	Low	Medium	Medium	Easy to apply, low reproducibility	No
AFLP[3]	DNA-Marker-Diversity (e.g., genetic distance)	High	High	High	Medium	High	Universal primer combinations can be applied to all species	Yes
SSR[4]	DNA-Marker-Diversity (e.g., genetic distance)	High	Medium	High	High	High	Highly specific, the marker of choice when available for the species under study	Tentative

[1] Restriction Fragment Length Polymorphism

[2] Randomly Amplified Polymorphic DNA

[3] Amplified Fragment Length Polymorphism

[4] Simple Sequence Repeats

Indicator type	Indicator	Variation detected	Sample throughput	Reproducibility / Reliability	Labour required	Technology required	Comment	Recommended for BioBio
SNP[5]	DNA-Marker-Diversity (e.g., genetic distance)	High	Very high	High	High	High	Sequence Information required, technically demanding	No
DArT[6]	DNA-Marker-Diversity (e.g., genetic distance)	High	Very high	High	High	High	Available only for selected species	No

[5] Single Nucleotide Polymorphism

[6] Diversity Array Technology

While the importance and the benefit of diversity at the species level is generally recognized, the role of diversity at the lowest level of organization, i.e., genetic diversity within populations and individuals, may be less apparent. Genetic diversity is indispensable for the response of species and populations to selection, either natural through environmental changes or human mediated through processes such as targeted selection (Reed and Frankham, 2003). This has long been recognized by plant breeders who routinely screen large germplasm collections for variation in specific traits or use ecotype populations to broaden their breeding germplasm (Allard, 1999). Genetic variation of quantitatively inherited, fitness related traits is essential for the adaptability of populations and may therefore contribute to ecosystem stability. Population fitness is a very complex trait and a direct relationship between genetic diversity and the various fitness components can often not be established (Booy *et al.,* 2000). However, preservation of diversity for genes controlling specific traits such as disease resistance has been shown to be of great importance for population survival (Foster Hünneke, 1991). In addition, reduced genetic diversity is often the result of partial inbreeding, which can directly influence population fitness (Oostermeijer *et al.,* 1995). On the ecosystem level, increasing crop genetic diversity has shown to be useful in pest and disease management and has the potential to enhance pollination services and soil processes in specific situations. It also contributes to the long-term stability of agroecosystems and helps to provide continuous biomass cover, aiding the ecosystem to sequester carbon and helping to prevent soil erosion (Hajjar *et al.,* 2008). With regard to the importance of genetic resources for the improvement of agronomically important traits and their value as a reservoir of biodiversity at the lowest level (FIG. 5.1), a detailed characterization of genetic diversity within crop and forage species is indispensable.

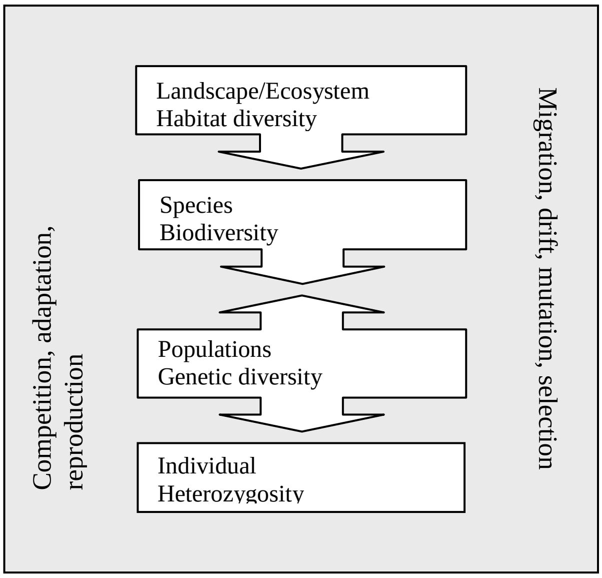

FIGURE 5.1. POTENTIAL CONNECTIONS AND FACTORS INFLUENCING DIVERSITY AT DIFFERENT LEVELS OF ORGANIZATION. (MODIFIED FROM VELLEND and GERBER, 2005)

In view of the importance of genetic diversity for crop production worldwide, extensive efforts have been undertaken to characterise and conserve germplasm resources of a broad variety of plant species (FAO, 1997). For example, the germplasm collections directory of Bioversity International (www.bioversityinternational.org) lists summary information on *ex situ* germplasm collections worldwide. Currently, summary information are available for more than 5 million accessions belonging to more than 20,000 species worldwide. For initial conservation efforts, collecting missions may focus on all accessions available in a particular region (Hammer, 1999). However, in order to utilize crop genetic resources and to minimize costs involved in maintaining large germplasm collection, a broad variety of approaches has been undertaken to characterise genetic diversity within various crop species (e.g., Achleitner *et al.,* 2008; Fjellheim *et al.,* 2007; Kumari *et al.,* 2008; Manninen and Nissila, 1997; Panahi and Talaie, 2001). Particular emphasis was given to the establishment of core collections, which harbor the majority of variation characteristic for a species (e.g., Kouamé and Quesenberry, 1993; Mosjidis and Klingler, 2006) Generally, genetic diversity may either be characterised by examining pedigrees and biparental relationships among individuals or based on methods which use markers that vary between the entities analysed. Despite the large number of markers available, there are a number of requirements that apply to all markers used for reliable estimates of genetic diversity. Markers should: (i) be heritable, (ii) discriminate between individuals, populations or taxa being examined, (iii) be easy to measure and evaluate, (iv) provide comparable results, and (v) be known to be either neutral or unlinked (FAO, 1997).

Although there are countless studies on genetic diversity within specific plant species and populations, so far no attempt has been made to develop general indicators of genetic diversity which can be used to compare genetic diversity across different plant species. Therefore, in order to estimate plant genetic diversity on individual farms and compare individual farming systems, a careful selection of the possible indicators outlined below has to be made, taking into account individual characteristics of the most prevalent species in the respective areas under investigation.

5.1.1. METHODS FOR CHARACTERISING GENETIC DIVERSITY

Genetic diversity within species and, therefore, the diversity among individuals and groups of individuals can be estimated in various ways (TABLE 5.1). Although environmental effects may be largely excluded through adequate measures and methods, estimates always reflect only a proportion of the overall genetic diversity of an individual, population or species.

5.1.1.1 Information based detection

5.1.1.1.1. On-farm survey

Collecting information on farm about landraces and cultivars grown is apparently one of the most straightforward approaches for assessing the genetic diversity of a species grown in a particular area. This strategy has consequently been widely used namely for establishing germplasm collections and detecting genetic erosion in a variety of plant species (Cebolla-Cornejo *et al.,* 2007; Hammer *et al.,* 1996; Teklu and Hammer, 2006). Such surveys are usually conducted using questionnaires directed at information related to genetic diversity of the populations grown on the respective farms (Teklu and Hammer, 2006). Potential indicators to be included in such surveys include the number of sub-specific taxa, population size, number and isolation as well as environmental amplitude or quantitative variation of particular characteristics (Brown *et al.,* 1997). In order to monitor differences in genetic diversity between farms and/or regions, indicators such as the number of landraces, vernacular names of landraces, main distinctive morphological or agronomic traits, changes in area under cultivation and estimates of phenotypic diversity may be also evaluated (FAO, 1997). Although it is generally recognized that indigenous and traditional knowledge is useful for analyzing genetic diversity (FAO, 1997), results from such surveys have to be treated with a reasonable amount of caution as long as no quantitative measures of genetic diversity are also surveyed. The mere fact that accessions or cultivars are grown under different names may not mean that they show large genetic differences. In addition, particularly in outbreeding species such as forage grasses or legumes, genetic variability within populations may be larger than between population variability (Herrmann *et al.,* 2005; Kölliker *et al.,* 1999). Consequently, only small differences in variability are often found among landraces and cultivars of such species (Kölliker *et al.,* 2003). In addition, genetic diversity within individual breeding programs may be quite narrow (Fang *et al.,* 2007) and the mere number of cultivated varieties may not be an indication for the on farm genetic diversity unless the origin of varieties is taken into account.

5.1.1.1.2. Pedigree analysis

Intra-specific diversity may be estimated by examining the pedigrees of different cultivars. Given certain assumptions, in particular that all parents contribute equally to the progeny, the degree of relatedness between individual cultivars can be estimated. This method of assessing diversity is a very useful characterization parameter that allows estimates of inbreeding or assessments of genetic variability. It is, of course, only possible for cultivars where pedigree records are available. It cannot be used for historically undocumented traditional cultivars such as landraces or for composites (FAO, 1997). However, for species such as rice, an extensive database is available on the genetic ancestry of rice varieties and of hybridizations made in national programmes. Pedigree information of numerous crop species is collected in the System-wide Information network for Genetic Resources (SINGER) which contains key data on the source, characteristics and distribution of the individual accessions in collection of various centres such as CIMMYT, IRRI, CIAT, CIP, ICARDA, ICRISAT, ICRAF, IITA and WARDA (Australian Centre for International Agricultural Research, 2009). Numerous studies have demonstrated the suitability of pedigree analysis for analysing crop diversity. In barley for example, genetic diversity was estimated by coefficient of parentage (CP) among 363 contemporary North American cultivars. The estimate of genetic diversity was found to be substantially lower among the 23 most widely grown cultivars that constituted 87% of the US harvested barley hectarage when compared to the overall diversity within the collection (Mikel and Kolb, 2008). In rice, genetic diversity based on CP was found to be increased in China from 1990-2003 when compared to 1976 to 1990, mainly due to incorporation of diverse breeding material from other countries (Wang and Lu, 2006). Thus, pedigree analysis is a powerful tool to estimate changes in genetic diversity, but target species for the survey have to be selected according to the availability of databases with pedigree information or other information on genetic diversity.

5.1.1.2. Marker based detection

The concept of using simple "marker" characteristics to select for more complex target traits was first proposed by Sax (1923) who observed an association between pigmentation and seed weight in *Phaseolus vulgaris*. The concept was consequently refined and expanded by many scientists and numerous marker systems targeting phenotypic, biochemical and molecular genetic characteristics have been developed and employed as tools to facilitate selection as well as to analyse genetic diversity in natural and breeding gene pools.

5.1.1.2.1. Phenotypic markers

Phenotypic markers describe observable, morpho-physiological characters of an organism. In plants, they include traits such as flower colour, fruit size and growth habit. Phenotypic markers are easy to detect and allow the agronomically most relevant approach to genetic diversity. However, in order to separate phenotypic plasticity from genetic diversity i.e., to account for environmental influences, laborious field experiments replicated across different environments are inevitable. This has been successfully applied in breeding and decade-long targeted phenotypic selection has resulted in crop and forage cultivars significantly improved for various traits (De Vita *et al.,* 2007; Humphreys, 2005). The diversity of a population with regard to phenotypic traits is usually described by its genetic variance and mean values (FAO, 1997). In order to standardize the collection of phenotypic data, Bioversity International (formerly International Plant Genetic Resources Institute, IPGRI) provides descriptor lists for a wide range of plant species which include fruit trees, arable crops, and vegetables (Bioversity International, 2009). In maize for example, such descriptors include day to tasseling, days to ear leaf senescence, ear height, tillering index, stem colour, kernel row arrangement or 1000 kernel weight. Despite the broad use of phenotypic characters for investigating genetic diversity, various studies also demonstrate its limitations. For example, in a study on diversity of phenotypic profiles in Vietnamese rice landrace populations, only small differences were found in agronomic characters among the populations. In addition, the authors observed significant differences in phenotypic profiles of farmers fields when compared to experimental sites, suggesting a significant influence of the environment (Fukuoka *et al.,* 2006). Phenotypic criteria to describe genetic diversity of forage cultivars are well established in a series of UPOV documents. Experiments are usually based on 60 individual plants, planted in at least 3 replicates, and a total number of 10 to 20 of UPOV accepted characters (UPOV, 2002; UPOV, 2009). UPOV characters, supplemented with some extra characters were used to show large phenotypic variation among natural populations which clearly exceeded variation among cultivars of *F. pratensis* in Norway (Fjellheim *et al.,* 2007). Some characters have particular high power in discriminating populations. In *F. pratensis,* 10 characters were identified by factor analysis as being responsible to distinguish populations, while in *L. multiflorum* only 8 characters were identified as most important (Peter-Schmid *et al.,* 2008a).

5.1.1.2.2. Biochemical markers (Isozymes)

Isozymes are multiple molecular forms of an enzyme sharing the same catalytic activity. They are analysed using electrophoresis and a color-producing reaction based on appropriate substrates and co-factors (Markert and Moller, 1959). Genetic diversity indices such as Euclidean squared distance are usually calculated based on presence or absence of particular bands on electrophoresis gels which vary between individuals or populations under study (Sammour *et al.,* 2007). Isozyme analyses revolutionized the analysis of genetic diversity when it was introduced 25 years ago. It remains a relatively inexpensive and reliable technique. However, the number of loci available is limited to 20 to 30 and only 5 to 6 variants are detected at each locus (FAO, 1997). However, isozymes have been shown to be particularly useful for the identification of cultivars in most important cereal crop species (Shewry and Halford, 2001). Gliadin electrophoresis has been proven to be particularly useful to investigate genetic diversity in old cultivars and landraces of Bulgarian wheat germplasm (Stoyanova and Kolev, 1996). Although seed protein markers detected a wealth of genetic divergence in local lentil accessions from Pakistan, they failed to differentiate accessions according to altitude or agroecological zones of the collection sites (Sultana *et al.,* 2006). This is a clear illustration of the limits of this method. On the other hand, isozymes have been successfully used to analyse genetic diversity in various forage species including red clover (Yu *et al.,* 2001) or creeping bentgrass (Warnke *et al.,* 1997) and they have been proven particularly useful to elucidate genetic relationships in forage grasses such as ryegrasses and fescues (Charmet and Balfourier, 1994). However, the number of isozyme assays is insufficient for many applications in plant breeding and isozyme markers often fail to distinguish between closely related individuals, limiting their use in survey of genetic diversity.

Application of protein electrophoresis for DUS variety testing is widely approved (UPOV, 2008). This technique has proven to be useful for the purposes of variety registration because the number of seeds which need to be tested for a particular variety is relatively small (OECD, 2001). The method is preferd for confirming the identity of individual seeds where other tests have been inconclusive (OECD, 2001).

5.1.1.2.3. Molecular genetic markers

Molecular markers are specific fragments of DNA that can be identified within the genome of the organism under study using a broad variety of techniques. The major advantages of molecular markers can probably be found in their nearly unlimited availability and their amenability to automation and high throughput analysis. Since the implementation of the first molecular marker systems (Botstein *et al.,* 1980), an enormous variety of marker systems have been developed and more than 30 different marker types have been described (Semagn *et al.,* 2006). Molecular genetic markers can reflect changes at the DNA level across the entire genome and they generally detect a large amount of genetic diversity among individuals and populations. Genetic diversity is usually expressed as molecular marker diversity by using a variety of different indices based on the marker system used, the species under study and the aim of the investigation. Mohammadi and Prasanna (2003) provided a comprehensive overview over the various statistical measures available. In the following, some of the most widely used molecular marker techniques are briefly described and samples of application for the investigation of plant genetic diversity are given.

RFLP – restriction length polymorphism markers were the first molecular markers to be extensively used in plant genetics research (Tanksley *et al.,* 1989) and significantly contributed to its advances. For detection of RFLP markers, genomic DNA is digested using restriction enzymes, restricted fragments are separated by electrophoresis and visualized after hybridization to a labeled probe. Polymorphisms between individual organisms arise due to gain, loss or relocation of restriction sites through point mutation, insertion/deletion, translocation, inversion or duplication resulting in size differences among the detected fragments. RFLP markers are highly reproducible, can be scored co-dominantly and may be conserved across species and genera. Although RFLP markers have been used for the analysis of genetic diversity or the construction of linkage maps in a broad variety of species (Maestri *et al.,* 2002; Pejic *et al.,* 1998), their extensive use particularly in poorly characterized species such as forage grasses or legumes has always been hampered by the fairly elaborate detection and the general lack of sequence information for many forage species.

RAPD ─ randomly amplified polymorphic DNA markers (Welsh and McClelland, 1990; Williams *et al.,* 1990) do not require *a priori* sequence information because they represent random DNA fragments which are amplified using short oligonucleotide primers of arbitrary sequence. Amplification products are separated on agarose or polyacrylamide gels and visualized using ethidium bromide or silver staining, respectively. Since a large number of markers can be generated in a short time, RAPDs rapidly became the markers of choice particularly for species where no DNA sequence information was available such as alfalfa (Yu and Pauls, 1993) or meadow fescue (Kölliker *et al.,* 1998) but also in crop species such as soybean or barley (Helms *et al.,* 1997; Manninen and Nissila, 1997). Despite their advantages, RAPD markers can only be scored dominantly for marker presence (i.e., heterozygous individuals can not be distinguished from homozygous individuals), they are often not transferable across different populations and they suffer from reproducibility problems (Jones *et al.,* 1997).

AFLP ─ amplified fragment length polymorphism (Vos *et al.,* 1995) are also anonymous DNA markers which are based on a more sophisticated technique of detection but are characterised by high reproducibility and better transferability when compared to RAPD markers (Jones *et al.,* 1997). To generate AFLP markers, genomic DNA is restricted using two different restriction enzymes and oligonucleotide adapters are ligated to the enzyme restricted sites. Consequently, random DNA fragments are amplified by PCR using primers complementary to the ligated adapter sequences. To reduce complexity of AFLP patterns, usually two consecutive PCR reactions are performed using primers complemented with one selective nucleotide in the first and three selective nucleotides in the second reaction. Fragments are separated on polyacrylamide gels and silver stained or analysed on an automated capillary electrophoresis system. Their reliability and suitability for high sample throughput have made AFLP markers very popular for the analysis of numerous plant species. Negri and Tosti (2002) were able to uniquely differentiate 36 *Phaseolus* landraces by using only three AFLP primer combinations. In wheat, AFLP markers have been found to valuably complement pedigree information for the estimation of genetic diversity (Soleimani *et al.,* 2002). In forage crops, AFLP markers have been widely used for the analysis of genetic diversity as well as for QTL and linkage analysis (e.g., Herrmann *et al.,* 2006; Skot *et al.,* 2002; Ubi *et al.,* 2003). Particularly, AFLP markers have been found to be useful for the distinction of landraces and wild accessions of red clover und to estimate the genetic diversity in these populations (Herrmann *et al.,* 2005; Kölliker *et al.,* 2003). Due to their universality and their high reproducibility, AFLP markers are predestinated for studies of genetic diversity where a number of different species are analysed and measures of genetic diversity are compared.

SSR ─ simple sequence repeat markers are sequence specific, PCR-based, co-dominant markers with a high level of reproducibility across populations within species. Short repeated sequence motifs of two to six nucleotides, which are ubiquitous in eukaryotic genomes, are amplified using primers targeting conserved flanking regions of the repeats (Tautz, 1989). Polymorphism arise due to variation in the number of repeat units caused by slippage during DNA replication and are detected by gel electrophoresis or using automated capillary electrophoresis systems. Their ability to detect polymorphism between closely related individuals and their co-dominant nature together with their sequence specificity and high reproducibility make SSR markers invaluable tools for genetic dissection of agronomic traits and analysis of genetic diversity in a broad range of species including ryegrasses (Jones *et al.,* 2000; Studer *et al.,* 2006) and clovers (Kölliker *et* *al.,* 2001a; Sato *et al.,* 2005). Using 23 SSR markers, Peter-Schmid *et al.,* showed distinct differences in the partitioning of genetic variation in *L. multiflorum* and *F. pratensis* with regard to altitude and managemgent practices (Peter-Schmid *et al.,* 2008a). Probably the only drawback of SSR markers is their laborious development which mostly involves construction and sequencing of genomic and/or cDNA libraries.

SNP — single nucleotide polymorphism markers are based on single base pair changes in a DNA sequence and are usually either detected by direct hybridization techniques or by generation and separation of allele-specific products through restriction, the detection of hetero-duplexes, primer extension, pyrosequencing or related technologies which have been described in detail by Kim *et al.,* (2007). SNP markers are extensively used for genetic characterisation of species such as maize where sequence information is abundant (Nelson *et al.,* 2008). However, the technique is technically demanding and currently only available for a selected number of species.

DArT — diversity array technology is a microarray hybridization based technique which allows simultaneous detection of thousands of markers without *a priory* knowledge of DNA sequence (Jaccoud *et al.,* 2001). A DArT array is prepared by enzymatic restriction of pooled genomic DNA of various organisms under study. Adaptors are ligated to the restricted DNA and fragments are amplified by PCR using primers with a selective overhang to reduce genome complexity. Fragments are cloned and arrayed onto solid support. The DNA samples for analysis are fluorescently labelled and hybridised to the DArT array. Successful hybridizations are dominantly scored for marker presence. Although DArT does not require *a priori* sequence information, interesting markers may be sequenced *a posteriori*, adding further value to the analysis. DArT arrays are available for a number of species including barley, *Lolium* and *Festuca* spp. (Kopecky *et al.,* 2009; Wenzl *et al.,* 2004), but the high cost involved in marker analysis will currently refrain users to apply it to large scale studies on genetic diversity.

5.1.1.2.4. Sample size for the analysis of genetic diversity

While the analysis of self-fertilizing species or cultivars based on single genotypes only requires one or few samples to be analysed per population, careful attention has to be given to the sample size when analysing outbreeding plant species or population based cultivars of genera such as *Festuca, Lolium* or *Trifolium*. For such species, to account for genotypes or alleles which occur at a frequency of at least ten percent, 40 plants have to be sampled, while 100 plants are needed to detect alleles occurring at a frequency of at least five percent (Crossa, 1989). However, practical limitations often limit extensive sampling of populations and one to 48 individuals have in the past been used to characterise genetic diversity in red clover populations (Dias *et al.,* 2007; Hagen and Hamrick, 1998). Analysis of bulked leaf material offers an efficient way to genotype a larger number of individuals per population with reduced effort (Greene *et al.,* 2004; Kölliker *et al.,* 2001b), but a loss of detection of rare alleles has to be taken into account. For AFLP analysis, markers present at frequencies of at least 20 % are still detectable in bulked samples consisting of DNA from 20 plants. By analysing two bulked samples per population (i.e., 40 individuals), Herrmann *et al.,* (2005) were able to investigate genetic diversity in a large red clover germplasm collection.

5.1.2. CONSIDERATION FOR THE SELECTION OF INDICATORS DEPENDING ON PLANT SPECIES

5.1.2.1. Crop Species

Crop species such as cereals, fruit trees, berries or vegetables are often grown as pure cultivars or landraces which are often based on only few genotypes and characterized by a substantial degree of uniformity within populations. For such species, genetic diversity may be evaluated through surveys and questionnaires. However, it is important to collect information not only on the number of different populations cultivated but also about characteristics of the populations such as their origin, their phenotypic appearance and their area of cultivation. Wherever possible, pedigree information must be used to complement such survey by detailed information on genetic diversity. In addition it may be appropriate to conduct isozyme or molecular marker based investigation on selected species.

5.1.2.2. Forage and Grassland Species

To the best of our knowledge, there are no standardized or internationally accepted methods for surveying genetic diversity of forage crops and grassland species on a farm scale. Such a method should be developed within BioBio. It could be applied to any production system, be it Bio / low-input or conventional, and would be one of the deliverables of the project. A distinction should be made between frequently re-sown and permanent grassland. Questionnaires should consider a part which assigns individual grassland surfaces to one of the following categories:

a) Leys as regular part of crop rotations
b) Grassland which has been re-sown within the past 5 years, following destruction/ploughing of the previous sward or following an arable crop
c) "Permanent" Grassland which has been re-sown more than 5 years ago, the re-sowing event being known, but with a variety of spontaneous plant species different from the species that had been sown
d) Truly permanent grassland which has never been sown, bearing a community of plant species which can be assigned to an alliance or lower unit of vegetation classification (association, sub-association)

Categories a) and b) may be considered together. The number of species and the number of cultivars per species sown should be surveyed and can be used as indirect indicators of genetic diversity of sown grassland. Seed trade regulations make sure that cultivars are genetically distinct from each other and that their within-cultivar genetic variability stays within certain limits. Therefore, there is no big incentive to try to determine genetic diversity among or within the cultivars used. The use of farm-saved seed or landraces is much more limited in forages compared to arable crops. It is probably the exception also in organic/low input situations. However, if farmers do maintain a landrace by long-term repeated use of their own seed to re-establish leys, this should be acknowledged as a specific contribution to genetic diversity. Category c) can be either treated like b) or like d). Botanical composition could be used as a criterion for assigning these surfaces to d) if the community of species can be assigned to a unit of vegetation classification, or to b) if this is not possible because the sown species still dominate. Category d) is at the same time the most important and the most difficult to assess in terms of genetic diversity. Research efforts of BioBio should focus on this category.

Surveys should also include indirect parameters of genetic diversity in permanent grassland. Kölliker *et al.,* (1998) concluded from a long-term experiment that an increase in management intensity led to a decrease of genetic diversity within population. This would imply that surfaces managed in a less intensive way would be valued higher in terms of maintaining genetic diversity within a species. However, this conclusion could not be confirmed when a number of different habitats with different management intensity were investigated (Peter-Schmid *et al.,* 2008a). There was no general trend towards a higher genetic diversity in more extensively managed grassland. Either there were no differences, or intermediate management intensity seemed to be favourable for a high within population diversity. However, for one of the species investigated (*F. pratensis*) the study showed that management intensity was significantly related to the grouping of populations based on molecular markers. Other habitat parameters like longitude, latitude, altitude, soil pH and Ca/Mg content were also involved. The authors conclude that maintaining permanent grassland of differing management intensity containing populations of the species of interest would contribute most to preserving its genetic diversity in different regions. These conclusions were supported by phenotypic data. The level of management intensity was significantly correlated with four phenotypic characters used for the assessment of genetic diversity among populations for both species investigated (Peter-Schmid *et al.,* 2008b).

L. multiflorum ecotype populations may be grouped according to the botanical composition of the swards of their origin (Boller *et al.,* 2009). Ecotypes from meadows belonging to the *Arrhenatherion* alliance performed generally poorer than ecotypes from non-*Arrhenatherion* meadows, but had more promising rust resistance. The authors conclude that classification of vegetation could be used as a criterion in selecting diverse sites for ecotype collections or conservation efforts.

Based on these results, the following hypothesis may be used for surveying genetic diversity in permanent grassland: "At a given location (farm), genetic diversity of a grassland species can be predicted by the number of distinct habitats in which the species occurs. Distinctness of habitats can be assessed by distinctness of the vegetation units to which their floristic composition is assigned".

This hypothesis would have to be tested within BioBio, using molecular genetic markers and/or phenotypic characteristics as outlined above. In all surveys of the case studies, vegetation units should be assigned based on botanical relevées and made plausible by relating the assignment to management history of the surfaces and other habitat parameters. Survey methods will be very similar to those used for assessing species diversity in permanent grassland. The questionnaires used to describe the management of permanent grassland should be complemented by some specific questions, particularly relating to the importance of over-seeding. The concept for *in situ* conservation of forage crops developed by Weyermann (2007) may be used as a basis to define criteria to be surveyed and to develop farmer questionnaires. Molecular genetic analysis will be performed on a representative subset of case study locations.

5.2. MEASURES OF THE SPECIES AND GENETIC DIVERSITY OF DOMESTICATED LIVESTOCK

The main mechanism by which grazing livestock could potentially influence biodiversity in pastures is through the creation and maintenance of sward structural heterogeneity, particularly as a result of diet choice (Rook *et al.,* 2004), and there is evidence from studies with both wild and domestic herbivores that different species of animal differ in their grazing habits. In general, the food requirements of mammals increase with increasing body weight as a result of increasing costs of maintenance and production, although crucially this increase is not linear. Consequently, a larger grazer requiring a greater absolute quantity of nutrients during a day will have less time per unit of nutrient to spend selectively foraging than does a smaller grazer with a lower absolute requirement. As a result, cattle and horses are generally less selective feeders than sheep or goats (Rook *et al.,* 2004; Fraser *et al.,* 2009). The diversity and species richness within a vegetation community can be affected in very different ways depending on the interaction between diet preferences of the grazing animal and the competitive abilities of the plants. A less selective diet is synonymous with an increased proportion of plant species generally avoided. In native pastures such plant species are often competitive, and there is potential for diversity to be maintained or enhanced by processes that reduce their dominance. However, the current knowledge base includes little direct evidence for the effect of grazing animal species on biodiversity (Rook *et al.,* 2004).

Differences in body size and the consequent allometric relationships with food intake, digestibility and selectivity could also potentially give rise to breed differences in grazing behaviour. In the past changes in nutrition and husbandry practices led to an increase in the use of larger, rapidly-maturing crosses within production systems at the expense of traditional breeds. While it has been postulated that a lack of adaptation to grazing biodiverse swards could result in grazing by such breeds having a detrimental effect on sward functionality, the evidence for effects on biodiversity is largely anecdotal (Rook *et al.,* 2004). Several recent comparisons of the dietary preferences and ingestive behaviour of different types of cattle have found few differences in the choices made by commercial and traditional breeds (Dumont *et al.,* 2007; Hessle *et al.,* 2008; Fraser *et al.,* 2009), implying that modern breeds could potentially deliver many of the biodiversity benefits of grazing currently associated with traditional breeds (Wallis de Vries *et al.,* 2007). While significant differences in diet composition have been reported between different breeds of sheep (Steinheim *et al.,* 2005; Fraser *et al.,* 2009), extrapolation to other breeds from the limited number of empirical studies done to date would be inappropriate (Dumont *et al.,* 2007) and the implications for biodiversity have not been tested. Furthermore, true genetic effects are often difficult to separate from the effects of rearing environment (Rook *et al.,* 2004). Likewise, age, sex and physiological status of the animals will also modify the grazing behaviour of a given species or breed, as will seasonality and associated changes in resource availability.

While diet choice can influence biodiversity through changes in sward composition and heterogeneity, the impacts of herbivores are not only realized through their foraging behaviour. Their numbers and densities must also be taken into account. Grazing intensity is a key management variable influencing the structure and composition of pastures (Scimone *et al.,* 2007) as well as the diet consumed (Dumont *et al.,* 2007). Grazing intensity also has a profound effect on invertebrates, although the response may differ depending on the vegetation type (Wallis de Vries *et al.,* 2007; Jauregui *et al.,* 2008). However, it is likely a range of grazing pressures is required to meet both species and functional diversity objectives (Milne, 1996; Jauregui *et al.,* 2008).

5.2.1. NUMBER AND PROPORTION OF INDIGENOUS BREEDS IN FLOCKS AND HERDS

Traditional and indigenous livestock breeds are often seen as being more appropriate for organic systems, due in part to their suitability to local conditions and their capacity to thrive on a forage only diet (Wanke and Boehncke, 2002). Most rare livestock breeds declined during agricultural intensification, when livestock selection and management was directed towards high productivity, often to the detriment of longevity, disease resistance, ease of parturition, meat quality and efficiency of feed conversion. These are characteristics, which, together with a high degree of adaptation to local environmental conditions, are the strength of many indigenous breeds (Fowler *et al.,* 2004).

Traditional breeds are also part of the genetic resource and their conservation is part of an EU Directive (Yarwood and Evans, 2002). Traditional or indigenous breeds of livestock are generally hardy and better adapted to living on poor quality vegetation without supplementary feeding and are better able to live out all year, at low stocking rates. They are, therefore, well adapted to manage semi-natural vegetation in an environmentally sensitive way (Yarwood and Evans, 2002).

Yarwood and Evans (2002) note that some traditional breeds of livestock are important in the conservation of specific habitats and that rare breeds have become useful in maintaining certain habitat conditions favoured by endangered plants and animals. They argue that grazing with rare breeds may be the most efficient option available for the farmer to maintain the nature conservation interest of a particular site. A potential problem with this as in indicator is clearly defining "indigenous" breeds and the fact that different classes and breeds of animals, even within the "indigenous" category, will have different impacts on different aspects of biodiversity.

5.3. SOIL ORGANISMS

Intact soil functions are fundamental for agriculture and soil fertility largely depends on biotic organisms in soils. Organic farming systems are more dependent on soil fertility than conventional farming due to the limited applicability of mineral fertilisers. Yields depend directly on the capacity of soils to sustain plant growth. In low-input farming systems, on the other hand, yield and production potential are often limited by restricted soil quality, i.e., shallow soils with low water retention and nutrient exchange potential.

Cency and Jones (2009) have summarised the state of the art of biodiversity indicators in soils. They conclude that "Bacteria, collembola and earthworms" are the most useful bio-indicators for appraising the evolution of biodiversity and assessing soil quality. TABLE 5.2 lists the soil biodiversity indicators discussed in this report. They are analysed in more detail in later sections (5.3.1 - 5.3.5).

