How much would it cost to monitor farmland biodiversity in Europe?

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How much would it cost to monitor farmland biodiversity in Europe?

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Running title: Farmland biodiversity monitoring scenarios

Summary

1. To evaluate progress on political biodiversity objectives, biodiversity monitoring provides information on whether intended results are being achieved. Despite scientific proof that monitoring and evaluation increase the (cost) efficiency of policy measures, cost estimates for monitoring schemes are seldom available, hampering their inclusion in policy programme budgets.
2. Empirical data, collected in twelve case studies across Europe, were used in a power analysis to estimate the number of farms that would need to be sampled per major farm type to detect changes in species richness over time for four taxa (vascular plants, earthworms, spiders and bees). A sampling design was developed to allocate spatially, across Europe, the farms that should be sampled.
3. Cost estimates are provided for nine monitoring scenarios with differing robustness for detecting temporal changes in species numbers. These cost estimates are compared to the Common Agricultural Policy (CAP) budget (2014–2020) to determine the budget allocation required for the proposed farmland biodiversity monitoring.
4. Results show that the bee indicator requires the highest number of farms to be sampled and the vascular plant indicator the lowest. The costs for the nine farmland biodiversity monitoring scenarios corresponded to 0·01%–0·74% of the total CAP budget and to 0·04%–2·48% of the CAP budget specifically allocated to environmental targets.
5. *Synthesis and applications*. The results of the cost scenarios demonstrate that, based on the taxa and methods used in this study, a Europe-wide farmland biodiversity monitoring scheme would require a modest share of the Common Agricultural Policy budget. The monitoring scenarios are flexible and can be adapted or complemented with alternative data collection options (e.g. at national scale or voluntary efforts), data mobilization, data integration or modelling efforts.

Key-words: agriculture, agri-environment schemes, biodiversity indicator, common agricultural policy, empirical data, farming system, habitat, sampling design, species trend, power analysis.

Introduction

Numerous scientific papers and research projects address the global biodiversity decline (Butchart *et al.* 2010). In response, political initiatives to reverse declines in biodiversity have increased in number and in their global coverage, e.g. the Aichi Biodiversity targets (CBD 2010) and the establishment of the Intergovernmental Platform on Biodiversity & Ecosystem Services (IPBES). The EU 2020 target of biodiversity enhancement in European agricultural areas was adopted in the greening of the European Common Agricultural Policy (CAP) for the period 2014–2020 (EU Regulation No 1307/2013). Positive effects of policies and adopted measures on biodiversity both at farm and landscape scales are, however, equivocal (Kleijn *et al.* 2011; Lindenmayer *et al.* 2012) and it is generally acknowledged that current monitoring of agri-environment schemes needs to be improved (Pullin *et al.* 2009; Scheper *et al.* 2013). Biodiversity monitoring is required to inform on possible positive or negative side-effects of management practices, external drivers (e.g. climate change), and of other policy measures such as the European renewable Energy Directive (EC 2009/28).

Europe is far from void of biodiversity monitoring schemes, but many operate at a national scale due to governance, language and institutional reasons (e.g. the UK Countryside Survey [http://www.countrysidesurvey.org.uk] or the National Inventory of Landscapes in Sweden [NILS] [Ståhl *et al.* 2011]). Pan-European monitoring schemes do exist but are much more rare, such as the European Land Use and Cover Area Frame Survey (LUCAS) which does not focus on biodiversity (EUROSTAT 2009). There are also citizen-science monitoring networks that provide excellent pan-European biodiversity data which are increasingly used in policy reporting, such as the Pan-European Common Bird Monitoring Scheme (http://www.ebcc.info/pecbm.html) and the European butterfly monitoring (Brereton*,* Van Swaay & Van Strien 2009). Whereas standardization of the sampling and data processing protocols within existing monitoring schemes can be well organized, the interoperability of indicators and data hamper the type of assessments that can be performed with data across monitoring schemes (Henry *et al.* 2008), making biodiversity assessments across taxa, countries and farming types currently precarious. To improve the interoperability of data and indicators, standardization and the implementation of a shared sampling design are considered crucial (Schmeller *et al.* 2015).

Biodiversity monitoring is often regarded as costly making budget constraints a common reason to avoid its implementation (Caughlan & Oakley 2001). However, Naidoo *et al.* (2006) showed that the effectiveness of policies is positively correlated with the presence of monitoring efforts. If decision makers are earnest in their concerns for biodiversity, biodiversity monitoring at multinational scale should be an integral part of the monitoring and reporting criteria of a European policy instrument like the CAP. The actual implementation of a shared farmland monitoring scheme would not only strengthen informed decision making, but it would also demonstrate political willingness to act, counteracting existing doubts on the current approach of the greening of the CAP (Péer *et al.* 2014). The need and willingness to invest in biodiversity information has been expressed at global and European level (Council of the European Union 2010), but a specific level of monitoring expenditure is not defined. Rieder (2011) argues that between 0.5 and 10% of a policy instrument budget should be allocated to evaluation and monitoring, whereas recommendations of the European Commission are at the lower end of this range (0.5%, EC 2004). Whilst cost estimates for the recording of some individual biodiversity indicators exist at regional or national level (see e.g. Mandelik, Roll & Leischer 2010), this information is lacking at international scales.

