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REVIEW

State-of-the-art practices in farmland biodiversity monitoring for North America and Europe

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Abstract Policy makers and farmers need to know the status of farmland biodiversity in order to meet conservation goals and evaluate management options. Based on a review of 11 monitoring programs in Europe and North America and on related literature, we identify the design choices or attributes of a program that balance monitoring costs and usefulness for stakeholders. A useful program monitors habitats, vascular plants, and possibly faunal groups (ecosystem service providers, charismatic species) using a stratified random sample of the agricultural landscape, including marginal and intensive regions. The size of landscape samples varies with the grain of the agricultural landscape; for example, samples are smaller in Europe and larger in North America. Raw data are collected in a rolling survey, which distributes sampling over several years. Sufficient practical experience is now available to implement broad monitoring schemes on both continents. Technological developments in remote sensing, metagenomics, and social media may offer new opportunities for affordable farmland biodiversity monitoring and help to lower the overall costs of monitoring programs.

Keywords Agricultural landscape · Ecosystem service · Essential biodiversity variables · Monitoring budget · Stakeholder need · Survey design

INTRODUCTION: PURPOSE OF MONITORING FARMLAND BIODIVERSITY

Biodiversity decline is a global issue, with potential negative consequences for all economic sectors (Perrings 2014). Agriculture is just one of them—albeit an important one. More than one-third of the global land area is currently under agricultural use (Foley et al. 2005). Various wild species have adapted to farming practices and depend on farmland habitats (Phillips 1998) and in fact most of the world's biodiversity will continue to exist outside protected areas (Dudley et al. 2005). Some of the most critical conservation issues today thus relate to ongoing changes in farming practices, which affect the wildlife on farms and adjacent habitats (Henle et al. 2008). Farmland biodiversity is threatened by the intensification and specialization of farming, which has led to a simplification of agricultural landscapes and a loss of (semi-)natural habitats (Benton et al. 2003; Hendrickx et al. 2007). At the same time, in other regions, marginal farmland is being abandoned and allowed to undergo natural succession, resulting in a loss of farmland habitats and associated species (Kampmann et al. 2012; Plieninger et al. 2014). Monitoring those trends can bring attention to these changes, engage the public, and trigger actions by policy makers (Table 1).

Pereira et al. (2013) proposed that "Essential Biodiversity Variables" (EBV) be monitored to inform the global status of biodiversity. There are three major groups of stakeholders requiring this information:

(i) Nongovernmental nature protection organizations (NGOs): Their objective is the conservation of wildlife and of habitats. Often they prioritize populations of charismatic wild species that are declining or threatened with extinction.
(ii) Governments and administrations: In both North America and Europe, policy measures for preserving biodiversity on farmlands have been implemented; these include the Conservation Reserve Program (USDA FSA no date) and the Wetland Reserve Program (USDA NRCS no date) in the U.S., the agri-environmental measures in European countries

Table 1 Recent examples of publicized findings relating to trends in biodiversity or to specific species

Region	Headline message	Policy implications	Source
1) USA	After the sting of vanishing bees, White House pollinates protection plan (CNN, May 2015)	The US Department of Agriculture and Environmental Protection Agency developed a national strategy (2015) to promote the health of honey bees and other pollinators (https://www.whitehouse.gov/blog/2015/05/19/announcing-new-steps-promote-pollinator-health)	http://www.cnn.com/2015/05/19/politics/white-house-bees-protect-pollinators/index.html
2) North America	Monarch butterflies decline; migration may disappear (USA Today, January 2014)	Supports the continental approach to conservation of this migratory species established in 2008 (http://www.fs.fed.us/wildflowers/pollinators/Monarch_Butterfly/conservation/conservation_plan.shtml)	http://www.usatoday.com/story/news/world/2014/01/29/monarch-butterflies-decline/5028977/
3) Europe	Europe's farmland birds in decline (The Guardian, August 2011)	In 2015, the European Union included a requirement of 5 % Ecological Focus Areas on farmland	http://www.theguardian.com/environment/2011/aug/24/europe-farmland-birds-on-decline
4) Europe	A plant called Stalin (translated from German) (Frankfurter Allgemeine Zeitung, August 2015)	A *Black List* of invasive neophytes has been put together for the European Union. Commerce with listed plants is forbidden, obligation to eliminate listed plants when they are observed	http://www.faz.net/aktuell/politik/neophyten-eine-pflanze-namens-stalin-13755250.html
5) Global	Earth is on brink of a sixth mass extinction, scientists say, and it's humans' fault (Washington Post, June 2015)	The UN Convention for Biodiversity formulates objectives, signatory states commit to mitigation measures and to reporting	http://www.washingtonpost.com/news/morning-mix/wp/2015/06/22/the-earth-is-on-the-brink-of-a-sixth-mass-extinction-scientists-say-and-its-humans-fault/
6) Global	Bee deaths from colony collapse disorder on the rise as researchers point to pesticides (Huffington Post, March 2013)	European Union: Moratorium for the application of three neonicotinoids (2014)	http://www.huffingtonpost.com/2013/03/29/bee-deaths-colony-collapse_n_2979959.html

(EU no date) and the so-called "greening" of the Common Agricultural Policy of the European Union. National and international objectives for the protection of biodiversity have been agreed upon, notably the Aichi Biodiversity Target 7 (Convention on Biological Diversity 2010). Information about the status and dynamics of biodiversity is needed to evaluate whether policy measures are effective and whether biodiversity targets are reached.

(iii) Private sector: Specific labels distinguish food products resulting from biodiversity friendly production (e.g., organic farming, Terra Suisse https://terrasuisse.migros.ch/). To be credible, biodiversity on those farms needs to be evaluated and compared to competing (nonlabeled) farms. Other economic sectors (tourism, nonrenewable and renewable energy) also affect agricultural landscapes and related biodiversity.