TABLE 5.2. INDICATORS OF SOIL ORGANISMS UNDER CONSIDERATION FOR SELECTION

Species indicators		
Soil microbiology	Mäder *et al.,* 2002 ; Philips *et al.,* 2006, Bardgett and McAlister, 1999 ; Elmholt, 1996	Soil enzyme activity; mycorrhiza colonisation; microbial diversity (BIOLOG substrate utilisation): increased by organic farming. Discusses the impacts of agricultural practices on soil ecology broadly. Interest in the measurement of soil fungal : bacterial biomass ratios as an indicator of grassland management intensity and natural fertility. High interest in soil fungi, mainly Penicillium spp. abundance, as bioindicators of soil health after transition to organic farming, and of the lapse of time after transition.
Soil enzyme activity	Garcia-Ruiz *et al.,* 2009	High interest in soil enzyme activity as an indicator of soil quality in organic and conventional farming systems (better than nematode communities).
Soil biochemical properties	Lagomarsino *et al.,* 2009	Interest in microbial biomass C and N, microbial respiration, N mineralization, dehydrogenase, chitinase, acid-phosphatase, arylsulfatase and beta-glucosidase activities, as indicators of soil organic matter biochemical alteration. Soil microbial biomass and enzymatic activities were successfully used to detect short-term changes in soil and, taking into account its role in soil functioning, beta-glucosidase resulted in the most suitable indicator to predict organic C accumulation in soil under organic management in a Mediterranean environment.
(Soil) fauna	Mäder *et al.,* 2002 ; Paoletti *et al.,* 2007; Sepp *et al.,* 2005	Carabids, staphylinids, spiders, earthworm biomass and abundance: increased by organic farming. Interest in detritivore invertebrates (through their life traits) for bioindication of agricultural landscapes and soil degradation. Interest in soil biota and their activity as bioindicators of human pressure and the effects of agri-environmental programmes.
Soil mites	Zacharda, 2001; Gulvik, 2007	Mite communities (Acari) are extremely sensitive to all types of soil disturbance. Interest in mites as indicators of soil biodiversity and land use monitoring (taking into account Oribatida to Actinedida ratio and some families or genera), BUT there is a need to develop standardised procedures for the collection of samples and analyses of data sets adapted to ecological soil acarology.
Soil nematodes	Schloter *et al.,* 2003;	Nematode community structure can be considered as a good indicator of soil functioning, hence

	Neher, 2001; Yeates, 2003; Ekschmitt *et al.,* 2001	indicator of the decomposition function. Theoretical high interest in nematodes (mainly maturity and trophic diversity indices) as bioindicators of soil health, condition or quality.
Earthworms	Paoletti, 1999ab; Büchs, 2003a ;	Earthworms are suitable indicators for soil structure or compaction, tillage practice, heavy metals and pesticides. High interest in earthworms as indicators of different farming practices and different landscape structures and transformations. Species number, abundance, biomass, ecological guilds can give easily measurable elements.
	Paoletti *et al.,* 1998	High interest in earthworms as bioindicators of negative effects of tillage and fungicide residues (Cu, Zn).
Soil dwelling Diptera	Frouz, 1999	Soil dwelling Diptera occupy five guilds: phytosaprophages, surface scrapers, microphages, mycophages and predators and are sensitive to the impact of human activities, such as agricultural practices (type of tillage, manure, fertilizer application, pesticide application, drainage, set aside, rotation, fire, heavy metals, liming etc…). Interest in soil dwelling Diptera as bioindicators of food web structure in soils and stress factors in agroecosystems.
Collembola - springtails	Gardi *et al.,* 2002 ; Rusek, 1998 ;	Collembola have well differentiated ecomorphological life-forms and feeding guilds which enable the functional role that Collembola play in ecosystems to be recognised in some degree. Collembola play an important role in plant litter decomposition processes and in forming soil microstructure. High interest in collembolan communities as indicators of soil quality (otherwise measured through organic carbon and aggregate stability which can be considered as ecological services: soil fertility and stability.
	Zeppelini *et al.,* 2009	Interest in collembola (species richness, endemism and diversity) as bioindicators of restoration in mined sand dunes.
Isopoda - woodlice	Paoletti and Hassall, 1999;	Provide information on functional aspects of decomposition processes showing clear reactions to tillage, to the supply of decaying organic materials and to pesticide input as well as to the concentration of heavy metals. Theoretical interest in woodlice life traits for bioindication of agricultural landscapes.
	Souty-Grosset, 2005	Interest of woodlice as indicators of grassland habitat quality
Carabidae - ground beetle adults and larvae	Büchs, 2003a; Sauberer *et al.,* 2008	Important soil predators, easy to trap due to their mobility on the soil surface; being widely distributed, they are considered sensitive indicators of environmental conditions.

Multifunctional indicator of soil quality	Velasquez *et al.*, 2007	Low interest in multifunctional indicator of soil quality, which only contains one subindicator of biodiversity.

5.3.1. SOIL MICROORGANISMS AND THEIR FUNCTIONS

Soil microorganisms are the predominant part of soil biological components, and are responsible for many important processes that support soil functioning (Nielsen and Winding, 2002). It has been assumed that 80 to 90% of all processes in soil are mediated by microorganisms (Coleman and Crossley, 1996). They are strongly involved in nutrient cycling processes such as decomposition of organic materials and mineralization of residues (Bloem *et al.,* 1997). Microorganisms are involved in transformation, degradation of various hazardous compounds such as pesticides and transformation and immobilization of heavy metals (Kumar *et al.,* 1996; Nakajima and Sakaguchi, 1986). Furthermore, they influence soil structure by producing extracellular compounds and thereby affect physical soil parameters such as soil aggregate stability, water holding capacity, water infiltration rate, crust formation, erosion, and susceptibility to compaction (Elliott *et al.,* 1996; Gupta and Germida, 1988; Lynch, 1984). Some of them are able to suppress soil-borne pathogens through antagonism, synthesize various enzymes, vitamins, and hormones that regulate population processes, and they directly interact with plant functioning, by providing nutrients and promoting nutrient uptake, or acting as plant pathogens or competitors of plant pathogens (Altieri, 1999).

5.3.2. MICROBIAL DIVERSITY

The tree of life is dominated by the huge diversity of microorganisms including all prokaryotes (Bacteria and Archaea) and a large part of eukaryotes. There are many different estimates on how many prokaryotic species are covering the ecosystems. Estimates reach from several ten thousands (Palleroni, 1994), half a million (Tiedje, 1994), or 2-3 millions (Trüper, 1992) up to 10^9-10^{12} (Dykhuizen, 1998) prokaryotic species. Based on recent data it has been hypothesized, that approximately 10^{30} prokaryotic cells consisting of billions of different species inhabit the global ecosystems (Dykhuizen, 1998; Whitman *et al.,* 1998). These are very large numbers when compared to the number of about 10'000 bacterial species validly described in the 'Approved List of Bacterial Names' (for updates see www.bacterio.net). The reason for this discrepancy is the extremely large numbers but also the fact that most microorganisms have not been cultured so far. There are estimates that only 1% of all bacteria may be culturable by standard techniques (Torsvik *et al.,* 2002). Optimal growth conditions may strongly vary among different microorganisms, e.g., requiring either very low or very high nutrient concentrations or temperatures. Some organisms may only grow as consortia requiring signaling of other organisms (Kaeberlein *et al.,* 2002).

The heterogeneous environments of soils provide the basis for extremely diverse communities and it has again been estimated that one gram of soil may harbour 10 billions of prokaryotic cells belonging to several thousands different species (Torsvik *et al.,* 2002). This makes analysis of these communities difficult and basically renders determination of their biodiversity (species richness) impossible. Therefore techniques have to be improved, to allow determination or to reliably estimatate microbial diversity in soils. This would of course be even much more challenging, if intra-specific genetic diversity were to be assessed in these microbial communities.

5.3.2.1. Cultivation-dependent and -independent detection of microorganisms

As indicated above, only a very small proportion of soil microorganisms can be cultivated under standard conditions (Colwell and Grimes, 2000). Therefore, cultivation-dependent techniques do largely fail to represent entire soil microbial communities and are thus unsuited for the general determination of soil microbial biodiversity or intra-specific diversity. Cultivation-independent techniques, in particular nucleic acid based methods, may overcome these limitations and may allow for a less biased characterization of soil microbial communities. The introduction of the polymerase chain reaction (PCR) represented a milestone in molecular biology, allowing for the specific detection and investigation of genetic material even at very small quantities (Mullis and Faloona, 1987). PCR-based genetic analyses have been applied to microbial ecology (Pace, 1997) and have opened a entirely new view on microbial organisms and their lives (Rappe and Giovannoni, 2003). With these techniques, organisms may be compared based on their genetic information, which allows for the classification of groups according to phylogenetic relatedness. Several genes have been identified as suitable for community studies and their selection depends on the research objective. Targeting functional genes (functional markers) may allow the assessment of microbial groups with specific functions such as nitrogen-fixation, nitrification, or denitrification (Bothe *et al.,* 2000; Widmer *et al.,* 1999). However, many functional genes do not allow for the comparison of entire communities. Therefore, phylogenetic markers such as the ribosomal RNA (rRNA) genes are promising targets for detecting and classifying soil microbial community components (O'Donnell and Gorres, 1999; Pace, 1997). Classification of organisms and their phylogeny have been assessed by sequence comparison of this gene.

5.3.2.2. Sequence analysis in microbial ecology

Currently, retrieval of sequence information from a representative marker such as the rRNA gene may probably reflect the most detailed approach to assess microbial community compositions. The sequence information would allow for phylogenetic affiliation to known organisms and to compare community composition among different samples. For this purpose, the marker gene is directly amplified from soil nucleic acid extracts by PCR with marker-specific primers, followed by sequencing of the amplification products (Dunbar *et al.,* 1999). Cloning and sequencing approaches for bacterial communities have been successfully applied in several studies (for review see Janssen, 2006) and may eventually be capable of resolving the entire species diversity in a given sample (Hughes *et al.,* 2001). However, although information content, resolution power, and throughput of this approach is very high, the method is highly laborious and time-consuming for diverse communities and has limited automation capabilities. Therefore, the approach is suitable for communities with moderate species diversity and low number of samples. Recent technical developments allow for large-scale sequence analysis (Tringe *et al.,* 2005; Tyson *et al.,* 2004; Venter, 2004), but these technical demands are not yet routinely accessible. Monitoring of soil quality characteristics have to rely on rapidly applicable and affordable tools, which currently excludes the routine use of DNA sequencing approaches.

Soil is an important component for monitoring of sustainability of land use in relation to both the conservation of natural resources and biodiversity of ecosystems. Moreover soils provide a wide range of goods and services, and the biota plays fundamental roles in the majority of them. Nevertheless, their inherent biological complexity makes it difficult to know which ones have to be monitored as logical candidates and effective indicators. A plethora of biological methods have been suggested as indicators for monitoring soils but few are used in national-scale monitoring or are published as international standards. A framework for selecting ecologically relevant biological indicators of soil quality, for national-scale soil monitoring, that cover the range of functions and services of soil has been devised by Ritz *et al.* (2009). The literature was surveyed to identify 183 candidate biological indicators which were then scored by experts and stakeholders against a wide range of scientific and technical criteria. The framework used the scores and weightings to rank, prioritise and select the indicators. This semi-objective approach using a "logical sieve" allowed repeated iterations to take account of end-user requirements and expert opinion. A ranked list of 21 indicators was produced that covered a range of genotypic-, phenotypic- and functional-based indicators for different trophic groups (TABLE 5.3).

TABLE 5.3. CONSOLIDATED LISTING OF DISTINCT INDICATORS USING COMBINED F_{SF} RANKED ACCORDING TO F_A ACCORDING TO DEPLOYMENT STATUS. Source: Ritz *et al.*, 2009.

Indicator	Indicator descriptor	F_A	Sub-cat. 1	Sub-cat. 2	Ref #
(a) **Deployment status = 2.** Cutoff point $F_A > 100$					
TRFLP—ammonia oxidisers/denitrifiers	Genetic profile—specific group	769	Genotype	Nucleic acid	115
PLFA profiles	Composition—total community	615	Phenotype	Biomarker	18
TRFLP—ITS fungal	Genetic profile—specific group	437	Genotype	Nucleic acid	118
Multiple substrate-induced respiration (MSIR) GC	Activity capability profile—total community	311	Function	Activity	158
Nematode Baermann extraction procedure	Numbers, composition and size of nematode community	302	Phenotype	Fauna	52
TRFLP—bacteria	Genetic profile—specific group	295	Genotype	Nucleic acid	117
Microarthropods Tullgren dry extraction	Numbers, composition and size of invertebrates community within soil	188	Phenotype	Fauna	50
On-site visual recording—flora and fauna	Numbers estimate of animals	173	Phenotype	Other	162
Microplate fluorometric assay—multi-enzyme	Enzyme potential activity—wide range	172	Function	Enzyme	30
TRFLP—Archaea	Genetic profile—specific group	146	Genotype	Nucleic acid	116
TRFLP—methanogens/methanotrophs	Genetic profile—specific group	123	Genotype	Nucleic acid	122
Invertebrates pitfall traps	Numbers, composition and size of invertebrates motile aboveground	123	Phenotype	Fauna	46
TRFLP—actinomycetes	Genetic profile—specific group	121	Genotype	Nucleic acid	113
(b) **Deployment status = 1.** Cutoff point $F_A > 100$					
TRFLP—nematodes	Genetic profile	437	Genotype	Nucleic acid	119
Multiple substrate induced respiration (MSIR) MicroResp	Activity capability profile	313	Function	Activity	160
TRFLP—protozoa	Genetic profile	291	Genotype	Nucleic acid	120
qPCR AM Fungi	Genetic profile	111	Genotype	Nucleic acid	92
(c) **Deployment status = 0.** Cutoff $F_A > 50$					
Functional gene arrays	Genetic profile	788	Genotype	Nucleic acid	84
Phylogentic gene arrays	Genetic profile	511	Genotype	Nucleic acid	91
FISH—keystone species	Genetic profile	138	Genotype	Nucleic acid	83
Soil proteomics	Phenotypic profile	51	Phenotype	Other	108

Deployment status defined, as at mid-2005, as follows: 2 = fully deployable; 1 = likely to be ready for deployment in the short-term; 0 = not ready, some years development still needed.

Some of these indicators belong to the sub-category of 'products' coming from biota activities, while others are organisms themselves. Given that references of the different indicators are not accessible neither in the article nor among the supplementary material on the website of the journal, some further information about some of these indicators is given below from published literature.

Sepp *et al.* (2005) selected indicators of soil biota and used for this purpose abundance, diversity, and ecological composition of earthworm communities and hydrolytical activity of the microbial community. They did not find any statistically significant differences in the abundance and number of earthworm species between intensive and extensive agriculture pilot areas, but found differences in the hydrolytical activity of the microbial community between these latter. They concluded that soil bioindicators are suitable for monitoring human pressure as well as the effects of agri-environmental programmes which can increase the activity of the microbial community.

Elmholt (1996) studied microbial activity, fungal abundance and distribution of *Penicillium* and *Fusarium* as bioindicators to characterize organically cultivated soils, in four farms (chosen carefully in an attempt to minimize the variations caused by differences in e.g., soil type, soil management, fertilizer practice, and crop rotation and development) which had been cultivated organically for 2, 8, 11, and 31 years, respectively. The importance of the crop was clearly demonstrated, with a significantly higher microbial activity in the ley soils than in the wheat soils. However, the wheat soils yielded the most consistent results and thus seem better suited for studies of the long-term development of a bioindicator. The results showed that the abundance of the mainly soil-borne Penicillia was significantly higher at the 'oldest' organically cultivated farm than at the other localities, indicating a temporal development during later years following transition to organic farming. The abundance of *Fusarium* was more variable at the genus level, but some of the species seem very promising as bioindicators, especially *F. solani* and *F. equiseti*, but also *F. culmorum* and *F. tabacinum*. The results also indicated a temporal development in species richness of *Fusarium* during the first years following transition.

Bardgett and McAlister (1999) tested the usefulness of measures of soil microbial biomass and fungal: bacterial biomass ratios as indicators of effective conversion from an intensive grassland system, reliant mainly on fertilisers for crop nutrition, to a low-input system reliant mainly on self-regulation through soil biological pathways of nutrient turnover. They showed that fungal: bacterial biomass ratios (measured by phospholipid fatty acid analysis; PLFA) were consistently and significantly higher in the unfertilised than the fertilised grasslands. There was also some evidence that microbial biomass, measured by chloroform fumigation and total PLFA, was higher in the unfertilised than in the fertilised grasslands. It was also found that levels of inorganic nitrogen (N), in particular nitrate-N, were significantly higher in the fertilised than in the unfertilised grasslands. However, microbial activity, measured as basal respiration, did not differ between the sites. So, they argue that the measurement of a significant increase in the soil fungal: bacterial biomass ratio, and perhaps total microbial biomass, may be an indicator of successful conversion to a grassland system reliant of self-regulation.

5.3.2.3. Genetic profiling approaches

Various genetic profiling techniques, which give a relative and simplified image of the microbial community structure, have been developed in the recent past. They represent a more practicable way to routinely assess differences in microbial community structures among soil samples. These techniques are based on PCR amplification of a specific genetic marker region, followed by resolution of the amplified genes based on specific sequence characteristics. In principal, the available genetic profiling techniques can be divided into three groups. A first type of methods was developed that relies on conformational changes and melting behavior of amplified sequences, e.g., denaturing and temperature gradient gel electrophoresis (DGGE: Muyzer, 1999) and single strand conformation polymorphism (SSCP: Schwieger and Tebbe, 1998). A second category is based on length polymorphism of amplified marker genes, e.g., ribosomal intergenic spacer analysis (RISA: Fisher and Triplett, 1999) or length heterogeneity PCR (LH-PCR: Suzuki *et al.,* 1998). The third category relies on analysis of restriction endonuclease-derived fragmentation patterns, where the marker genes are differentiated based on the location of specific enzymatic restriction sites, e.g., PCR restriction fragment length polymorphism (RFLP: Massol-Deya *et al.,* 1995) and terminal RFLP (T-RFLP: Hartmann *et al.,* 2005; Liu *et al.,* 1997). These methods are well developed and widely applied, and all have their own advantages and limitations (for review see: Hill *et al.,* 2000; Kirk *et al.,* 2004). Whereas DGGE, TGGE, and SSCP may differentiate at very low phylogenetic levels, e.g., species, and allow to efficiently accessing phylogenetic information of the operational taxonomic units (DGGE or SSCP bands), these techniques show only a moderate resolution power for highly complex communities. In addition, these techniques have a low automation capability, which interferes with high throughput of samples, and do not allow for comparison among larger batches of samples. Furthermore, DGGE and TGGE are not compatible with automated capillary electrophoretic systems. RISA represents a rapid, high-throughput and high-resolution technique, with a similar phylogenetic sensitivity and identification capability as DGGE and SSCP, but currently is lacking the extensive sequence database required for comparison of data among studies. Application of capillary electrophoretic systems directly converts banding patterns into digital data and therefore represents an optimal analysis for following statistical data analysis. Finally, T-RFLP has proven to be a consistent and rapid high-resolution profiling technique for highly diverse communities, but may have a lower phylogenetic resolution when compared to the other approaches.

5.3.2.4. Application of genetic diagnostics in soil microbial ecology research

Genetic profiling analyses have been shown to detect changes in complex soil microbial community structures and in different habitats and environmental conditions. Soil microbial community structures were successfully analyzed in rice field soils (Noll *et al.,* 2005; Weber *et al.,* 2001), grassland soils (Kuske *et al.,* 2002), forest soils (Hackl *et al.,* 2004), and agricultural soils (Hartmann *et al.,* 2006; Hartmann and Widmer, 2006; Widmer *et al.,* 2006). The methods have also been applied to study effects of metal contamination (Frey *et al.,* 2006; Hartmann *et al.,* 2005; Tom-Petersen *et al.,* 2003), hydrocarbon pollution (Denaro *et al.,* 2005), 4-chlorophenol pollution (Jernberg and Jansson, 2002), different crops (Kuske *et al.,* 2002; Pesaro and Widmer, 2006), transgenic plants (Rasche *et al.,* 2006), compost amendment (Perez-Piqueres *et al.,* 2006), dry-rewetting stress (Pesaro *et al.,* 2004), flooding stress (Graff and Conrad, 2005), or CO2 exposure (Klamer *et al.,* 2002). These are only few selected examples and there are numerous more in the scientific literature. However, based on the complexity of the systems analysed it has not yet been possible to reliably determine and compare diversities or effects on diversities of microbial communities in soil. This remains a challenge to soil microbial ecology which may become solved in the future when next generation sequencing approaches supplemented with automated data analysis become applicable to routine soil analyses.

5.3.3. SOIL ENZYME ACTIVITY

According to Garcia-Ruiz *et al.* (2009), soil enzyme activities show a natural temporal variability which could mask the variability due to the type and timing of soil management practices. They insisted on this fact even if the GMea (geometric mean of enzymes activities) was significantly higher in organic than in conventionally managed plots, independently of the sampling and, moreover, showed significant correlation with the first axis of the PCA (principal component analysis). In addition, the GMea, and scores on the first axis were highly correlated with some of the soil nematode indices. Therefore, the GMea was a suitable tool to condense the whole set of soil enzyme values in a single informative numerical value, which was more sensitive to management practices than nematode community indicators.

Lagomarsino *et al.* (2009) used biochemical properties of soil, such as microbial biomass C and N (MBC and MBN), microbial respiration, N mineralization, dehydrogenase, chitinase, acid-phosphatase, arylsulfatase and beta-glucosidase activities, as indicators of soil organic matter biochemical alteration, to compare conventional vs. young (4 years) organic management systems. They found that soil microbial biomass and enzymatic activities were successfully used to detect short-term changes in soil and, taking into account its role in soil functioning, beta-glucosidase was the most suitable indicator to predict organic C accumulation in soil under organic management in a Mediterranean environment.

5.3.4. SOIL MICROBIAL COMMUNITIES - CONCLUSIONS

Analyses of soil microbial communities have tremendously profited from the recent advance in genetic diagnostics, which have opened entirely new perspectives (TABLE 5.2). An unexpectedly large number of microorganisms live in soils; many of them are still unknown and perform unknown functions. Currently available techniques allow exploration of these microbial communities but are resource intensive and do not allow the determination and comparison of actual diversities of soil microbiota. Therefore, it currently seems unreasonable to propose using these tools for routine monitoring of biodiversity indicators for farmland soils.

5.3.5. SOIL INVERTEBRATES

Soil invertebrates have been shown to respond sensitively to land management practices and to be correlated with beneficial soil functions (Paoletti, 1999a; Büchs, 2003a; b). General requirements for using this biological group as indicators of biodiversity in soils:

- they must form a dominant group occurring in all soil types
- they must have a high abundance and high biodiversity
- they must play an important role in soil functioning, e.g., in food webs
- they should exist in numbers that can be effectively handled by investigators, and have to be identifiable without undue effort and time.

5.3.5.1. Soil mites

Mites (Acari) have apparently seldom been used as indicators of biodiversity. However, it seems that they could be used as soil biodiversity indicators as well as indicators of epigeic/aerial biodiversity survey. Mite communities are extremely sensitive to all types of soil disturbance.

Gulvik (2007) reviewed some studies on mite communities in order to discuss whether the diversity and numbers of mites present in the soil can reflect human impact on ecosystems and landscape, and whether mites can be used in monitoring systems. The conclusion was that: (1) Most oribatid mites with their long life span, low fecundity, slow development and low dispersion ability can be robust indicators of the environment. Changes in the dominance structure of mite communities (Oribatida to Actinedida ratio) are suggested to be an 'early warning' of stressed mite communities, which could indicate adverse conditions for other taxa. (2) Both the number of species and the percentage abundance of Nothroidea and Ptyctimina decrease following human impact on the landscape. Even if individuals cannot be determined to species levels (e.g., *Ptyctimina*), the percent contribution and frequency of these taxa in relation to stress gradient (input levels) within the landscape provide valuable data. (3) *Tectocepheus velatus* (Michael 1880) and other oribatid taxa with a similar life-history strategy should be evaluated as potential bioindicators for impoverished ecosystems. (4) National and local reference data sets on the biodiversity of mite communities in diverse habitats and along stress gradients need to be collected. (5) Residual natural and semi-natural habitats (such as old woodlands, riparian ecosystems, old hedges and grasslands) with species-rich mite communities found in rural and urban landscapes should be preserved as refuges for dispersion of soil fauna. (6) Comparison of mite communities in traditional, low-input farmland in Norway with those from human-dominated landscape in other European countries can contribute to a better understanding of how human activity alters biodiversity along land-use gradients. This will aid the development of a soil bioindicator system. (7) There is a need to develop standardised procedures for the collection of samples and analyses of data sets adapted to ecological soil acarology. Thus it appears from the latter point that even if mite communities may be a good group of biodiversity indicators, significant work remains to be done and further knowledge must be acquired before they can widely be used as such.

Zacharda (2001) studied communities of predatory phytoseiid mites, collected from orchards and vineyards in the Czech and Slovak Republics, and their responses to the stresses of conventional agricultural management. Testing their susceptibility to a variety of pesticides using toxicological bioassays, he could show that predatory phytoseiid mites are sensitive bioindicators of pesticide stress in farmland ecosystems, and outlined their role in farmland revival.

Based on the above, our conclusion on the issue of aerial bioindication with mites is that they could be tested in BioBio on condition that there are standardised procedures for the collection of samples and analyses of data sets, as well as feasibility of 'easy' identification. Such a test could either involve a between-farm comparison (in diachronic or synchronic studies) or be conducted in a theoretical reference comparative way if national and local reference data sets exist on the biodiversity of mite communities in diverse habitats and along stress gradients. Nevertheless, due to the paucity of information about mites, particularly regarding the relationship between mite diversity and wider biodiversity, we are sceptical to the use of Acari in BIOBIO.

5.3.5.2. Soil nematodes

Since it has become appreciated that soil nematode assemblages are abundant, diverse and contribute to soil nutrient turnover, they have been increasingly used as indicators of soil condition (Schloter *et al.,* 2003; Neher, 2001). Nematodes fulfil quite well the requirements for being used as bioindicators and seem to be at present state of knowledge the most promising group among soil bioindicators. For instance, microbial communities (soil fungi and bacteria) are known to play critical roles in ecological processes but they present inherent logistical and interpretative challenges. Being higher in the food chain than soil microbes, nematodes serve as integrators of physical, chemical, and biological properties related to their food resources. Their generation time (days to years) is longer than metabolically active microbes (hours to days), making them more stable temporally and less prone to fluctuate with ephemeral nutrient flushes.

Nematodes may be the most useful group of soil organisms for community indicator analysis because more information exists on their taxonomy and feeding roles than does for other mesofauna. Nematodes play an important role in essential soil processes and possess several attributes that make them useful ecological indicators. Soil nematodes can be placed into at least five functional or trophic groups, and they occupy a central position in the detritus food web. A small fraction of soil nematode fauna depends directly on primary producers, feeding on plant roots and their exudates. However, most of the soil nematode species actually have beneficial roles in ecosystem processes and are not parasites or pests. For example, microbial-grazing nematodes affect growth and metabolic activities of microbes and alter the microbial community, thus regulating rates of decomposition and nutrient mineralization. The direct contribution of nematodes to nitrogen mineralization and distribution of biomass within plants has been demonstrated in controlled experiments. Predatory nematodes also regulate nitrogen mineralization by feeding on microbial grazing nematodes, a conduit by which resources pass from bottom to top trophic levels.

Unlike earthworms, nematodes are ubiquitous and certain species are frequently the last animals to die in polluted or disturbed areas, partly because they can survive desiccation and revive with moisture. Relative to other soil fauna, trophic or functional groups of nematodes can be separated easily, primarily by morphological structures associated with their various modes of feeding. The relative abundance and size of nematodes typically make sampling and extraction easier and less costly than for other soil fauna.

Therefore, nematodes may be the most useful group for soil mesofauna and health analysis. Nevertheless, use of nematodes as functional indicators relies on the allocation of nematodes to feeding groups and reproductive strategies. Species within feeding groups vary in their food resources and response to environmental variables. Therefore species-level discrimination is necessary to permit further advances in understanding the role of nematodes in soil processes and thus in ecosystem resilience (Yeates, 2003) and to be comparable with other biota. In many regions such identification will be difficult due to inadequate systematic knowledge of the nematode fauna (Yeates, 2003). Initially, simple indices of abundance, proportions, or ratios of nematodes by trophic group were proposed. Subsequently, diversity indices were employed and a Maturity Index (MI) was developed for terrestrial nematodes. Rigorous statistical analyses reveal that maturity and trophic diversity indices better differentiate the ecological condition of soils on a regional scale than do individual or ratios of trophic groups.

Several challenges remain to be overcome before it is possible to fully understand and interpret maturity and trophic diversity indices (see Neher (2001) for more details):

- the ability to resolve trophic or functional groups must be improved and many species have yet to be assigned to trophic and functional groups
- a more thorough understanding of the sequence of community succession relative to soil function dynamics would be useful for establishing the kind of community composition associated with ecologically sound agricultural systems
- quantitative associations that reveal cause/effect relationships or mechanisms between nematodes and ecosystem functions
- alternative indices for describing how nematode communities respond to environmental stress.

Nevertheless, according to Ekschmitt *et al. (*2001), nematode community structure can be considered a good indicator of soil functioning as "a high nematode richness can generally be seen as a good indicator of an active nematofauna and microflora in mineral grassland soils, and hence as an indicator of the decomposition function." And "a rough guess" of soil faunal diversity can be cost-effectively derived from environmental data (climate, soil and vegetation characteristics) while an estimate of moderate quality can be obtained with reduced taxonomic effort. The precise richness of a soil community, however, is subject to autogeneous community dynamics, to biotic interactions with other populations, and to conditions in the past, and can therefore only be retrieved by immediate investigation of the community itself." (Ekschmitt *et al.,* 2003).

Finally, if nematode community structure seems to be a good indicator of soil functioning and health, Garcia-Ruiz *et al. (*2009) demonstrated that "the geometric mean of enzymes activities was a suitable tool to condense the whole set of soil enzyme values in a single informative numerical value, which was more sensitive to management practices than nematode community indicators."

5.3.5.3. Earthworms

Earthworms (Anellidae, Oligocheta) are key soil detritivores, essential for composting and recycling soil nutrients whilst contributing to the maintenance of soil structure. About 430 species occur in Europe, 90 species in Italy (Paoletti, 1999b; Edwards, 2004; Peres *et al.*, 2006). Earthworms are classified into three main ecophysiological categories: (1) leaf litter/compost dwelling worms (epigeic), (2) topsoil or subsoil dwelling worms (endogeics); and (3) worms that construct permanent deep burrows through which they visit the surface to obtain plant material for food, such as leaves (anecic). Anecic species which are large, vertically burrowing earthworms building up stable burrows play an important role in conservation and improvement of soil structure. Earthworm populations depend on both physical and chemical properties of the soil, such as soil temperature, moisture, pH, salts, aeration and texture, as well as available food, and the ability of the species to reproduce and disperse. One of the most important environmental factors is pH, but earthworms vary in their preferences. Most earthworms favour neutral to slightly acidic soil.

Most of the larger species (over 18-25 cm long) have disappeared from cultivated areas and are mainly only found in woodland and unimproved grassland. This is mainly due to high predation rates occurring after tillage operations (Paoletti, 1999). Earthworms form the base of many food chains. They are preyed upon by many species of birds, e.g. starlings, thrushes, gulls, crows. Earthworms are eaten by some mammals such as foxes, hedgehogs and moles, as well as by many invertebrates such as ground beetles, snails and slugs. The role of earthworms in enhancing soil fertility is well known and farming practices have considerable effects on both earthworm abundance and species composition. Earthworms contribute to physical, chemical and biological soil processes such as soil structure formation, e.g., formation of stable aggregates, and organic matter dynamics through nutrient cycling, decomposition of residues, and soil pore water dynamics through their burrowing activities, which provide soil pores for aeration and water infiltration. Consequently, the productivity of arable farming systems can be improved by the presence of abundant earthworm populations.