The objective of this paper is to stimulate the development, the discussion and eventually the implementation of a European farmland biodiversity monitoring system by proposing a sampling design to detect changes in species richness in four taxonomic groups (vascular plants, earthworms, spiders and bees). Measures of agro-environmental schemes are aimed and implemented on individual farms. Therefore, the farm was considered to be the relevant scale for monitoring changes in farmland biodiversity. As specific measures often target specific farm types, a distinction in major farm types was used.

Combining information from a pan-European data set on the variability of species richness for four taxa across major farm types and the spatial distribution of farm types in Europe, enabled an estimation of the number of farms that would need to be sampled to detect changes in species richness. The proposed sampling design for a European farmland biodiversity monitoring scheme is complemented with estimates of the related costs presented in Targetti *et al.* (2014), which were then compared with the CAP budget (2014–2020) to estimate a possible budget allocation for the monitoring scheme. To the best knowledge of the authors, this is the first attempt to provide cost estimates for large-scale monitoring for European policy instruments, using statistical estimates of the number of farms that should be sampled to reliably detect changes in biodiversity.

Materials and methods

Method outline

This study aimed to develop a monitoring scheme in which a 10% change in species richness in 5 years could be identified with a 10% probability error for farmland biodiversity per dominant farm type per region in Europe. To achieve this, the study combined results from four different components. First, we obtained an estimate for the number of farms that should be sampled per region in Europe, by applying a power analysis on empirical data of species richness of four taxa for 12 case studies. Second, we delineated regions in Europe based on the country boundaries, environmental conditions and farm composition. Third, we applied the farm sample size estimates to all regions of Europe with different indicator set options. Fourth, we computed the costs for these monitoring scenarios and compared them to the CAP budget (2014–2020).

The four steps are explained in brief hereafter, a more detailed explanation of methods and uncertainties can be found in Appendix S1 in Supporting Information.

Source of empirical data

In 12 European case studies (i.e. specific farm type in one region), 10–20 farms were sampled. These case studies were part of the BioBio project (Fig. 1, full project description in Herzog *et al.* 2012).

Within each case study region, farms were randomly selected. For the purpose of this paper, the farm types (*sensu* EC 1985) were aggregated into four categories, namely (i) field crops and horticulture, (ii) specialist grazing livestock, (iii) mixed crops and livestock and (iv) permanent crops.

Farms were sampled using an indicator set which was developed with stakeholders and includes minimal information redundancy (Herzog *et al.* 2012, ch. 2). The indicator set contained 23 indicators spanning four categories: genetic, species and habitat diversity and farm management, of which the species category included sampling of four taxa: vascular plants (from here on referred to as plants), earthworms, spiders and bees (Herzog *et al.* 2013). Farmer interviews and habitat mapping were done for all the land managed by the farmer. Per farm, each habitat type was randomly sampled once for all of the four taxa on the same location. Vegetation samples (10 m × 1 m in linear and 10 m × 10 m in areal habitats) consisted of recording all plant species and allocating cover estimates at 5% precision. Earthworms were sampled via extraction for 10 minutes with an expellant solution (diluted allyl isothiocyanate: AITC) and then hand sorted for 20 minutes. Three subsamples were taken (30 cm x 30 cm x 20 cm deep) during one visit. Spiders were suction-sampled from soil surface and vegetation using a modified leaf blower (Stihl SH 86-D). On three different days, five areas of 35.7 cm diameter were sampled within each selected habitat. Bees were sampled during good weather conditions with a handheld net along a 100 m × 2 m transect for 15 min. Bees were sampled on three different days. A more detailed description of the standardized sampling protocols can be found in Dennis *et al.* (2012).

Species richness was computed per taxa per farm (Fig. 2). Means and standard deviations of the observed species richness were computed using species rarefaction-extrapolation curves (Chao *et al.* 2014). An example of the variation in species richness within a case study region is shown in Figure 3. Species accumulation curves for all case studies and all four taxa are presented in Appendix S1.

Budgetary cost calculations and estimates

The costs and the number of hours spent preparing fieldwork, collecting data and processing field samples (i.e. taxonomic sorting and identification) were recorded and used to compute the average efforts required for sampling a standardized farm (Targetti *et al*. 2014). The costs of monitoring farms throughout Europe were estimated using labour cost differences between European countries (Targetti *et al*. 2012). The estimation of the total budget required per sampled farm considered five different components: data collection, supervision, data processing and reporting, data quality assurance and administration (Busch & Trexler 2003). The quantification method for each component can be found in Appendix S2.

Required number of farms that need to be sampled

Based on the variability of the empirical data for the four taxa, estimates could be made of the number of farms required to be sampled to detect statistically significant trends in species richness per major farm type: the required farm sample size.