Often the major stakeholders commission and fund the monitoring programs. Their practical implementation is, however, always affected by budget restrictions that limit the number of indicators that can be measured, the number of samples which can be taken, the frequency of surveys, and so forth.

This practical limitation leads to design choices, which are summarized in Fig. 1. Based on a review of major existing (farmland) biodiversity monitoring programs, we summarize the state of monitoring related to those design choices.

OPERATIONAL BIODIVERSITY MONITORING PROGRAMS IN NORTH AMERICA AND EUROPE

We identified 11 major monitoring programs, relevant in the context of farmland biodiversity in North America and in Europe (Table 2). This selection is not exhaustive; it is based on our knowledge of existing programs through working relations with some of their managers and on a complementary literature survey. The selection is limited to operational programs with a regional, national, or international extent, conducting repeated measurements of habitats and species to detect trends over time and to programs addressing "common" species and habitats. Specific programs for threatened species (Red Listed), for invasive species and for rare habitats are not covered, nor are programs which are limited to tracking land use/land cover change.

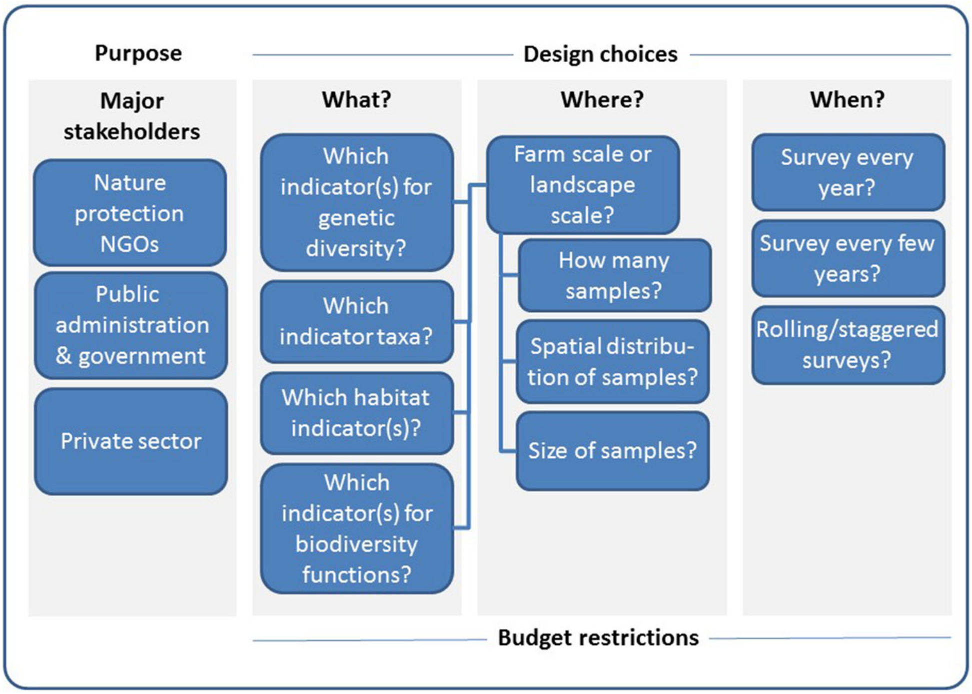

Fig. 1 Design choices needing to be addressed when a farmland biodiversity monitoring program is conceptualized. Major stakeholders are usually also involved in funding. Once the indicators have been selected, the number, spatial distribution, and size of the sample have to be defined as well as the temporal resolution. Those choices are usually made under budget restrictions

Bird and arthropod citizen science monitoring programs

Biodiversity monitoring was first initiated by nature protection NGOs in response to a perceived decline in populations of wild species. Monitoring data should allow this decline to be quantified and provide a basis to lobby governments for more nature conservation efforts. Birds were the first species group to be monitored from 1966 onwards on both continents across the entire rural landscape (Table 2a). The Pan-European Common Bird Monitoring collates nationally collected data from 40 European countries and computes overall trends (European Bird Census Council 2015). In most countries, the majority of data are generated by volunteers. Unlike the European common bird monitoring, the survey transects of the North American breeding bird survey have been centrally selected in a stratified random approach (Sauer et al. 2014). However, the census is also based on voluntary participation of field surveyors, who are free to choose their transects. This monitoring is complemented by more recent initiatives, also depending on volunteer participation, such as the North American Monitoring Avian Productivity and Survivorship Program (http://www.birdpop.org/maps.htm), the Waterfowl Breeding Population and Habitat Survey (http://www.flyways.us/surveys-and-monitoring/waterfowl-population-surveys/may-breeding-population-and-habitat-survey) and the Feeder Watch citizen scientist initiative (http://feederwatch.org/). The strengths and weaknesses of the first two programs are discussed in a report by the U.S. North American Bird Conservation Initiative Monitoring Subcommittee (2007), with weaknesses attributed mostly to “the need to keep the costs of those vast programs manageable” (p. 31). Based on national data, Birdlife International reports on the “State of the world’s birds” (http://www.birdlife.org/datazone/sowb). Birds are the only indicator group for which such a global overview on their status is achieved.

Ten years after bird monitoring, butterflies were the second charismatic species group for which large-scale monitoring efforts were initiated. Today, Butterfly Conservation Europe coordinates monitoring activities in 14 European countries. Data are mostly collected by volunteers, who are largely free to select transects of their choice (preferred sampling, van Swaay et al. 2012), transects that have been allocated across the entire territories, including farmlands.