Earthworms are considered suitable indicators for soil structure or compaction, tillage practice, heavy metals and pesticides (Paoletti, 1999a,b; Büchs, 2003a). According to Paoletti (1999b), endogaeic earthworms are more suitable for monitoring pesticide or heavy metal residues than anecique species. Biomass, species numbers and ecological guilds (e.g. Epigaeic, anecique, endogaeic) can be used as key indication parameters for assessing earthworms in agro-ecosystems (Paoletti, 1999a,b). Knüsting *et al.* (1991, in Büchs, 2003a) showed that for farming systems with different pesticide input levels (beside litter supply) tillage is the main factor steering the performance of earthworm communities in agro-ecosystems. Positive effects of specific farm practices on earthworms such as direct drilling and minimal cultivation of cereals have been found. In vineyard agrosystems, some authors demonstrated that earthworm biomass and abundance were correlated to microbiological biomass, and that, comparing conventional management to integrated management, these two biological components could reveal the anthropic constraints (Paoletti, 1988; Paoletti, 1999b; Peres *et al.*, 2006).

The activity performed by earthworms allows the soil to reach a condition that hosts many other sorts of organisms, hence enhancing the overall soil biodiversity. Rich soil biodiversity and biomass, in turn, means a supply of higher amounts of resources for greater above ground trophic levels, so contributing directly to enhance the overall biodiversity of agroecosystems (Lee, 1985; Edwards, 2004).

Earthworm sampling should preferably be carried out during cool and wet seasons. Although earthworms can live in litter, soil, wet mud, submerged mud, organic manure, composts, dung, under bark and on rotten wood, most earthworm species are adapted to a particular habitat. One active collection system consists of hand sorting from soil cores of 25 x 25 or 30 x 30 cm^2 dug to a depth of 20–50 cm with a spade. Digging deeper than 20–30 cm into the soil yields few specimens but sometimes reveals interesting deep-burrowing species. To assess populations of deep-burrowing and larger specimens, irritant solutions can be used to stimulate the earthworms to come to the soil surface, thereby facilitating collection. One particularly effective technique involves the application of aqueous formaldehyde solution onto 50 x 50 cm^2 of soil.

5.3.5.4. Soil dwelling Diptera

Soil dwelling Diptera are insects which spend either their entire life cycle or the immature stages of their life in the soil. There are several reasons why soil dwelling Diptera are suitable as bioindicators. The larvae of Diptera form an important part of the edaphon in various ecosystems, and may represent the most abundant part of soil macrofauna in some ecosystems. The soil dwelling Diptera represent a very diverse group that includes animals of various sizes and shapes (Fig. 5.2) at various levels in the food webs (e.g., saprophages, algaevorous or fungivorous species, predators, etc.).

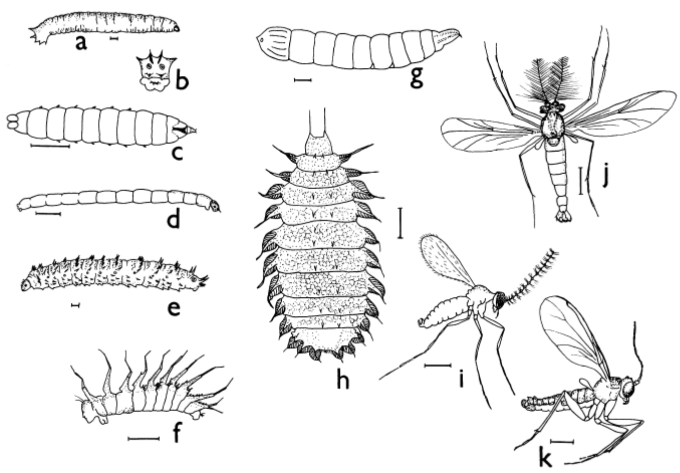

Fig. 1. Some morphological types of soil dwelling Diptera, scale bars represent 1 mm; a–h larvae, a—Tipulidae, b—Tipulidae, end of body, c—Cecidomyiidae, d—Chironomidae, e—Bibionidae, f—Ceratopogonidae, g—Empididae, h—Fanniidae; i–k—adults, i—Cecidomyiidae, j—Chironomidae, k—Sciaridae.

FIGURE 5.2. SOME MORPHOLOGICAL TYPES OF SOIL-DWELLING DIPTERA (Source: Frouz, 1999).

Some species of soil dwelling Diptera play a significant role in the decomposition of dead organic matter and nutrient release. The life cycle of soil dwelling Diptera combines slowly moving larvae and highly mobile adults which obviously respond differently to environmental changes. The larvae dwell in conditions similar to those of other entirely soil dwelling animals. Contrary to the majority of other soil animals which migrate very slowly, Diptera can colonize new habitats quickly by the flying adults. One disadvantage for the use of soil dwelling Diptera as bioindicators relates to difficulties in their determination (particularly for larval stages). These difficulties are probably the main reason why this important group of soil fauna has been frequently neglected. Frouz (1999) made a review on this group to give a brief outline of their ecological background in order to facilitate interpretation of data. In particular, the data concerning the ecology and distribution of soil dwelling Diptera and their reaction to stress and man-managed environments are addressed.

Soil dwelling dipteran larvae are more or less abundant according to various temperate ecosystems, being more numerous in grasslands and forests (up to 14 000 individuals m^{-2} for beech forests) than in cultivated fields (where estimates range from less than 100 to more than 600). Different families (Cecidomyiidae, Chironomidae, Ceratopogonidae and Sciaridae) dominate according to the average climate of sites and type of ecosystems. It seems that their biomass reflects the input of plant litter into the ecosystem. The lowest values of biomass were found in agroecosystems under annual crops, where almost all litter is removed, and in ecosystems of low productivity such as alpine meadows. Although less numerous, other families like Tipulidae, Bibionidae, Rhagionidae and Empididae, Dolichopodidae and Hybotidae frequently form an important part of the total biomass of dipteran larvae in various ecosystems. It can be assumed that 50–150 species of soil dwelling Diptera can commonly be reached in a given plot (i.e., in one vegetation type in one locality) in the temperate zone. The general pattern of seasonal dynamics of soil dwelling dipteran larvae in temperate habitats is characterized by increasing abundance in late autumn and winter, and decreasing abundance in late spring and summer. Conversely, the peak of adult emergence appears in spring. The highest numbers of larvae can be found on the bottom of the litter layer; numbers decrease with increasing soil depth and towards the surface of the litter, which is usually drier. The vertical distribution of particular families and groups (surface scrapers, predators) can differ from this scheme.

Soil-dwelling Diptera have a variable position in food webs and Hövemeyer (1984) suggested classification of feeding types of dipteran larvae into five groups: phytosaprophages, surface scrapers, microphages, mycophages and predators (cf. TABLE 5.4).

TABLE 5.4. CLASSIFICATION OF FEEDING GROUPS OF SOIL-DWELLING DIPTERAN LARVAE (Source: Hövemeyer, 1985).

Table 5
Classification of feeding groups of soil dwelling dipteran larvae according to Hövemeyer (1984), modified. Original terms of Hövemeyer are given in quotes

Feeding groups	Mode of feeding	Food utilized	Families included
Phytosaprophages 'Phytosaprophage'	Consume large particles of plant tissue (alive or dead), including associated microflora and fine soil particles	Dead or live plant tissue, leaves, roots, litter	Trichoceridae, Tipulidae, Limoniidae (part), Bibionidae, Sciaridae, Scatopsidae
Surface scrapers 'Oberflächenschaben'	Scrape fine particles from surface of plant litter	Associated microflora, fungi, algae, protozoa, nematodes, fine organic detritus	Lonchopteriade Phoridae (part) Drosophilidae (part) Otiitidae Lauxanidae Fannidae
Microphages 'Microphage'	Consume fine particles in soil	Fungi, algae, moss, protozoa, nematodes, fine organic detritus	Chironomidae Ceratopogonidae (part) Stratiomyidae
Mycetophages 'Mycetophage'	Consume hyphae of fungi	Fungi	Cecidomyiidae
Predators 'Zoofagous'	Predation	Soil oligoachaeta, other soil insects, particularly larvae, etc.	Limoniidae (part) Ceratopogonidae (part) Empidoidea Tabanidae Rhagionidae Therevidae Muscidae (part - Phaoninae)

The main natural environmental factors affecting the distribution of Diptera in soil are input of dead organic matter into soil and soil moisture. While young larvae are sensitive to moisture deficiency, before pupation the larvae search for dryer spots in the soil. High water content in soil can negatively affect development of older larvae and pupae in soil. Thus it is also necessary to consider annual and seasonal climatic aspects, and even geographical aspects of environmental factors, because the impact of a particular factor can be different in various stages of dipteran life cycles.

Soil dwelling Diptera are also sensitive to the impact of human activities, such as agricultural practices (type of tillage, manure, fertilizer application, pesticide application, drainage, set aside, rotation, fire, heavy metals, liming etc.). These insects can be used as indicators at different levels:

- Level of individuals: can reveal effects of stress factors (e.g., mortality of individuals, absorption of xenobiotic substances, and changes in certain physiological or morphological parameters). Nevertheless, there is little information on this use of the group in terrestrial ecosystems.
- Level of species: some species are characteristic for particular ecological conditions and may be used as indicators, but there is a low level of knowledge about the ecology of many dipteran groups and ecological requirements have only been documented for very few families. Also this approach is complicated by taxonomic difficulties.
- Level of higher taxa: Diptera are a very heterogeneous group and evaluation at the level of order is not recommended. The ecology of lower taxonomic levels (families, subfamilies, tribes etc.) is more uniform; nevertheless even here interpretation should be made with caution because strong differences may occur within the same genus. Thus species level is the only one that is really efficient for defining a bioindicator value.

- Level of community: Changes in community structure under the influence of stress factors are suitable for evaluation of the influence of stress factors in field conditions. The problem with evaluating the whole dipteran community is the difficulty of species identification, particularly of larval stages; this can be partly solved by limiting the study to the families that are more easily identifiable or by using a morphologically distinguishable 'morphospecies'. As with other insects, communities of soil dwelling Diptera on more disturbed plots usually display a higher level of dominance of the most dominant species in comparison with more stable biotopes.
- Level of functional groups and food webs: The analysis of community structure and position in the food web can be very fruitful, especially when evaluating organic matter input manipulation. However, not only feeding demands may vary among species of one family and even within a species depending on environmental conditions and larval age, but also Diptera, particularly larvae, are frequently identified only to the order level. Thus, reliable data about food sources can be obtained only by gut content analysis.
- Adults of some Diptera need particular landscape elements to express their behaviours (swarming markers, breeding or resting sites, etc.) (Delettre *et al.,* 1992). Thus changes in population or community of soil dwelling Diptera may reflect modification of landscape elements which are necessary for adult behaviour. The necessity to consider various forms of interactions of larval and adult stages at the landscape level can complicate interpretation at the local level. Nevertheless, consideration of these interactions when evaluating functional changes in a landscape can be a very promising approach for future research.

NB. *All the information above, when not mentioned, comes from Frouz (1999).*
To conclude, several limitations appear at all the levels considered for using soil dwelling Diptera as indicators of biodiversity in agricultural systems. In particular, the ecological knowledge about them is still limited, and the information that exists is scattered between many miscellaneous literature and experts, and therefore quite inaccessible for common use. Ideally, this knowledge and information should first be gathered into an ecological database. Finally, even the identification of adults, a fortiori the one of larvae, seems to be a problem. This would imply dissecting the gut of larvae to get the content for analysis in order to be able to classify them into the different functional groups of the food web.

5.3.5.5. Collembola – Springtails

Gardi *et al. (*2002) used two indicators of biological soil quality (BSQ-ar, based on arthropods; and BSQ-c, based on Collembola species) to compare permanent grasslands of the Po valley to other agricultural land uses. The quality of soils was higher in permanent grasslands with respect to the arable lands, as evidenced by the standard soil quality indicators (organic carbon, aggregate stability) and confirmed by BSQ. This research proposes BSQ as a synthesizing soil quality indicator.

Rusek (1998) exposes several reasons why collembola are potentially good indicators of soil management regimes in agroecosystems: More than 6500 species of Collembola are known from throughout the world and these are only a small part of the still undescribed species. There are many checklists and catalogues of Collembola for smaller territories and entire continents. Biogeographical analyses have been made for some genera and smaller territories. The most serious problems for a global biogeographical analysis is the lack of enough records from immense territories of all continents. Local biodiversity of Collembola can be very high, reaching over 100 species in small mountain ranges. Sampling methods would allow the documention of Collembola biodiversity on a global scale. Collembola have well differentiated ecomorphological life-forms and feeding guilds which enable the functional role that Collembola play in ecosystems to be recognised in some degree. Collembola play an important role in plant litter decomposition processes and in forming soil microstructure. They are hosts of many parasitic Protozoa, Nematoda, Trematoda and pathogenic bacteria and in turn are attacked by different predators. They utilise as food Protozoa, Nematoda, Rotatoria, Enchytraeidae, invertebrate carrion, bacteria, fungi, algae, plant litter, live plant tissues, and some plant pathogens. Soil acidification, nitrogen supply, global climate change and intensive farming have greatly impacted collembolan diversity.

5.3.5.6. Isopoda – Woodlice

Woodlice (Arthropoda, Isopoda) are key soil detritivores, which have been shown to have potential as indicators of the sustainability of soil management practices. There are about 500 species in Europe (Armadilliidae: 244; Porcellionidae: 229), possibly 400 species in Italy (Paoletti and Hassall, 1999).

Woodlice provide information on functional aspects of decomposition processes showing clear reactions to tillage, to the supply of decaying organic materials and to pesticide input as well as to the concentration of heavy metals (Paoletti and Hassall, 1999). Büchs *et al. (*2003a) recorded Isopoda in fields only in considerable abundances when extensive farming or set-aside was applied. They are sensitive to pesticide applications, marked differences in density being found between conventional and organic cultivation regimes. Isopoda biomass is higher under no-tillage or minimum tillage regimes which leave crop residues near the soil surface. Isopods tolerate some heavy metals by accumulating them in vesicles in the hepatopancreas. They are thus, potentially useful for monitoring bio-accumulation of such contaminants and can serve as bioindicators of heavy metal pollution (Paoletti and Hassall, 1999).

5.3.5.7. Coleoptera, Carabidae - ground beetle adults and larvae

Carabids (Arthropoda, Coleoptera) are important soil predators that are easy to trap due to their mobility on the soil surface. They are widely distributed and are considered sensitive indicators of environmental conditions. There are about 2700 species in Europe and 1300 in Italy (Büchs, 2003a; Rainio and Niemelä, 2003).

The suitability of carabids as indicators of the ecological status of arable fields is widely accepted. However crop management intensity strongly influences the presence of carabids, selecting for the generalist ones and overriding the influence of soil types on the field carabid community (Büchs, 2003a). Some authors showed that *C. auratus* indicates extensive cultivation by increasing activity density and body weight. Beyond that, *Amara. similata*, *A. aenea*, *A. familiaris*, *Ophonus rufipes* and *Harpalus affinis* benefit mostly from organic agriculture and so they can be assumed as the most important ground beetle species able to characterise low input agro-ecosystems (Büchs, 2003a). Refer also to Section 5.4.2.2 for an analysis of carabids as idicators of species diversity.

5.4. WILD PLANT, ANIMAL AND FUNGAL DIVERSITY ON FARMLAND

The selected speciesdiversity indicators should also give as representative a picture as possible of organismal diversity as a whole. In this context, as arthropods make up about 65% of the species number of all multicellular organisms (Hammond, 1992) and probably make up even higher percentages in cultivated areas (Duelli, 1998), they potentially represent good candidates for biodiversity correlates. Examples of commonly used species groups in biodiversity monitoring schemes are vascular plants, birds and butterflies. These groups have basically been implemented as indicators because they score highly on many of the broad criteria defined for selecting indicator taxa (TABLE 5.5; Pearson, 1995) e.g., they are relatively easy to monitor, provide relevant information on general environmental conditions, include emblematic species, react quickly to environmental changes, and from this, large datasets are usually available. In the context of organic and low-input farming, however, indicators may be selected according to other or additional criteria. In addition, many new approaches and terms have been developed to refine the indicator species concept. These include focal species, umbrella species, flagship species, or guilds as indicators (e.g., Simberloff, 1997; Noss, 1999).

The list of desirable properties makes it possible to select individual indicators. However, because none of the individual indicators will *a priori* possess all these properties, a set of complementary indicators is required. It will be appropriate to select a representative 'shopping basket' of taxa that together serve as a composite focal group (sensu Hammond, 1995). Indeed, it has been shown that (i) indicator taxa comprising a greater number of species tend to perform better than indicator taxa with fewer species, (ii) most indicator taxa perform worse than indicator groups consisting of a comparable number of species selected among all taxa, and (iii) it is difficult to predict which taxa are efficient biodiversity indicators when selected only on distributional properties such as mean range size (Larsen, 2009). Selected indicators should represent multiple levels of biological organization, spatial and temporal scales, in particular of the habitat use and of the food resources.

TABLE 5.5. PROPERTIES FOR THE SELECTION OF SPECIES DIVERSITY INDICATORS (ADAPTED FROM PEARSON, 1995; STORK, 1995).

Well known taxonomy and easy identification
Biology and life history well understood
At higher taxonomic levels (order, family, etc.) occurrence over a broad geographical range and breadth of habitat types so that results will be broadly applicable
At lower taxonomic levels (species), specialization of each population within a narrow habitat to detect habitat change
Populations readily surveyed
Some evidence that pattern observed in the indicator taxon are reflected in other taxa
Large random samples encompassing all species variation are possible
Predictable, rapid, sensitive, analysable and linear response to disturbance
High taxonomic and ecological diversity (many species in each locale or system)
Potential economic importance of some populations (agricultural relevance)

Concerning the properties of the indicators, two kinds of criteria should be taken into account in addition (developed and illustrated with indicators in TABLE 5.6):

- Criteria related to the physical compartments of the agricultural activity where candidate indicators are occurring, i.e., the macro- and microhabitats, and the spatial dimension needed by indicators to complete their life cycle;
- Criteria related to the function of the candidate indicators within the agro-ecosystem, the trophic level.

The use of these criteria allows representation of the impact on the different niches of agro-ecosystems affected by the agricultural activities and to take into account functional aspects of biodiversity.

5.4.1. SPATIAL SCALE

Spatial requirement: species require different conditions to complete their life cycle that may be spatially distributed at different scales. In searching indicators, it is important to define at which spatial scale which indicator will be asked to indicate the impact of agricultural activities. In this context, we may distinguish between three categories of an indicator's spatial requirement:

- a small plot = a few cm^2 to a few m^2
- a combination of habitats = a few m^2 to a few hectares
- a landscape = a few hectares to a few km^2.

These equate to the levels of field plot, complex of habitats and landscape presented in TABLE 5.6. Species indicators might be surrogated by indirect indicators in any physical compartment and at any scale, e.g., land use intensity of a region for species requiring a few hectares and many macro-habitats, soil tillage intensity for below ground species, pesticide application and fertilization intensity for above ground species, etc.

5.4.2. FUNCTION

Trophic level indicates the position of an organism in the food-web. The trophic level determines an important part of the function in the agro-ecosystem. Ecosystem services derived from ecological functions (which are the utilitarian human interpretation of them) are natural processes acting within and among ecosystems including natural- and agro-ecosystems. They are of particular importance since they include biological control of pests, pollination and decomposition processes beside the crop production itself (Le Roux *et al.*, 2008). It is recognized that simplification of agro-ecosystems caused by intensification of agricultural practices will affect important ecosystem services via the loss of biodiversity (Le Roux *et al.*, 2008). In organic/low-input farming, services may be preserved by particular management practices and this has to be investigated with appropriate indicators including beneficial organisms for pest control such as predatory and parasitoid arthropods, pollinators such as wild bees and decomposers such as oribatid mites in soil.

TABLE 5.6. INDICATOR SPECIES GROUPS RELEVANT TO AGRO-ECOSYSTEMS, MACRO- AND MICROHABITATS, SPATIAL SCALES, TROPHIC LEVELS AND ECOSYSTEM SERVICES.

			Indicators	Grassland and woody habitat flora[1]	Crop weeds	Birds	Small Mammals	Amphibians	Snails and Slugs[3]	Spiders	Carabid beetles	Butterflies	Grasshoppers	Bees and Bunble bees
Macro- and micro-habitat:	Agro-ecosystem category	Grassland		x		x	x	x	x	x	x	x	x	x
		Crops			x	x	x	(x)	x	x	x			x
		Special crops[2]		x	x	x	x	(x)	x	x	x	x	x	x
		Semi-natural habitats		x	x	x	x	x	x	x	x	x	x	x
	Habitat	hypogeous					x		x		x			
		epigeous	Litter			x	x	x	x	x	x		x	x
			Herb layer	x	x	x	x	x	x	x		x	x	x
			Shrub layer and trees	x		x	x	(x)	x	x		x	(x)	
		Water bodies				(x)		x	x					
Scale:	Life area (spatial requirement)		Field scale	x	x									
			Complex of habitats						x	x	x	x	x	x
			Landscape			x	x	x						
Function:	Trophic level and ecosystem service		Primary producer	x	x									
			Detritivore						x					
			Herbivore			x	x		x		x	x	x	x
			Pollinator									x		x
			Predator			x	x	x	(x)	x	x		(x)	
			Parasitoid											
Correlation with the overall species richness				sig.[3]		-	-	-	-	sig.	n.sig.[3]	sig.	-	sig.

FOOTNOTE: X: Illustrates habitat indicators which are mainly sensitive to ecosystem conditions (habitats), at which scale they may indicate changes in ecosystem conditions (scale) and the kind of ecosystem services that they provide (function). 1 hedgerows, groves and trees; 2 orchards, vineyards and vegetables; 3 sig. = significant correlation ad n. sig. = non-significant correlation with overall biodiversity, after Duelli and Obrist (1998).

Regarding the diversity of species, whilst authors have shown that some species groups may serve as surrogates for the whole biodiversity (Heteroptera, plants; Duelli, 1998) in certain circumstances, many studies revealed poor correlations between species richness in one taxonomic group and species richness in other groups (e.g., Billeter, 2008; Gaston, 1996; Lawton, 1998; Wolters, 2006). However, Larsen *et al.* (2007) have shown that selecting indicator groups containing threatened, endemic or range-restricted species, improves bioindication effectiveness of the groups, especially towards other threatened and range-restricted species. In this context, the approach of Duelli and Obrist (1998) is still very useful because they have considered the effort for sorting and identification beside the correlation coefficient of single indicator groups to the overall number of species.

In the specific case of biodiversity indicators for organic and low input farming (compared to the conventional farming baseline), indicators should indicate the impact of current management operations that characterize these farming systems on biodiversity. Obviously, these indicators have first to be largely distributed in the cultivated landscape. A full list of proposed candidate biodiversity indicators is given in TABLE 5.7.

TABLE 5.7. DIRECT INDICATORS OF WILD PLANT, ANIMAL AND FUNGAL DIVERSITY ON FARMLAND

Species indicators	References	Indicator characteristics and suitability
TERRESTRIAL PLANT SPECIES		
Flowering plants of semi-natural habitats	Sauberer *et al.*, 2008;	Appropriate indicator for biodiversity in agriculturally dominated landscapes of eastern Austria
	Wittig *et al.*, 2006;	Interest in selected indicator species as indicators/surrogates of the total species richness of vascular plants and of the number of endangered species.
	Manhoudt *et al.*, 2005	Potentially high interest in this plant sampling method: minimum sampling area determined for different types of habitats, allowing to compare the species richness of non cultivated versus cultivated habitats, both for conventional and non conventional farming systems. Indicator threshold values are defined for conventional farms.
Vascular plant species on 25m2 plots according to Braun-Blanquet (1964)	Kampmann *et al.*, 2008	Pastures were the mountain grassland management type with the highest plant diversity
Flowering plants of cultivated forage and food crops	Mäder *et al.*, 2002; Noe *et al.*, 2005; Aavik and Liira, 2009;	Increased by organic farming. Weed cover in cereal fields' used as an indicator of floral and faunal diversity. High interest in functional groups (agrotolerant and nature-value species) instead of global species richness: the distinction between agrotolerant and nature-value plant species, and the estimation of habitat structure would increase the effectiveness of biodiversity monitoring in agricultural landscapes in comparison with classical methodology based on the assessment of total plant species richness.
	Hyvonen and Huusela-Veistola, 2008;	Interest in trophic interactions between 25 common arable weeds and individual groups of farmland birds, pollinators (wild bees), phytophagous insects and insect pests. Each weed species was weighted based on the number of reported linkages with each animal group. The results suggest that the ecological consequences of changes in the intensity of agriculture can be explored with the aid of a biodiversity indicator based on species interactions.
	Suarez *et al.*, 2001	Interest in weed community characteristics as indicators of soil degradation levels.

Cryptogams	Sauberer *et al.,* 2008	An appropriate indicator for biodiversity in agriculturally dominated landscapes of eastern Austria.
Plant functional types	Noe *et al.,* 2005	Based on Grimes classifications (stress tolerant, ruderal, competitor, high conservation value).
EPIGEAL INSECTS	Longcore, 2003	Interest in terrestrial arthropods as indicators of ecological restoration since they seem more sensitive (through their diversity) than vegetation.
Auchinorrhyncha (Hemiptera) – Planthoppers	Nickel and Hildebrandt, 2003	High interest of Auchenorrhyncha as indicators of biotic conditions in grasslands.
Coleoptera, Carabidae - Ground beetles	Rainio and Niemela, 2003;	Theoretical interest in carabids as indicators but as crucial understanding of their relationship with other species is incomplete; they should be used with caution.
	Irmler, 2003;	Low interest in ground beetles as indicators of agricultural characteristics because they mainly responded to yearly climate conditions.
	Sauberer *et al.,* 2008	In combination with Mollusca an appropriate indicator for biodiversity in agriculturally dominated landscapes of eastern Austria.
Coleoptera, Coccinellidae (Ladybird beetles)	Iperti, 1999	Interest in coccinellid life traits for bioindication of agricultural landscapes.
Coleoptera – pollinators	Westerkamp and Gottsberger, 2000; Rathcke and Jules, 1993; Shuler *et al.,* 2005	Wild pollinators are sensitive to local and landscape floral resources, habitat fragmentation and farming practices.
Coleoptera, Scarabidoidea – dung beetles	Davis *et al.,* 2004	Interest in Scarabaeine dung beetles as indicators of biodiversity, habitat transformation and pest control chemicals in agro-ecosystems. They use these dung beetles as biodiversity, ecological or environmental indicators at each of three spatial scales: regional, local, and pasture.
Coleoptera, Staphylinidae - rove	Bohac, 1999	Interest in Staphylinids as indicators (ecological features highlighted, but practical features seem still to be improved).

beetles		
Diptera – pollinators	Speight, in press	There is substantial information on flower visitation by flies, although knowledge is still patchy regarding whether particular fly species collect pollen or nectar, whether both sexes visit flowers, etc.
Hymenoptera, bees and wasps	Anderson *et al.,* 2005;	Interest in parasitic wasps (15 families and 75 genera of Hymen. Parasitica, compared with 15 species of spider (Araneae), 16 species of true bugs (Hemiptera), 72 species of beetles (Coleoptera), 25 families of flies (Diptera) to be used as indicators of grassland management and surrogates of grassland arthropods.
	Tscharntke *et al.,* 1998	High interest in trap-nesting bees and wasps, and their natural enemies, as indicators of habitat quality, because they represent different trophic levels and their interactions.
Hymenoptera, Formicidae - ants	Lobry de Bruin, 1999 ;	Low interest in ants as indicators of soil function in rural environments since there is still a need for specific experiments to test the hypothesis that ants can be used as indicators of soil quality.
	Sauberer *et al.,* 2008	In combination with Mollusca an appropriate indicator for biodiversity in agriculturally dominated landscapes of eastern Austria.
Lepidoptera – butterflies	Rundlöf *et al.,* 2008 ; Wenzel *et al.,* 2006	Butterfly species richness and abundance were significantly increased by organic farming. Classification into different ecological groups according their general habitat requirements, requiring a structured landscape, dispersal ability, space demand, population density trophic range of caterpillars, etc. Twenty-one butterfly and four burnet species found in 1972 not observed in 2001, two butterfly species of 2001 not found in 1972.
Lepidoptera – moths	Taylor, 1986; Woiwod and Stewart, 1990; Littlewood, 2008	The wide range of caterpillar host plants and the ease of identification of adult moths collected by light traps provides a strong case for using moths as an indicator for farmland under different management systems.
Orthoptera, Acrididae – grasshoppers	Kampmann *et al.,* 2008; Sauberer *et al.,* 2008	Pastures were the mountain grassland management type with the highest plant and grasshopper species richness (100m2 plots, plots were covered by area transects).

Syrphidae (Diptera) – Hoverflies	Burgio and Sommaggio, 2007; Sommaggio, 1999; Speight, 2008	High interest in Syrphids as indicators of habitats richness and integrity in agricultural landscapes. Theoretical high interest in Syrph life traits for bioindication of habitats richness and integrity in agricultural landscapes High interest in Syrphids as indicators of habitats richness and integrity in agricultural landscapes.
OTHER INVERTEBRATES		
Araneae –spiders	Sauberer *et al.,* 2008 ; Marc *et al.,* 1999	Indicator for biodiversity in agriculturally dominated landscapes of eastern Austria. High interest in spiders as indicators of habitat quality as well as beneficial predators in agroecosystems.
Mollusca, Gastropoda – land snails	Sauberer *et al.,* 2008	Indicator for biodiversity in agriculturally dominated landscapes of eastern Austria
VERTEBRATES		
Birds	Sauberer *et al.,* 2008	Appropriate indicator for biodiversity in agriculturally dominated landscapes of eastern Austria.
Large mammals		
Small mammals	Bates and Harris, 2009	Hedgerow size no effect on species richness or species diversity, no significant interaction between hedgerow size and farm type on either abundance or diversity of small mammals. (Longworth trapping, all captures were identified to species, weighed, sexed, aged)
Chiroptera - bats		
OTHER SPECIES CATEGORIES		
Taxonomic groups of special conservation concern	Larsen *et al.,* 2007	High interest in indicator groups with special status species (threatened, endemic...) which improve bioindication properties, especially for other species groups also with special status.

Top predators	Roth and Weber, 2008	Low interest in top predators as indicators/surrogates since relationships among higher taxa are complex and depend on the species group and the scale of analysis. They recommend the use of more than one indicator species from different taxonomic groups when identifying areas of high biodiversity.
Functional arthropods in agroecosystems	Paoletti *et al.*, 1999	Interest in functional arthropod groups in identifying the role of different rural landscape units (or mosaic rural landscape) for creating an ecologically favorable agricultural environment.
Indicator groups	Larsen *et al.*, 2009	High interest in indicator groups: (i) high number of species outperform indicator groups with few species, (ii) the majority of indicator taxa are less effective than groups with the same number of species belonging to different taxa, (iii) effectiveness of indicator taxa correlates poorly with selected distributional properties such as mean range size of the indicator taxa.
AQUATIC ORGANISMS OF STANDING AND RUNNING WATER ON FARMLAND		
Odonata	Foote and Hornung, 2005	High interest in larval dragonfly and damselfly community as indicator of intactness and as surrogate of overall aquatic macro-invertebrate community.

5.4.1. TERRESTRIAL PLANT SPECIES

Terrestrial plant species include several species groups known to use various cultivated and non-cultivated habitats. The choice of the taxonomic level that will be recorded to monitor the impact of change in agricultural practice will be determined by the properties of the environmental factors associated with the change in management practice and the taxa that are exposed to their effects. Invertebrates have notably been used at the level of trophic group to study impact of agricultural management on biodiversity (Hawes *et al.,* 2003). Pragmatic, logistic and financial considerations have to be taken into account too, as the difficulties or time consummation linked to taxonomic identification. In the great majority of following cases, the metric indicators will be the abundance of species or group of species. Another common approach is to use a diversity index which is independent of sample size and combines species richness and the evenness of their abundances, as Shannon-Wiener index H (Storkey *et al.,* 2008).

5.4.1.1. Flowering plants of semi-natural habitats

There are many arguments for using flowering plants (angiosperms) as indicators. These primary producers dominate most terrestrial ecosystems, shaping our physical environment and forming the basis of food chains. They constitute an important part of agricultural landscape biodiversity and provide food, shelter, breeding sites, refuges, etc. for a wide range of other organisms. Most mammals, birds, invertebrates and insects are directly or indirectly dependent on one or more species of flowering plants and diversity of flowering plants may therefore indicate diversity of other organisms. Since flowering plants are dependent on particular conditions for growth, each species can indicate the occurrence of a specific set of environmental conditions, such that changes in the environment are reflected in changes in plant abundance or distribution. Compared to many species groups, there is a relatively high level of knowledge about flowering plants, regarding their taxonomy, habitat requirements, present distribution and trends of change. There is a vast and accessible literature on flowering plants in most countries, and there are many people, both professionals and amateurs, who are experienced in species identification. Plants can be identified for a large part of the growing season and are virtually sessile, therefore easier to sample than animals and with greater replicability of results. Finally, many flowering plants are attractive and easily appreciated by the general public, a desirable characteristic of an indicator from a stakeholder point of view.

