Chapter 4
Monitoring Essential Biodiversity Variables at the Species Level

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Abstract The Group on Earth Observations Biodiversity Observation Network (GEO BON) is developing a monitoring framework around a set of Essential Biodiversity Variables (EBVs) which aims at facilitating data integration, spatial scaling and contributing to the filling of gaps. Here we build on this framework to explore the monitoring of EBV classes at the species level: species populations, species traits and community composition. We start by discussing cross-cutting issues on species monitoring such as the identification of the question to be addressed, the choice of variables, taxa and spatial sampling scheme. Next, we discuss how to monitor EBVs for specific taxa, including mammals, amphibians,

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butterflies and plants. We show how the monitoring of species EBVs allows monitoring changes in the supply of ecosystem services. We conclude with a discussion of challenges in upscaling local observations to global EBVs and how indicator and model development can help address this challenge.

Keywords Species · EBV · Monitoring · Population abundance · Distribution

4.1 Introduction

People have monitored and managed species for thousands of years, but national and international biodiversity monitoring is a relatively recent phenomenon. By the end of the 1800s, some governments had established monitoring agencies, mostly taxon-specific. In the United States, for example, Congress established the U.S. Fish Commission in 1871 to recommend ways to manage the nation's food fishes, and the Division of Biological Survey in 1885 in order to promote 'economic ornithology, or the study of the interrelation of birds and agriculture.' In 1940, these divisions were combined into the U.S. Fish and Wildlife Service. Later, the U.S. Endangered Species Act of 1966 mandated species monitoring. At the international level, the multilateral CITES Treaty, established in 1973, required that the

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international trade of potentially vulnerable species be monitored by countries. Starting in the 1960s and during the following decades, conservation-focussed Non-Governmental Organisations (NGOs) also became involved in monitoring schemes, such as the Common Bird Census of the British Trust for Ornithology. Since the 1990s, the Habitats and Birds directives further stimulated species monitoring in European countries, although even today major gaps remain (Schmeller 2008; Henle et al. 2013). The global change discourse has increased the demand for biological monitoring. The Aichi Targets for 2020 by Parties to the United Nations Convention on Biological Diversity affirm an international desire to curb the rate of biodiversity loss (Leadley et al. 2014) and their assessment requires an expansion of current species monitoring efforts (Pereira et al. 2012; Tittensor et al. 2014).

Ecological monitoring in the early 20th century was largely organised around estimating population sizes of specific species. Capture-recapture methods were developed for fish by the Danish biologist Carl Petersen in the 1890s. In the mid-20th century, technologies developed in the world wars, including radioisotopes and radio-tracking collars, revolutionised ecological monitoring, and broadened the scope of monitoring from individual populations to ecosystem level processes. Part of this trend was reflected in the development of the Long Term Ecological Research (LTER) network (Aronova et al. 2010). In the last few decades, the development of extensive monitoring schemes based on trained volunteers or citizen scientists has allowed for the tracking of entire taxonomic groups over national and continental scales, for example, the Breeding Bird Survey in the USA or the Pan-European Common Bird Monitoring Scheme (Pereira and Cooper 2006). At the same time, remote sensing technology has started to make incursions into species level monitoring (see Chap. 8), including population counts of birds and mammals or the detection of invasive species (Pettorelli et al. 2014). In the last decade, the development of websites, such as ebird.org, ispot.org, inaturalist.org and observado.org, which allow for the global recording and sharing of species observations, has led to a new wave of citizen science engagement (see Chap. 9).

Studies of biodiversity remain unevenly distributed across the globe. One review of papers published in ten leading journals from 2004 to 2009 found that approximately 75 % of studies are conducted in protected areas (Martin et al. 2012). Studies were also disproportionately conducted in temperate, wealthy countries. Similarly, Amano and Sutherland (2013) found that a country's wealth, language, geographical location, and security explain variation in data availability in four different types of biodiversity databases. At a global scale, biodiversity monitoring is also biased towards consideration of certain taxa. For example, systematic IUCN Red List assessments have been carried out for only a few taxonomic groups, and the proportion of species assessed in each group is unrelated to its representation in global diversity (Pereira et al. 2012). Such geographical biases and historical contingencies have led to mismatches between prioritisation and protection (Jenkins et al. 2013).

In the past, gathering data for biodiversity management involved querying colleagues and conducting extensive literature reviews. But in the past two decades, vast quantities of ecological data have been made digitally accessible. Nevertheless, aggregating relevant knowledge often remains difficult and inefficient. A key challenge for the future is the development of tools for aggregating local studies to generate broader-scale patterns. International conservation projects are seriously limited by spatial gaps in biodiversity monitoring data, and geographical biases must be taken into account when extrapolating from single-site studies.

The Group on Earth Observations Biodiversity Observation Network (GEO BON) is developing a monitoring framework around a set of Essential Biodiversity Variables (EBVs) which aims at facilitating data integration, spatial scaling and contributing to the filling of gaps. EBVs have been inspired by the Essential Climate Variables (ECVs) framework of the Global Climate Observing System developed by Parties to the UN Framework Convention on Climate Change (Pereira et al. 2013). Here we build on this framework to explore the monitoring of EBV classes at the species level: species populations, species traits and community composition. We start by discussing cross-cutting issues on species monitoring such as the identification of the question to be addressed, the choice of variables, taxa and spatial sampling scheme. Next, we discuss how to monitor EBVs for specific taxa, including mammals, amphibians, butterflies and plants. We show how the monitoring of species EBVs allows monitoring changes in the supply of ecosystem services. We conclude with a discussion of challenges in upscaling local observations to global EBVs and how indicator and model development can help address this challenge.