In Canada, a network of contributing partners helps coordinate biodiversity monitoring across many taxa using standardized protocols (Table 2a). Its foundations were laid by the Canadian Ecological Monitoring and Assessment Network, EMAN (1994–2010), which supported the development of standardized monitoring protocols (Roberts-Pichette 1995).

Table 2 Operational biodiversity monitoring programs in North America and Europe with relevance for farming landscapes, in the sequence of their year of establishment

Monitoring scheme	Why? Motivation and purpose	For whom? Main stakeholders and funders	What? Indicator groups	Where? Scope, scale, and sampling design	When? Temporal intervals	Source
(a) Bird and arthropod citizen science monitoring programs						
Pan-European bird monitoring scheme	Track bird populations as indicators for the state of nature	Volunteer surveys and composite funding from NGOs, foundations and mostly Ministries of Environment	Abundance of common bird species	Preferably 1 km squares, systematically and/or randomly sampled from the rural landscape	Annual repetitions, starting between 1966 (UK) and 2009 (Luxemburg)	http://www.ebcc.info/pecbm.html
North American Breeding Bird Survey	Estimate bird population change	Volunteer surveys and composite funding from U.S. Geological Survey, Canadian Wildlife Service	Abundance of common bird species	Entire territory of the US and of Canada, stratified random selection of 5000 transect starting points and directions, 39.2 km average length	Annual repetitions, starting in 1966 with 600 transects, now 2900 operational transects	http://www.mbr-pwrc.usgs.gov/bbs/
Butterfly monitoring (Europe)	Provide information on the status of butterfly species to prevent extinction	Volunteer surveys and composite funding from NGOs, private foundations and some governmental institutions	Abundance of butterfly species	Preferably transects of 1 km, mostly free choice	Annual repetitions starting between 1976 (UK) and 2010 (Sweden. Luxemburg)	http://www.bc-europe.eu/index.php?id=339
Biological Survey of Canada BSC	Discover, survey, and inventory Canada's biological diversity; detect changes and provide policy advice	NGO, foundation	Arthropods	Total national territory, sampling depending on taxon	Started 1977, intervals not specified	http://biologicalsurvey.ca
(b) Government-led farmland biodiversity monitoring						
UK Countryside Survey	Provide evidence about the extent and condition of the UK's countryside. Estimate changes to help form policies that influence management of the countryside	Ministry for Environment, Food and Rural Affairs & Natural Environment Research Council	Habitat categories, vascular plant species lists, add-ons, e.g., veteran trees, ponds, soil cores, etc.	Rural landscape; 591 1-km squares, stratified random distribution across England, Scotland, and Wales	1978, 1984, 1990, 1998, 2007	http://countrysidesurvey.org.uk/
Northern Ireland Countryside Survey	Provide reliable information on habitats to give context to discussion on biodiversity and land use issues and indicate progress in meeting national and international obligations on wildlife conservation and sustainable development	Environment Agency and Ministry of Agriculture and Rural Development Northern Ireland	Habitat categories	Rural landscape; 628½-km squares, stratified random distribution	1992, 2000, 2007 (288 squares)	http://www.science.ulster.ac.uk/nics/PUBL/nicstech.pdf

Table 2 continued

Monitoring scheme	Why? Motivation and purpose	For whom? Main stakeholders and funders	What? Indicator groups	Where? Scope, scale, and sampling design	When? Temporal intervals	Source
Germany HNV farmland monitoring	Provide a baseline indicator to incorporate environmental concerns into the EU Common Agricultural Policy	Ministry of Environment	Habitat categories, ecological quality	Farmed rural landscape; 900 1-km squares, stratified random distribution	2009, 2010–2013, 2014–2017 (rolling survey)	http://www.bfn.de/0315_hnv+M52087573ab0.html
Norway 3Q	Document and analyze landscape change	Ministries of Agriculture and of Environment	Land cover types, vascular plants, birds	Farmed rural landscape, 1000 1-km squares, systematic distribution (until 2002), stratified random distribution (since 2003)	1998–2002, 2003–2007, 2008–2012, 2013–2017 (rolling survey)	http://www.nibio.no/en/topics/landscape-monitoring
Switzerland BDM & ALL-EMA	Evaluate whether farmland biodiversity goals are reached and whether policy measures are effective (ecological focus areas)	Ministry of Environment (BDM), Ministries of Agriculture and of Environment (ALL-EMA)	Species lists of vascular plants, birds, butterflies, land snails, context information (BDM); Habitat categories, vascular plants (ALL-EMA)	Total national territory; 500 1-km squares, regular grid; 1600 10-m^2 point samples, regular grid (BDM); Farmed rural landscape; 170 1-km squares (habitat) and 3000 10-m^2 point samples (plants), nested stratified random distribution (ALL-EMA)	2001–2005, 2006–2010, 2011–2015 (BDM rolling survey) 2015–2019 (ALL-EMA rolling survey)	http://www.biodiversitymonitoring.ch/; www.ALL-EMA.ch
Sweden NILS	Monitor the conditions and changes in the Swedish landscape	Environmental Protection Agency and Board of Agriculture	Land cover & land use; vascular plant species, lichen, context information	Total national territory; 631 1-km squares, stratified random distribution	2003–2007, 2008–2012, 2013–2017 (rolling survey)	http://www.slu.se/nils
(c) Public private partnership						
Canada, Alberta, ABMI	Track changes in Alberta's wildlife and their habitats to inform land use decision-makers, land stewards, and the public at large	Public private partnership	Land classes: species lists of vascular plants, lichen, mammals, birds, mites; context information	Entire territory of Alberta, 1656 nested sites of 20 km squares (aerial photo, mammal transect) and 1 ha, regular grid	2007–ongoing (rolling survey planned, first repetitions starting 2015)	www.abmi.ca