Required farm sample sizes were computed for detecting a change in the average species richness for each of the four taxa between two consecutive sampling rounds. The variance of the estimated average difference $V(\bar{d})$ in species richness between two sampling rounds is given by the summed variances of estimated average species richness found in each sampling round minus their covariance (Brus & Noij 2008):

$$V(\bar{d}) = Var(\bar{y}_2 - \bar{y}_1) = Var(\bar{y}_1) + Var(\bar{y}_2) - Covar(\bar{y}_2, \bar{y}_1) \qquad \text{eqn. 1}$$

The variance of the estimated average species richness ($Var(\bar{y}_1)$ and $Var(\bar{y}_2)$ in equation 1) is determined by the variation of the species richness per farm in the sampled population of farms, the sample size (number of observed sampling units [farms]) and the type of sampling design (e.g. simple random or stratified random). Since farms were selected fully randomly within case study regions, the variance of the estimated average species richness in sampling round 1, can simply be estimated by:

$$Var(\bar{y}_1) = \frac{S_1^2}{n} \qquad \text{eqn. 2}$$

With S_1^2 being the population variance of the species richness per farm in sampling round 1, and *n* the sample size (number of observed farms per sampling round). Using the means and standard deviations of species richness per farm, derived from the rarefaction procedure, 1000 random sets of species richness for each farm were drawn from a normal distribution. For each set, the population variance per case study region was computed.

The covariance of the two estimated averages (third term in Equation 1) depends on the correlation of the species richness per farm in the two sampling rounds and the proportion of farms that is revisited and observed at both times, referred to hereafter as the matching proportion. The stronger the correlation and the larger the matching proportion, the larger the covariance and the smaller the variance of the estimated change in average species richness. For simple random sampling, the covariance of the estimated average species richness in the two sampling rounds equals (Brus & Noij 2008):

$$Covar(\bar{y}_2 - \bar{y}_1) = \frac{S_{1,2}^2 \cdot p}{n} = \frac{r_{1,2} \cdot S_1 \cdot S_2 \cdot p}{n} \qquad \text{eqn. 3}$$

With $S_{1,2}^2$ being the population covariance of the species richness per farm in sampling round 1 and 2, *p* the matching proportion, and $r_{1,2}$ the correlation of the species richness per farm in sampling round 1 and 2. The population standard deviations in two sampling rounds S_1 and S_2 were assumed to be equal. The matching proportion was assumed to be 80% and the correlation between the first and the second sampling round, $r_{1,2}$, was estimated to 0.9 for plants and 0.75 for the three invertebrate groups based on empirical time series of species richness of previous projects (Aviron *et al.* 2009; M.W. Lüthi, unpublished results). Since these values are based on relatively few data, an uncertainty bandwidth of 0.1 was assumed. This was incorporated by drawing, for each of the 1000 random sets of species richness, a temporal correlation from a uniform distribution of 0.85–0.95 for plants and 0.7–0.8 for the invertebrates.

Finally, the following requirements on the quality of the statistical tests were defined: the probability of wrongly identifying a 10% change in the total number of species should be smaller than 10% (type I error); the probability of not identifying an actual change of 10% of the average species richness should also be smaller than 10% (type II error). Given these requirements, the sample sizes can be computed by a power analysis (Brus & Noij, 2008) with a 95% confidence interval that includes the uncertainty of the original species richness data and the uncertainty bandwidth of the temporal correlation. The power analysis is based on two key equations (Brus & Noij 2008; Brus *et al.* 2011), one to compute the critical value for the difference beyond which the null-hypothesis H_0 $\bar{d}=0$ is rejected (i.e. there is a 10% change in species richness):

$$d_{\text{crit}} = \Phi^{-1}\left(1 - \alpha/2; 0; V(\bar{d})\right) \qquad \text{eqn. 4}$$

And a second one to compute the power of the test (required to be above 90%):

$$1-\beta=\Phi\left(d_{\mathrm{crit}};\bar{d};V(\bar{d})\right) \qquad \text{eqn. 5}$$

Here α refers to the type I error, β to the type II error. $\Phi^{-1}\left(1-\alpha/2;0;V(\bar{d})\right)$ (equation 4) is the quantile corresponding with a cumulative lower probability of 1- $\alpha/2$ for a normal distribution with mean 0 and variance $V(\bar{d})$. $\Phi\left(d_{\mathrm{crit}};\bar{d};V(\bar{d})\right)$ (equation 5) is the cumulative lower probability of d_{crit}, for a normal distribution with mean $\bar{d}$ and variance $V(\bar{d})$.

Sampling design at European level

To allow for stratified sampling of dominant farm types, 25 countries in Europe were divided in homogeneous regions (Fig. 1) (Jongman *et al.* 2012). Region delineation was determined by the farm types (according to the Farm Accountancy Data Network, FADN), country boundaries, and the environmental zone (Metzger *et al.* 2005). The spatial units used were the European Nomenclature of Territorial Units for Statistics level 2 (NUTS2). FADN farm types were ranked based on their surface. Country boundaries were used to take into account national differences in the agro-environmental schemes. Each NUTS2 region was described by one to four dominant farm types (based on a cover of at least 75% of the total utilized agricultural area). Per country, comparable NUTS2 regions were merged while respecting boundaries determined by different environmental zones. A maximum of five regions per country was set to avoid having too many small regions (Jongman *et al.* 2012). The composition of dominant farm types per region can be found in Appendix S3.

The smallest reporting unit for monitoring was the “Farm type per region” with the required number of farms to be sampled being expressed as the percentage of the total number of farms per farm type per region. In compliance with existing recommendations (Elbersen *et al.* 2010), a minimum sample size of 15 farms per farm type per region was retained.

Percentages of the total number of farms of that farm type per region could only be derived for the nine case study regions for which FADN data was available on the regional farm composition, namely Austria, France, Germany, Hungary, Italy, the Netherlands, Spain (Dehesa), Spain (olives) and Wales.