Whilst most criteria for indicators are clearly fulfilled by flowering plants, demands for cost effectiveness may nevertheless limit their use as indicators. Field sampling is costly, yet large species variation means that many samples are needed to provide reliable indications of status and changes. Cost effectiveness can be increased by focusing on certain species or functional groups, or on certain habitats (e.g., field margins, hedgerows, or specific types of grassland). Appropriate methodology and sampling strategy are essential to ensure reliable results and avoid problems of misinterpretation.

There are a number of issues of scale that must be taken into account when using species as biodiversity indicators. Firstly, species richness at any given geographical location is limited by the species pool (sensu Zobel *et al.,* 1998). This can complicate interpretation of data (Wilson and Anderson, 2001; Grace, 2001), and particularly comparisons across sites at regional and European scales. Similarly, some rare habitat types may be species poor, but nevertheless contribute to the total biodiversity at a regional, national or European level (Duelli and Obrist, 2003). Temporal scale is also important, because although some plant species may respond rapidly to changes in land management, many species respond slowly to, for example, cessation of traditional management. This time lag may mean that present day species composition reflects historical rather than current land use (Auestad *et al.,* 2008), again complicating interpretation.

There are some plant species that may function as indicators over a large geographical range. For example, *Anemone nemorosa* and *Polygala vulgaris* have been confirmed as indicators of old grasslands in Norway (Austrheim *et al.,* 1999), Germany (Waldhardt and Otte, 2003; Waesch and Becker, 2009), and Sweden (Gustavsson *et al.,* 2007). These studies and others have shown that old grasslands tend to be more species rich and contain more species of high conservation value than newer grasslands, such that the presence of these species may be a good indicator of biodiversity in agricultural landscapes. Generally, however, the indicator value of individual species varies with geographic location and habitat type and cannot be easily transferred from one site to another. Even if single species consistently indicated a specific set of conditions, they can have only limited use as biodiversity indicators, since only a small sub-set of the total biodiversity can be expected to share the same habitat requirements.

A more general approach is to use the total number of plant species as an indicator of overall biodiversity. Duelli and Obrist (1998) concluded that flowering plants made excellent correlates of overall biodiversity, taking into account both correlation coefficient, but also the relatively low sampling effort involved compared with invertebrate taxa. Sauberer *et al. (*2004) also found vascular plants to have the highest correlations with overall species richness. Other studies (Kremen, 1992; Prendergast *et al.,* 1993; Prendergast, 1997; Hooper *et al.,* 2000; Billeter *et al.,* 2008) have found poor correlations between flowering plants and other taxa. This discrepancy may be due to issues of scale (Pearson and Carrol, 1999; Hooper *et al.,* 2000; Økland *et al.,* 2006; Auestad *et al.,* 2008), either due to the relative importance of different factors in determining species numbers at different scales or simply due to methodological details such as number of observations. Prendergast (1997) points out that there is little reason to expect covariance of entire taxa or taxonomically defined sub-groups, but that subsets of taxa of the same habitat type are more likely to be correlated.

Thus, whilst single plant species may be too specific in their habitat demands to use as meaningful biodiversity indicators, and total numbers of plant species may be too general, functional groups could provide an appropriate intermediate level. The idea of plant functional groups is not new (Raunkiær, 1907), however it has been considerably refined in recent years and has been applied in the context of agricultural landscapes and in pan-European studies (Liira *et al.,* 2008). A wide range of functional traits of flowering plant species has been documented (Ekstam and Forshed, 1992; Ellenberg *et al.,* 1992; Noble and Gitay, 1996; Grime *et al.,* 1997; Hill *et al.,* 2002; Lososova *et al.,* 2006; Liira *et al.,* 2008).

Although functional groups may provide more robust indicators than using any single indicator species, functional groups are also context specific. The traits that are relevant for functional responses may be different in different regions, due to different climates, evolutionary histories and management (Lavorel *et al.,* 1997). Currently most research using functional groups has focused on understanding and predicting how plant species respond to their environment. Liira *et al. (*2008) found that "various functional traits can be combined into an emergent group of nature quality indicator species, and that such a group has the highest prognostic power to describe the status of conditions for biodiversity in the agricultural landscape" (Liira *et al.,* 2008, p.11). It should also be noted that, although any single group may fail to serve as a surrogate for total biodiversity, we can probably select a set of taxa with different ecological requirements to indicate different aspects of biodiversity (Ricketts *et al.,* 1999; WallisDeVries *et al.,* 2002).

5.4.1.2. Flowering plants of cultivated forage and food crops (including 'arable weeds')

Historically, more than 7000 plant species were cultivated or collected for human food (Wilson, 1992) but only around 150 plants are under extensive cultivation today (FAO, 1997). Although multi-cropping systems and agro-forestry support more biodiversity than monocultures, crop species are nevertheless unsuitable as indicators of wider diversity due to the great variation in management intensity that can occur in systems with identical crops. To assess developments in the biodiversity of the crop plants themselves, numbers of species can be recorded, as well as analyses at the level of genetic diversity e.g., numbers of landraces (see section 3).

A more promising candidate biodiversity indicator within the cultivated area may be the occurrence of weeds (Albrecht, 2003). Albrecht (2003) suggests that weeds are key species in arable fields, with strong correlations to total species diversity, and points out that the high percentages of dormant seeds produced by weeds means that this group can indicate long-term management conditions, avoiding problems due to the extreme variation found within arable fields during the growing season. To increase indicator sensitivity, Albrecht (2003) suggests excluding ubiquitous generalist weeds and noxious perennials that often occur outside fields, focusing on those weed species that have their main habitat in the fields ("characteristic species"). In many countries, such weeds have declined considerably in recent decades (Ture and Bocuk, 2008; Van Elsen *et al.,* 2006; Van Calster *et al.,* 2008). Braband and Van Elsen (2006) describe a method that farmers can carry out to record easily identifiable target arable weed species and report good correlation between occurrence of the selected species and total arable plant species in grain fields.

5.4.1.3. Cryptogams

Bryophytes include mosses (phylum Bryophyta), liverworts (phylum Marchantiophyta) and hornworts (phylum Anthocerotophyta). As for all "lower plants", the bryophytes lack roots and a vascular system for taking up and transporting water and nutrients. This limits their capacity to compete with vascular plants in many habitats, although they can dominate in certain environments where vascular plants do not thrive or in early successional stages where there is little competition from other plants (Hassel 2004).

By regularly exposing bare soil, agriculture can create suitable habitats for bryophytes and may historically have contributed to increased abundance of some species (Porley, 2000). In central and northern Europe, some species may be found almost exclusively on cultivated land (Porley, 2000; Bisang, 1992) and agricultural habitats may be important for the conservation of some threatened species. In Norway, for example, 20% of mosses on the Norwegian red list are associated with cultural landscapes (Hassel, 2004). In Austria, a study of agricultural landscapes recorded a total of 506 bryophyte species, of which 135 were endangered (Zechmeister *et al.,* 2002).

Both total bryophyte species number and number of threatened species have been shown to be significantly higher in habitats and landscapes where land use intensity is low (Zechmeister and Moser, 2001). Intensification of agriculture in Europe is thought to have lead to declines for many bryophytes (Porley, 2000). Bryophyte occurrences have been shown to be closely linked to crop type and the timing of cultivation and harvesting, with untreated stubble-fields providing the best habitat (Bisang, 1998; Porley, 2000; Hassel, 2004). Bisang *et al.* (2009) pointed out that soil conservation directives of agri-environmental schemes in Switzerland have a negative impact on bryophytes by reducing the availability of autumn stubble. Little is known about the effects of organic agriculture on bryophytes (Porley, 2000).

Zechmeister *et al.* (2002) reported that moderately and less intensively used meadows (including fens), low intensity vineyards, field margins and fallow lands were important habitats for endangered bryophytes. They found a significantly higher number of endangered species in upland landscapes dominated by moderately intensive cattle farming compared with lowland landscapes with a wide range of mainly intensive farming styles. The percentage of species that were endangered was higher in intensively than in moderately used areas (Zechmeister *et al.,* 2002).

Several studies (Pharo *et al.,* 1999; Löbel *et al.,* 2006; Santi *et al.,* 2009) have shown that, at a general level, bryophyte species diversity is poorly correlated with the species diversity of other taxa such as vascular plants, birds or arthropods. Closer correlations seem probable at a more specific habitat level, since availability of stubble-fields is important both for other groups such as arable weeds (Pinke and Pal, 2009) and granivorous birds (Perkins *et al.,* 2008).

Some bryophytes may be able to persist for many years, even decades, as diaspores or tubers in the soil, germinating if conditions should become suitable (During and Horst, 1983; Porley, 2000; Hock *et al.,* 2008). This may mean that only above-ground populations can reliably indicate current farming conditions. This long-term survival in the soil may mean that bryophytes are insufficiently sensitive to management changes to be used as indicators for wider biodiversity.

Being relatively inconspicuous, bryophytes have often been overlooked in agricultural habitats and, although there has been considerable progress in recent years, there is still a considerable lack of knowledge about this group (Bisang, 1998; Hock *et al.,* 2008). In addition, the lack of emblematic species or popular appeal may make bryophytes less acceptable to stakeholders. Although bryophytes have been used very successfully as indicators for monitoring air pollution (Krommer *et al.,* 2007), they may not be the best choice as European indicators of biodiversity on farmland at the current time.

There are relatively few species of ferns associated with agriculture in Europe. One very dominant species, bracken (*Pteridium aquilinum*), is an invasive species in areas of marginal, low intensity grassland and heathland, and could be considered an indicator of a lack of agricultural biodiversity. A number of *Equisetum* species are associated with disturbed soil. Their presence may indicate a certain type of agricultural habitat, but they are not particularly relevant as indicators of wider biodiversity. Several rare *Botrychium* ferns are associated with species-rich grasslands. Being low-growing and poorly competitive, they are sensitive to soil fertilisation and agricultural abandonment, and they have insufficient time to set spores in grasslands that are cut early. *Botrychium matricariifolium* is one of the best studied species (e.g., Marttila *et al.,* 1999; Muller, 1992; 1999; 2002). However, it has been shown to be sensitive to spring drought, which somewhat reduces its suitability as an indicator since absence from vegetation may thus reflect weather rather than management practices. *Ophioglossum vulgatum* is another rare fern that disappears from meadows following fertilization. However, this species does not decline with land abandonment (Muller, 2002).

5.4.2. EPIGEAL INSECTS

5.4.2.1. Auchinorrhyncha (Hemiptera) – Planthoppers

The froghoppers (Cercopidae), planthoppers (Delphacidae) and leaf hoppers (Cicadellidae and related families) of Auchenorrhyncha are a group of insects that are widespread across Europe (Eyre, 2005) and make an important contribution to the arthropod diversity of many habitats of farmland (Biedermann *et al.,* 2005). Hollier *et al. (*2005) advocated the use of species composition of Auchenorrhyncha as an ecological indicator because this measure is sensitive both to plant species composition and architecture. Auchenorrhyncha can be sampled with standard sampling methods and collected in reasonable abundance (Gibson *et al.,* 1992). Crucially, annual variability of Auchenorrhyncha is small in comparison with variability in response to vegetation composition and geographical location. Froghopper species in particular have been reasonably well studied and the distribution of these species in relation to altitude and latitude (Whittaker, 1971; Whittaker and Tribe, 1996) has been investigated and these results used to develop models predicting their extent at zoogeographical scales under different climatic conditions (Masters *et al.,* 1998; Whittaker and Tribe, 1998).

5.4.2.2. Coleoptera, Carabidae - Ground beetles

Carabid beetles have traditionally been studied in agro-ecosystems (synthesis in Holland, 2002a), mainly because they are predators and therefore play a role as biocontrol agent against agricultural pests (e.g., Kromp, 1999), although ecological investigations demonstrated their polyphagy (Luff, 1987). Carabid beetles occur in all kinds of cultivated fields and farming systems (Holland, 2002b) and are influenced by them (e.g., Schreiter, 2001).

Carabid beetles principally live on the ground (adults) or in the first soil centimetres (adults and larvae). Assemblages are determined by farming operations that change abiotic conditions (temperature, humidity and light). In crop fields, a basic assemblage of about 30 species varies according to regional conditions (Luff, 2002). Farming operation such as soil tillage influences the carabid beetles and no tillage systems have shown the highest species diversity (Kromp, 1999). In addition, the crop rotation favours or decreases the species richness according to the crop type and the period of occurrence (Holland *et al.,* 2002). Insecticides and fungicides applied in conventional farming systems have detrimental effects either directly or by reduction of the female fertility (e.g., Thiele, 1977; Basedow, 1990). Furthermore, herbicide applications reduces the weed flora resulting in an impoverishment of the carabid fauna due to unsuitable micro-climatic conditions (Kromp, 1999). In grasslands, intensive grazing or mowing activities modify the soil structure and humidity with negative impact on carabid diversity (e.g., Tietze 1985; Eyre *et al.,* 1989). More recently, investigations showed that the management and the land-use diversity in the surroundings of fields influenced carabid assemblages in grasslands but not in cereals (Batary *et al.,* 2008). Semi-natural habitats and field margins in the agricultural landscape have positive effects on the carabid fauna (e.g., Lys and Nentwig, 1992; Aviron *et al.,* 2009). Organic farming systems usually harbour higher species number and individual abundance than conventional ones (e.g., Kromp, 1989; Pfiffner and Luka, 2003).

As ground dwelling arthropods, carabid beetles have mostly been captured with pitfall traps. However, they can also be collected by hand searching, Tullgren funnel (species inhabiting the soil) and suction samplers. Methods to optimize the collecting procedure with pitfall traps have been developed so that spring and summer carabid diversity can cost-efficiently be estimated (Duelli *et al.,* 1999). Carabid beetles can be collected with pitfalls (together with spiders for instance) by technician without particular knowledge (in contrast to e.g. butterflies that have to be identified during field observation). Disadvantages are that pitfall traps do not measure the carabid abundance since carabids are not sampled within a defined area (activity density). After collection, they have to be identified by taxonomists. The modern taxonomy of European carabid beetles is well known (Freude, 1976) and subject to only minor changes that do not influence their reliable identification.

5.4.2.3. Coleoptera – pollinators

Wild pollinators are sensitive to local and landscape floral resources (Westerkamp and Gottsberger, 2000), to habitat fragmentation (Rathcke and Jules, 1993) and to farming practices (Shuler *et al.,* 2005). Moreover, their populations can enhance production of some crops and are, in this way, an important natural resource (Klein *et al.,* 2006). Recently new agri-environment schemes have been proved to enhance pollinator diversity in nearby intensively managed farmland (Albrecht *et al.,* 2006). Wild pollinator (including many Hymenoptera species and some Diptera species) abundance and richness are therefore potentially very good indicators of local habitat quality and of farm practices. See 5.4.2.5 (Diptera) and 5.4.2.6 (Hymenoptera) for further discussionis of pollinators.

5.4.2.4. Coleoptera, Staphylinidae - rove beetles

The family Staphylinidae is one of the largest families of beetles, with about 32,000 known species. The family is distributed worldwide and is found in practically all types of ecosystems, in which it shows a strong ecological specialization and generally strong dispersal abilities. About half of the staphylinid species are found in litter, forming one of the most common and ecologically important insect components of the soil fauna. Knowledge of the broad habitat requirements of common staphylinid species and the fact that the family is distributed in practically all semi-natural and man-made habitats are two features that make staphylinids attractive as potential indicators. In spite of this, staphylinids are used less often in bioindicative studies compared with ground beetles, primarily because of the practical difficulties associated with staphylinid taxonomy. Staphylinid adults are usually easily distinguished from other beetles by their short truncate elytra, which leave more than half of the rather flexible abdomen exposed (FIG. 5.3).

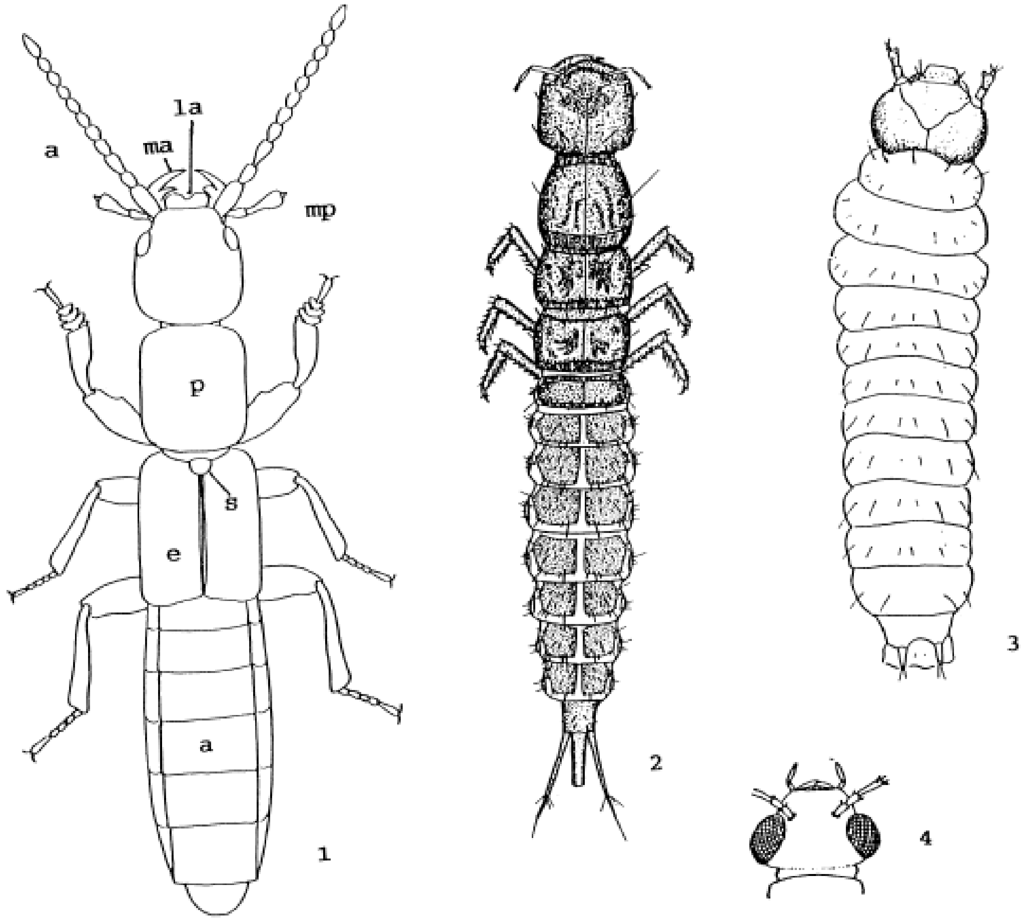

FIGURE 5.3. ADULT AND LARVAL MORPHOLOGY OF STAPHYLINIDAE (SOURCE: Bohac, 1999).

Larvae of staphylinid beetles have been poorly studied despite the fact that they are a relatively common component of the soil fauna. Most staphylinid larvae can be distinguished from most other beetle larvae by the presence of a pair of articulated appendages (urogomphi) at the apex of the ninth abdominal tergum (FIG. 5.3, 2-3). The trophic groups of staphylinids serve as the basis for the hierarchic classification of their life forms which is used in monitoring:

- Zoophagous (predators and parasitoids)
- Mycetophagous
- Saprophagous
- Phytophagous
- Myrmecophilous

The majority of staphylinids are known as non-specific predators, active mainly during the day and feeding on various soil arthropods such as nematodes, mites, Collembola, small insect imagos and larvae, etc. Few predator species are specialised on ants and termites, and the genus Aleochara is specialised in parasitizing fly puparia. Other Staphylinids feed on various organic substances, or algae, pollen, mycelium, and most often there is a feeding specialization among sub-families or rather genera. An investigation of the life form spectrum of staphylinid communities of 155 biotopes showed that the number of life forms can vary from 4 (sandlands) to 11 (cultivated meadows). The greatest variety of life forms was found in staphylinid communities living in natural or semi-natural ecosystems (forest, steppe, non regulated riversides and brooksides, subalpine meadows, pond borders).

Communities of staphylinids can be used as indicators of the environmental status and particularly of human influence on ecosystems. Staphylinids can be collected by pitfall trapping or by taking soil quadrat samples. Bohac (1990) developed an index of staphylinid communities for the evaluation of the degree of human influence on ecosystems which is calculated on the basis of dividing beetles into ecological groups according to their relation to the naturalness of biotopes. The value of this index ranges from 0 (only eurytopic species are present and the community is highly affected by man) to 100 (only species of undisturbed, climax communities are present). Ecological analysis for evaluation of community structure was employed in a study of beetle communities in biotopes with different degrees of anthropogenic effects (Bohac and Fuchs, 1991).

5.4.2.4.1. Staphylinid assemblages in farming areas

Staphylinids are the second most important group of epigeic invertebrates in agricultural landscapes in terms of activity and abundance. They represent about 19% of all beetles in terms of number of individuals. The number of staphylinid species is often higher than that of carabids and in some biotopes staphylinid abundance can be 15 times greater than that of carabid specimens. Staphylinids are important predators of some pests e.g., aphids, caterpillars, wire worms and other invertebrates. In central and western Europe, the staphylinid fauna of fields is strongly influenced by the surrounding biotopes. Generally, the number of staphylinid species in fields increases from one-year cultures to cultures growing for several consecutive years. Agricultural measures (tillage, manure, chemical NPK and pesticides) have a lower and more short-term influence on staphylinid communities compared with other factors such as relief of agricultural landscape, surrounding biotopes, soil humidity and crop change. The change of crop from wheat to maize was shown to influence the dominance of staphylinid species in communities. The higher humidity of the soil in the maize field facilitated its colonization by hygrophilous staphylinid species from the surrounding biotopes. Observations made to date do not indicate a negative influence of tillage on staphylinid beetles living in fields. On the other hand, shortly after tillage the activity of staphylinid imagos and larvae was found to decrease by approximately 20-fold in comparison with their activity before tillage. Manure and NPK fertilizers influence staphylinid community structure: their abundance was stimulated by manure; this effect was also observed for other predators (Carabidae, Chilopoda) and reflected an increase in the quantity of their prey. The manure also raised the soil moisture content and therefore produced an increase in the number of hygrophilous species. The lowest number of staphylinid species was found on the plot with the highest dose of NPK and a given species was found only on plots without fertilizers. Insecticides negatively influence staphylinid communities but their effects depended on the vegetation cover and presence of litter. In contrast, herbicide treatment did not appear to influence activity of staphylinid beetles in a field experiment.

To conclude, in some cases staphylinids are more suitable and sensitive indicators than carabid beetles, but their importance for monitoring is currently limited because of difficulties in their identification. Furthermore, many species are not easily found using quantitative sampling methods (pitfall trap, soil samples). Future refinements in identification and sampling methods and additional information regarding the interaction of staphylinids with other insects and their environment should result in their increased use as indicators of environmental quality. Nevertheless, this family presents very many interesting biological and ecological features that are also useful in indicating environmental conditions, and may therefore be worth testing in BioBio.

NB: unless otherwise mentioned, all the information above comes from Bohac (1999).

5.4.2.5. Diptera – pollinators

Some flies will visit flowers primarily for their pollen, others will take both pollen and nectar and others nectar alone. Flower pollination may not be the objective of flower visits by flies, but is frequently one consequence. Flies are increasingly being recognised as performing a significant "ecological service" through their pollinating activities in crops and orchards. Wind pollinated flowers do not produce nectar and are only visited by pollen feeding flies; female flies consume pollen more often than do males, because it is a source of protein needed for egg maturation. Pollen can also be a useful energy source for male flies, because it is rich in carbohydrates, lipids, vitamins and minerals. However nectar, which is composed almost entirely of sugars in solution, is even more attractive to them, so both sexes are equally attracted to flowers which carry pollen and nectar. Since pollen is difficult to digest, its food content is inaccessible to many would-be competitors.

Since many insects visiting flowers do so only to consume nectar, plants have also evolved mechanisms to ensure their nectar stores cannot be tapped without the visitor either getting dusted with pollen or brushing against the stigmata at the same time. As flower types radiated (evolved in different directions) so did the mouthparts of the flies exploiting them, so that today members of various fly families have mouthparts specialized for feeding at flowers of structurally different types.

There is substantial information on flower visitation by flies, but knowledge is still patchy regarding whether particular fly species collect pollen or nectar, whether both sexes visit flowers, etc. There remains a need for more detailed, accurate observations on flower visitation by flies. Flowers of certain plant families are visited more frequently and by a wider range of flies than are others, e.g., Ranunculaceae, Rosaceae, Umbelliferae and Compositae. These flowers are mostly either white or yellow, with the yellow flowers predominating. Yellow flowers tend to be unspecialised, with easily available nectar. Red flowers, conversely, tend to be specialised, e.g., with a tubular corolla and their nectar is characteristically hidden in some way. Red flowers tend to be visited only by flies with specialised mouthparts (Knuth, 1906-1909).

Representative species of nearly all families of Nematocera have been recorded feeding at flowers, males of bloodsucking culicids and ceratopogonids. Pollen is the protein source for females of some non blood-sucking *Atrichopogon* spp. (Ceratopogonidae) (Downes, 1955) which pierce pollen grains and suck out the contents. Even females of blood-sucking mosquitoes may gather nectar from small tubular blooms, usually after dark. Non-biting Nematocera have unspecialised mouthparts, so flower-feeding is restricted to plants with more exposed nectar, like umbellifers. There are a few records of *Nephrotoma* and *Tipula* feeding on umbels during daylight hours though night feeding may be more usual. The Bibionidae are frequently noticed at flowers of those plant families that exhibit exposed nectaries. Flowers visited by cecidomyiids, mycetophilids and psychodids are mostly low-growing and found in damp and shady places, e.g., golden saxifrage (*Chrysosplenium*). Some Sciaridae and Keroplatinae, however, occur commonly on umbels and a few of the latter have elongate probosces adapted for flower-feeding, e.g., *Macrorrhyncha*, *Asindulum*, *Antlemon*.

Among Brachycera, many Bombyliidae (beeflies) have a long proboscis, rigid tube tipped by the labellae, which are blade-like (and able to pierce soft plant tissues) as opposed to the inflatable membranous flaps of many Diptera. They tap nectar sources in tubular flowers, e.g., primrose (*Primula vulgaris*), which cannot be tackled by most flies, while still in the air - they hover, humming-bird fashion, in front of the flower, enabling rapid escape when danger threatens.

Some bombyliids are known to collect pollen but this has not been investigated in any detail. Asilids, dolichopodids, empids and tabanids all feed at flowers, but according to Downes (1958) their greatly modified mouthparts have been developed primarily for blood-sucking or predation and are only secondarily used for flower-feeding. Dolichopodid mouthparts differ markedly from those of most predatory flies, being short and rarely tubular; they are largely confined to flowers with well-exposed nectaries. Non-blood-sucking male tabanids feed mainly at flowers but may also obtain sugar from honey-dew. They also both suck up nectar and pierce plant tissues to suck the juices. Some empids definitely feed on pollen (ingesting entire grains) as well as nectar (Dowries and Smith, 1969) and are apparently not predatory. There are some records of Rhagionidae, Stratiomyidae and Therevidae feeding at flowers, usually composites or umbellifers with readily accessible nectar.

Among Cyclorrhapha Aschiza, the Syrphidae (hoverflies) are arguably the most important of all the Diptera families in relation to pollination. Long lists of flowers visited by them are given in de Buck (1990). Both sexes are known to ingest pollen grains and nectar. Some, however, also collect pollen from wind pollinated flowers, e.g., *Melangyna quadrimaculata* feeds from hazel (*Corylus*) catkins and *Melanostoma* and *Platycheirus* feed on pollen of grasses (Gramineae), sedges (Cyperaceae) and plantain (*Plantago lanceolata*). *Melanostoma* is recorded as a pollinator of timothy (*Phleum pratense*) and cocksfoot (*Dactylis glomerata*). Fully 96% of the British syrphid species are known to visit nectar-bearing flowers, and 27% have been found at pollen-only flowers. These are the proportions of the syrphid fauna that are known to feed on the flowers of plants characteristic of different habitat strata (data from the StN database, see Speight *et al.,* 2008). Nectar is mopped up by the labella as in many Diptera. Pollen grains are also picked up by the labella and ingested. Hoverflies are thus able to probe hidden nectar sources in a wide range of flowers, the flies with a longer proboscis being able to reach the more deeply hidden nectaries. However, syrphids which specialise in feeding at tubular flowers (e.g., *Anasimyia*, *Rhingia*, *Sphegina*, *Volucella*) have a more sophisticated apparatus. Thus for instance, *Rhingia* can feed from bluebell (*Endymion*) as well as from a wide range of composites and umbellifers, *Sphegina* from herb robert (*Geranium robertianum*), *Anasimyia* from bog-bean (*Menyanthes*) and *Volucella* from *Buddleja*. There have been various studies comparing the pollinating activity of flies and bees, but, until recently, those studies have effectively been based on assessing "how good a bee a fly can be". Latterly, attempts have been made to establish more directly what role flies play in pollination, that give a different perspective. It transpires that, while bees may effect pollination, syrphid flower visits augment seed quality and viability in crops, through multiple pollination effects (Frank and Volkmar, 2006; Jarlan *et al*, 1997). A second effect of syrphid flower-visiting activities can be to ensure the pollination of the flowers of plants which, because they are present as only a few individuals, do not get pollinated by bees, in mixed stands of flowering plants (Gibson *et al.,* 2006). Conopidae specialise in flower-feeding; the different genera have mouthparts modified to differing extents for flower-feeding and this is strongly reflected in the flowers visited.
Among Cyclorrhapha Schizophora, flower-feeding is the exception rather than the norm, only becoming important in the calypterates. Chloropidae, Lauxanudae. Psilidae. Sepsidae and Tephritidae also use flowers, chiefly Umbelliferae, as food sources but few have specialised in this habit. Many calypterates including *Scathophaga* feed at flowers, consuming both pollen and nectar - these flower products are probably the sole source of food for many of them. Some have been seen collecting pollen from flowers with concealed nectar sources which they could not tap. Most do not have suitable mouth-parts to exploit tubular flowers, the exceptions being the dexiine and phasiine tachinids. Although they are unspecialised and largely confined to flowers with easily available nectar, the Muscidae are the next most important family to the Syrphidae as pollinators.

There are other reasons that flowers get visited by flies. Males of various flies locate females on flowerheads for mating. *Scathophaga* has been observed to practice ambush predation on or beside flower heads, as is practiced by some empids. Flowers can also provide good vantage points to watch for the approach of potential enemies or to shelter when the weather is cold, wet or windy. Large tubular flowers provide shelter from the rain and flowers are often warmer than surrounding substrates, allowing flies to increase their metabolic rate.

NB. All of the information above comes from Speight (in press).

To conclude, it seems that flower-visiting flies have never been used as indicators and it may be difficult to link their occurrence to specific ecological conditions or to other aspects of wider biodiversity. Nevertheless, they could be used in a relative way, either in a diachronic study on one farm or in a synchronic study on several farms. In both cases, that would imply either to identify at the species level or higher taxa levels, or to describe the various morpho-species, of the flies found feeding at flowers of different shapes. Those flowers with flies could be used for a ratio with all the flowers (visited/non visited). As we do not aim to propose new bioindication methods, however, this group of insects will most probably not be tested in BioBio.

5.4.2.6. Hymenoptera, bees and wasps

Bees and wasps (Apidae, Sphecidae, Eumenidae, Pompilidae) have recently been used as indicators in ecological studies and were recognized as promising indicators for ecological change of habitat quality (e.g., Tscharntke *et al.,* 1998; Steffan-Dewenter *et al.,* 2002). They are characterized by complex life histories and have specific requirements for nutrition and nesting (see Kremen *et al.,* 2007 for a synthesis). They need habitats rich in flowering plants (e.g., Banaszak 1992), as a large proportion of the species only collect pollen from certain plants. In addition, bees and wasps have specific nesting sites, such as dead wood, bare soil, plant stems or small rock cavities which should be close to feeding sites.