Government-led farmland biodiversity monitoring

In the US, the Environmental Monitoring and Assessment Program (EMAP) aimed to advance the science of ecological monitoring and to lay the foundation for a national resource monitoring program. Today, aquatic resources are routinely monitored according to the EMAP guidelines (McDonald et al. 2002). For agroecosystems, biodiversity monitoring coordinated with the regular farming census (5 year intervals) was proposed (Stevens 1994). This proposal, however, has not been adopted and to date, there is no government-led farmland biodiversity monitoring in North America. The evaluation of the effectiveness of agri-environmental measures (Ferris and Siikamäki 2009) is based on case studies. For example, New Mexico State University assesses effects of prescribed grazing, brush management, and upland wildlife management on biodiversity in the Southwest. The findings of case studies are then synthesized at the national level (latest synthesis report by Haufler 2007).

Government-led biodiversity monitoring is limited to five European countries (Table 2b). The UK Countryside Survey (Carey et al. 2008) was the first systematic habitat and biodiversity monitoring for all land uses (Bunce and Shaw 1973). The open countryside was classified into 32 land classes according to major environmental factors (Bunce et al. 1996). A stratified random sample of 1-km squares was then drawn, to be representative for those land classes. For each square, a habitat map is established, according to a habitat key based on plant life forms (Maskell et al. 2008) and ~30 randomly assigned vegetation sampling points are located in each square where species lists of vascular plants are recorded. The UK Countryside Survey pioneered systematic habitat and species monitoring and other programs, notably the Northern Ireland countryside survey, have adapted its approach.

In the Norwegian 3Q program, 1000 1-km squares were selected as a systematic sample of farmed rural landscapes (Dramstad et al. 2002; Økland et al. 2006). After two monitoring periods, the systematic sample was replaced by a stratified random sample in order to achieve better coverage of marginal agricultural lands where more rapid changes occur (Stokstad et al. 2013). While the UK and Northern Ireland Countryside surveys operate with campaigns concentrated in an individual year, the 3Q program is conducted as a rolling survey. Each year, 20 % of the sites are surveyed and the sample is complete only after 5 years.

The German monitoring of High Nature Value (HNV) farmland (Oppermann et al. 2012) aims at reporting the status and extent of HNV farmland to the European Union (PAN, IFAB and INL 2011). HNV farmlands comprise areas where agricultural land use dominates and supports either a high diversity of species or habitats and/or species of conservation concern (Lomba et al. 2014). Nine hundred 1-km squares were selected through stratified random sampling based on land cover and on natural conditions. Field recording is limited to 16 coarse habitat categories. The ecological quality of each habitat is then evaluated and judged (four levels according to predefined criteria and target species). Quality criteria and species can differ between federal states in order to account for biogeographic gradients across the country.

The UK, Irish, Norwegian, and German programs focus on rural or farmed rural landscapes. In Switzerland, the national Biodiversity Monitoring surveys two regular grids covering the entire national territory (BDM Coordination Office 2014). On 500 1-km landscape squares, vascular plants, birds, and butterflies are recorded in standard transect walks and on an additional grid of 1600 10-m^2 plots, vascular plants, and land snails are surveyed. From 2015 onwards, on a subsample of 170 1-km landscape squares, habitat information is also recorded. The subsample was randomly stratified in order to reflect farmland biodiversity of major biogeographical regions and of national agricultural census zones (www.all-ema.ch).

The National Inventory of Landscape in Sweden (NILS, Ståhl et al. 2011) also covers the entire national territory of Sweden, including farmland, and has adopted a nested systematic sample design with 1-km squares at the core (land cover and land use mapping from aerial photographs), which are located at the center of 5-km squares (extensive context information). At the interior of the 1-km squares, 12 transects and 12 circular sample plots are located according to a systematic design to capture linear and point features, plant species, and more context information.

In the USA, a nation-wide National Environmental Observatory Network (NEON) is being deployed (starting in 2010). The 20 sites, stratified by ecoregion, will be monitored for organisms and biogeochemical fluxes using a variety of methods including ground-based and airborne instruments (Keller et al. 2008). While aimed at monitoring and forecasting ecosystem response to global change, like the nation-wide programs in Germany and Sweden, this program will encompass agricultural as well as other land uses.

Public–private partnership

The Alberta Biodiversity Monitoring ABMI (Canada) covers the entire territory of the province of Alberta, encompassing all major land uses (from urban to natural, Table 2c). This unique example of a public–private partnership is run by a nonprofit institute that receives support and funding from member institutions comprising public institutions (e.g., the University of Alberta, Royal Alberta Museum), nature conservation NGOs, and private companies of the energy, forestry, agricultural, and tourist sector. Those stakeholders have an interest in evaluating the consequences of their activities on biodiversity. The ABMI operates on a regular grid with nested plots and transects, monitoring land cover classes, vascular plants, and several faunal groups. The size of the observation plots is adapted to the mobility of the species groups (Burton et al. 2014).

DESIGN CHOICES AND STATE-OF-THE-ART PRACTICES

What is farmland?