A five-yearly frequency of monitoring (sampling interval) was assumed following the recommendation of the European Biodiversity Observation Network project (Brus *et al.* 2011). According to the temporal sensitivity of the Essential Biodiversity Variables (Pereira *et al.* 2013), this frequency is in line with the dynamics of important biodiversity variables such as «Species distribution», «Ecosystem structure» and «Community composition». Instead of sampling all farms once per five years, each year, 20% of the farms would be sampled over a five year period to ensure a continuous stream of data, to allow for a more resource-efficient approach and to reduce the effect of annual climate variability.

Indicator scenarios

Nine scenarios were developed to allow for comparison between different options for information output based on three different indicator sets and on three levels of biodiversity data robustness (Fig. 4). The scenarios were applied to all identified regions in Europe. This implies the underlying assumption that the sampled farms were on average representative for all of Europe and ignores regional variability in species richness across Europe. This crude assumption was necessary because no other data sets were available to allow for a more sophisticated extrapolation method. For more reflection on the impact of this assumption see Appendix S1.

[Include Figure 4 here]

There are three scenarios to consider: a full indicator set, a full indicator set without bees and a reduced indicator set (only plants). For each indicator set, three additional scenario options were developed using the estimates of the required farm sample size per species indicator per farm type. For the High, Average and Low scenario options, respectively, the highest, the average and the lowest sample size percentage of all four taxa per farm type were applied. Whereas the High scenario option offers a first estimation for an effective monitoring scheme, the Low scenario reflects a case in which minimal monitoring is organized at European level. It is assumed that countries will then develop complementary monitoring at national or regional level.

The combination of options leads to nine cost scenarios for a European farmland biodiversity monitoring scheme with different percentages of farms of a farm type that should be sampled per region and with different information outputs.

Comparison of cost scenarios with the CAP budget

To compute cost estimates, the required farm sample sizes of the scenarios were multiplied by the monitoring costs for a standardized farm for each country (Table S2.2 in Appendix S2). The computed costs were placed into the context of the budget allocated to environmental and biodiversity objectives of the CAP for 2014–2020.

The total CAP budget for First and Second Pillar measures is 408 billion Euro for the period of 2014–2020. The “green” budget, which are the funds allocated for environmental and biodiversity targets, makes up 30% of the total budget (the "greening" package of Pillar 1 and earmarked budget of Pillar 2 [Péer *et al.* 2014]). The total "green" budget was estimated at 122·5 billion Euro for the whole period with an indicative annual budget of 17·5 billion Euro.

Results

The estimated number of farms that should be sampled for the detection of a 10% change in species richness per farm type over a five year period differed between case studies and between farm types from 19 to 465 farms. In general, monitoring bees required the largest, and monitoring plants the smallest number of farms to be sampled. On average the Permanent Crops farm type required the largest farm sample sizes (Table 1).

The required farm sample size in the High scenarios mostly followed the percentage of farms that should be sampled for the bee and plant indicators respectively (Table 2). Only in the case of the High scenario for Specialist grazing livestock, the earthworms showed the highest variability, requiring a higher number of farms to be sampled for a representative and reliable estimate.

Depending on the scenario chosen, approximately 184k (High scenario, full indicator set), 38k (Medium scenario, full indicator set exclusive bees) and 5·6k (Low scenario, reduced indicator set) farms would need to be sampled, which corresponds to 6·3%, 1·3% and 0·2% respectively of the total number of European farms. The difference between the full set with and without bees for High and Low scenarios is 77k and 15k farms, respectively.
An implementation of the full indicator set for the High scenario, would require 0·74% of the CAP budget and 2·48% of the "green" CAP budget (443 Mio € annually) (Table 3). Not monitoring the bees would reduce the costs considerably (a cost reduction of 79-126 Mio € per year), namely to 0·53% of the CAP budget and to 1·75% of the "green" CAP budget. The reduced indicator set for the Low scenario would require 0·01% of the total CAP budget and 0·04% of the "green" CAP budget (7 Mio € annually).

In general, the estimated CAP budget allocation in seven of the nine scenarios remained below the lowest budget allocations proposed in the literature (i.e. the European Commission proposed 0·5% [2004]). When considering the "green" CAP budget, five of the nine scenarios fulfilled this criterion.

Discussion

The results provide an informed estimate of the required sampling design, sample size and costs for farmland biodiversity monitoring for Europe. Depending on the scenario chosen, between 6·3% and

0·2% of the total number of European farms would need to be sampled, which would require between 0·74% and 0·01% of the CAP budget (Table 3). Of the three fauna indicators, the bees demonstrated the highest data variability and therefore required the largest farm sample size.

Estimates are contingent on several assumptions and simplifications which do not necessarily cover the expected complexity of reality. The proposed sampling design is not presented as the ideal monitoring scheme, but rather as a starting point for discussions and further refinements. For this purpose, the estimates are presented at the regional scale (Appendix S1), to provide input for the development of or to complement existing monitoring schemes at national or regional scales.

Validity of the results

The first important methodological limitation is that the estimates of the required farm sample sizes are based on species data from only four taxa, which serve as proxies for the numerous other species depending on European farmland. The choice of plants, earthworms, spiders and bees as farmland biodiversity indicators was based on scientific robustness, iterative stakeholder consultations and feasibility (Herzog *et al.* 2013). These criteria increase the potential acceptance and implementation of the indicator set (Danielsen *et al.* 2010). Future monitoring could increase the number of taxa included or invest in data integration between existing monitoring schemes to increase the sensitivity for specific changes in agricultural management practices (Henry *et al.* 2008).