Locally, species richness and abundance depend on plant species richness and cover as well as on the habitat composition and diversity in the surrounding landscape (Tscharntke *et al.,* 1998; Schweiger *et al.,* 2005). Furthermore, Schweiger *et al. (*2005) showed in an extensive sampling across Europe that wild bee communities are first influenced by the land use intensity in a region, then by the landscape structure, i.e., the proportion of semi-natural elements in the landscape. Other investigations demonstrated a response of bees to field margins and boundaries which suggests that they may be good indicators of agri-environment schemes (Sepp *et al.,* 2004; Marshall *et al.,* 2006). This response was considered to likely reflect better floral resources as mentioned by other authors (e.g., Carvell *et al.,* 2007). By recently reviewing the decline in species richness and crop visitation rate for pollination in response to the distance to natural habitats for several crops world wide, Ricketts *et al. (*2008) emphasized the importance of conserving and managing sufficient resources for wild pollinators within the agricultural landscape to maintain the pollination services.

Bees and wasps provide crucial ecological service in the agricultural landscape; they are considered to be predominant and most economically important group of pollinators in most geographical regions (e.g., Klein *et al.,* 2007). A decline in bee diversity will affect the pollination of many insect-pollinated crops and wild plant species. While pollination by bees significantly increases the crop yield (e.g., Hoehn *et al.,* 2008), wasps can be considered as indicators of beneficial interactions because they may be effective predators of other insects. The role of the landscape context and of the land-use change on pollination has been comprehensively synthesized by Kremen *et al. (*2007).

With respect to farming systems, Holzschuh *et al.* (2007) demonstrated that organic farming increases bee diversity by enhancing flower availability. In addition, bee diversity was influenced by the landscape context and the interaction of both, organic farming being more effective in homogeneous landscapes.

Bees and wasps can be captured by sweep netting or with a butterfly net along transects or in areas. They also can be passively caught with yellow pan traps and window traps (e.g., Duelli *et al.,* 1999). Tscharntke *et al.* (1998) recommend the use of trap nests composed of internodes made of e.g., common reed or with paper tubes of different sizes because communities of trap-nesting bees and wasps (Apidae, Sphecidae, Eumenidae, Pompilidae) comprise an important group of pollinators as well as potential natural enemies of insect pests. In addition, the diversity of these communities is highly correlated with total bee and wasp diversity. In a test of the different methods to catch bees in agricultural and semi-natural habitats across Europe, Westphal *et al.* (2008) found that UV-bright pan traps of different colours is the most efficient trapping method to estimate bee diversity. Moreover, this method is fairly low in cost, reliable and simple to use. Disadvantages are that pan traps do not measure the bee abundance since bees are not sampled within a defined area (activity density like pitfall traps), and that there are relatively high post-sampling processes to identify (which can also be an advantage).

5.4.2.7. Lepidoptera – butterflies

Butterflies are one of the main insect groups to have attracted interest in monitoring programmes (Butterfly Conservation Europe website). They have attracted the widespread interest of naturalists throughout history and many records have been made of the geographic location and population size of many of the species (Dennis, 1992). Many species have complex life histories with caterpillars either being entirely dependent as herbivores on a particular food plant (specialist) or a wide range of native plants (generalist). Other species have sophisticated life histories with intermediate caterpillar stages requiring a scavenging phase in the nests of meadow or heathland ant species (Thomas, 1991). The adult butterflies feed on numerous nectar producing flowers, many being territorial within specific habitats and where courtship and egg laying requires specific habitat structures along with the presence of the caterpillar food plant. Butterfly caterpillars can also be sensitive to microhabitats in which the required thermal regime accompanies the availability of a suitable food plant.

The value of monitoring of butterflies was recognised by Pollard *et al.* (1986) and a standard method for a national monitoring programme was developed from this to determine the diversity and abundance of species across standard butterfly walks (UK Butterfly Monitoring Scheme set up to monitor changes in butterfly abundance in the UK since 1976; UK BMS website). The UKBMS method has been adopted by many Butterfly Conservation bodies across Europe (Butterfly Conservation Europe website), and is second only to the Pan-European Common Bird Monitoring scheme in scale. Recently, The Helmholtz Centre for Environmental Research (UFZ website) was successful in developing butterfly monitoring schemes in Israel, Australia and China.

The method requires the setting up of a standard transect through the habitats of represented at a particular site, often following existing boundaries or paths. The transect walk should be between 0.5 to 1.5 hours at a steady pace. The route should be walked 26 weeks a year when weather conditions are benign (rules about time of day, temperature range, sunlight and rainfall must be met before a transect can be walked). Counts of butterflies are made in a 5 x 5 x 5 m box of airspace in front of the recorder for each section of the transect (Hall, 1981).

5.4.2.8. Lepidoptera – moths

Moths have been the focus of widespread monitoring in many European countries over long periods, initially because of their popularity with amateur naturalists and the availability of portable, low priced light traps for overnight surveys. Pictorial field identification manuals have been available for many years to enable relatively easy identification of adult moths. There have been many studies linking aspects of farming systems with the species composition of moths on different kinds of farmland (Littlewood, 2008; Poyrey *et al.,* 2004; Woiwod and Stewart, 1990). National monitoring schemes standardised the light trap design and sampling periods, such as the Rothamsted light trap network in the UK (Taylor, 1986). This method was adopted at terrestrial sites of the UK Long Term Ecosystem Research network, the UK Environmental Change Network (ECN website) that was initiated in 1993. Many of the sites are situated in arable or pastoral farmland and the data have demonstrated significant trends in populations and species richness of moths over the 15 years of the monitoring (Conrad *et al.,* 2004). The advantages of this standardised sampling method are the 'attraction' of moths from the mix of habitats on farmland as a product of the intensity and light frequency of the light, the simplicity of the of collection of material each morning and the associated central service for the processing of samples and identification of material. The disadvantages are the labour requirements throughout the period of adult emergence and flying and the need for a mains electricity supply which can rather bias or limit the locations that can be sampled. These considerations may limit the practicality of this sampling method on organic and especially low input farmland, despite the advantages of monitoring such a species-rich group which has complex interactions with habitat diversity and structure, the availability of particular host plants and management practices. The inclusion of moths as a biodiversity indicator is desirable but may therefore not be feasible.

5.4.2.9. Orthoptera, Acrididae – grasshoppers

Grasshoppers are often used to estimate the nature conservation value of agricultural areas (Detzel, 1992; Ingrisch and Köhler, 1998). Most species are easy to identify and detect. They are taxonomically well known and the life history of many species is well understood (Harz, 1969; 1975; Harz and Kaltenbach, 1976; Ingrisch and Köhler, 1998; Detzel, 1998). They occur over the whole of Europe and even up to 3000 m a.s.l. Most species inhabit grasslands from very dry to very wet conditions, ruderal areas and different formations of shrubs and shrublands - few are forest species. Most species are herbivorous, a considerable number is carnivorous or both and several are detritivorous and feed on decaying plant materials or excrements.

Grasshoppers are very sensitive to land use changes like the inclusion or absence of structural elements e.g., bushes, shrubs, stony or sandy surfaces. They are also very sensitive to the intensity of agricultural land use like e. g. the number of meadow-cuts per year and the life-stock management of a pasture (Walter *et al.,* 2007). Further relations between grazing intensity and grasshoppers have been reported, e.g., by Batáry *et al. (*2007) and Kruess and Tscharntke (2002). In general extensively grazed pastures support a richer species assemblage of grasshopper and bush crickets than intensively grazed ones. Mowing has been shown to have a major negative impact on grasshoppers (e.g., Fartmann and Mattes, 1997; Humbert *et al.,* 2008). The type of equipment used is a major determinant of the mowing impact. Furthermore, the subsequent treatment of the mown hay can have even more severe impacts on grasshopper populations than mowing itself (see Humbert *et al.,* 2008). In arable fields grasshoppers occur only in low numbers or are completely absent, which is due to mechanical soil operations that destroy eggs and nymphs (most species are ground breeders; e.g., Fartmann and Mattes, 1997; Laußmann, 1998; Gottwald *et al.,* 2005; Marshall *et al.,* 2006). The existence of field margins, hedgerows or fallow land can increase diversity and density considerably, as well as smaller field sizes (e.g., Hill *et al.,* 1995; Laußmann, 1998; Marshall *et al.,* 2006).

The effects of organic farming on grasshoppers is poorly studied, probably due to the fact that the majority of work on organic farming took place on arable land, where grasshopper occurrence is limited (see above). Merely the work of Jones and Sieving (2006) found an indirect positive effect for grasshoppers in organic fields: foraging activity of birds was increased significantly in organic fields with sunflowers planted as an intercrop, and grasshoppers were the most common prey type.

Based on these findings grasshoppers may serve as indicators in grassland agriculture. It will be appropriate to judge the impact of farming regimes at the level of single species, species groups and assemblages. Some difficulties might occur in intensively managed grassland especially in northern Europe, where the species richness is not that high like in central or southern Europe and species numbers may vary e.g., only between 0 and 3 in a ha-plot. For arable fields grasshoppers seem to be valuable indicators only if the study includes field margins and non-cropped habitats.

Species numbers at different scales can vary considerably e.g., between 0 and about 25 on one ha. Most used investigation-methods are synoptical and accustical identification along transects and relevees with biocoenometers (Ingrisch and Köhler, 1998), which allow a standardised comparision of the results. In general all the species of a study plot or area can be found with only 2-3 samplings per year. More samplings will be needed if the abundances over the year are required.

5.4.2.10. Syrphidae (Diptera) – Hoverflies

5.4.2.10.1. Why use syrphids?

The literature on syrphid identification is now reasonably accessible and most species can be identified with confidence. A few syrphids new to science are still described each year from Europe, but they are mostly from south-eastern parts of the continent. Sufficient habitat, microhabitat and traits information is available for more than 90% of the European syrphid species to make it worthwhile to code that information into a database. Syrphids can provide information about all habitat strata, from grass-root zone to the canopy of dominant forest trees. They also occur in most of the non-marine habitats of Europe, except large or deep water bodies (i.e., aquatic habitats of lakes and rivers), cliffs and caves. A final, unusual syrphid attribute worthy of mention is that, among the larvae of this one family of flies, all three trophic groups are represented, namely, herbivores, predators and saprophages.

Logistically, it is of concern whether a standardised sampling technique exists for an insect group that might be used in field surveys. Equally, the amount of time that must be spent in field campaigns can be significant. It is advantageous to select organisms that can be sampled quickly and reliably. Syrphids can be collected in a standardised way using Malaise traps (see FIG. 5.4, and although trap installation is best carried out with the active participation of specialists, specialists are not required for collection of the sample bottles from the traps during a field survey. Further, the large catchment area of an individual Malaise trap ensures that the material it collects provides information about more than the immediate vicinity of the trap, making Malaise traps suitable for farmland-scale investigations. The sample bottles collected from Malaise traps can also be used for storage of samples and all that has to be done to extract the syrphids from them is to sort the samples under a microscope, the extracted specimens then being immediately available for identification. Finally, there is one advantage to using syrphids that makes them almost unique among European terrestrial invertebrates: information needed for the interpretation of species lists has already been organised into a database and is available to those who wish to use it for analytical procedures, in the form of a Database of European Syrphidae (Syrph the Net (StN)).

FIGURE 5.4. MALAISE TRAP, INSTALLED ON AN ABANDONED MEDITERRANEAN DRY PASTORAL FOREST

5.4.2.10.2. Assessing the performance of the biodiversity maintenance function of a habitat by means of Syrph the Net

Maintenance of biodiversity can be regarded as a property of habitats that can be assessed and managed. The biodiversity maintenance function of a site can be assessed by comparing the expected syrphid species for the combination of habitats present on that site and the observed syrphid species. The the StN database provides the expected lists of syrphid species for a site, to allow such a comparison.

The "macro-habitat" categories used in the StN database are generally consistent with the EU's "CORINE" habitat classification system and in the more comprehensive "EUNIS" system derived from CORINE, hence they correspond to the habitats of the Habitats Directive. The database also lists the affinity of syrphids for micro-habitats, which are also coded into the database. Here, the term "microhabitat" is applied to a structural feature of a macrohabitat, with which the developmental stages of syrphids are associated. The larvae of each syrphid species are associated with one or more microhabitats, the different species occupying different parts of the same macrohabitat. On an overmature tree, for instance, the larvae of some syrphid species will be found among the foliage, the larvae of others will occur in sap-runs, larvae of another set of species will be found in trunk rot-holes and yet another set will occur in rotting tree roots. In the same way, herb-layer vegetation, the litter zone, shrubs and under-storey trees each have their own distinctive complement of syrphid species, as do the various aquatic and sub-aquatic microhabitats.

Syrphid species exhibit considerable habitat fidelity at both macrohabitat and microhabitat levels allowing both macrohabitat and microhabitat associations of the species to be coded into the database. Coding is based on information from both published and unpublished sources (unpublished information gathered from European syrphidologists). Macrohabitat and microhabitat data are coded into the database using a simplified “fuzzy-coding" system. Species may occur in association with more than one category of macrohabitat, in which case they are coded accordingly. Inevitably, different people will have their own perceptions of what constitutes a habitat like “humid *Fagus* forest” or “lowland unimproved dry acidophilous grassland” (or whichever the 316 habitat categories described in StN) and to maximise conformity of interpretation every category used in the database is provided with a written definition.

The list of syrphid species expected to occur on a site can be produced from the macrohabitats file of the StN database after the habitat survey has been completed (refer to BioHab methods, section 5.5). It is important to understand that only part of a country fauna can be expected to occur on a site, due to limitations imposed by regional climate, geology and latitude. Hence, comparison with the syrphid species list for the region in which the site is located. A regional syrphid list is an expression of the maximum syrphid biodiversity that can be expected to occur anywhere within that region. The Range, Status and Distribution spreadsheets provide regional distributions for selected countries but there are other reliable sources of data such as Syrfid in France (Sarthou and Monteil, 2006). The list of the syrphid species expected to occur in the site is produced by combining the species/habitat association data for each observed macro-habitat with the regional syrphid list.

A comparison between the expected and observed list of syrphid species for the site and specific macro-habitats can be represented as the percentage of expected species present. The percentages indicate the relative contribution of each macro-habitat to the ‘biodiversity maintenance function’ of the site. This could be a useful way to assess how agricultural management, including habitat conversion, may have affected syrphid species and their associated microhabitats in open farmland habitats (Haenke *et al.*, 2009).

5.4.3. OTHER INVERTEBRATES

5.4.3.1. Araneae –spiders

Spiders (Araneida) are strictly predators and have intensively been investigated in agro-ecosystems where they contribute to the control of agricultural pests (e.g., Symondson *et al.,* 2002; Lang, 2003; Nyffeler and Sunderland, 2003). Spiders are abundant and form species-rich taxon occurring in (nearly) all terrestrial ecosystems including agro-ecosystems (e.g., Christophe *et al.,* 1979; Marc *et al.,* 1999) where they can remarkably be found from the first soil centimetre up to the tree canopy. Their very broad micro- and macrospatial distribution, and sensitivity to environmental conditions such like temperature, humidity and light (which are strongly determined by agricultural practices in agro-ecosystems) makes them appropriate as indicators for e.g., management intensity of farming systems (e.g., Büchs *et al.,* 1997; Marc *et al.,* 1999; Lang *et al.,* 2008).

In agricultural fields, responses of farmland spiders to agricultural practices and management intensity are well known and documented (e.g., Basedow *et al.,* 1985; Gibson *et al.,* 1992; Büchs *et al.,* 1997; Holland, 1998; Downie *et al.,* 1999; Dennis *et al.,* 2001; Churchill and Ludwig, 2004; Thorbek and Bilde, 2004). In crop fields a basic spider assemblage is composed of about 20 species which is almost invariably found across Europe (Luczak, 1979; Blick *et al.,* 2000). This assemblage is regionally completed by species to reach more than 60 species in e.g., wheat fields. Strongly disturbed habitats like annual crops or intensively managed grasslands can be quickly re-colonized by spiders after harvest or perturbation from surrounding biotopes noticeably by ballooning (e.g., Bishop and Riechert, 1990; Schmidt and Tscharntke, 2005a). In this context, landscape features play an important role in determining spider assemblages at local fields (e.g., Jeanneret *et al.,* 2003; Batary *et al.,* 2008a) and in particular, perennial habitats are important source pools for spiders in the agricultural landscape (Schmidt and Tscharntke, 2005b). Recently, spiders have been successfully used in several studies investigating the effect of agricultural management and impact on biodiversity across Europe to show differences among farming systems and management practices including organic farming (Pfiffner and Luka, 2003; Schweiger *et al.,* 2005; Clough *et al.,* 2007; Batary *et al.,* 2008b).

Knowledge about the ecological demands of spiders has developed rapidly over recent years and habitat preferences of mid-European spiders including farmland species are well known (e.g., Hänggi *et al.,* 1995; Platen, 1996). Spiders are relatively easy to collect using various sampling methods (e.g., Duffey, 1974), i.e., pitfall (ground-dwelling spiders) and sticky (ballooning spiders) traps, hand searching (ground dwelling spiders, etc.), Tullgren funnel (spiders in soil), suction samplers (D-vac, ground dwelling spiders, etc.), sweep net (spiders in low vegetation) and beating method (spiders in middle to high vegetation). Methods to optimize the collecting procedure with pitfall traps have been developed so that spring and summer spider diversity can cost-efficiently be estimated (Duelli *et al.,* 1999). Spiders can be collected (with pitfalls together with carabid beetles, for instance) by technician without particular knowledge (in contrast to butterflies that have to be identified during field observation). After collection, however, they have to be identified by taxonomists. The modern taxonomy of European spiders (Platnick, 2009) is well known and is subject to only minor changes that do not disturb their reliable identification.

5.4.3.2. Mollusca, Gastropoda – land snails

Snails are sensitive to land use change both to extensification (Cremene *et al.,* 2005) and intensification (Boschi and Baur, 2007; Baur and Baur, 1995) and are therefore potentially good indicators for changes in ecological conditions and biodiversity, specifically agro-biodiversity. Many snail species show distinct habitat specificity which is a prerequisite for bioindication: There are open land and forest species and specific species assemblages occur in wetland habitats and grassland types (Falkner *et al.,* 2001). Although snails are not widely established as indicators, e.g., the biodiversity monitoring of Switzerland uses them as one of four organism groups together with plants, birds and butterflies. They represent organisms with low mobility and small home ranges and they are – in contrast to e.g., birds, butterflies and plants – a relatively unprotected group. This is an advantage because indicators which are in the focus of ecological measures may pretend a too optimistic dynamic of biodiversity. Faunistic data are available for most European countries (Falkner *et al.,* 2001) and bioindication values of snails are indicated on the red list of Switzerland (Duelli, 1994), in Falkner *et al.* (2001), covering shelled gastropoda of Western Europe) and in Wells and Chatfield (1992, for Europe).

The standardized sampling procedure of snails used in the Biodiversity Monitoring of Switzerland (Biodiversitätsmonitoring Schweiz) is consisting of 1) soil sampling in the field, 2) washing out of shell fractions, 3) sorting of shells from dry matter and 4) identification and provides a good workability as it is not dependent on specific weather conditions and not restricted to a narrow time frame (sampling in late summer and autumn is ideal for identification). As there is little variation in mollusc populations between years (Bishop, 1977), one field assessment provides good information for analyses. While the sorting of the shells from the soil is the most time consuming part (with binocular, partly very small shells of a few mm), the identification procedure is in most cases possible by external features of the shells, dissection is only necessary in few cases. In contrast to slugs, snails enjoy public appreciation, especially because of their beautiful shells. However, a fast resettlement because of ecological improvement cannot be expected, because of the low mobility/small home range characters of snails.

5.4.4. VERTEBRATES

5.4.4.1. Birds

Gregory *et al. (*2005) argue that the farmland bird indicator is a useful surrogate for trends in other elements of biodiversity in this habitat. Birds are relatively easy to detect, identify and census. Their taxonomy is well resolved and the general level of our understanding of their population biology and behaviour is high. Birds are wide ranging in habitat distribution, moderately abundant, are of moderate body size and have moderate life spans. These characteristics result in population responses to environmental change at moderate spatial and temporal scales. Birds tend to be at, or near, the top of the food chain and are thus responsive to signals that accumulate through the chain (the most obvious examples being persistent pollutants). There are often good historical and contemporary data on bird population changes and these data are realistic and relatively inexpensive to collect. In some situations, at least, birds can reflect changes in other biodiversity and are responsive to environmental change.

Expert ornithologists selected 24 native bird species typical of agricultural habitats in Europe. Information on species-specific national population sizes of these species can be obtained for a particular year from the European Bird Database (Tucker and Heath, 1994; BirdLife International/European Bird Census Council, 2000). Bird indices could be produced using TRIM (TRends and Indices for Monitoring data—Pannekoek and van Strien, 2001), based on time-series of counts.

5.4.4.1.1. Density of territories – single species

Density of single bird species is strongly recommended as an indicator. The most clear-cut results relating density to farm type were found for quite abundant field breeding birds like Skylark *Alauda arvensis* or Lapwing *Vanellus vanellus*. For other species in fields results have been more ambiguous, but when all birds were counted, the majority of cases showed beneficial effects of organic farming. With repeated standardised counts, many bird species in fields as well as in adjacent habitat like hedgerows, can be recorded, enabling comparison of organic and conventional farms.

The density (abundance) of bird species occupying territories in fields has been studied in the majority of papers on birds and organic farming. At the level of single species most of the work has been done with Skylarks, probably one of the best studied farmland birds (e.g., Donald and Vickery, 2001). For this species significant differences in densities on organic versus conventional farmed land have been shown on several occasions (Braae *et al.,* 1988; Christensen *et al.,* 1996; Wilson *et al.,* 1997; Fuchs and Saacke, 2006; Neumann *et al.,* 2007; Piha *et al.,* 2007; Kragten and de Snoo, 2008). Chamberlain *et al.* (1998), however, found a significantly higher density of Skylarks only in one of the three years studied. For other field breeding birds differences in density were not found in all cases. Negative effects of organic farming have been reported also, but in the majority of cases organic farming has been found to be favourable for birds' densities. A higher territory density of Lapwings on organic versus conventional farmland was found by Braae *et al.* (1988), Christensen *et al.* (1996), Kragten and de Snoo (2007 and 2008; in both cases statistical significance was not always reached) and Piha *et al.* (2007). Contrastingly, Neumann *et al.* (2007) found no differences in density of Lapwing territories in northern Germany. Christensen *et al.* (1996) reported higher densities for all abundant field breeding birds, while Kragten and de Snoo (2008) found that, except for Skylark and Lapwing, seven other species showed no significant differences in territory density.

Some studies reported results on all bird species recorded in sufficient numbers to perform statistical tests. For this purpose no distinction between birds breeding in the fields, breeding in surrounding habitat or using the fields as foraging habitat was made. For the majority of species densities did not differ, but among species with significant differences the by far highest part had greater numbers on organic farmland (Braae *et al.,* 1988: 36 out of 39 species with higher density on organic farmland; Christensen *et al.,* 1996: 31 out of 34 species; Freemark and Kirk 2001: eight out of ten species; Beecher *et al.,* 2002: all eleven species). A study on species of field boundaries in Britain found higher densities on organic farms only in a minority of the investigated species (Chamberlain *et al.,* 1998).

5.4.4.1.2. Density of territories – species groups/all species

Positive effects of organic farming on overall density of birds outnumber the cases where no effect has been found. Density of territories is therefore regarded as a suitable indicator and overall density needs no extra counting effort if standardised counts of birds are applied (see density of territories – single species). Several scientific papers documented a higher overall density of birds in organic farming (Braae *et al.,* 1988; Christensen *et al.,* 1996; Freemark and Kirk, 2001; Beecher *et al.,* 2002; Belfrage *et al.,* 2005). A study on the effect of intercropping sunflowers *Helianthus annuus* in organic vegetables found higher bird densities than in control fields (Jones and Sieving 2006) and an analysis on specialised birds, according to habitat and diet, found a higher overall density on organic farmland (Filippi-Codaccioni *et al.,* 2008). Species of field boundaries on organic farms were more abundant in one of three years studied in Britain (Chamberlain *et al.,* 1998). A study in Germany monitored the development of the bird assemblage on a farm after switching the management from conventional to organic and integrated production, and reported a steady increase of overall density (+ 60 % in five years; Laussmann and Plachter, 1998). A significant enhanced overall density in organic farming systems also was found in a meta-analysis conducted by Bengtsson *et al.* (2005). On the other hand there is some evidence of similar overall bird density in organic and conventional farming at field/farm level (Lokemoen and Beiser, 1997; Kragten and de Snoo, 2008) as well as on landscape level, where landscape structure and agricultural land-use have been the principal determinants of bird assemblages (Piha *et al.,* 2007).

5.4.4.1.3. Species richness

A couple of research studies found a higher species richness on organic than on conventional farmland (Christensen *et al.,* 1996; Lokemoen and Beiser, 1997; Laussmann and Plachter, 1998; Freemark and Kirk, 2001; Beecher *et al.,* 2002; Belfrage *et al.,* 2005). In the meta-analysis carried out by Bengtsson *et al.* (2005) summarized data on species richness and diversity revealed higher values for organic farming. A study in Germany did not find different species richness on organic and conventional farmland (Neumann *et al.,* 2007) in accordance with work done at the landscape level in Finland (Piha *et al.,* 2007). Additionally, Fuller *et al.* (2005) found no difference in winter species richness between organic and conventional farms in Britain. Given the far more positive findings, species richness of birds on organic farmland should be included as indicator.

5.4.4.1.4. Density outside the breeding season

The overall density of birds on organic farms in winter was shown to be higher in Great Britain (Chamberlain, 1998; Fuller *et al.,* 2005); which was also reported for autumn (Chamberlain *et al.,* 1998). In Germany, some species groups had higher densities in autumn and winter (Hötker *et al.,* 2004). On the contrary work from North Dakota/USA did not find different densities of birds in organic and conventional fields (Lokemoen and Beiser, 1997). The few studies on density outside the breeding season found generally positive results, but (1) counts in autumn and winter need extra counting effort and (2) the habitat bond of birds is strongest and more continuous during the breeding season.

5.4.4.1.5. Diversity

Studies on the diversity of birds in the breeding season (Neumann *et al.,* 2007; Piha *et al.,* 2007), autumn (Chamberlain *et al.,* 1998) and winter (Chamberlain *et al.,* 1998; Fuller *et al.,* 2005) did not find a higher diversity of bird species on organic farmland. Chamberlain *et al.* (1998) report a higher diversity in one out of three breeding seasons. Bengtsson *et al.* (2005) combined data on species richness and diversity in their meta-analysis and found higher values for that parameter in organic farming. In orchards managed organically bird diversity was significantly higher than in conventional ones (Fluetsch and Sparling, 1994; Genghini *et al.,* 2006). The ambiguous scientific results hinder a clear decision in favour of diversity as an appropriate indicator for biodiversity of organic/low input farming. Nevertheless, diversity (e.g., expressed via the Shannon Index) can easily be calculated based on the data gathered for overall density and species richness, and the use of diversity for the purpose of this study can be tested empirically.

5.4.4.1.6. Biomass

The total biomass of field-dwelling farmland birds showed a significant positive relation to the area of organic farming at landscape level. The effect diminished after the removal of Skylarks, the most abundant bird, from the data set (Piha *et al.,* 2007). No other results on biomass are available for comparison, however. Interestingly, in this study organic farming did not show a positive correlation with overall bird density (which is correlated with biomass). This indicator is easy to calculate on the basis of abundance and published data on bird weights.

5.4.4.1.7. Frequency of occurrence

Freemark and Kirk (2001) report higher frequencies of occurrence for birds on organic fields. We found no studies of frequency except Genghini *et al* (2006) for orchards. This indicator is simple to calculate from standardised count data necessary to estimate density and species richness.

5.4.4.1.8. Foraging intensity

The foraging intensity of birds, measured as the number of birds per hour and foraging bout lengths, was higher on organic compared to conventional fields (Jones and Sieving, 2006).

Organically grown wheat was used significantly more by foraging Yellowhammers, *Emberiza citrinella* than conventional wheat (Morris *et al.,* 2001). In an experimental design the average weight gain of foraging Grey Partridge, *Perdix perdix* chicks was highest on organic fields (Herrmann and Fuchs, 2006). Data hint on suitability as an indicator; but this needs extra count effort. Foraging intensity is dependent on surrounding habitat/neighbouring breeding birds and there would be a need to correct the data for that, or to have sites with almost identical habitat characteristics.

5.4.4.1.9. Density of nests

At least for field breeding birds nest density can be regarded as indicator for biodiversity of organic fields, but it shall not be considered further because of the high intensity of work that is necessary to locate nests in the fields (see method sections of the cited references studies). The number of Skylark nests on organic arable farms has been considerably higher than on conventional farms, especially in spring cereals, lucerne and grass leys, all of which were predominantly or exclusively grown on organic farmland (Kragten *et al.,* 2008). A study on Lapwings found the number of nests in organic farmland being almost twice as high as in conventional farmland, but the differences were not statistically significant (Kragten and de Snoo, 2007). At the multi-species level Lokemoen and Beiser (1997) found a significantly higher nest density on organic farmed fields. On the other hand, studies conducted by Lubbe and de Snoo (2007) and Kragten *et al.* (2009) found similar numbers of Swallow *Hirundo rustica* nests at farm buildings of organic and conventional farms.

5.4.4.1.10. Brood parameters

Several results on brood parameters do not support the suitability of parameters on breeding success as indicators for organic biodiversity. Brood parameters are time-consuming to obtain (see references cited in this section) and thus less appropriate in terms of efficiency. Hatching success of field breeding birds in North Dakota/USA was the same in organic and conventional managed arable fields (Lokemoen and Beiser, 1997). Bradbury *et al.* (2000) reported the same finding for survival rates of Yellowhammers in the United Kingdom, although breeding started slightly earlier on organic farms. In England, no significant difference in the daily nest survival rate of Skylarks was found (probably due to small sample sizes), although the overall nest success was higher on organic farms (Wilson *et al.,* 1997). Lapwings in The Netherlands were found to have a lower breeding success on organic than on conventional farmland in one of the two years studied, which was due to nest losses by farming operations on the organic farmland (Kragten and de Snoo, 2007). Yellowhammers breeding on organic farms showed significantly larger clutches than birds on conventional farms (Peterson *et al.,* 1995) and the breeding success of the Mourning Dove, *Zenaida macroura* and American Robin, *Turdus migratorius* was greater in organic than in conventionally managed orchards in a study in Pennsylvania, USA (Fluetsch and Sparling, 1994).

5.4.4.3. Small mammals

This group, which is ecologically important by constituting a prey base for many terrestrial and avian carnivores, has been shown to be sensitive to the fragmentation of the hedgerow network at landscape scale (Michel *et al.,* 2006), and to the local habitat quality (width of hedges and the tree species richness, Michel *et al.,* 2007). Indicators which are used are biomass and abundance estimated from animal trapping in permanent habitats using baited live-traps checked at 24 and 48h after installation in permanent habitats (eight hedges per landscape unit). A standardized method consists of a baited 100-m trap-line (Spitz *et al.,* 1974) in hedgerowa, with one trapping session (of 4-5 days) every month from April to October.

5.4.4.4. Chiroptera - bats

Bats are distributed almost worldwide. There are 45 species in Europe, which – because of severe declines throughout the last decades - are all protected under the Bern Convention, the Convention on Migratory Species and the EC Habitat Directive. These declines have been linked to habitat loss and fragmentation, to agricultural intensification and extensive use of pesticides and to increased mortality rates through other human activities. During the last years a lot of scientific and conservational attention was put on this species group and lead to an increased public awareness and enormous progress in knowledge about the ecology of bats and research methods.

One aspect in the ongoing research about bats is their applicability as indicators for biodiversity. In this context Lund and Rahbek (2002) showed that bats are suitable to represent species richness of other groups. Also Jones *et al.* (2009) discuss the suitability of bats as indicators for climate change and habitat loss and highlight the high potential of this species group. The EUROBAT working group on “Bats as indicators” which reviewed and evaluated the suitability of bats as indicators on a national level for SEBI2010 (EUROBATS, 2006) concluded that the advantages of implementing bats as indicators outweigh the disadvantages and that there is high potential for implementing this indicator. Within some national monitoring schemes - e.g., the UK Biodiversity Indicators (Stevenson *et al.,* 2009) bats are already implemented as biodiversity indicators.