Some of the programs listed in Table 2 are restricted to farmland, while others are generic and cover all major land uses. Farmland biodiversity indicators are then extracted. This raises the question of the actual definition of farmland. According to the Oxford Dictionary, this is land used for farming, which corresponds to "The activity or business of growing crops and raising livestock" (http://www.oxforddictionaries.com/de/definition/englisch/farming). While this appears to be intuitively clear, low-intensity farming often blends into other land uses. Examples are forests, salt marshes or marginal mountain grasslands used for grazing. Is bison ranching actually a farming activity? Which seminatural habitats such as hedgerows or small woods are still part of the farmland? Basically there are two approaches to delimiting farmland. From their observations, the bird and butterfly surveys (Table 2a) extract the species that are related to farmland, based on their knowledge of the ecology of the species. Without spatially delimitating farmland, this allows these surveys to provide information about farmland birds (e.g., Eurostat, no date) or butterflies related to grasslands (EEA 2013). The monitoring programs that also record habitats (Tables 2b, c) have to define the spatial extent of farmland. They usually opt for a broad definition, arguing that, for example, semi-natural habitats on farmland, even if they are not used for growing crops or raising livestock, are part of the agricultural landscape and are a prerequisite to the existence of major farmland species. More important than the actual definition, however, is that it is not changed between surveys. The value of monitoring results stems from consistent time series, which allow for straightforward comparisons.

Purpose and major stakeholders

Biodiversity monitoring was implemented by three main groups of stakeholders with somewhat differing motivations:

(i) Nature protection NGOs, whose members have an inherent motivation to protect biodiversity (Table 2a);

(ii) Ministries for Environment (their collaborators often share the ethics of nature protection NGOs) and Ministries for Agriculture (often because of the need to justify environmental payments to farmers) (Table 2b);

(iii) A public–private partnership between government organizations and private industries with business activities in rural areas, interested in evaluating the environmental consequences of their activities (Table 2c).

Usually, the stakeholders also provide the resources for the monitoring programs, either through volunteer work or by making funding available. The government-led monitoring programs often integrate data from the citizen science programs, such as the European Union reporting on farmland birds and butterflies (EEA 2013; Eurostat, no date).

What indicators to monitor?

There is a rich literature on biodiversity indicators, from conceptual foundations (e.g., Noss 1990) to more specific work on farmland biodiversity indicators (e.g., Büchs 2003). The first indicators to be monitored were those faunal groups (birds, butterflies) that were charismatic enough to capture the attention of volunteers and NGOs (Table 2a). Smaller initiatives have subsequently been launched for example for bumblebees, amphibians, reptiles, and specific mammal species (Lengyel et al. 2008). Government-initiated schemes appear to be less dependent on the criterion of inspiring public interest in a biodiversity indicator. They have all opted for land use/land cover and habitat indicators, most of them in combination with vascular plants. This choice coincides with the stakeholder priorities reported by Targetti et al. (2015), who expect highest efficiency of farmland biodiversity monitoring from habitat and vascular plant diversity monitoring, efficiency being measured as the ratio between information benefit and cost of measurement.

Is it sufficient then to restrict farmland biodiversity monitoring to habitats and vascular plants? In Nordic countries, lichen and mosses may contribute considerably to biodiversity, and at the same time they are the main fodder resource for reindeer herding over the winter. As a consequence, they are recorded by the Swedish NILS program. Animal taxa represent 80 % of the known multicellular species on earth (The World Conservation Union 2010). Arthropods alone make up 66 % of the world's known species and in an agricultural context many of them are either pests or service providers. Among the ongoing programs (Table 2), only in Canada do arthropods (other than butterflies) seem to be a priority, namely in grasslands (BSC) and in soils (mites, ABMI). Some of the government monitoring programs also include butterflies (BDM Switzerland) and birds (3Q Norway, BDM Switzerland), mostly taking advantage of the volunteer monitoring already in place. Land snails (BDM Switzerland), spiders, springtails (Collembola), and mammal groups (ABMI Canada) are occasionally added. The Canadian ABMI observes the largest range of taxonomic groups.

Unlike the other schemes, the Swiss BDM originally was restricted to species diversity and did not monitor habitats; only recently has a habitat indicator been added. The German HNV monitoring, on the other hand, includes no species indicator and is limited to habitat categories. It is also the only scheme that directly collects interpreted data on the "ecological quality" of habitats instead of primary raw data as recommended by Metzger et al. (2013). While stakeholders ultimately require interpreted information, its direct collection in the field is prone to expert bias. Also, criteria for ecological quality are likely to evolve over time as a result of global change and shifting value systems. It will not be possible to adapt the quality criteria to those new realities without compromising the time series, whereas primary data can be reaggregated or interpreted according to new criteria.

Many of the monitoring programs record additional (context) information e.g., on soil, management practices (e.g., grazing observed), or specific landscape elements (veteran trees, etc.). This information is helpful for the interpretation of biodiversity indicators. Other monitoring activities (e.g., soil quality, water quality) are sometimes combined with the biodiversity monitoring to take advantage of the sampling design and create synergies through combined data recording.

None of the monitoring programs included genetic diversity indicators (of crops, husbandry animals, or wild species). This can be attributed to the methodological complexity and to the still comparatively high cost of measuring genetic diversity (Last et al. 2014).

Where to monitor? Sampling design, scale, and landscape grain

We found no monitoring program that operates at the farm scale. The few scientific articles on farm-scale biodiversity monitoring argue that this would facilitate the communication with farmers (Lüscher et al. 2014) and that management could be taken into account, which would support the interpretation of biodiversity indicator trends (Herzog et al. 2013). Farmland management indicators came up as a stakeholder priority in the investigation by Targetti et al. (2015). At the farm scale, biodiversity and management records could be linked to farm accountancy monitoring, which is carried out in all North American and European countries. This would allow monitoring in the framework of pressure–state-response (OECD 2003). Yet, there are also important drawbacks of farms as monitoring units. Unconsolidated farms are spatially disaggregated; individual plots are intermingled with plots managed by other farmers and by other land uses. Such farms are not suitable sampling units for mobile species such as birds. Also, farms tend to change over time (in size, ownership, and farming activities). This increases the variability of measured indicators as compared to those measured in stable landscape units or transects.