A second limitation is that the data used were gathered using a single sampling approach whereas the sampling techniques could be revised to reduce data variability (for bees see e.g. Fortel *et al.,* 2014). Additionally, the proposed monitoring scheme uses the farm as a monitoring unit to focus on the scale at which agricultural management decisions are taken. For many biodiversity and ecosystem service estimates, the inclusion of information on larger-scale processes requires also monitoring at a landscape scale (Geijzendorffer & Roche 2013; Schneider *et al.* 2014). The cost estimates indicate that even if additional monitoring efforts at landscape scale doubled monitoring costs, six out of the nine scenarios would still remain below the 0·5% boundary of the total CAP budget.

The third important limitation is that the empirical data base stems from only twelve case studies, collected in one year. For the extrapolation, the variability of species diversity was assumed to be similar per farm type throughout Europe. As a consequence of the small empirical data base in comparison to the total number of farms in Europe, the estimated farm sample sizes should be considered as coarse rather than precise indications, and monitoring cost estimates should be treated with caution. Still, the existence of an empirical data base – albeit small – is a major asset to evaluate the effort needed to implement a monitoring scheme. The presented findings should be considered as a starting point for the urgently needed debate on the feasibility of a European biodiversity monitoring scheme (Council of European Union 2010).

Monitoring scenarios

The full indicator set (four taxa, habitat and farm management indicators, including genetic diversity) was developed based on minimum information overlap in the BioBio project. It covers five of the six Essential Biodiversity Variables (EBV) classes (Pereira *et al.* 2013), namely Genetic Composition, Species Populations, Community Composition, Ecosystem Function and Ecosystem Structure. This indicates good overall coverage of farmland biodiversity, in comparison to the reporting for the Habitat Directives which covers 3 EBV classes (Geijzendorffer *et al*., 2015). The proposed monitoring scheme was developed to capture broad biodiversity trends to assess the influence of large scale changes such as adaptations of European policies like the CAP reform. With 0·74% of the CAP budget (2·48% of the budget allocated to "green targets"), information about the biodiversity status on 6·3% of all farms in Europe could be obtained (the High scenario and full indicator set option).

The proposed farm sample sizes would allow to detect a 10% change in species richness per farm type per region over five years, which is a rather crude in comparison to the annual change of 1% required for the monitoring of red list species and threatened habitats according to the European Habitats Directive (EC 2005). However, whereas the red list monitoring focuses on the monitoring of individual species, the presented sampling design aims to detect large changes in species richness per taxa across many different habitat types on farmland under dynamic farm management practices per region and the 10% change in species richness should therefore be considered as a starting point rather than an aim per se. Although this study focused on species richness, as a sole indicator for trends in biodiversity it is obviously limited and further work such as on the EBVs (Schmeller *et al.* 2015) could identify other indicators of importance for farmland biodiversity. Some of these indicators, involving e.g. species identity, could already be quantified from the data gathered with this monitoring protocol, others might require complementary data and/or monitoring. According to the results, the 10% error probability is only achieved for all four taxa under the High scenario. The required farm sample size estimates could be further adjusted by taking into account regional species pool patterns, by adjusting for the spatial biodiversity patterns within Europe (e.g. earthworm distribution patterns [Entling *et al.* 2012]) or by including alternative sampling methods.

Ideally the proposed monitoring scheme would not be implemented standalone, but serve as a backbone for the integration of data from existing monitoring scheme to further strengthen the interpretation of trends on farmland. Especially the presented Low scenarios and the reduced indicator set options should be complemented by additional targeted monitoring; for instance by focusing on endangered species, or on biodiversity hotspots or sinks (Kleijn *et al.* 2011), by using remote sensing information (Duro *et al.* 2007) or by integrating them with existing monitoring schemes. Still, the focus of the proposed monitoring scheme, namely detecting the impact of changes in management (resulting from policy measures) on farmland biodiversity should be considered. For instance, the proposed monitoring design can be combined with bird data, but the high mobility of birds and their dependence on landscape patterns instead of individual farms, restrict the potential of data integration.

The three invertebrate groups included in the proposed full indicator set (earthworms, spiders, bees) are related to major ecosystem services (decomposition, pest control, pollination) which are particularly relevant in an agricultural context. The reduced indicator set obviously lacks this information. It is nonetheless a commonly used combination of indicators (i.e. habitat and plant data) as proxies in biodiversity monitoring (the UK Countryside Survey (http://www.countrysidesurvey.org.uk), the Swedish NILS [Ståhl *et al.* 2011]). The reduced indicator set still comprises farm management information, which allows analysis of causal relationships between changes in species richness and agricultural practices. Although methods for cross monitoring scheme assessments are not yet well developed (Henry *et al.* 2008), already the reduced indicator set including environmental and management information, plant and habitat data could provide a central backbone for data integration of existing monitoring schemes and could be linked to alternative fauna indicators.