With reference to the objectives of the BioBio project bats seem to be promising candidates as biodiversity indicators, too. EUROBATS (2006) highlights the importance of bats – representing 23% of the European mammal fauna - as biodiversity indicators. Wickramasinghe *et al.* (2003; 2004; 2007) found that there is a significant effect of agricultural intensification on nocturnal insect prey and therefore on bats. They showed differences in overall and foraging activity of several bat species related to agricultural intensification by comparing organic and conventional farm types. The main reasons for that seem to be higher amount of structure and better habitat quality in terms of prey availability on organic farms. Contrary to the results of Wickramasinghe *et al.,* Pocock and Jennings (2008) found that bats were less sensible to agrochemical inputs but recorded a strong effect of boundary loss on small bat species. Nevertheless, the results show that bats have a great potential for indicating the richness and connectivity of farmland structures and of local habitat quality. Another important advantage is the functional aspect with European bats being nocturnal foragers on aerial insects making bats important for pest control. Regarding the application of the BioBio concept in non European countries – e.g., Uganda – other functional aspects like pollination through bats might also be interesting.

Bats can be sampled in their roosts in winter and summer or through sound detection sampling techniques in summer. The sampling method could be a critical factor for choosing bats as an indicator group. Regarding the BioBio objectives an easy and affordable but meaningful sampling method needs to be found. Another problem that needs to be addressed is the scale - farm or landscape scale - at which bats can be sampled with significant accuracy and how these measures could be aggregated to higher levels.

5.4.6. AQUATIC ORGANISMS OF STANDING AND RUNNING WATER ON FARMLAND

A possible method for surveying vegetation in running waters is the Kohler-method (Kohler, 1978; Kohler and Janauer, 1995), which is in accordance with the respective European Standard (EN 14184:2003) and the regulations of the Water Framework Directive. Contrary to the classic vegetation surveying methods the vegetation is not examined in homogenous areas, but a whole unit is regarded uniformly. The borders of the continuous survey units are determined with the aim to have sections with almost uniform vegetation and approximately equal environmental conditions, so the length of the units is different. Artificial structures such as sluices and bridges could be ecological barriers and therefore sharply delineate the unit borders. For each survey unit every plant, moss and alga visible to the naked eye is recorded and abundance estimated on a scale from 1 to 5 (1-rare, 2-ocasional, 3-frequent, 4-abundant, 5-very abundant). Survey units are briefly characterized based on shadiness, the parameters of the riverbed, dynamic of water flow, width and depth of the water, turbidity, the riverside vegetation, and land use type. Due to the standardised field survey methodology and data processing the results are comparable for the investigated waters and also for the further monitoring period.

For the assessments three categories of life forms for the macrophytic species can be distinguished: hydrophytes, amphiphytes and helophytes. The calculation of various characteristic parameters and the ecological evaluation of the waterways is done using hydrophytes and amphiphytes only. For the quantitative characterisation, the following indices were calculated by Kohler and Janauer (1995) and Pall and Janauer (1995):

- The Relative Plant Mass (RPM) gives the plant mass of a given species in relation to the total plant mass of all species for the respective stretch of waterway. The RPM is suitable to describe the diversity and dominance structure in the surveyed river. All the species with value less than 1% are summed up and depicted as residual.
- The Mean Mass Indices (MMT, MMO) provide information about the distribution of plants in the area investigated. In case of the MMT (T: total) the plant mass is calculated with respect to all mapped survey sections of the regarded stretch. In the case of the MMO (O: occurrence) only those survey sections in which the respective plant is present are taken into consideration. Therefore, the MMO is always bigger than the MMT value. Conclusions can be drawn from the relation of both values of the Mean Mass Indices. If both values are nearly identical the given species is present in the respective stretch with similar plant masses. Increasing differences of both values indicate a more and more spotted occurrence of the given species. Based on the field data distribution diagrams containing the species lists were prepared. In the distribution diagram the cells are relative to the real length of the unit so we get a more true representation. The number of summarized species is also given so the diagram also provides information about the total number of plant species.

The most widely used method for surveying macrophytes in lakes is based on belt transects. This approach allows mapping of the distribution of individual species and abundance of aquatic macrophytes, it provides robust data sets that can be used to generate indices and metrics, and it is a cost effective means of data collection. Also recommended, however is an investigation following the shorelines of the survey lakes and using the same method as in the case of running waters (CEN). With the monitoring of macrophytes the eutrophication of watercourses connected with different landuse types and agricultural activities could be detected. Regarding the usefulness of aquatic biodiversity indicators for the objectives of BIOBIO (farm scale analysis for organic and low-input farming systems) we have to be cautious because of the difficulty to relate observations in the watercourse to the management of specific plots / farms. Water quality at a given location is not only influenced by the management of the adjacent plot, but integrates the effects of land management of the upstream watershed. It may thus be difficult to establish links between biodiversity measurements in waterbodies and specific farming practices.

5.5. HABITAT ASSESSMENT AND MONITORING IN THE WIDER COUNTRYSIDE

5.5.1. SCOPE OF HABITAT INDICATORS UNDER CONSIDERATION

Reduction of diversity and complexity of habitats at different scales is a critical process underpinning loss of biodiversity on agricultural land (e.g., Benton *et al.,* 2003). Organic farms may have higher levels of habitat heterogeneity than non-organic farms. This is because basic standards for organic agriculture include provisions to "maintain a significant portion of farms to facilitate biodiversity and nature conservation", including (among others) wildlife refuge habitats and wildlife corridors that provide linkages and connectivity to native habitats (IFOAM 2002). Mansvelt and van der Lubbe (1999) showed that the diversity of landscape and farming systems was greater in organic farms, regarding land use types, crops, livestock and plantings (hedges, shrubs, trees). Organic crop rotations are more diverse (Agreste 2007) and arthropod diversity has been shown to relate to crop diversity (Schweiger *et al.,* 2005). In terms of landscape diversity, the organic types of agriculture may potentially offer one route to restoring farmland biodiversity (Krebs *et al.,* 1999) although data are missing that confirm this statement. At the landscape scale, low-input farming systems concentrated in HNV regions are supposed to provide a wider mosaic of different arable, grass and semi-natural habitats and landscape elements, such as field margins, hedges and grass strips, patches of uncultivated land, used at different levels of intensity (the presence of semi-natural habitats is a defining feature of HNV farmland).

The most comprehensive project to develop statistically reliable habitat indicators is the GB Countryside Survey (www.countrysidesurvey.org.uk). This survey has had four sampling dates since 1978 and reports on about ten indicators linked to habitats and environmental strata. In other countries comparable actions have been started in recent times, but most still lack time series (Jongman and Bunce, 2008).

In this section we review and propose indicators that characterize organic and low-input farming systems at the farm and landscape scale, including unfarmed features which are related to the farming systems. We examine indicators that measure habitat occurrence (quantity and spatial arrangement) and habitat quality (environmental, site and management conditions). By habitat occurrence we mean both the quantity of habitat of different types (landscape composition) and how this is arranged in the landscape (landscape configuration). Habitat quantity and habitat quality can both be measured from “indirect sources” (remote sensing/ information on farm practices from agricultural databases) or from field measurements. Remote sensing is an effective method for measuring larger and simple habitat occurrence (preferably supplemented with field work) such as continuous forest, large scale agricultural fields, steppes and deserts, whilst field recording is more effective for measuring complexes and quality. It is expected that farm systems in Europe are consisting of complexes. Indicators of habitat occurrence and quality can be used to measure changes over time and can be applied to examine links to biodiversity.

5.5.2. SYSTEM, STRATA, HABITAT AND INDICATOR DEFINITIONS

5.5.2.1. Agro-ecosystems

In the Utilized Agricultural Area (UAA), agro-ecosystems can basically be divided into four main categories, i.e., grasslands, crops, special crops (vineyards, orchards, vegetables), and semi-natural habitats (partly agriculturally managed). Aquatic ecosystems, e.g., ponds and streams as well as temporary water bodies can also be included. However, in some countries such as Switzerland, aquatic ecosystems are not considered as part the UAA. Unfarmed features such as hedgerows or stone walls may be part of the UAA or may not be part of it, depending on the ownership and the cadastre. Thus, when deciding upon indicators of habitat occurrence and habitat quality it will be necessary to make decisions regarding the limits of our system (Jongman and Bunce 2008). In BioBio we will include semi-natural habitats and unfarmed features in the immediate vicinity of (adjacent to) farms (and thus potentially affected by farming practice) regardless of ownership and legal status.

5.5.2.2. Strata

Terrestrial ecosystems are vertically divided into three main strata: the hypogaion (below ground, humus and endogenous layers), the epigaion (soil surface, litter) and hypergaion (above ground, herb, shrub layers and trees). Decisions will need to be made, particularly when selecting habitat quality indicators, which stratum should be included in the measurements. The habitat mapping that follows the BioHab protocol will focus on the hypergaion.

Between southern and northern Sweden the landscape changes from an open nearly semi-desert through intensive large corn fields into small scale mountain landscapes, back to large arable fields, extensive grasslands with dairy farming and in the north into bogs and extensive Boreal heathlands. The Environmental gradient in Europe is not only characterised by natural vegetation, but also by different forms of farming (FIG. 5.5). Mountain areas in the Mediterranean are characterised by terraces, while they are in the north part of Europe rough grazing land. Lowland areas vary from open marginal arable, to intensive pastures and extensive grazing land. Therefore it is needed to make a distinction between the different European environments when exploring the distribution of agroboiodiversity and farmland features.

Farmland near Almeria (Spain) with solitary olive trees and a water collecting system in the hills

Hedgerow and stonewalls in the Lake district, UK

Grassland in northern Estonia on the edge of abandonment. The trees in the grassland are young spruce

FIGURE 5.5. EUROPEAN FARMLAND EXAMPLES

The official biogeographical zones of Europe as legally binding divisions are a proper tool to divide countries in groups according to their main environmental characteristics, but it is not sufficient to divide the European landscapes into more or less comparable units. Mountains do occur in Spain, Austria, Germany and Scandinavia. They have comparable features as they all have mountain farming, but they also have differences, e.g., due to differences in climate. These differences have to be covered in the inventories. Terraces do occur in Mediterranean mountains, but not in Scandinavia. However, hedges and stonewalls do occur in all Europe, from Greece to Ireland, but with a dominance in pasture landscapes. Some landscapes, such as the semi-desert of Almeria and the Mani in Greece have very specific features (FIG. 5.5).

The Environmental Stratification of Europe (Metzger *et al* 2005) covers the environmental variation in the European continent the best. It has proven its value in several European studies. For the different databases that are being used in this study national stratifications have been made. Most of these can be transformed into the European stratification without big problems. Therefore this stratification has been used as a unifying approach for a European overview.
The Environmental Stratification of Europe (FIG. 5.6) has been constructed using tried and tested statistical procedures to link European environments as well as field data. It shows significant correlations with principal European ecological data sets. As shown in comparative studies, such stratification can be used for strategic random sampling for resource assessment and for measurement of change (Metzger *et al* 2005, Jongman *et al* 2006). The hierarchy of the Environmental Stratification (EnS) allows regional applications to be aggregated into continent-wide assessments, thus facilitating the growing demand for coherent European ecological data to assist EU policy and global state of the environment assessments such as the EU State of the Environment Report and the Millennium Ecosystem Assessment. The EnS does not replace existing classifications, but has proven to provide a framework for integration between them and subsequent estimates of habitat and vegetation when field data become available.

Environmental Stratification of Europe

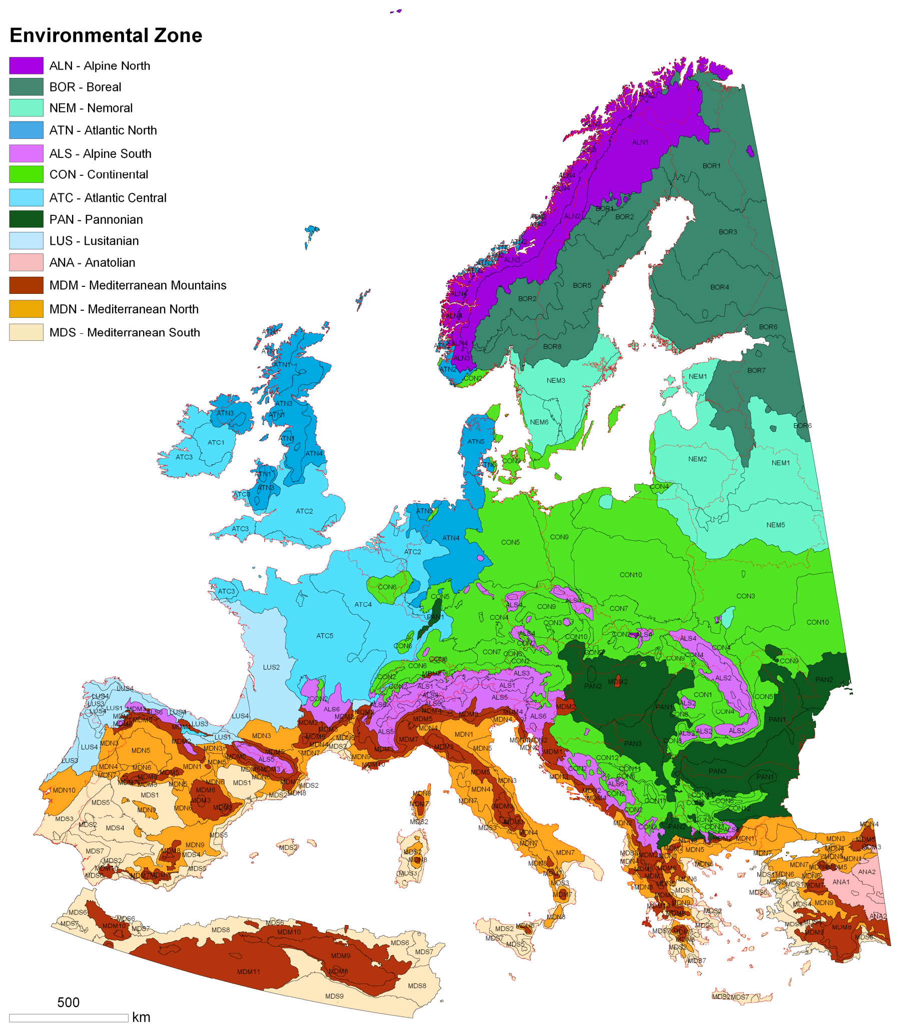

FIGURE 5.6. THE ENVIRONMENTAL STRATIFICATION OF EUROPE IN 13 ZONES AND 84 STRATA. WHERE THE SIZE OF THE STRATUM PERMITS, THE INDIVIDUAL STRATA ARE LABELLED WITHIN THE MAIN ENVIRONMENTAL ZONES. THE STRATIFICATION EXTENDS FROM 11° W TO 32° E AND FROM 34° N TO 72° N. IT IS PROJECTED IN A LAMBERT AZIMUTHAL EQUAL AREA PROJECTION. CERTAIN STRATA DO NOT NECESSARILY FIT TRADITIONAL EXPERIENCE AS IN THIS STRATIFICATION STRICT STATISTICAL RULES HAVE BEEN MAINTAINED, LEADING TO THESE APPARENT INCONSISTENCIES, E.G. PANNONIAN ZONE IN GERMANY AS A DRY RAIN SHADOW AREA (METZGER *ET AL.*, 2005).

The Environmental strata provide a convenient set for monitoring and assessing change for a continent as diverse as Europe and are appropriate for stratified sampling and analysis of environmental data. However, there are too many strata for summary reporting and presentation of the principal characteristics of Europe. An aggregation of the strata into a limited number of Environmental Zones (EnZs) was created to facilitate communication based on the experience of a similar situation in Great Britain, where 32 land classes were reduced to six zones for reporting purposes. The main environmental regions mentioned above (Alpine, Boreal, Continental, Atlantic, Mediterranean and Anatolian) were subdivided on the basis of the mean first principal component score of the strata in the regions. All Mediterranean strata with altitudes above 1000 m were assigned to Mediterranean Mountains.

5.5.2.3. Habitat

Every species has its own demands with respect to habitat, thus it is important to clearly define "habitat". Generic habitat concepts as indicators of the habitat level of biodiversity simplify this process. Often habitats are defined in terms of vegetation cover types, yet there may be ecological attributes other than vegetative cover type that are more important in recognizing patterns of species occurrence (Williams *et al.,* 1997). Recently, the EU project BioHab developed a standardized habitat mapping methodology that has been meticulously tested at the European scale (Bloch-Petersen *et al.,* 2006; Bunce *et al.,* 2008). BioHab offers a viable procedure for consistent data collection. The BioHab project defines habitat as "An element of land that can be consistently defined spatially in the field in order to define the principal environments in which organisms live." BioBio will utilise the mapping system proposed by BioHab as it provides a standardised habitat mapping methodology at the European level (see 5.1.3.1).

It should be stressed at this stage that the term "habitat" may lead to confusion in the communication with stakeholders. For them, "habitat" is often associated with conservation value and in their work they do not consider e.g., a maize field as a "habitat". This distinction is also reinforced by European legislation, e.g., the "Habitat Directive", which emphasizes habitats with conservation value. In a scientific context, however, habitat is used in a neutral way, relating to all patch, line and point features which make up land use / land cover.

5.5.2.4. Spatial Configuration of habitat

It is not just the quantity of habitat that is important but also the spatial arrangement of the habitat in the landscape. Invariably, landscapes comprise of a heterogeneous mosaic of landscape elements. Effectively, the landscape mosaic can be broken down into a spatial pattern of patches, corridors and matrix (Forman and Godron, 1986). The patches consist of relatively distinct, homogeneous, non-linear areas (e.g., woodland, grassland, moor), and corridors are distinct linear strips of a particular type (e.g., hedgerow, field margin), both of which differ from the adjacent landscape elements. The matrix is in effect the background ecosystem or land-use type, characterized by its extensive cover, high connectivity and/or major control over dynamics (e.g., arable land in agricultural landscapes).

Whereas the patch-corridor-matrix concept is an effective landscape model in the central and western European lowlands, it is less applicable in mountain regions or in parts of the Mediterranean. Mountain regions are often dominated by grasslands, the composition of which changes gradually (Hofer *et al.*, 2008). In the Mediterranean gradients can occur from grassland to shrub and from shrub to Dehesa and Montado. In BioBio we may face this situation in the HNV case studies (semi-natural grasslands in Hungary and Bulgaria, Dehesas in Spain and Tunisia, and mountain grasslands in Switzerland, Norway and Wales).

Different types of landscapes can be defined by the intensity of the human influence in space and time. Boundaries between different types of landscapes are dependent on regional, physical and cultural circumstances. Landscape elements can be divided by form characteristics or structure characteristics and are often classified into linear elements and patch elements. In this research a division by structural characteristics is used. Landscape elements can be divided into (I) woody features, (II) grassy features and (III) wet features and artificial features. These three groups of landscape elements are defined by structural characteristics (Jongman and Bunce, 2008).

An agricultural landscape is characterised by a dominance of agricultural land use and management with a strong human influence, introduced elements and remnants of the original natural landscape. The presence of historic elements and the presence of man made features in general determine the character of cultural and the artificial landscapes. From the perspective of form characteristics, most farmland features are linear or point elements. However, some elements can be of such a size that they could be defined as areal elements. Different countries in Europe have different approaches towards this. Categories of landscape elements are often combinations of different elements: a verge with a hedgerow, a verge with a tree row, a verge with a hedge, a verge with a stone wall, a verge with a ditch, a grass strip with a tree row, a ditch with a hedgerow, a ditch with a tree row, a ditch with a hedge, a ditch with a stone wall.
Landscape elements do not exist just for scenery but have, often historically, a function within the agricultural production system. It is expected that in organic farming systems and the landscapes formed by them landscape features might be more frequent and still be used.
Examples of such agricultural functions are (1) hedgerows that act as cattle fence, as wind shelter, as a border, against erosion and supply of fodder and fuel and farm wood, (2) ditches used as cattle fence, for irrigation or drainage, (3) terraces to prevent erosion, (4) woodlots to supply fodder and fuel and farm wood, (5) bogs to supply fodder and fuel and so on. Together with the agricultural function and physical conditions agricultural management defines the type, structure, configuration and size of the landscape element and composition and abundance of species present in the element together constituting the landscape.

5.5.2.5. Scale

Scale is the spatial or temporal dimension of an object or process and is characterized by both its extent and grain size (Turner *et al.,* 2001).The extent is the size of the study area or the temporal duration of the phenomenon under observation. The grain is the finest level of spatial resolution possible within a given dataset. Together they define the upper (extent) and lower (grain) limits of the spatial resolution in the study. The thematic resolution is a further scale aspect which defines the number of different habitat/land use classes to be used to define the landscape. The spatial resolution is a measurement of precision as it dictates the smallest possible feature that can be detected in the study. The thematic resolution defines the level of detail by which the landscape is defined. The choice of scale is dependent upon the organisms or process under consideration and an appropriate scale must be chosen to avoid erroneous conclusions.

In BioBio we will work at the farm scale. An individual farm may be a discontinuous spatial unit consisting of individual plots intermingled with other plots owned by other farmers and / or with unfarmed land-use types. This will probably apply to the case studies investigating organic farms (France, Austria, Germany, Norway, Switzerland, Wales, Netherlands, Italy, Uganda). The HNV case studies (Bulgaria, Hungary, Spain, Tunisia) are more likely to consist of larger, spatially coherent farm units. In the Ukraine, due to the large size of farms (>1000 ha) even individual (organic) farms are likely to form coherent spatial units.

5.5.2.6. Habitat Indicators

Habitat indicators can be directly measured in the field or indirectly by using remote sensing and farm surveys. Both methods can be used to examine the spatial arrangement and quantity of habitats in a landscape as well as the quality of habitats. These should generate indicators that either link to aspects of biodiversity or reflect changes in the habitats and landscapes over time. The spatial pattern and quantity of habitat is measured most effectively using remote sensing data. Commonly, these measurements are made at the landscape scale. Maps of the habitats are commonly created for a defined spatial scale using either satellite images or aerial photographs and supporting material such as cadastral maps (Gustafson, 1998). The habitats are commonly defined according to a particular habitat classification method (e.g., the EUNIS system, Davies and Moss 2002) and are often verified in the field where supporting data may be collected (e.g., habitat quality data). However increasingly it will be necessary to report on the habitats described in Annexe 1 of the Habitats Directive of the EU because these are the basis of habitat conservation in European law. The problem with the EUNIS classes is that there are not rules for mapping them in the field, hence the BioHab system described below. Effectively categorical maps are made up of individual land use/habitat patches, which can be combined to form classes of habitat/land use, or entire landscape mosaics. These are used to calculate landscape metrics, a great number of which exist that describe the characteristics of the individual patches, classes or the entire landscape (McGarigal and Marks, 1995). Metrics fall into two basic categories, those that describe composition (e.g., habitat amount, diversity) and those that describe configuration (e.g., habitat density, isolation, shape complexity, proximity).

Indicators generated through field measurements are more effective at examining the quality of habitat and are often based on environmental, site and management aspects. Regardless of whether habitat indicators measure quality or quantity it is important that they are simple and cost-effective to measure, easy to understand and interpret, reflect changes in habitat over time or aspects of biodiversity and are relatively robust (see section 3.1.2).

5.5.3. APPROACH FOR INDICATORS OF HABITAT OCCURRENCE

The approach to selecting indicators of habitat occurrence will be to choose both an appropriate habitat/land use classification system and scale at the European level.

5.5.3.1. Defining and quantifying habitat types

In order to develop a biodiversity indicator dataset at the European level it is necessary to have an appropriate generic system of habitat definition. Recently, the EU project BioHab has developed a standardized habitat mapping methodology that has been meticulously tested at the European scale (Bunce *et al.,* 2008). BioHab offers a viable procedure for consistent data collection. The Field Handbook published in 2005 has now been up-dated to incorporate subsequent experience especially in Italy, Southern France and Israel. The habitat qualifiers, which characterize individual habitats with respect to their ecological features and quality also need further work and could also include categories specifically related to organic farming and HNV areas. The challenge for BioBio resides in the adaptation of landscape oriented habitat assessment methods to organic/low-input farm scale assessments for – often non-consolidated – farm holdings of different sizes, which may be intertwined with other farming systems. We propose that the definitions of BioHab are adapted for this purpose.

To this end, a classification of farmed and unfarmed land has been made (TABLE 5.8) which builds on the work developed within a research project on unfarmed features carried out for the EU in 2008 (Jongman and Bunce, 2008) and has been tested in the EU FP6 SEAMLESS project. This document is now available on the BioBio website and includes information on the occurrence of areal features, point and linear features throughout Europe from field records. The tables in this document can be used to indicate the likely number of classes to be present in different zones. The application of this typology is essential as much biodiversity is restricted to linear features which are not directly managed by farmers but still influenced by farming practices (Bunce *et al.,* 2005). There are also many references in the scientific literature concerning the importance of linear features in the maintenance of biodiversity. It is proposed to assign vegetation recording plots (see section 5.1.5.3) to all the farmed categories as well as those indirectly affected by farming. Land uses such as urban and extensive forests will be excluded.

TABLE 5.8. OVERVIEW OF FARMED AND UNFARMED CATEGORIES. VEGETATION PLOTS IN BIOBIO WILL BE PLACED IN CATEGORIES 1,3,4,5 AND 6

1. *Fields managed only for agricultural objectives.*

Such fields are usually intensively used but may also involve extensive systems. Usually there is a division between:

a. *Cultivated land used for arable (e.g., wheat) or perennial or woody crops (e.g., fruit trees, vineyards)*
b. Grasslands used directly (grazing) or indirectly (hay, silage) by livestock

2. *Fields managed regularly for non-agricultural objectives.*

Usually these fields are used for horses or donkeys held for recreational purposes but could also include fields and mesotrophic grasslands managed for nature conservation and landscape objectives.

3. *Unenclosed land used regularly by stock, usually sheep and goats but also cattle and horses for meat.*

This category has a wide range of intensity of use and varies in character both regionally and locally. It would include many upland grasslands and heathlands but also dehesas, montados and wood pastures elsewhere. There is a potential overlap here with forests grazed by domestic stock where the tree cover is over 30%, so such land should be included here as the structure and character of the ecosystems present are determined by grazing.

4. *Unenclosed land used occasionally by sheep or goats but not in regular agricultural use and minimally affected by grazing* (e.g., some blanket bogs and mountain summits in Britain).
5. *Linear or point features on, or adjacent to, farmland that are managed directly or are likely to be highly influenced by farming activities e.g., hedges on farmland and grass strips between fields*[7].
6. *Linear or point features on, or adjacent to, farmland that are indirectly influenced by current agriculture but are not*

[7] The separation of categories 5 and 6 is to some degree arbitrary. But was determined on the basis that class 5 actually had deliberately inputs from farmers, e.g.,cutting hedges. Class 6 will have only indirect effects from farming, e.g. spray drift.

managed actively (e.g., field corners and small woodlands surrounded by agricultural land).

7. *Land not used by agriculture (usually urban herbaceous using the BioHab definition) and managed usually by mowing, e.g., roadside verges, recreation areas and sport fields.*

8. *Land not used by agriculture but maybe managed for forestry, nature conservation except where grazing is involved or urban objectives*

   (a) *Abandoned fields and unenclosed land no longer used by agriculture.* Long term set-a-side could be included here. This category would also include habitats under nature conservation management e.g., wetlands, some salt marshes and heathlands.

   (b) *Land which has never been used by agriculture or managed e.g., steep roadside banks, cliffs and scree.*

   (c) *Forests.* These could be divided into three categories if a relationship was required with intensity of management

      (i) Forests managed regularly often for nature conservation objectives using active management e.g., coppice woods for vernal flowers and for firewood

      (ii) Commercial forests of planted species e.g., Sitka spruce in the UK and Norway Spruce in northern and central Europe. Small recent amenity plantations are not included here as they are still indirectly affected by agricultural practices

      (iii) Forests that have not been managed in recent times, say about 50 years

   (d) *Urban land within the definition provided by the BioHab project (Bunce et al., 2005; 2008)*

It should be realised that the patterns of the different farmed and unfarmed categories in the landscape may vary over distances of a few metres in some regions to hundreds of metres elsewhere. Similarly they may occupy whole landscapes, as in high mountains, or may only be in fragments of only tens of square metres, as in cereal prairies. The typology described above (TABLE 5.8) therefore operates at a landscape level and is to be used to separate the allocation of farmed from unfarmed vegetation plots.

Testing this typology in SEAMLESS firstly showed that the different classes had inherently different vegetation present and that any comparison of biodiversity had to be carried out within relatively homogeneous units. In the present case the analysis would be by the farmed/unfarmed categories and then by vegetation class, which is comparable to the habitat level. The relevance of these conclusions for BioBio is the confirmation of the value of the framework used in the SEAMLESS project.

5.5.3.2. Defining scale

In order to develop a biodiversity indicator dataset at the European level it will be necessary to chose an appropriate spatial and thematic scale. As described in 5.1.2.5 the choice of spatial and thematic resolution defines the limits of the study, i.e., the smallest possible features that can be identified, the area or temporal extent of the landscape and the level of land use detail. Many landscape/habitat indicators are known also to be sensitive to scale and changes in scale (e.g., Bailey *et al.,* 2007a; b; Wu, 2004; Wu *et al.,* 2002). This means that certain habitat indicators may lose their appropriateness with changes in scale. Furthermore, scale affects the appropriateness of habitat indicators for particular organisms or processes. In BioBio, we propose to limit the indicator development to the plot and farm scale. Landscape scale will not explicitly be considered (but may become relevant in HNV case studies, i.e., Spain, Hungary, Bulgaria, Norway, Tunisia). The inclusion of unfarmed habitats will be limited to adjacent hedges, stone walls, etc.

5.5.4. CASE FOR SELECTION OF INDIRECTLY MEASURED HABITAT INDICATORS

"Indirect indicators" in this context are derived from remote sensing (satellite/aerial photographs) or official databases (e.g., farm input statistics).

5.5.4.1. Indicators of habitat quantity - habitat occurrence

The patterns and mosaics that form landscapes are often more easily appreciated when seen from above than when seen from the ground, and it is perhaps not surprising that the term Landscape Ecology was first coined by a geographer who used aerial photographs in his research (Troll, 1939). Since those days, remote sensing has become a standard tool for mapping and monitoring land cover and land use, including both aerial photographs (Avery, 1977; Macaulay Land Use Research Institute, 1989; Cousins and Ihse, 1998; Dramstad *et al.,* 2002; Ihse, 2007) and satellite data (Jones and Wyatt, 1988; Haines-Young, 1992; Wrbka *et al.,* 1999; Jones *et al.,* 2001). Maps created from remote sensing information will have a given spatial scale; with land use types delineated according to specific mapping rules and classification system (see 5.5.3.1). To extract and communicate information from the maps, hundreds of quantitative measures of landscape pattern - landscape metrics - have been developed (McGarigal and Marks, 1995) which provide indicators of landscape composition and configuration.

For landscape indicators to be useful as *indicators of habitat occurrence*, the map classes must relate to identifiable habitat types and the maps must be at an appropriate resolution. Even if habitats can be reliably identified, landscape metrics can only provide truly meaningful *indicators of biodiversity* if the pattern of habitats in the landscape can be linked to biodiversity (Gustafson, 1998; Corry and Nassauer, 2005). This link is often difficult because of limited understanding of the effects of landscape pattern on ecological processes (Wu and Hobbs, 2002). Thus, the decision about what and how to map the landscape may not be straightforward (Arnot *et al.,* 2004).

As landscape metrics respond to changes in scale, their usefulness may depend to a large extent on how the landscape is defined. Bailey *et al.* (2007b) showed that, for simple landscapes, density of habitat patches[8], largest patch index and habitat amount were the most useful metrics to distinguish between different European landscapes. For landscapes with around 14 classes, the largest patch index, edge density, nearest neighbour, proximity index, a shape metric (circle) and a landscape diversity index were most useful. Bailey *et al.* (2007a) explored the role of thematic resolution for a suite of species groups and showed that metrics describing the grain and area occupied by the largest patch (dominance metrics) were suitable to distinguish between landscapes at coarse thematic scales, whereas shape, configuration and diversity indices were more useful at finer scales. At intermediate scales metrics that represented all of these components of landscape pattern were appropriate as landscape descriptors.