The programs investigated here operate with landscape units and transects. They aim at generating representative data for larger regions, nations, or continents. This requires a probability-designed sample that allows for statistical generalizations. Systematic sampling on a regular grid has the advantage of being more robust and independent of stratifications based on altitude, land use/land cover, or climate, some of which may change on the long run. Systematic sampling is the best design for long-term time series, allowing for the highest flexibility in data analysis (poststratification) (Stevens 1994). Among the three programs established using systematic sampling, the 3Q program (Norway) has in the meanwhile switched to stratified sampling (Stokstad et al. 2013) and in Switzerland, the original BDM program (BDM Coordination Office 2014) is complemented with a stratified random sample of open agricultural landscapes (www.all-ema.ch). For both programs, a new sampling strategy was adopted in order to improve cost efficiency and address stakeholder needs. The ABMI program in Alberta pursues sampling on a regular grid.

The other programs reviewed have adopted a stratified random sampling approach, based on environmental and/or administrative strata. Stratified random sampling has long been advocated for biodiversity surveys (Austin and Heyligers 1989; Franklin et al. 2001) and ensures an adequate representation of smaller regions or strata, which would not be sufficiently represented in a random sample to allow for statistical analysis of trends (Metzger et al. 2013). At the same time, the sampling effort in larger regions can be reduced to limit the collection of redundant data.

Several programs did not initially adopt a statistically based sampling strategy. For the bird and butterfly programs (Table 2), it is more challenging to adopt stringent probability sampling as these programs rely on field work by volunteers, who are often motivated by preferences for certain sites and may be restricted to regions that are easily accessible. The bird monitoring programs have made considerable efforts, though, to improve sampling and statistical representativeness. In Europe, free choice sampling is now limited to three countries (Voříšek et al. 2015). In North America, transects were selected centrally according to a stratified random process. However, there is no volunteer for each transect and the 60 % which are actually surveyed (Ziolokowski et al. 2010) are chosen according to observer preference.

In the European programs, habitats are usually mapped in 1-km squares, a unit first introduced by the UK Countryside Survey and later also adopted in Norway, Switzerland, Sweden, and Germany. Interestingly, Northern Ireland has opted for smaller 0.5-km squares, better adapted to the finer grain of its agricultural landscapes (Carey et al. 2008). In North America, on the other hand, the size of landscape samples tends to be larger (e.g., 20-km squares, ABMI in Canada), as does the length of bird recording transects (39.2 km, North American Breeding Bird Survey). These are adaptations to the coarser landscape grain and larger extent. In fact, the average field size, as a proxy for landscape grain, is significantly larger in North America than in Europe (Fritz et al. 2015). Median field size in two U.S. case studies (Yan and Roy 2014) was 12.0 and 47.4 hectares, respectively, as compared to 1.8 hectares across 25 European test regions (Herzog et al. 2006).

When to monitor? Temporal resolution

The bird and butterfly surveys are repeated annually. The majority of the other monitoring programs operate with rolling (staggered) surveys, mapping e.g., 20 % of the total sample each year until it is complete after 5 years. The U.K. and the North-Ireland Countryside Surveys are the only monitoring programs that concentrate the observations in a single year, with intervals of 6–8 years between surveys (on–off surveys).

Rolling surveys are considerably less expensive than annual surveys but still allow for annual estimates and trend analysis (although with lower statistical confidence), provided that the annual subsamples are randomly chosen (Urquhart and Kincaid 1999; McDonald 2003). Compared to surveys that are repeated every 5–10 years, they have the practical advantage of a more continuous distribution of the workload so that well-trained field teams can be maintained, resulting in consistently high data quality. Also, years of extreme weather conditions affect the data less. Allocating rolling surveys across 5 years fits the time span proposed for species and habitats by Pereira et al. (2013).

If monitoring is performed in a policy context, the timing of availability and communication of results needs to be in line with the timing of political processes. There are windows of opportunity, when such results can actually influence the legislative and decision-making process. They will then have significantly more impact than at other periods, when no biodiversity-related decisions are being considered.

Some monitoring programs also investigate relationships between cause and effect—e.g., between habitat change and species diversity. Identifying such relationships can strongly facilitate the interpretation of observed trends, provided that they can be differentiated from other factors acting on species populations (e.g., annual weather conditions) and account for time-lags between changing pressures and biodiversity response (Essl et al. 2015). Annual monitoring data may be required to elucidate such relationships, although targeted case studies may be a cost-effective strategy to investigate cause-effect relations.

Budget restrictions

If monitoring programs could be designed solely based on scientific criteria, design choices (Fig. 1) would be easy: Select indicators which cover genetic diversity, plants, animals, habitats, and functions; both farms and landscapes would be monitored with a large number of samples on a systematic grid; surveys would be repeated every year. However, because resource restrictions apply, survey design should maximize the gain of useful information while minimizing monitoring costs.

Resource restrictions therefore affect design choices (Fig. 1):

(i) Among the indicators that can be measured, habitat and vascular plant diversity and farm management indicators yield the most efficient cost-benefit relation (Targetti et al. 2015). If resources allow, they should be complemented with fauna taxa of important ecosystem service providers. This may, however, increase the cost considerably.

(ii) Farm-scale surveys allow for synergies between biodiversity observations, farm management, and accountancy indicators. However, monitoring at farm scale has some disadvantages (unconsolidated farm holdings, farm structural change) as compared to fixed landscape squares or transects (coherent spatial unit, stable over time). We are not aware of investigations that looked at the trade-offs between those two options.