Recommendations for future monitoring

Monitoring is not only needed to determine progress towards an objective, but can also render investments more effective, like in the case of controlling invasive species (Bogich, Liebhold & Shea 2008), the protection of nature areas (Balmford & Gaston 1999) or in avoiding costly (irreversible) losses (Armsworth *et al.* 2012). The presented farmland biodiversity monitoring scheme provides a starting point for further refinement and planning purposes at European, national or regional scale. The full indicator set originated from an extensive stakeholder consultation process followed by an information redundancy analysis. Therefore, decisions to include fewer indicators or lower sampling densities should be done only after extensive additional analysis.

There is potential to use the proposed sampling design to integrate data from different monitoring schemes, as well as that the outputs of the monitoring are likely to inform multiple policy objectives rather than just the CAP. Regardless of the potential, the implementation of the proposed monitoring scheme seems already economically feasible and sharing of its costs across policy instruments politically attractive, especially for a land use sector that is supposed to provide important ecosystem services for the future.

Adaptation of monitoring schemes over time is common practice (see for instance LUCAS [EUROSTAT 2009] or the NILS [Ståhl *et al.* 2011]) to improve data collection efficiency and to ensure the relevance of data collected with regards to new changes in policies, agricultural management or new biodiversity trends, e.g. the recently identified bee mortality. Whereas such adaptations potentially cause problems in terms of interoperability of data over years, it is unlikely that everything can be foreseen in detail in advance and proposed monitoring schemes should have a certain degree of flexibility (Lindenmayer & Likens 2009). The monitoring scheme proposed in this paper can be adapted by changing methods, adding or removing indicators, adding or removing regions or countries and by adjusting the number of farms to be sampled.

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Supporting Information

Additional supporting information may be found in the online version of this article.

Appendix S1. Method description, errors and uncertainties

Table S1.1. Summary of the sampling effort over the BioBio case studies.

Table S1.2. Overview of required farm sample size estimates per case study.

Fig. S1.1. Overview of the input data and assessments performed in this study.

Fig. S1.2. Overview of the different steps in the collection of biodiversity data.

Fig. S1.3. Accumulation curves for plant, earth worm, spider and bee species richness in the 12 case studies.

Appendix S2. Cost estimations for monitoring

Table S2.1. Adaptation of costs from the monitoring pilot phase.

Table S2.2. The costs for biodiversity monitoring of a standardized farm (in Euro).
Fig. S2.1. Estimated cost allocation for five budget components.
Appendix S3. Spatial distribution of farms and regions delineation within Europe
Table S3.1. Number of farms per farm type per European region.

Data Accessibility: The species richness data used in this study are available from the Dryad Digital Repository: http://dx.doi.org/10.5061/dryad.kc688 . The used cost data can be found in Appendix S2 and in Targetti et al. 2014.

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Tables

Table 1. Required sample size (number of farms to be sampled) per case study per species indicator to identify a 10% change in species richness in five years

Case study region			Estimated number of farms to be sampled within the case study region and of the indicated farm type to allow for the detection of a 10% change in species richness in five years (confidence interval 95% included in brackets).			
No	Country	Farm type	Plants	Earthworms	Spiders	Bees
1	Austria	Field crops and horticulture (n=16)	52 (36 -68)	87 (48 – 126)	105 (77 – 133)	427 (274 – 580)
2	Bulgaria	Specialist grazing livestock (n=16)	46 (35 – 57)	142 (61 – 223)	65 (38 – 92)	148 (95 – 201)
3	France	Field crops and horticulture (n=16)	37 (27 -47)	22 (12 – 32)	22 (13 – 32)	137 (101 – 173)
4	Germany	Mixed crops and livestock (n=16)	27 (18 – 36)	24 (10 – 38)	42 (26 – 58)	465 (239 – 691)
5	Hungary	Specialist grazing livestock (n=18)	37 (27 – 47)	356 (250- -462)	239 175 – 303)	247 (115- -379)
6	Italy	Permanent crops (n=18)	25 (16- -34)	22 (101 – 341)1	144 (76 – 212)	167 (105 – 229)
7	The Netherlands	Field crops and horticulture (n=14)	29 (19 – 39)	110 (39 – 181)	197 (132 – 262)	164 (31 – 297)
8	Norway	Specialist grazing livestock (n=12)	20 (14 –b26)	38 516 – 60)	42 (25 – 59)	50 (22 – 78)
9	Spain	Specialist grazing livestock (n=10)	19 (13 – 25)	123 (35 – 211))	47 (27 – 67)	77 (30 – 124)
10	Spain	Permanent crops (n=20)	140 (113 – 167)	226 (148 – 304)	172 (133 – 211)	279 (164 – 394)
11	Switzerland	Specialist grazing livestock (n=19)	50 (39 – 61)	27 (12 – 42)	97 (67 – 127)	129 (82 – 176)
12	Wales	Specialist grazing livestock (n=20)	22 (16 – 28)	22 (11 – 33)	39 (28 – 50)	59 (22 – 96)

Table 2: Required farm sample percentage for each of the four farm types for the full and the reduced indicator sets and for the High (H), Average (A) and Low (L) scenarios

	Full indicator set			Full indicator set excl. Bees			Reduced indicator set		
	H	A	L	H	A	L	H	A	L
Field crops and horticulture (n=3) [%]	5·12	1·96	0·59	3·45	0·87	0·16	0·62	0·32	0·16
Grazing livestock (n=3) [%]	10·77	2·72	0·87	10·77	2·72	0·57	1·12	0·42	0·23
Mixed (n=1) [%]	4·91	4·91	4·91	0·44	0·44	0·44	0·28	0·28	0·28
Permanent crops (n=2) [%]	1·70	0·75	0·52	1·38	0·75	0·52	0·85	0·28	0·06