A further point to consider when selecting and using habitat indicators is that despite considerable analysis of the topic, many studies have been unable to distinguish clearly between the biodiversity effects of habitat quantity and habitat spatial configuration. This is because the processes that change landscape usually change both quantity and spatial configuration simultaneously (Harrison and Bruna, 1999). For example, fragmentation of natural biotopes is usually caused by a loss of habitat (amount) that makes remaining patches both smaller and more isolated (spatial structure), exposing them to edge-effects (quality). Thus, determining the effects

[8] See TABLE 5.3 for details of specific landscape metrics.

of fragmentation per se is extremely challenging (Fahrig, 2003). Even studies of experimentally fragmented landscapes, where patch size, shape and position in the landscape are controlled, produced results that vary across taxa and across experiments (Debinski and Holt, 2000) and are as likely to have positive effects on biodiversity as negative (Fahrig, 2003). Although lacking consistent field evidence to support ecological theories about the effects of fragmentation per se, the fact that fragmentation in the wider sense is usually tightly linked with habitat loss and habitat degradation (Harrison and Bruna, 1999; Ewers and Didham, 2006) means that the effects of habitat fragmentation on biodiversity are more likely to be negative. Thus, habitat indicators such as habitat patch density, habitat amount, matrix amount, average distance between habitats and habitat proximity indicators may be useful to measure this effect (see TABLE 5.9). However, spatial indicators relating to landscape configuration may not be practical for application at farm scale studies of non-consolidated farm holdings. If BioBio is limited to the farm scale, the spatial arrangement of habitats, which belong to a specific farm is not meaningful. Measuring e.g., the distance between two semi-natural grassland plots of an organic farm makes no sense if between those two plots there is another semi-natural grassland which belongs to another farm and may have a similar quality. Such a measure would have no ecological meaning.

Several authors have documented time lags in the response of species to changes in habitat occurrence (Burel, 1993; Gu *et al.,* 2002; Lindborg and Eriksson, 2004; Helm *et al.,* 2006; Orsini *et al.,* 2008). It is therefore possible that the effects of landscape configuration on species distribution may be under-estimated, since historical landscape data are often not available and species distribution is compared with current landscape structure. If this is the case, indirect indicators of biodiversity based on habitat occurrence may be more sensitive than direct species indicators, providing an earlier warning of changes to come. This is because indirect indicators will show the amount of habitat present in the landscape which may indicate both habitat loss and fragmentation even though indicator species may currently be present in the remaining habitat remnants.

Whilst the term fragmentation bears negative connotations, its corollary may be habitat diversity, which is often viewed as positive for biodiversity – and in fact is one of the aspects of biodiversity to be assessed in BioBio (in addition to genetic and species diversity). Again, there are issues of scale (Whittaker *et al.,* 2001). What may be a "small" fragment for one species could support a large population of another species. The role of different elements in the landscape will vary from species to species, depending on body size, habitat demands and dispersal characteristics (Bunce and Howard, 1990). Although some studies have identified increasing species diversity with increasing landscape diversity (e.g., Meyer *et al.,* 2009), other studies have found contrasting effects for different species groups (Dauber *et al.,* 1993). Jeanneret *et al.* (2003) concluded that there are no general models relating overall species diversity to landscape diversity but that the relationship strongly depends on the organism examined.

A review by Gustafson (1998) concluded that "Pattern analysis techniques are most useful when applied and interpreted in the context of the organism(s) and ecological processes of interest, and at appropriate scales, although some may be useful as coarse-filter indicators of ecosystem function". Indicators based on habitat occurrence are likely to indicate different things for different species and will have to be interpreted carefully. Measures of different aspects of biodiversity (e.g., number of individuals, density, species richness, α diversity, β diversity, γ diversity) may vary in different ways, even for the same taxon in the same region over the same period (Meyer *et al.,* 2009; Weber *et al.,* 2004; Hendrickx *et al.,* 2007). These challenges can make habitat indicators difficult to communicate to stakeholders. However, habitat diversity is an aspect of biodiversity in its own right and will need to be measured in BioBio by means of indicators.

The effects of landscape structure on biodiversity are dependent not only on the amount and distribution of semi-natural habitats, but also on the characteristics of the matrix in which these habitat patches are embedded. Different types of matrix may offer different degrees of resistance to movement (Knaapen *et al.,* 1992; Ricketts, 2001; Fahrig, 2001) and this effect may be very different even for closely related species (Pither and Taylor, 1998, Jauker *et al.,* 2009). In some cases, that which may be considered "matrix" – i.e., the cultivated crop, may provide valuable resources for a short period of time or for a certain stage of the lifecycle. For example, generalist pollinators, such as some bumblebees, require semi-natural habitats to provide nesting sites and a continuous food supply throughout the season (Corbet, 2000), yet may benefit greatly from mass flowering crops such as oilseed rape (*Brassica napus*) (Westphal *et al.,* 2003). Similarly, red clover (*Trifolium pratense*), with its extended flowering season, can be a major forage source for many European longer-tongued bumblebee species (Carvell, 2002). To capture the different uses of different land types within agricultural landscapes, including both cultivated and non-cultivated areas, the OECD have recommended a "habitat-species matrix" indicator (OECD, 2001). To calculate the indicator, each separate use of a habitat type by a species is recorded as one habitat use unit, weighted according to how critical the habitat is for the species in question (Neave *et al.,* 2000).

There has been considerable focus on the spatial arrangement of small patches and corridors of semi-natural habitat in agricultural landscapes, perhaps not least because these systems provide semi-experimental study systems, with a relatively clear patch-matrix structure. Some relationships between species richness and landscape metrics have been documented, in particular, regarding the shape of patches (Heegaard *et al.,* 2007). The consensus nevertheless seems to be that habitat amount and habitat quality are the most important determinants of species richness (Harrison and Bruna, 1999; Liira *et al.,* 2008; Fahrig, 2001; Billeter *et al.,* 2008).

To sum up regarding indirect indicators of habitat occurrence, technological advances have made it relatively easy for us to describe and quantify major land use patterns in the landscape. In addition we have theoretical understanding of various ecological processes that may link habitat patterns to the occurrence of different species or species groups. However, the choice of indirect indicators depends on the ecological process and organisms under investigation. Thus, maps used to generate the indirect indicators must have an appropriate scale and habitat classification system. This requires decisions to be made regarding how the landscape should be classified for the map, the area that the map should cover and the size of the smallest features to be mapped.

5.5.4.2. Potential habitat occurrence indicators for use in organic farming and low-input systems

TABLE 5.9 provides a summary of indicators of habitat diversity and /or of indicators which are linked to biodiversity particularly in organic farming systems. TABLE 5.10 examines each of these indicators in more detail, considering their potential for BioBio and likely disadvantages of their application. A common disadvantage is that there can be scaling issues. However, in BioBio it is proposed that BioHab mapping principles will yield standardised maps from which indicators can be derived, thus minimising scaling problems.

From consideration of the indicators in TABLE 5.10, it is likely that the simplest and most obvious measures are the best. For example, the total area of habitat of the different types, habitat diversity, density of patches, edge density and patch shape heterogeneity. Together they could be used to indicate the complexity of the agro-ecosystem at the regional, landscape or farm scale.

TABLE 5.9. SUMMARY OF INDICATORS THAT RELATE TO HABITAT DIVERSITY OR TO BIODIVERSITY IN CONVENTIONAL, ORGANIC AND LOW-INPUT SYSTEMS

Indicator Type	**Potential Indicators**
Measures of: Landscape or Farm-Scale Complexity	No one measure. Could include: • Complexity of crop structure/field use, • (No. of crops?), • Diversity of land-use types • % of non-crop habitat, • % of arable land, • % of permanent grassland, • % ECA, • Linear element density (hedge/field boundaries) • Continuous hedge length, • Hedge width/height, • Field size , • Patch Number • Patch Size, • Perimeter to Area Ratio, • Patch Shape.
Measures of Landscape/Habitat Heterogeneity	• Number of patches • Landscape/habitat diversity (e.g., Shannon, Simpson, Patch richness)
Measures of Quantity	• Habitat Amount • % Area of Arable Land[9] • Proportion of Arable Land • % Area of Permanent Grassland • Area of Cut Hedgerow • Proportion of Hedgerow Planting • Hedgerow Size
Measures of Linear Feature	• Total Edge Length (e.g., hedges, tree lines, field boundaries) • Edge Density (e.g., hedges, tree lines, field boundaries)
Measures of Patch Characteristics	• Number of Habitat Patches • Patch Size • Patch Size Heterogeneity • Patch Shape

[9] Rotational grassland: sown grassland as part of the crop rotation, which is in place between several months and 2-3 years. Interrupted grassland: is regularly ploughed and re-sown (every 3-6 years), i.e., the plot is continuously used as grassland but it is not permanent grassland.

TABLE 5.10. INDICATORS OF HABITAT DIVERSITY, POTENTIAL APPLICABILITY FOR BIOBIO AND PITFALLS. MEASURES OF COMPLEXITY

Indicator	How Measured	BioBio use?	Pitfalls
Landscape Scale Complexity	Authors suggest: Complexity of crop structure/field use, No. of land-use types No. of crops (including rotational and interrupted grassland),[10] % of non-crop habitat, % of arable land (including rotational and interrupted grassland), % of permanent grassland, % ecological compensation area (ECA) or % AES management, Linear element density (hedge/field boundaries), Continuous hedge length, Hedge width/height, Field size. Additionally, Concepcion *et al.,* 2008	An indicator that shows the contribution of organic farms to landscape complexity which can be associated with higher levels of biodiversity (e.g., Bengtsson *et al.,* 2005). Possibility to construct a landscape or farm scale complexity index. Possibility to look at spatial arrangement of ECA on organic compared to conventional farms. Organic farms have a greater complexity of both crop structure and field use at the landscape scale even in more heterogeneous landscapes (Norton *et al.,* 2009). Non-crop habitats, grassland and high-stem fruit trees cover a greater extent in the locality of organic farms and on the farms themselves (Norton *et al.,* 2009; Steiner and Pohl, 2009). Species richness of ground-dwelling spiders in crop fields was linked to large-scale landscape complexity (percentage of perennial non-crop habitats) regardless of faming system (Schmidt *et al.,* 2005). Species richness increases with percent cover of grassland in the surrounding landscape (Purtauf *et al.,* 2005). Densities (length per unit area) of linear features, both hedges and boundaries, were higher on organic farms (Norton *et al.,* 2009; Steiner and Pohl, 2009). Hedges on organic farms were higher, wider and had fewer gaps than those on their non-organic counterparts, providing well maintained continuous hedges (Norton *et al.,* 2009). The field size was smaller on organic farms (Norton *et al.,* 2009). The diversity of land use types and crops are higher on organic farms (Mansvelt and van der Lubbe, 1999).	No one measure. Quality of maps dependent upon accurate identification of land use/habitat types and will probably require field mapping. How to tell the difference between permanent and rotational grassland without vegetation surveys? Is beyond farm scale. Scaling issues: Indicator values can vary depending upon the thematic and spatial resolution of the map. Thus, the complexity of the landscapes will vary depending on the thematic and spatial resolution. Some measures (see Wu, 2004) are robust and predictable to changes in both grain and extent (patch number, edge length) or predictable to changes in grain but not extent (Mean patch size, edge density) or not very predictable to changes in scale (% landscape occupied by x).

[10] Rotational grassland: sown grassland as part of the crop rotation, which is in place between several months and 2-3 years. Interrupted grassland: is regularly ploughed and re-sown (every 3-6 years), i.e., the plot is continuously used as grassland but it is not permanent grassland.

	found that in landscapes of increasing complexity: Mean patch size decreases, Patch number increases, Field boundary length increases, Patch shape complexity increases, Area occupied by linear elements increase.		
Farm-Scale Landscape Complexity	Measures used to examine small-scale landscape complexity include: Patch size/field size, Edge density/hedge density, Habitat diversity, Perimeter to area ratio.	May provide an indicator of small-scale complexity at the farm scale? Possibility to construct a farm-scale complexity index? Possibility to use individual measures as listed here as habitat level measures of diversity on organic farms? Possibility to include other measures such as those suggested for the landscape-scale complexity? Smaller field sizes and higher densities of hedges provide a high perimeter to area ratio which is correlated to higher species richness of plants, butterflies and carabids; The effect of landscape features was larger in the disturbed habitats (cereal fields and ley) than in the more stable semi-natural pastures (Weibull *et al.,* 2003). Perimeter to area ratio is suggested as an indicator of ecological change for long-term monitoring (Olsen *et al.,* 2007). Small scale complexity (measured as habitat diversity) was associated with butterfly diversity (Weibull *et al.,* 2000).	There is not one simple measure of complexity. Quality of maps dependent on accurate identification of land use/habitat types which may require field mapping. Scaling issues: Indicator values can vary depending upon the thematic and spatial resolution of the map. Thus, the complexity of the farm will vary depending on the thematic and spatial resolution. The metrics, mean patch size and edge density, are predictable to changes in grain but not extent (Wu, 2004).

TABLE 5.10. (CONT.) MEASURES OF HETEROGENEITY

Indicator	How Measured	BioBio use?	Pitfalls
Number of Habitat Patches	Number of patches of the corresponding habitat type	Easy to measure. Can be used to compute other indicators, e.g., density of habitat patches per hectare or habitat diversity. Suggested as an indicator of ecological change for long-term monitoring (Olsen *et al.,* 2007). **Increase** in number of woody habitat patches led to increase in woody species richness **(**Bascompte and Rodriguez, 2001).	The measure is of limited interpretive value by itself because it conveys no information about area, distribution, or patch density. Patch number is affected by changes in matrix amount and will show initially increases in number of patches with increasing matrix amount due to fragmentation effects before then declining. Scaling issues: Indicator values can vary depending upon the thematic and spatial resolution of the map. Thus, habitat patch number will vary depending on the thematic and spatial resolution. Patch number is a relatively robust measure with predictable responses to changes in both grain and extent (Wu, 2004).
Landscape/Habitat Heterogeneity	Many methods, e.g., Simpson's Diversity Index Shannon's Diversity Index Patch Richness (no. of patches of different types within a farm) Patch Richness Density Relative Patch Richness	Easy to calculate. Measurable at the landscape scale, so between farms and across regions, or at the farm scale to examine the mosaic of fields connected by non-cropped habitat, or at the patch scale to examine within patch heterogeneity. The patch scale requires field mapping. Used as a measure of regional, landscape and farm-scale complexity. More bird species richness with increasing mosaic heterogeneity (richness of l types) (Haslem and Bennett, 2008). Habitat heterogeneity is associated with richness of taxonomic assemblages (Bennett *et al.,* 2006), spider species richness (Clough *et al.,* 2005), higher bee diversity (Holzschuh *et al*, 2007), bird species richness or abundance (Atauri and de Lucio, 2001; Freemark and Kirk, 2001; McGarigal and McComb, 1995), lepidopteran species richness (Atauri and de Lucio, 2001) and increases in generalist insect herbivores (Jonsen and Fahrig, 1997). Habitat heterogeneity is considered the key to restoring and retaining biodiversity in temperate agricultural systems (Benton *et*	Scaling issues: Habitat/landscape diversity will vary depending upon the thematic and spatial resolution of the map. Diversity will partly be dependent upon the number of habitat/land-use classes that are defined by the user (Bailey *et al.,* 2007a, b). Changes to diversity values are difficult to predict with alterations to the maps grain size or extent (Wu *et al.,* 2003).

		al., 2003).	

TABLE 5.10. (CONT.) MEASURES OF QUANTITY

Indicator	How Measured	BioBio use?	Pitfalls
Habitat Amount	The total amount of the defined habitat, normally expressed as a percentage, in the farm.	Easy to calculate. Robust and can be measured at the farm, landscape or regional scale. Commonly found to correlate to biodiversity. Evidence that there is more habitat on organic farms and that this may increase biodiversity. Habitat amount had been suggested as an indicator of ecological change for long-term monitoring (Olsen *et al.,* 2007). Used as measure of the complexity of the farm or landscape At a pan-European scale species richness of vascular plants, birds and arthropods (bees, true bugs, hover flies, spiders, carabid beetles) found to increase with the share of semi-natural habitat (Billeter *et al.,* 2008). More woodland birds were found with more native vegetation cover and more open-tolerant birds if more scattered trees (Bennett *et al.,* 2004; Haslem and Bennett, 2008). More bird species richness in general (Haslem and Bennett, 2008; Freemark and Kirk 2001), bumble bee richness (Hatfield and LeBuhn, 2007; Sepp *et al.,* 2004), richness and abundance of perennials, sedges, pteriodophytes and higher nature quality indicator species (Liira *et al.,* 2008), generalist and specialist plant species richness (Krauss *et al.,* 2004) and insect herbivore species richness (Clough *et al.,* 2007) has been found with more habitat in the landscape. Habitat loss in fragmentation studies appears to have large and consistently negative effects on biodiversity (Fahrig, 2003). The amount of organic farming in the landscape may have an additive (species richness) or interactive effect (abundance) on butterflies (Rundlof *et al.,*	What is habitat? Habitat is different things to different organisms and this may lead to definition issues. Can be resolved by rigorous application of the BioHab methodology. The amount of habitat may be more important in less complex modern agricultural landscapes? Scaling Issues: The amount of habitat will be affected by the how the landscape is defined, i.e., number of land use classes (Bailey *et al.,* 2007a, b). Course definitions of habitat may lead to exaggerated amounts of habitat. The amount of habitat is also affected by the spatial definitions of the grain and map extent (Wu, 2004).

		2008). Some species were only found in landscapes with a high proportion of organic farming and this system may reduce the negative effects of semi-natural habitat fragmentation and improve matrix quality especially in homogeneous intensively farmed landscapes (Rundlöf *et al.,* 2006). There is a greater quantity of habitat availability (e.g., woodland, hedge, field margins) on organic farms (Anon, 2005; Gibson *et al.*(2007) and Noe *et al.,* (2005) suggest the use of uncultivated biotope area on the organic farm as a general measure of wildlife habitats. However, Gibson *et al.,* (2007) did not find higher amounts of plant species richness, abundance or diversity in these elements on organic when compared to conventional farms.	
Percentage area of arable land (including rotational and interrupted grassland), Proportion of arable land (including rotational and interrupted grassland), Percentage area of permanent grassland		Easy to calculate. More arable land found on conventional farms. More permanent grassland found on organic farms. Used as measures of the complexity of the farm or landscape. The percentage area of arable land is higher on conventional than organic farms but the area of grass is significantly higher on organic farms (Bates and Harris 2009). Sauberer *et al.,* (2008) found the proportion of arable land to be an appropriate indicator for biodiversity in the agricultural landscapes of eastern Austria.	Permanent grassland is difficult to identify from aerial photographs and will require field work, potentially with vegetation surveys. This information is likely to result from BioHab habitat mapping.
Area of cut hedgerows, Proportion of hedgerow planting, Hedgerow size	Hedgerows directly mapped in the field or from aerial photographs and management and planting noted in the field or through farm interviews.	In some regions, hedgerows are a typical element on organic farms and uncut hedgerows form a larger area of organic than conventional farms (Bates and Harris, 2009). A higher proportion of organic farms carried out hedgerow planting and the size of hedgerows was greater (Bates and Harris, 2009).	

TABLE 5.10. (CONT.) MEASURES OF LINEAR FEATURES:

Indicator	How Measured	BioBio use?	Pitfalls
Length of Edge, Edge Density, Length of linear features		Easy to measure. Could use as a measure of farm-scale landscape complexity (see complexity indicators). Much research that suggests linear features to be higher on organic farms and relates linear features to relate to biodiversity. Have been suggested as indicators of ecological change for long-term monitoring programmes (Olsen *et al.,* 2007). The length of field margins is suggested as an indicator for biodiversity potential on organic farms (.Siebrecht and Hüsbergen, 2009). Woodland dependent birds have been related to the extent of hedges (Bennett *et al.,* 2004), a positive relationship has been found between length of ecotones between cultivated land and forest and the distribution of bumblebee species (Sepp *et al.,* 2004), positive effects of amount of woody border in the landscape have been observed on overall insect species richness in alfalfa fields (Holland and Fahrig, 2000) and species richness has been observed to be affected by corridors and connectivity (Debinski and Holt, 2000) However, increases in field edge density have also been found to explain a decrease in richness of high nature quality species and an increase in richness of annual graminoids (Liira *et al.,* 2008). Therefore, linear elements may not compensate for habitat loss and support disturbance tolerant generalist species.	Scaling Issues: Linear feature measures will be affected by the how the landscape is defined, i.e., number of land use classes (Bailey *et al.,* 2007a, b). Must be standardised.

TABLE 5.10. (CONT.) MEASURES OF PATCH CHARACTERISTICS:

Indicator	How Measured	BioBio use?	Pitfalls
Number of Habitat Patches	The number of patches of a given habitat type within the farm	Easy to measure. Can be used to compute other indicators, for example the density of patches per hectare. A simple measure of landscape or farm-scale heterogeneity. Suggested as an indicator of ecological change for long-term monitoring (Olsen *et al.,* 2007) Bascompte and Rodriguez (2001) found **increases** in woody habitat patches led to increase in woody species richness.	Of limited interpretive value by itself because it conveys no information about area, distribution, or density of patches. Fragmentation can lead initially to increases in patch number before the fragmentation process ultimately results in a reduction in patch number. (See Matrix Amount). Will be affected by the how the landscape is defined, i.e., number of land use classes (Bailey *et al.,* 2007a,b). Course classification will result in habitats being grouped together and forming larger patches.
Patch Size	Many possibilities, e.g min, max, mean, median, standard deviation, coefficient of variation. Can measure mean size agricultural fields.	Easy to calculate. Suggested as an indicator of ecological change for long-term monitoring (Olsen *et al.,* 2007) Siebrecht and Hüsbergen (2009) suggest field size as an indicator for biodiversity potential on organic farms. Wenzel *et al.* (2006) found species number (butterflies, burnet moths) to be higher in larger (>10 ha) habitat remnants in 1972 and 2001.	Will be affected by the how the landscape is defined, i.e., number of land use classes (Bailey *et al.,* 2007a, b)
Patch Size Heterogeneity	This measures the coefficient of variation of the patch size	Easy to calculate. Bascompte and Rodriguez (2001) found increases in patch size heterogeneity led to increases in woody species richness.	Will be affected by the how the landscape is defined, i.e., number of land use classes (Bailey *et al.,* 2007a,b)
Patch Shape	FRAGSTATS (McGarigal *et al.,* 2002) or using the number of shape characterising points (NSCP, Moser *et al.,* 2002)	Shows the landscape complexity. Sauberer *et al.* (2008): found patch shape an appropriate indicator of biodiversity in Austrian agricultural landscapes.	Will be sensitive to scaling issues May be difficult to interpret

5.5.4.3. Indirect indicators of habitat quality

Indirect indicators of habitat quality may include some environmental, site and management factors that can be derived from remote sensing, existing map data or from farm census data. For discussion of farm management practices as indicators of habitat quality – refer to section 4 and to section 5.4 where this is discussed at the species level.

Indirect indicators of habitat quality from remote sensing include the use of red–green–blue (RGB) colour tonal values from false-colour infrared (FCIR) aerial photographs, combined with information on the age of grassland stands, their topographic position, and pH value of the soil (Waldhardt and Otte, 2003). Similarly, spectral information in satellite data are increasingly being used to identify areas of high species richness (Lauver and Whistler, 1993; Rocchini *et al.,* 2005; Rocchini, 2007; Gillespie *et al.,* 2008). However, although technological advances and improved availability are making these methods more successful, they nevertheless require a final stage of validation in the field. They can make field work very much more effective, but cannot replace it.

5.5.5. CASE FOR SELECTION OF DIRECTLY MEASURED HABITAT INDICATORS (FIELD MEASUREMENTS)

"Direct indicators" in this context are derived from field measurements and will include indicators of habitat quantity and habitat quality.

5.5.5.1. Indicators of habitat quantity

In GB the direct measure of habitat quantity used by government is the length of hedgerows /km square. In BIOBIO, habitat quantity will result from the application of the BioHab mapping procedures. Whilst the CORINE Land Cover is often used for estimating habitat extent in Europe the mapping unit of 25 hectares is too large for many habitats and it does not correspond the farm scale approach required in BioBio. There are also issues of quality control between countries although the general patterns across Europe are readily interpretable.

At the farm level that is required in BioBio it seems that there is no alternative to field survey but this can also be integrated with recording habitat quality using vegetation plots. Estimates of the time required can be obtained from experience in Italy, GB, Flanders and Austria. The habitat categories of BioHab can be supplemented by the recent work that enables simultaneous recording of EU Habitats' Directive, Annex 1 habitats.

5.5.5.2. Indicators of habitat quality

Issues of habitat quantity and quality are closely linked. If "habitat" were defined strictly at a species-by-species level, you would ideally include only those areas that met all criteria for suitable habitat, both in terms of local environmental factors and spatial layout. This is impossible in practice, however, both because of the numbers of species, but also lack of knowledge about habitat requirements for individual species. Hence the need to indicate gradients of quality – on the basis that different quality will support different species. According to the OECD (2001), indicators of habitat quality provide information on the quality of different habitats types across agro-ecosystems in terms of their structure, management and use/requirement by wild species.

Direct structure measurements (or "site factors"), according to several studies on non-cutivated habitats such as hedgerows and field margins (OECD, 2001; Michel *et al.,* 2007; Burgio and Sommaggio, 2007) include:

- Physical structure; width of the hedges (m), average height of the canopy (m), cover of the tree layer (index from 0 to 5).
- Vegetation composition; plant species present in each habitat. Calculation of herbaceous, shrub, and tree species richness indices or definition of vegetation types.

Direct management measurements consider farming practice (OECD, 2001). It is difficult to directly monitor farming practices as they occur, therefore, precise land cover descriptions are gathered by undertaking farm interviews. This enables farm practice to be deduced by establishing what has been done to the fields and when (see section 4 on Indirect / farm management indicators).

Some practices can leave evidences in the field that can be directly measured, for example:

- Density of grazing animals.
- Grass burned by herbicide spray.
- Hedge trimming.
- Weeds in crops (-> low herbicide usage).
- Proportion of area covered by permanent vegetation.

However, their actual observation is to some extent subject to arbitrary factors related to the timing of field work (e.g., traces of herbicide which disappears over time) and whilst these observations may be used to corroborate indications made in farm interviews, they cannot be systematically recorded.

According to Dornelas *et al.* (2009), it can be difficult to evaluate the effects of management on communities in different localities due to regional differences in species richness and composition. Because species abundance distributions (SADs) express assemblage structure in comparable units, they can be used to characterize communities irrespective of species composition, and for this reason provide a novel means of assessing the effects of management. Their study is based on the weed community and shows that SADs are informative indicators of environmental heterogeneity in modified landscapes. Weed communities include both rare species, which are often of conservation interest and pose little threat to crops, and abundant species, which can be problematic.

Dornelas *et al.* (2009) recommend using the weed seed bank communities in fields because they are the potential expression of the underlying weed species assemblage and they represent all plants that have successfully reproduced under past management. Due to the crop rotation and management strategies which continuously change field conditions, changes of the weed community in fields is assumed to be quicker and more profound than in natural ones. Therefore, the seed-bank composition is a better representation of the long-term impact of environmental heterogeneity on the vegetation than the actual field flora, the former being more robust to the 'background noise' caused by changes in seasonal weather patterns than the latter (Bàrberi *et al.,* 1998). Information on vegetation characteristics can be obtained from any species lists by applying databases holding information on CSR strategies and Ellenberg values or aliens.

5.5.5.3. Direct measurements related to the use/requirement of habitat by wild species.

Certain directly measurable habitat variables are related to the use or requirement of wild species (TABLE 5.11). For example,

- Avian richness has been observed to be correlated with both shrub richness and percent of tree cover (Luther *et al.,* 2008).
- Organic crop rotations are more diverse (Agreste, 2007) and crop diversity has been related to arthropod diversity (Schweiger *et al.,* 2005).
- Higher bee diversity, flower cover and diversity of flowering plants were recorded in organic compared with conventional fields systems (Holzschuh, 2007). Bee diversity was related both to flower cover and diversity of flowering plants, suggesting plant-mediated effects of the farming system.
- The cover of flowering plants, height of herb layer, percent of bare ground, shrub cover layer, wind protection and inclination have all been suggested as habitat quality measures for butterflies (Krauss *et al.,* 2005). A monophagous habitat specialist was found to be dependent on large habitats with large food plant populations rather than on quality measurements.
- Weed cover in cereal fields has been suggested as an indicator of fauna and flora biodiversity on organic farms (Noe *et al.,* 2005).
- Ellenberg scores of vegetation communities act as indicators of habitat quality and are more effective indicators than landscape structure in less fragmented landscapes (Petit *et al.,* 2004). In fragmented lowlands, the species richness was related to the area of the woodland patch, length of hedgerows, area of woodland whilst in the less fragmented uplands it was related to the Ellenberg score.

TABLE 5.11. DIRECTLY MEASURED HABITAT INDICATORS

Structure Measurements		
Indicator	BioBio Potential	Cons
• Hedge Width (m) • Presence of herbal strip adjacent to hedges. • Average Canopy height (m) • Vegetation composition	Can be derived from BioHab habitat mapping by defining qualifiers accordingly.	
Management Measurements		
Indicator	BioBio Potential	Cons
• Land cover • Density of grazing animals. • Grass burned by herbicide spray. • Hedge trimming. • Weeds in crops (-> low herbicide usage). • Proportion of area covered by permanent vegetation.	Can be derived from BioHab habitat mapping by defining qualifiers accordingly.	Consistent recording of grazing animals and herbicide application is doubtful.
Direct measurements related to the use/requirement of habitat by wild species.		
Indicator	BioBio Potential	Cons
• Cover of flowering plants. • Height of herb layer. • Ellenberg scores.	Can be derived from BioHab habitat mapping by defining qualifiers accordingly.	

• Percent bare ground. • Cover of Shrub layer. • Shrub richness. • Percent of tree cover. • Percent of weed cover. • Number of crops in rotation. • Wind Protection. • Inclination.		

5.5.5.4. Vegetation sampling in farmed and non-farmed land

The vegetation sampling method will be adapted from the protocol of the EU research projects BioHab and EBONE. There are three widely used terms for recording vegetation; relevees, quadrats and plots. Relevees is used in the phytosociological literature, quadrats in the quantitative literature and plots for recording at the landscape level. The EBONE project has three tiers of recording of biodiversity:

A. The landscape level: km squares in EBONE, whole farms in BioBio.
B. The habitat level where complexes of habitats form landscapes.
C. The vegetation level; where different types of vegetation make up the habitats.

In the BioBio project it is suggested that biodiversity recording is undertaken at the habitat and vegetation level and that the landscape unit is the farm. Vegetation plots in BioBio will only be recorded in the following types of land as defined in Table 5.8:

1a) Cultivated land
1b) Enclosed grassland used by stock
3) Open land used regularly by agriculture
4) Open land used occasionally by agriculture
5) Features directly affected by farming
6) Features indirectly affected by faming.

6. FILTERING CRITERIA

6.1. SCIENTIFIC EVALUATION OF THE CANDIDATE LIST OF BIODIVERSITY INDICATORS

The full list of reviewed indicators of the general groups: genetic, species, habitats and indirect – farm management was reviewed by applying scientific selection criteria (TABLE 6.1) in four workshop groups during the Work Package 2 workshop, Aberystwyth University, September 2009. The lists included potential indicators sieved from existing European schemes, namely, Malahide headline indicators (EEA, 2004), IRENA indicators (EEA, 2005), Common Monitoring and Evaluation Framework for Rural Development Programmes/ LEADER (CMEF website) and Streamlining European Biodiversity Indicators (SEBI website). In general, with the exception of selected IRENA indicators, the other indicator sets were developed for landscape, regional and national scales whereas BioBio focuses on field, farm to district (where farms are large or comprise multiple, dispersed holdings) scales. Opportunities for cross-linkage and upscaling to these existing indicator sets will be considered throughout the validation and analysis of candidate indicators, 2010-12.

TABLE 6.1. SCIENTIFIC SELECTION CRITERIA APPLIED TO REVIEWED INDICATORS

A. Genetic diversity indicators	**B. Species diversity indicators**	**C. Habitat diversity indicators**	**D. Farm management indicators**
A. General scientific criteria 1. Sensitivity 2. Reproducible 3. Comparable across different species B. Specific scientific criteria for BioBio 1. Applicable for crops, trees, vegetables, vines 2. Applicable for fodder plants 3. Applicable for husbandry animals	A. Taxonomy (well known at species level) B. Collection method (Standard method (as simple as possible) C. Scale (Set of indicators to cover the scales: habitats, plots, landscape and farms)	A. General scientific criteria 1. Sensitivity 2. Reproducible 3. Comparable across different ecosystems B. Specific scientific criteria for BioBio 1. Applicable at plot and farm scale 2. Relates to ecological function and quality 3. Relates to habitat change (monitoring)	A. Information available in official records (local, national and international) B. Possible to obtain reliable information through farmer interviews C. Proven relationship with direct biodiversity indicators

A three star scoring system was applied for each of the scientific criteria (TABLE 6.1). These were averaged within and across workshop groups. Those with average star ratings > 2.5 (green) and > 2.0 (amber) were selected for further scrutiny by the original Task Group leaders and, applying summary criteria for Stakeholder requirements and an estimate of effort required to collect the data and calculate the indices, the four lists were further evaluated (TABLE 6.2).