(iii) The number of samples must yield the statistical power to detect changes in biodiversity indicators at the required precision and over a predetermined time scale (e.g., *X* % decrease of the population of species A over Y years; see Legg and Nagy 2006; Franklin et al. 2011). Provided that prior knowledge about the variability of the indicator exists, a power analysis can inform on the required sample size (Nielsen et al. 2009; Geijzendorffer et al. 2016).

(iv) Although systematic sampling (on a grid) is the best design for monitoring, stratified random sampling allows increased efficiency of data collection by allocating proportionally more samples to small regions or rare strata that otherwise would not be sampled often enough to allow for statistical analysis, and fewer to large regions that would be oversampled. Most programs have therefore adopted this approach.

(v) Compared to the other cost factors, the size (area) of the samples may be less important than location as a significant portion of the cost incurs by transport to and from sampling site (Targetti et al. 2014). This allows the surveyed area to be increased in coarse-grained landscapes without increasing costs dramatically.

(vi) Among the different options for temporal resolution, rolling/staggered surveys allow programs to considerably reduce their costs as compared to annual surveys, while presenting logistic and analytical advantages over on–off surveys. Programs involving volunteers can afford annual repetition, while most government-led programs operate with rolling surveys.

In practice, often a budget is made available and scientists are then asked to design a monitoring within those restrictions (Targetti et al. 2014). This optimization exercise is often based on incomplete information.

RESEARCH NEEDS

Technological innovation

Monitoring programs are by nature conservative. Methods need to be constant and reproducible in order to yield consistent time series. Still, new technologies can potentially help to decrease the costs of monitoring and/or to increase the insights gained.

Remote sensing is already being used to prepare maps for fieldwork, for example, for habitat mapping. The standard approach consists of subdividing the landscape into discrete patch, line, and point elements (wall to wall map). If clear rules are followed on land-cover definitions, minimum mapping units, etc., this type of mapping works well in more intensively used agricultural landscapes (e.g., arable landscapes with clearly visible field boundaries). In gradient-dominated landscapes (e.g., rangelands), defining habitat boundaries is more difficult and sometimes arbitrary. Point sampling is a possible alternative (www.all-ema.ch). Remote sensing information may in the future offer alternative techniques to map and monitor gradient-dominated landscapes (Lausch et al. 2015). Progress is being made to estimate biodiversity metrics from remotely sensed information (Rocchini et al. 2015). Skidmore et al. (2015) review variables that can be operationally monitored from satellite data including ecosystem structure indicators (land cover, fragmentation, heterogeneity) and species traits (leaf area, leaf nitrate content). There are limitations, however with respect to remote sensing of individual species, faunal groups in particular.

Metagenomics are to date mostly applied in the analysis of microbial biodiversity. In the future, genetic analysis of bulk samples, notably of arthropods, might facilitate their identification and even reduce the costs of monitoring (see e.g., Shokralla et al. 2015; Tang et al. 2015 and references therein). This field of research is rapidly developing and—similar to new remote sensing methods—should be pursued in parallel to established monitoring methods.

Citizen science was at the origin of biodiversity monitoring and the most comprehensive monitoring programs on birds still rely on data recorded by volunteers. New approaches emerge, which are made possible by e.g., social media and online computer/video gaming (Newman et al. 2012). This will create a new challenge: How to handle the increasing amount of data that can be generated? However, the main challenges remain the quality control of the data and the representativeness of the samples (Bonter and Cooper 2012; Hochachka et al. 2012). The well-established bird monitoring programs have made huge efforts to address those issues by training their participants and by evolving from preferential to stratified random sampling. Those issues need to be tackled by all citizen science programs in order to produce quality information, representative for a specified region or country.

What about ecosystem services?

In an agricultural context, ecosystem services are particularly important (Clergue et al. 2005; Tscharntke et al. 2005; Van Zanten et al. 2014). Many species interact with crops and farm animals and either provide services (pollination, predation, decomposition, etc.) or dis-services (pests, diseases, weed competition, etc.; Zhang et al. 2007). Geijzendorffer and Roche (2014) scored seven monitoring schemes as to their potential ability to provide ecosystem service indicators. Schemes that include a range of sampling methods, scales, and also habitat data, had the highest potential to inform about ecosystem services. Among the schemes evaluated in Table 2, the UK and North Ireland Countryside Surveys, 3Q (Norway), BDM/ALL-EMA (Switzerland), NILS (Sweden), and ABMI (Canada) meet those criteria, while monitoring schemes limited either to land use/land cover or to specific species groups (birds, butterflies) do not.

The measurement of ecosystem services can be extremely difficult, let alone repeated measurement in a monitoring context. Liss et al. (2013) reviewed 121 studies of pollinator services and found 62 combinations of metrics used to quantify this comparatively well-defined ecosystem service. Seppelt et al. (2012) propose a "blueprint" to harmonize future ecosystem service studies, facilitating generalizations based on individual investigations. Their proposed framework involves active stakeholder participation to drive a design process and addresses similar choices as those listed in Fig. 1.

However, the inherent complexity of biodiversity leads to such a multitude of interactions between species (including their genetic variability), habitats, and the abiotic environment, that ecosystem services monitoring approaches at this stage will be exploratory and adaptively modified, rather than consist of set protocols. Moreover, distinct ecosystem services and dis-services operate at specific scales and have a strong spatial component that also needs to be taken into account (Iverson et al. 2014; Mastrangelo et al. 2014). In an agricultural context, the existing monitoring programs for yield as a provisioning service (national yearbooks, statistics of the Food and Agriculture Organization) and for dis-services (e.g., pests and diseases http://aphmon.fera.defra.gov.uk/, http://euroblight.net/) can be a starting point. We see a potential here for involving farmers in data collection (as they are in farm accountancy networks) to reduce the cost of monitoring (Targetti et al. 2014). In Switzerland, for example, the Ministry of Agriculture collects environmental indicators on a subsample of the farms involved in the accountancy network (http://www.blw.admin.ch/themen/00010/00070/index.html?lang=fr). The joint analysis of environmental and financial data allows for a comprehensive evaluation of the sustainability of farm enterprises (Jan et al. 2012). Biodiversity at the farm scale is addressed by means of a life cycle assessment approach (Jeanneret et al. 2014). In parallel, farmland habitats and vascular plants are monitored at the landscape scale (the Swiss ALL-EMA program listed in Table 2b).