Table 3: Monetary and relative cost estimates for the nine sampling scenarios, in relation to the total CAP budget (2014–2020; 408·3 billion Euro) or to the part allocated to environmental and biodiversity targets, the "green" CAP budget (122·5 billion Euro). Numbers in grey present budget shares below 0·5%, the lowest allocation found in literature (EC 2004)

Reference budget	**Scenarios options**	High farm sample size option	Average farm sample size option	Low farm sample size option
Annual cost estimations for the 5 years rolling survey	Full indicator set	Mio € 433	Mio € 179	Mio € 103
	Full set excl. Bees	Mio € 307	Mio € 85	Mio € 24
	Reduced indicator set	Mio € 28	Mio € 13	Mio € 7
Percentage of the total annual CAP budget	Full indicator set	0·74%	0·31%	0·18%
	Full set excl. Bees	0·53%	0·15%	0·04%
	Reduced indicator set	0·05%	0·02%	0·01%
Percentage of the annual CAP budget allocated to green targets	Full indicator set	2·48%	1·02%	0·59%
	Full set excl. Bees	1·75%	0·15%	0·14%
	Reduced indicator set	0·16%	0·08%	0·04%

Figures

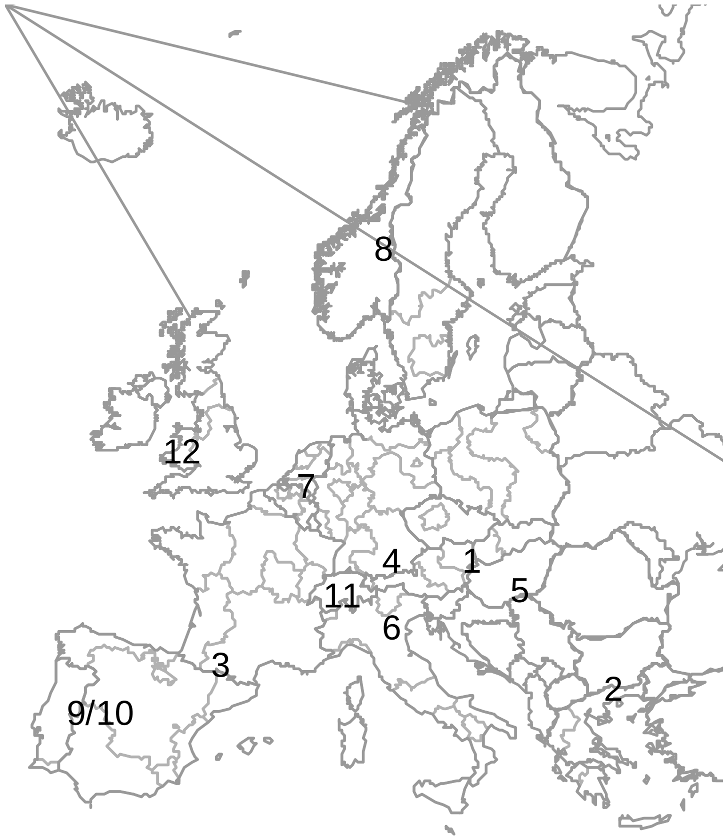


Figure 1: Overview of the case study regions and the zones that served to develop the spatial sampling design. Numbers of case studies correspond to those in Table 1.

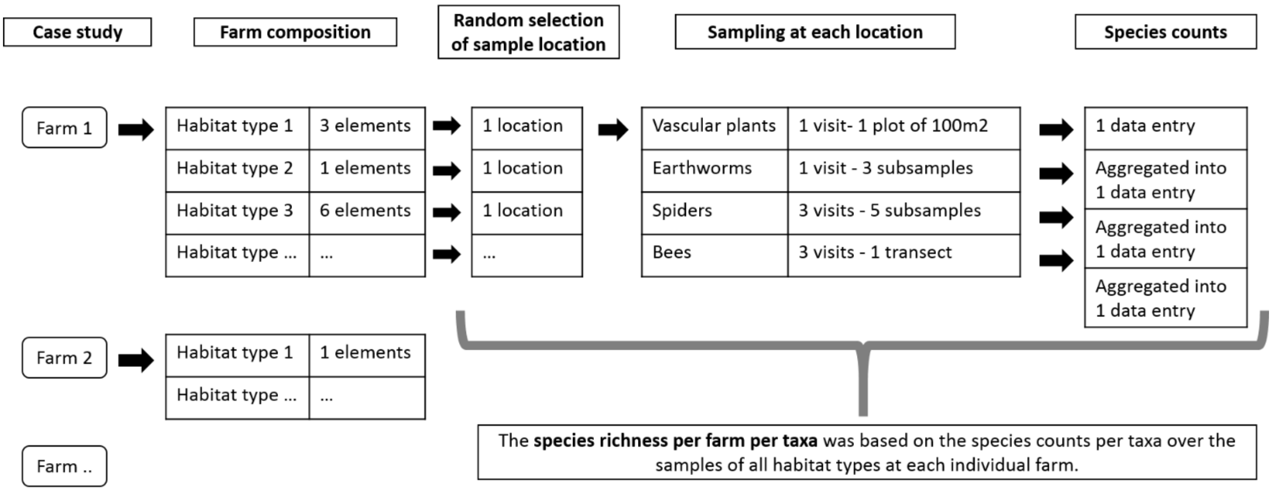


Figure 2. Overview of the computation of the species richness per taxa per farm.