TABLE 6.2. SELECTION CRITERIA REFLECTING STAKEHOLDER REQUIREMENTS AND EFFORT TO COLLECT DATA

Stakeholder requirements	Effort Level and Cost Specifics for:
1. Easy to develop, easy to use, not too expensive to apply 2. Appropriate by farmers, consumers and administration 3. Assess the farmer progress and project progress	1. Labour time 2. Equipment type 3. Transport Use 4. Consumables Use

Once collated, the consequent lists of indicators under the four sub-headings were agreed and reviewers were asked to prepare a two page Fact Sheet of the selected indicators for circulation to the Stakeholder Advisory Board members so that they could complete the next stage of the selection by applying the full 18 selection criteria (Table 3.4, see Section 6.3). A full cost effectiveness analysis will be conducted on the short-listed indicators during the field validation in 2010, using actual data from 12 case study regions. The methods of socio-economic analysis are next reviewed.

6.2. COST-EFFECTIVENESS MODELLING. COST OF COLLECTING, ANALYSING AND STORING DATA VERSUS VALUE TO SOCIETY, E.G., MORE EFFECTIVE AGRI-ENVIRONMENT POLICY FORMULATION

6.2.1. ECONOMIC EVALUATION OF BIODIVERSITY: PRIVATE (PRODUCERS AND CONSUMERS) AND PUBLIC (EXTERNALITIES).

There are a number of issues to be discussed under this theme, including cost of biodiversity increasing features and measures, contribution (synergies vs. trade offs) of biodiversity with agricultural production, consumer perception and willingness to pay through product purchase (and issues in mediation, communication by the other actors in the value chain, such as big retailers) and citizens willingness to pay from externalities related to biodiversity. Monetary and non-monetary evaluation techniques are also considered: monetary evaluation of non-market goods; Multi-criteria and participatory-based (e.g., expert panels) techniques used to compare different systems with different biodiversity performances.

6.2.1.1. Indicator selection criteria and choices

The valuation of biodiversity is a crucial issue for three reasons. First, it is able to provide information for the comparison of land use or development options which have an impact on biological diversity. Second, by aiding the comparison of different land use or development options, it supports policy decisions at different – local, regional, national, EU – levels. Finally, as the valuation of biodiversity highlights the many ways through which biological diversity contributes to human life, it is able to raise the awareness of the society. Biodiversity in the agricultural context, however, differs a lot from many other contexts, because it is a joint product of human and natural processes and because the maintenance of the specific biodiversity found in agricultural landscapes requires further human activities (Soini and Aakkula, 2007). Biodiversity can be considered a global (mixed) public good, that is, most aspects of biodiversity are characterised by non-rivalry and non-excludability while in the case of marketed goods and services derived from biodiversity rivalry and excludability prevails (Ostrom, 2005; Bela *et al.,* 2008). Moreover, biodiversity works at different levels, such as genes, species and ecosystems. This makes the valuation of biodiversity more complicated, and necessitates the value of biodiversity to be assessed at different hierarchical levels: from the value realized in market exchange through the total economic value to the potential value provided for the whole humanity and the value stemming from the ability of biological diversity to maintain the long term stability of the biosphere (Gowdy, 1997; Bela, 2008; Nijkamp *et al.,* 2008). Thus, beside economic values, ecological and social/psychological values – or in other classifications instrumental and intrinsic (Nunes and van den Bergh, 2001) or ecological and subjective (Straton, 2006) values – should be also taken into account. Nunes and van den Bergh (2001) identify nine general aspects of biodiversity valuation, which allows to compare different valuation approaches (TABLE 6.3).

TABLE 6.3. NINE GENERAL ASPECTS OF BIODIVERSITY VALUATION

General aspects	Economic valuation	Ecological valuation	Psychological valuation
Perspective	Instrumental	Intrinsic	Intrinsic
Indicators	Monetary	Biological	Non-monetary (attitudinal)
Values	Direct and Indirect	Indirect	Direct and Indirect
View	Resource	Diversity	Resource and Diversity
Subject of valuation	Change	State and Change	Change and State
Geographical context	Local and Global	Local and Global	Local
Level of investigation	All	All	All
Approach	Reductionist	Holistic	Holistic
Participants	Expert and Lay	Expert	Lay and Expert

The BioBio project focuses especially on the ecological valuation of biodiversity which is mirrored by the fact that most of the indicators identified so far come from the biological domain. One of the tasks of the Economic Cross Cutting Theme is, however, to provide an alternative to ecological valuation. To this end, it is worth comparing economic versus psychological – or monetary versus non-monetary – valuation in more detail, based especially on the study of Nunes and van den Bergh (2001).

As the table above suggests, monetary valuation is based on the instrumental perspective which argues that biodiversity is used for instrumental purposes in terms of production and consumption, while non-monetary valuation accepts that biodiversity has a value on its own (intrinsic value). Both monetary and non-monetary valuation aim to take into account direct and indirect values of biodiversity (that is, values stemming from the direct use of biodiversity and values stemming from the ability of biodiversity to provide options for direct use), although monetary values usually underestimate the indirect values. While monetary valuation usually grasps the value of biodiversity through the value of certain biological resources (e.g., endangered species), non-monetary valuation can focus on the variety of life as well. Economic valuation, as is based on neoclassical welfare economics, focuses on the changes of biodiversity when valuing it (e.g., WTP is based on the monetary compensation of a change). Psychological valuation, on the other hand, is able to understand the constant value perceptions people attach to biodiversity. Both monetary and non-monetary valuation can value the diversity of genes, species and ecosystems, although non-monetary valuation is hardly applicable at larger spatial scales but used mainly at the local level. Economic valuation applies a reductionist approach when it assumes that the total value of biodiversity can be disentangled into direct use, indirect use and non-use values. Psychological valuation, however, applies a more holistic approach focusing on the values lying in the integrity, stability and resilience of complex systems. Both monetary and non-monetary valuation involves participants from expert communities and the wider public, however, non-monetary valuation often has a clearer focus on public engagement and participation which might lead to social learning.

If we consider the philosophies behind economic and psychological valuation, further differences can be identified. Monetary valuation methods consider people who value biodiversity as consumers who are rational or boundedly rational and have perfect information; however, many studies have proven so far that these assumptions lead to distortion (e.g., protest answers in WTP studies) (Spash and Hanley, 1995; Spash *et al.,* 2009; Martín-López *et al.,* 2007). Furthermore, economic valuation of biodiversity is based upon the aggregation of individual decisions (values) and often applies discounting, which takes on the problem of inter- and intra-generational equity instead of handling it (Martínez Alier, 2002; Wilson and Howarth, 2002). Psychological valuation, on the other hand, propose a more comprehensive way of valuing biodiversity and the goods and services provided by it, because "when we focus on cultural, memory and linguistic variables we are appraising not only the intrinsic value of ecosystem services, but also their effects on human health or social structures, their aesthetic contributions, and their significance for future generations (O'Hara, 1996)" (Kumar and Kumar, 2008, p. 814).

The next table summarizes the pros and cons for economic and psychological valuation of biodiversity (TABLE 6.4).

TABLE 6.4. ADVANTAGES AND DISADVANTAGES OF ECONOMIC AND PSYCHOLOGICAL VALUATION OF BIODIVERSITY

Economic (monetary valuation)	**Psychological (non-monetary) valuation**
+ Relatively cheap and quick	+ Addresses more dimensions of the value of biodiversity
+ Easily used in decision-making at any of the spatial levels (from local to global)	+ Deals with social equity and cultural / psychological aspects
+ Allows for a direct comparison with monetary values of alternative land use / development options	+ Supports conflict resolution
– Ethical questions concerning discounting and social equity questions	– Participatory / deliberative decision making processes are needed to apply the results
– Methodological questions concerning simplification, aggregation, homo oeconomicus approach	– Relatively timely and costly
– Monetary indicators can point to the opposite directions as biological indicators	– Results can be ambiguous and used mainly at the local or regional level

Two types of indicators can be measured if one would like to value biodiversity in economic or psychological terms: landscape level indicators and farm level indicators. Landscape indicators and models have been used to assess multiple ecosystem services, and their relationship to economic drivers, under different alternative landscape futures (Baker *et al.,* 2004; Naidoo and Ricketts, 2006). Both monetary and non-monetary indicators can be defined at landscape level, although there are only few indicators of this type where data is available for a comparison across Europe (TABLE 6.5).

TABLE 6.5. SUMMARY TABLE OF CANDIDATE INDICATORS AT LANDSCAPE LEVEL

Candidate indicators name	**Candidate indicators details (e.g., measures, scale of applicability etc.)**	**Comments**	**Bibliographic references**
Extra payments for homes in high biodiversity value (green) areas	Hedonic pricing (monetary valuation) method has to be applied to compile this indicator. Existing data based on the sales of real estates at country or regional level can be used, although the price difference for homes in average and high biodiversity value areas can also differ according to other factors (size, age, style etc.) which are difficult to filter out.	Analysis and comparison of existing data bases can be relatively cheap. Although data bases can be very different concerning how detailed they are and which areas are covered (both being rich and poor in biodiversity having similarities in all the other features), which makes the analysis more difficult and methodologically poor.	Nunes *et al.,* 2001. Nijkamp *et al.,* 2008.
Extra payments for travel and stays in high biodiversity value (protected) areas	Travel cost (monetary valuation) method has to be applied to compile this indicator. Existing research results can be used but they are usually made in smaller non-agricultural areas from which general consequences are difficult to draw.	Comparison of available results is difficult as the methodology applied in different countries is rarely similar. To make an original research in all the countries participating in BioBio (or all over the EU) is quite time consuming as detailed questionnaires has to be made and data has to be analysed.	Nunes *et al.,* 2001. Nijkamp *et al.,* 2008.
People's willingness to pay for preserving a protected landscape or species	Contingent valuation (monetary valuation) method has to be applied to compile this indicator. There are a few existing research results but they are quite sporadic and cannot cover Europe. Moreover, existing research results differ in their scope, methodology, way of questioning and focus	It seems quite difficult to compare existing research results (perhaps the method of benefit transfer can be used for this), especially because existing researches do not cover the studied geographical area. To carry out an original research is really resource and time consuming; moreover, the methodology should	Nunes *et al.,* 2001. Nijkamp *et al.,* 2008. Nunes and Van den Bergh, 2001. Turpie, 2003. Garrod and Willis, 1997. Christie *et al.,* 2006.

	of research (e.g., the species which is valued).	be adjusted to the social background. Thus, the societal context can distort the result even if the similar methodology is used.	
Number and extent of seed networks in an area	Non-monetary indicator for indicating the diversity of land races.	This is a new indicator which should be measured in the participating countries.	
WTA values for diverse gardens	A monetary indicator measured by choice experiment methodology. Farmers are asked about their demand for diverse gardens.	Existing results from three rural areas of Hungary (Őrség-Vendvidék, Szatmár, Dévaványa) could be applied to other countries by using the benefit transfer method.	Bela *et al.,* 2007.
The contribution of biodiversity to human well-being (measured qualitatively)	Different non-monetary valuation methods can be used (e.g., focus groups, multi-criteria analysis, citizens' jury, scenario workshop).	Only a few studies have used qualitative valuation methods across Europe. As the applicable methods can be very diverse, the comparative analysis of existing results (and the establishment of a general data base) seems impossible. To carry out this research is time consuming, but the research methodology can be very well adapted to the social context, the comparison of results is relatively easy, although generalization from a few case studies is not possible.	Soini and Aakkula, 2007. Christie *et al.,* 2006. Lindemann-Matthies and Bose, 2008.
Public awareness of inhabitants and farmers	By using qualitative research techniques (interviews, focus groups) it is possible to measure how many times interviewees mention biodiversity and what kind of associations do they attached to it.	Public awareness is an indicator proposed already by SEBI 2010, however, there are no direct indicators developed and measured across countries.	SEBI, 2010 Buijs *et al.,* 2008.

The Millennium Ecosystem Assessment raised the need for indicators, particularly those that are linked to biodiversity and its role to sustain ecosystem services. Very few references directly address the indication of ecosystem services. Facing the challenge of SEBI 2010 and the Millennium Ecosystem Assessment, ecosystem (service) valuation has become an integral component of ecosystem indication towards halting the loss of, and sustaining, biodiversity at the level required to maintain their service provision. Gren *et al. (*1995), for instance, calculated the value of the entire Danube floodplain, mainly with respect to its regulative function, i.e., nitrogen and phosphorous retention: the value was at least 650 million Euros per year (Feld and de Bello, 2006).

6.2.2. ECONOMICS OF MEASURING BIODIVERSITY (COST AND BENEFIT OF MEASURE)

This includes methodologies for cost of measuring indicators and approaches to understand their information content and contribution to decision making. A few papers offer similar analysis even in different fields. Concerning costs, the best (and likely unique) example directly concerning biodiversity is provided in Qi *et al.* (2008). They measure the direct costs of the ecological measurement protocols used in the Farm Scale Evaluation project (impact of herbicide tolerant GM crops on farmland biodiversity). The resulting cost was comprised between 217 and 4548 £ per site depending on the protocol adopted. Other papers attempt to approximate such measures. For example, Cantarello and Newton (2008) sought to identify indicators and evaluate their suitability for assessing the conservation status of forested habitats. Cost-effectiveness of monitoring methods is assessed, where costs are measured in terms of time required, while effectiveness is discussed in terms of distribution of measures and correlation between results of different methods. The issue of benefit or effectiveness is rather more complicated and scarcely treated in the literature. The benefit of indicators can be assessed only in connection with the value of their contribution to decision making. This implies a formalisation and knowledge of a) The monetary values behind biodiversity and its changes; b) The mechanisms of decision in which the information is used.

TABLE 6.6. ASSESSMENT OF BIODIVERSITY INDICATOR PERFORMANCE

Candidate indicators name	**Candidate indicators details (e.g., measures, scale of applicability etc.)**	**Comments**	**Bibliographic references**
Effort (labour)	Working time (hours) per site and protocol	Includes field and laboratory; Cantarello and Newton (2008) propose a different disaggregation	Qi *et al.,* 2008; Cantarello and Newton, 2008
Effort (labour cost)	Labour cost (euro) per site and protocol	Complementary to the previous, but dependent on local salary	Qi *et al.,* 2008
Travel cost	Includes costs of travel time and vehicle	It could be broken down in three (labour time, labour cost and vehicle costs) like in the references, but it seems too detailed	Qi *et al.,* 2008
Overall cost per site and protocol	Aggregate of the previous ones	Can also be aggregated per site	Qi *et al.,* 2008

		only (summing protocols)	

6.2.2.1. Indicator selection criteria and choices

This part is not intended to provide biodiversity indicators, but rather "indicators about the indicators" used to measure biodiversity. The indicators here are intended to quantify ease of understanding/ease of measurement. Refinements may concern either the detailed decomposition of resource used/costs and the reference unit for measure (e.g., site, farm, trial). In fact, for the aim of the project we propose a more detailed measurement. However, this will depend on the choice of the other indicators and related protocols. The issue is treated more in detail in Viaggi and Cuming (2009). The issue of effectiveness is skipped here, but see again Viaggi and Cuming (2009).

6.2.3. ANALYSING, SIMULATING, PREDICTING AGENTS BEHAVIOUR IN BIODIVESITY CONSERVATION IN AGRICULTURE

This is a wide stream of literature in economics and includes:

a) *Ex post* analysis of determinants of the outcome of conservation policy or technology/practice adoption by farmers (e.g., through econometric/statistic models);
b) Simulation tools at farm, regional or state level (e.g., programming models, area models).

Studies vary in terms of instruments used, but have in common a focus on decision mechanisms, either through the analysis of actual (or stated) behaviour (a) or through simulation based on economic models (b). From the perspective of the project they have in common a focus on the decision mechanisms at farm level, rather than on biodiversity. As a result, they usually adopt very simplified measures of biodiversity, most often referring to areas of specific crops or landscape elements. For these reason we consider this field as beyond the scope of the project BioBio.

6.2.4. BIODIVERSITY POLICY AND OPTIMAL POLICY INSTRUMENTS (IN AGRICULTURE)

6.2.4.1. Description of scope and literature

This includes:

a. statistics, monitoring or evaluation information about policy implementation;
b. studies about policy instruments and evaluation of alternative policy instruments e.g., flat rate payment may not require sophisticated indicators while score based payments require very robust and easy to measure indicators.

Point a) is relevant to understand what human action is active in the system under evaluation. It is not necessarily treated in the scientific literature, but rather in policy statistics (e.g., about rural development) and evaluation documents (e.g., evaluation of rural development plans). Point b) is beyond the scope of BioBio, though some considerations about present and perspective policy relevance of biodiversity indicators could be relevant. A classical content is the comparison of the efficiency of different policy designs and/or their impact in terms of effectiveness. The information concerning point a) can be obtained from various sources, the most relevant of which is the past and current framework for the evaluation of rural development measures used by the European Commission (2006). This document identifies the main components of the evaluation framework and defines related indicators. According to this framework, indicators are classified into: 1. Outputs; 2. Results; 3. Impacts; 4. Baselines.

6.2.4.2. Indicator selection criteria and choices

Though most of the literature in this field would not fit for BioBio, some indicators illustrating policy incentives towards specific practices could be relevant. For example, having a contract for organic production under axis 2 of the Rural development plans (RDP) may provide specific incentives to defined farm practices. We rely mainly on the list of indicators available from the guidance documents for the rural development policy presently in force in the EU (European Commission, 2006), as this is the most directly relevant policy area for BioBio (TABLE 6.7). As for Output indicators, only those related to Measure 214 Agri-environment payments are reported. Other measures could be relevant, but output indicators are mostly the same. Indicators are drawn from the guidance note F. The document also provides a detailed description of each indicator.

TABLE 6.7. SUMMARY TABLE OF CANDIDATE INDICATORS PLUS REFERENCES

Candidate indicators name	**Candidate indicators details (e.g., measures, scale of applicability etc.)**	**Comments**	**Bibliographic references**
Number of farm holdings and holdings of other land managers receiving support	Output indicator - Regional level	This is mostly used to area-wide studies and statistics as an indicator to policy action or response. Also by OECD and local sources	European Commission (2006)
Total area under agri-environmental support	Output indicator - Regional level, could be used at farm level		European Commission (2006)
Physical area under agri-environmental support under this measure	Output indicator - Regional level, could be used at farm level		European Commission (2006)
Total Number of contracts	Output indicator - Regional level, could be used at farm level		European Commission (2006)
Number of actions related to genetic resources	Output indicator - Regional level, could be used at farm level		European Commission (2006)

Area under successful land management contributing to: (a) bio diversity and high nature value farming/forestry	Result indicator contributing to axis 2: Improving the environment and the countryside through land management. Regional level, could be used at farm level		European Commission (2006)
Reversing Biodiversity decline: Change in trend in biodiversity decline as measured by farmland bird species population	Impact indicator. Regional level, could be used at farm level		European Commission (2006)
Maintenance of high nature value farmland and forestry: Changes in high nature value farmland and forestry	Impact indicator. Regional level, could be used at farm level		European Commission (2006)
Biodiversity: Population of farmland birds: Trends of index of population of farmland birds	Baseline indicators (objective-related, axis 2)		European Commission (2006)
Biodiversity: High Nature Value farmland and forestry: UAA of High Nature Value Farmland	Baseline indicators (objective-related, axis 2)		European Commission (2006)
Biodiversity: Tree species composition: Distribution of species group by area of FOWL (%coniferous/% broadleaved/%mixed)	Baseline indicators (objective-related, axis 2)		European Commission (2006)
Biodiversity: Protected forest: % FOWL protected to conserve biodiversity, landscapes and specific natural elements (MCPFE 4.9, classes 1.1, 1.2, 1.3 and 2)	Baseline indicators (context-related, axis 2)		European Commission (2006)

6.3. ASSESS THAT CANDIDATE LIST INCLUDES BIODIVERSITY INDICATORS OR INDICES THAT ARE APPROPRIATE FOR POLICY,

MANAGEMENT AND PUBLIC USE AND ARE EASY TO UNDERSTAND

The reviewed indicators that had been ranked with an average of > 2 out of 3 stars and been accepted following further application of summary stakeholder requirements and estimated effort for collection (section 6.1) had been summarised in Fact Sheets for the SAB to review. These were for 10 of 16 genetic, 11 of 36 species, 15 of 58 habitat and 12 of 22 indirect indicators (TABLES 6.8 and 6.9).

TABLE 6.8. DIRECT INDICATORS PRESENTED TO the Stakeholder Advisory Board (SAB) FOR SCRUTINY

A. Genetic diversity indicators	**Animal husbandry**: A1) number and amount of different breeds per species A2) information on breeding practices ("on-farm" bull, artificial insemination,...) A3) where available: pedigree of the herd **Arable crops, legumes and trees** A4) number and amount of different cultivars / landraces / accessions per species (**CultDiv)** A5) information on the origin of cultivars / landraces / accessions (**CropPedDiv)** A6) information on seed propagation practices (on farm multiplication, sharing with neighbours etc) A7) where possible: description of the cultivars based on IPGRI descriptors (through the farmer) A8) where available: pedigree information on the cultivars grown **Grassland species** A9) where available number and amount of different cultivars A10) information on seed propagation practices and amount of re-seeding
B. Species diversity indicators	B1) Flowering plants of cultivated forage and food crops B2) Flowering plants of semi-natural habitats B3) Lepidoptera – butterflies B4) Earthworms B5) Hymenoptera – ants B6) Bird species richness B7) Small mammals B8) Araneae –spiders B9) Hymenoptera, bees and wasps

	B10) Carabid beetles B11) Diptera, syrphidae, hoverflies
C. Habitat diversity indicators	C1) Habitat density C2) Habitat richness C3) Habitat diversity C4) Number of crops in rotation C5) Percentage area of arable land C6) Percentage area of permanent grassland C7) Percent of tree cover C8) Cover of shrub layer C9) Availability of nitrogen, humidity, etc. (Ellenberg values) C10) Weeds in crops C11) Cover of flowering plants: flowers of different colours C12) Vegetation composition: share of valuable habitats C13) Hedgerows; grassy strips between fields; streams, rivers and lakes; stone walls, terrace walls C14) Multispecies grassland swards C15) Grassland quality

TABLE 6.9. CANDIDATE FARM MANAGEMENT INDICATORS (adapted after SABII Meeting, Brussels, 2 Nov 2009)

Code	Factsheet	Candidate Indicator	Unit of measurement	Comments
D1	**DivEnt**	**Diversity of Enterprises**	Number and relative land area of enterprises at the farm level	
D2	**AvStock**	**Average Stocking Rate**	Livestock units per ha UAA	
D3	**MinFert**	**Area without Use of mineral fertiliser**	% UAA without use of mineral-based fertilisers	
D4	**NitroIn**	**Nitrogen - input (or N-Balance)**	kg nitrogen per ha	"input" or "balance" t.b. decided after analysis of data requirements. Account for fertilizer, manure and green manure.
D5	**EnerIn**	**Energy Input**	GJ/ha	
D6	**CertOrg**	**Organic farming**	Certified as organic yes/no	
D7	**AgrEnv**	**Area under agri-environment support**	Agri-environmental measures and area covered	
D8	**IntExt**	**Intensification/Extensification**	Expenditures on fertiliser, crop protection and concentrate feed stuff (€ per ha.) FADN Farm Classification.	
D9	**PestUse-TFI**	**Pesticide use**	Treatment Frequency Indicator (TFI)	

D10	**PestUse-Area**	**Reduced use of chemical pesticides**	Area of UAA without or with reduced use of chemical pesticides	
D11	**FieldOp**	**Field operations**	Frequency and timing of field operations (by operation type)	Operation types: tillage & tillage system, mechanical weed control & plant protection, harvesting, mowing.
D12	**GrazInt**	**Grazing Intensity**	Frequency and intensity	
D13		**Productivity (cereal, milk or meat)**	Tonnes per ha or per LU per year	
D14		**Irrigation**	Practiced yes/no	
Indicators to be derived from Candidate Indicators				
(B1)		**Presence of grass-clover and legumes in the rotation**	Percentage of crop rotation	
(D4)		**Manuring & Green Manuring**	Tonnes per ha and year	
(B1)		**Intercropping and Undersowing**	Percentage of crop rotation	
(D11)		**Mowing**	Frequency and timing of operations	
(D11)		**Mechanical weeding**	Frequency and timing of operations	
(D11)		**Soil cultivation**	Tillage system, timing and frequency	
(B1)		**Crop Diversity**	Number and relative land area of crops at the farm level = Crops in the rotation and their percentage	
B !!		**Relative proportions of livestock species on farm**	Percentage of species	

Farm management indicators will not suffice to characterise the farms and allow for a comprehensive interpretation of the (direct) biodiversity measurements. They will therefore be complemented by background information (TABLE 6.10).

TABLE 6.10. SUGGESTED ADDITIONAL BACKGROUND INFORMATION FOR FARM QUESTIONNAIRES (to be expanded based on requirements for data analysis)

Parameter	**Unit of measurement**	**Comments**
Location of farm	GIS data	
Climate	MAT (mean annual temperature), MAP (mean annual precipitation)	
Soils	typical soil unit for farm	(on plot level more data necessary: pH, clay content, stone content, organic matter content …)
Period since conversion to organic farming	Years	
Farm size	Ha	

Average field size	Ha	
Feeding system		
Farm ownership		FADN/FSS classification
Share of income from agriculture	e.g., full time, part time, …	FADN/FSS classification
Types of marketing	trader, retailer, farmers' cooperative, processing on the farm, direct marketing (farmers' market, farm store, other…) etc.	FADN/FSS classification
Attitude of farmer/ farm family		towards role of farmers in society, importance of biodiversity conservation, role of farms in conservation

The SAB review provided the Project Co-ordination Committee with recommendations for indicators to take forward for evaluation in the WP 3 Case Studies during 2010. The aim was to considerably reduce the number of indicators from 57 but also to retain:

- Scientific rigour and credibility
- Coverage of different spatial scales
- Represent Ecosystem Services
- Potential for cross-validation (e.g., farm management vs species or habitat indicators).

It was also necessary to recognise the differential times from data collection to indicator calculation and presentation for different types of measurement. The selected indicators are chosen as sensitive to year to year changes in farming systems. However, data for some indicators will have a time lag before analysis and reporting:

- Interview and direct observations (3 – 6 months)
- Field removal of samples (6 – 12 months)
- FADN/FSS data (at least two years old).

The final selection must also take into account the need to reflect the broadest possible aspects of spatial scale and trophic level with interactions with farmland habitats (FIG. 6.1).

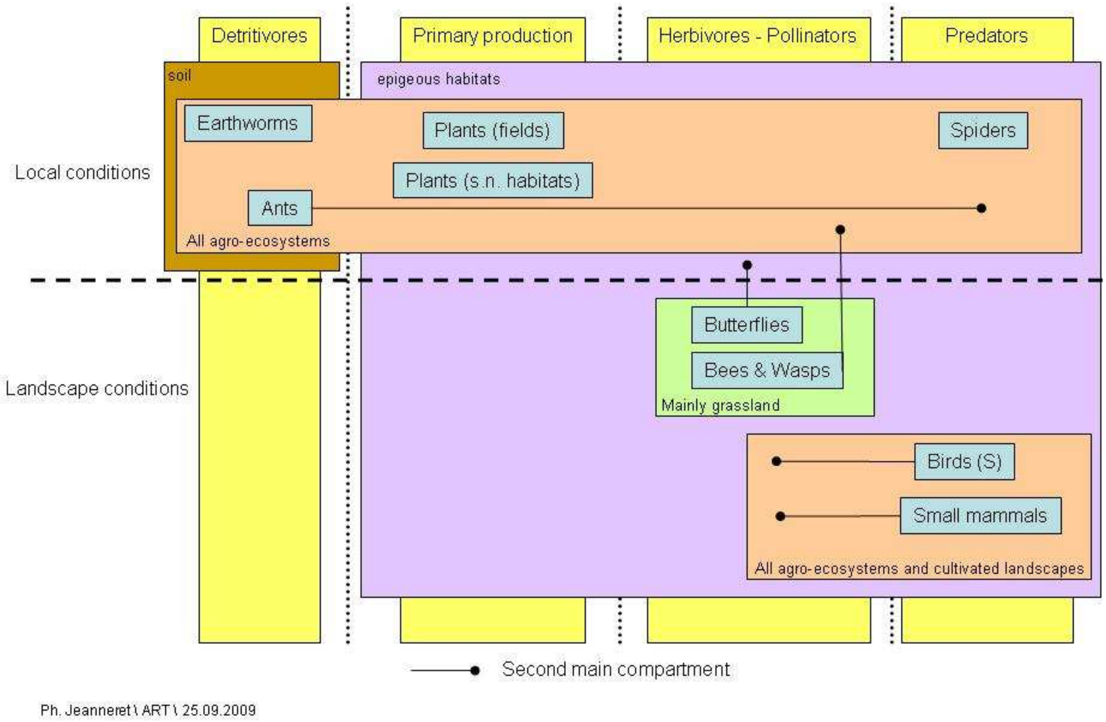

FIGURE 6.1. CANDIDATE INDICATORS (SPECIES GROUPS), HABITATS, SCALES OF INDICATION AND FUNCTION (FOOD CHAIN)

TABLE 6.11 lists the indicators which will be carried forward for field testing in 12 European case study areas in 2010, followingthe deliberatins of the SAB and the subsequent decisions by the Project Coordination Committee (Bruxelles, October 2009). For details on the SAB workshop see the respective report and Deliverable D.7.1.

TABLE 6.11. CANDIDATE BIODIVERSITY INDICATORS FOR EVALUATION IN 12 CASE STUDY REGIONS IN 2010.

A. Genetic diversity indicators	**Animal husbandry**: A1) Number and amount of different breeds per species (**Breeds**) A2) Information on breeding practices ("on-farm" bull, artificial insemination,...) (**Liveprac**) A3) Where available, pedigree of the herd (**LivePedi**) **Arable crops, legumes and trees** A4 + A5) Number, amount and origin of different cultivars / landraces / accessions per species (**CultDiv)** A6) Information on seed propagation practices (on farm multiplication, sharing with neighbours, etc) (**seedmulti**) A7) Where possible, description of the

	cultivars based on IPGRI descriptors (through the farmer) (**CropCuPheDiv**) A8) Where available, pedigree information on the cultivars grown (**CropPedDiv)** **Grassland species** A9) Where available, number and amount of different cultivars (**GrassGenDiv**) A10) Information on seed propagation practices and amount of re-seeding (**ReSeed**)
B. Species diversity indicators	B2) Flowering plants of semi-natural habitats B4) Earthworms B6) Bird species richness (candidate without field validation) B8) Araneae –spiders B9) Hymenoptera, bees and wasps
C. Habitat diversity indicators	C1) Habitat Patch density (**HabDensity**) C2) Habitat richness C3) Habitat diversity (**HabDiv**) C4) Number of crops in rotation (**CropRot**) C5) Percentage area of arable land (**ArableArea**) C6) Percentage area of permanent grassland (**GrassArea**) C7) Percent of tree cover (**Tree**) C8) Cover of shrub layer (**Schrub**) C9) Availability of nitrogen, pH, moisture as Ellenberg values (**Ellenberg**) C10) Weeds in crops (**Weed**) C11) Cover of flowering plants: flowers of different colours (**Quality**) C12) Vegetation composition: share of valuable habitats (**ValueHab**) C13) Linear elements: hedgerows, grassy strips between fields, streams, rivers and lakes, stone walls and terrace walls (**Linear**) C14) Multispecies grassland swards (**Multigrass**) C15) Grassland quality (**GrassQ**)
D. Farm management indicators	D1) Diversity of enterprises at the farm (**DivEnt**) D2) Average stocking rates (grazing livestock units ha^{-1}) on farm (**AvStock**) D3) Area of land without use of mineral-based fertilisers (**Minfert**) D4) N input (**NitroIn**) D5) Input or Direct and Indirect Energy

	for crop production (**Enerln**) D6) Certified as Organic (**CertOrg**) D7) IRENA Indicator 1: area under agri-environment support (**AgrEnv**) D8) IRENA Indicator 15: intensification/extensification (**IntExt**) D9) Pesticide Use – Treatment Frequency Indicator (**PestUse-TFI**) D10) Area of land without or with reduced use of chemical pesticides (**PestUse-Area**) D11) Frequency and timing of field operations (**FieldOp**) D12) Frequency and intensity of livestock grazing (**GrazInt**) D13) Productivity (cereal, milk or meat) D14) Irrigation (practiced or not?)

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Bibliographies prepared as part of the original Task Group review are presented in a supplementary document.

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