Global biodiversity monitoring

Pereira and Cooper (2006) and Schmeller et al. (2015) conceptualize global biodiversity monitoring and identify key requirements. The farmland monitoring programs from North America and Europe essentially meet those requirements and could contribute significantly to global monitoring. With the ongoing expansion of farmland in biodiverse tropical regions, farmland biodiversity monitoring will continue to gain importance. Methods will, however, need to be adapted. For example, Herzog et al. (2013) tested 23 biodiversity indicators, developed for European farms, in Tunisia and in Uganda. Major limitations were the lack of a habitat classification, dearth of taxonomic expertise, and unstable taxonomy of arthropod groups. Monitoring methods therefore need to account for local conditions, including the socio-economic context. Yet they should be integrated in a global framework that allows for interoperability due to common monitoring protocols and harmonized data structure and standards, making it possible to inform essential biodiversity variables (Schmeller et al. 2015).

Geijzendorffer et al. (2016) show that biodiversity monitoring is "affordable" compared to other public expenses that support farming. They argue that the efficiency of agri-environmental schemes and of cross-compliance requirements such as ecological focus areas could be increased if their effects were monitored. Those arguments need to be heard outside academic circles. We see two main pathways to achieve this:

(i) Bottom-up: Raising public awareness through the media with messages as the ones collated in Table 1.
(ii) Top-down: Raising the awareness of high-level policy makers, similar to the awareness building which took place for climate change. The Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) has taken on this challenge.

Both approaches require scientists to get involved with the public and policy issues.

CONCLUSIONS

A diversity of sometimes conflicting recommendations for monitoring have recently been made in the literature. Kosztvi et al. (2014) insist that monitoring programs need to be able to detect changes of species and habitats over time, otherwise they are a waste of resources (see also Legg and Nagy 2006). They therefore call for a power analysis at the planning stage of any new monitoring program. Lindenmayer and Likens (2011) question the validity of the concept of indicator species. They list 56 groups proposed as biodiversity indicators in the literature—but their actual indicator value (to act as surrogates for other species) has rarely been investigated. These authors expect indicators and monitoring to ultimately detect causal relationships between observed biodiversity and problems to be addressed. Metzger et al. (2013), on the other hand, argue for the need to include biodiversity in broad monitoring endeavors of environmental change, similar to the arguments made by Pereira and Cooper (2006) and Pereira et al. (2013).

The programs reviewed here operate in the broad biodiversity monitoring context addressed by the latter authors. They are not driven by ecological hypothesis but by requirements of policymakers to support decision making. This involves, of course, links to management practices and policies, but those links will mostly be limited to correlative evidence. Causal testing of hypothesis would be too demanding for the available resources and would often not be practical. For example, one would need to exclude a set of farmers (or farmers of several regions) from a policy instrument (often associated with funding) to compare biodiversity trends on farms (in regions) with and without policy instrument.

In this context, we summarize the state-of-the-art of broad farmland biodiversity monitoring programs with respect to the design choices (Fig. 1) as follows:

(i) What? Habitat categories and vascular plant diversity are a suitable baseline for monitoring, but should be complemented by faunal groups that are charismatic and/or related to major agricultural ecosystem services. Indicator measurement has to involve standardized collection of raw, un-interpreted data (species lists, habitat categories), which are a prerequisite for consistent time series.

(ii) Where? A stratified random sample allows for comparatively higher replication of measurements on smaller strata while reducing the amount of redundant data. Landscape sectors and transect lengths need to be adapted to the grain of the landscape and the mobility of the species.

(iii) When? Rolling surveys/staggered surveys allow for annual trend analysis while reducing costs as compared to annual repetitions. Rolling surveys have operational advantages over on–off surveys (consistent staffing and training). Timing of availability and communication of results should be coordinated with the policy agenda.

If those requirements are met, monitoring results can be integrated across programs (Pereira and Cooper 2006; Henry et al. 2008; Pereira et al. 2013; Schmeller et al. 2015). Measuring and monitoring ecosystem services and dis-services is the next big challenge. Much still needs to be learned about the functioning of agroecosystems. At this stage, monitoring programs can be extended to encompass major providers of ecosystem services (e.g., pollinators, pest predators), while monitoring the actual service (pollination, predation and effects on yield) should be investigated with experimental approaches.

Monitoring should not be confounded with long-term ecological research projects, which have a stronger experimental character but do not aim to generate trend data representative of larger regions. However, mandated monitoring programs (sensu Haughland et al. 2010; Lindenmayer and Likens 2010) can be combined with experimental research and/or the evaluation of policy instruments in a modular way. A mandated monitoring program provides an infrastructure for related case studies to investigate ecological hypotheses or pressing questions, such as investigations of pollinators, immigration of neobiota, or evaluations of agri-environmental payments.

It is impossible to know today how monitoring data will be used in the future. Climatologists who started systematically recording weather data centuries ago (Bradley and Jones 1992) had no idea that their records would later be used for investigating climate change. This example illustrates the importance of collecting un-interpreted raw data, with well-described methods and based on a statistically sound sampling design.

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