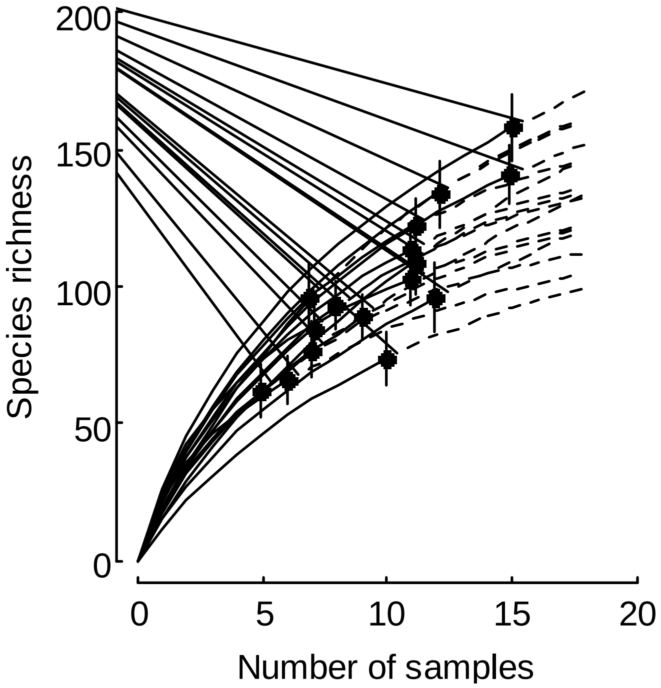
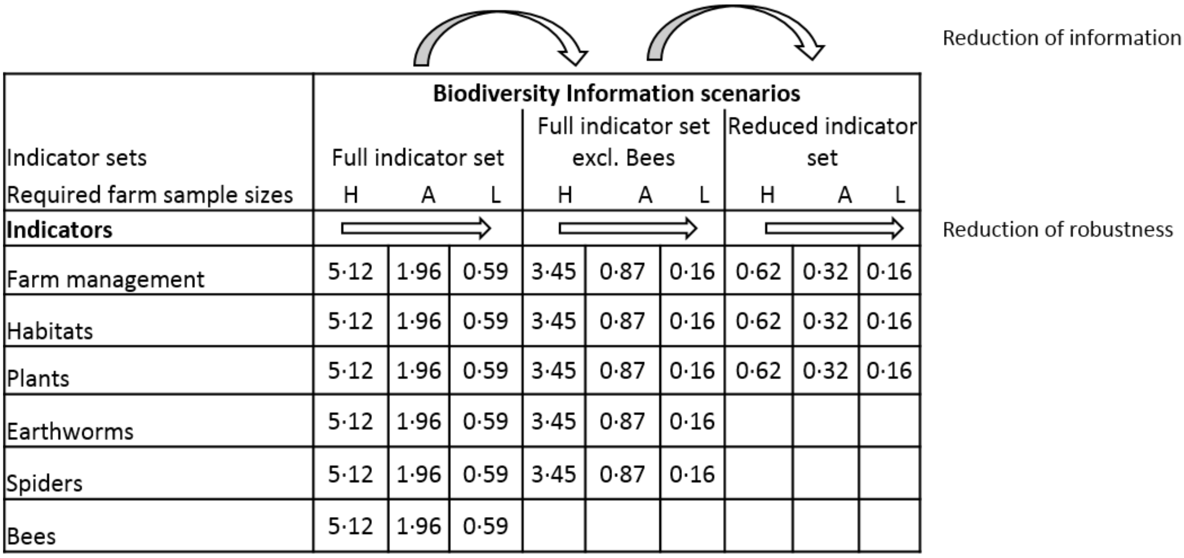


Figure 3: Example of accumulation curves for plant species richness in 16 farms in the French case study. Dots with bars are observed species richness with 95% confidence interval. Solid curves are species rarefaction curves, dotted curves are extrapolation curves. As the taxa were sampled using a stratified sampling approach, the number of samples (x-axis) is identical to the number of habitat types found per farm.



Indicator sets	Biodiversity Information scenarios: Full indicator set			Full indicator set excl. Bees			Reduced indicator set		
Required farm sample sizes	H	A	L	H	A	L	H	A	L
Indicators									
Farm management	5·12	1·96	0·59	3·45	0·87	0·16	0·62	0·32	0·16
Habitats	5·12	1·96	0·59	3·45	0·87	0·16	0·62	0·32	0·16
Plants	5·12	1·96	0·59	3·45	0·87	0·16	0·62	0·32	0·16
Earthworms	5·12	1·96	0·59	3·45	0·87	0·16			
Spiders	5·12	1·96	0·59	3·45	0·87	0·16			
Bees	5·12	1·96	0·59						

Figure 4: Indicators included per scenario as well as estimates of the farm sample sizes for the Field crops and horticulture farm type (% of the total number of farms of that farm type per region). The information output is reduced between the indicator sets from left to right and the robustness of the data output decreases from high (H) over average (A) to low (L).