4.2 Defining the Scope of the Monitoring Program

When designing a monitoring scheme, one needs to keep in mind three main questions: why monitor, what to monitor, and how to monitor (Yoccoz et al. 2001)? Addressing the first question is important to define the monitoring goals. The second question leads to the identification of which biodiversity variables should be monitored. Finally, the third question leads to the assessment of different sampling schemes and methods (often taxon specific). This is a process that needs to be done with great care, as once a monitoring system is established, changing it can, in some instances, invalidate all the previous monitoring efforts.

4.2.1 Surveillance and Targeted Monitoring

We can classify monitoring in two broad categories: surveillance monitoring and targeted monitoring (Nichols and Williams 2006). In surveillance monitoring, the goal is to have baseline data for one or multiple biodiversity variables. For instance, one may want to know how species population abundances are changing across as many taxa as possible. There are no a priori specific questions to be addressed.

Instead the goal is to obtain as much data as possible about that biodiversity variable over time. Data obtained by surveillance monitoring can be used for a multitude of research and management questions, with many of them defined years after the monitoring program started.

In contrast, targeted monitoring addresses specific research or management questions. For example, if the main management goal of a reserve is the protection of a specific species, monitoring the population of that species, as well as vital forage and habitat for that species, will be a necessary part of any monitoring design. Another type of targeted monitoring addresses the impact of specific drivers on biodiversity change. For instance, one may want to compare areas that receive relatively low impacts from a driver of concern to those that receive high levels of impact from that same driver and to measure all the EBVs that are likely to change with exposure to that stressor. Thus, for example, if timber harvest is the driver of concern, comparing unlogged and logged areas is likely to show a difference in the abundance of tree and other plant or animal species.

4.2.2 Choosing Which Variables, Taxa and Metrics to Monitor

Based on the available list of candidate EBVs (see www.geobon.org), we chose seven variables to discuss in this chapter that are relevant at the species level (Table 4.1). Monitoring any of these variables requires that one or more particular taxonomic group is chosen (e.g., mammals). Next, for the variables in the species population class, a key sampling design question is how many species of a given taxonomic group shall be monitored for abundance or occurrence. For instance, one may be interested in monitoring as many species as possible and therefore choose methods that assess simultaneously a wide range of species in as many locations as possible. Monitoring species population variables across entire assemblages also provides a community level overview of biodiversity change (Dornelas et al. 2014). Such broad surveys may capture population trends of abundant species, but may fall short of providing precise abundances for rare species. Instead, rare species may require targeted sampling schemes both from the point of view of spatial sampling and field methodology (Thompson 2013).

For the community composition variables, the choice of metrics to measure taxonomic diversity or species interactions become paramount (Table 4.1). For instance taxonomic diversity can be measured by many metrics, including (Magurran 2004): species richness, Simpson's diversity index, phylogenetic diversity, functional diversity, beta diversity, among others. In some cases (e.g. richness), only the presence or absence of the species is needed to calculate the metric. In others, relative abundance is required (Simpson's index), or turnover over gradients (β diversity), or cladistic information (phylogenetic), or trait information (functional).

Table 4.1 Essential biodiversity classes, essential biodiversity variables, and associated sampling design questions

Essential biodiversity class	Essential biodiversity variable	Main design choice	Metrics or taxa groups (examples)
Species populations	Species abundance	How many taxa to monitor?	Common versus rare species
	Species distribution		
	Species age structure		
Species traits	Phenology	Which metrics and how many taxa to monitor?	Metrics are taxon-dependent: flowering time, migration time
	Body mass		Harvested versus non-harvested species
Community composition	Species interactions	Which metrics to monitor?	Connectedness, length of trophic chain, interaction strength
	Taxonomic diversity		Species richness, species α and β diversity, phylogenetic diversity, etc.

For variables in the species traits class, both the general identification of which variable should be measured, what particular metric of that variable, and which species should be monitored, have to be considered (Table 4.1).

In any case, metrics and taxa to be monitored should follow a range of required and desirable criteria. Required criteria include: (1) monitoring should have a low impact on the targeted organisms over time; (2) the monitoring protocol should be reliable and repeatable with different personnel; (3) for targeted monitoring, the variable should have a strong correlation with the driver of concern; and (4) the variable should be ecologically important, that is, impacts on the variable have meaning at an ecosystem level or localised impacts are significant enough to warrant concern. The variables or metrics that meet the four required criteria are then evaluated for the desired criteria. Desired criteria include: (1) a quick response to the stressor so that effects are detectable in a short time frame; (2) a quick response to management actions so the efficacy of actions can be determined in a short time frame; (3) minimal stochastic variability so sample number can be small and effects can be clearly connected to the stressor of concern; (4) ease of measurement; (5) extended sampling window so scheduling and staff time can be more effectively allocated; (6) cost effectiveness; (7) ease of training personnel; (8) baseline data is available so effects seen are known to be stressor-caused and not a natural fluctuation; and (9) a response to the stressor can be seen when the impacts are still relatively slight; if the change cannot be detected until a large decline in resource condition occurs, alteration to the systems may be impossible or difficult to repair. The metrics that meet all the required criteria and most of the desired criteria can be chosen and then ranked, based on the number of desirable criteria they meet.

If some metrics obtain similar rankings, budgetary considerations can be used to prioritise measures to be included in the final program. A two-tier system may be adopted: Tier 1 measures can be carried out more frequently (e.g., yearly) and are either very important or less expensive. Tier 2 metrics are done less frequently (e.g., every 5 years), generally because they are expensive, destructive (e.g., material has to be collected), or require expertise that is not readily available. In addition, Tier 2 indicators can act as a check on more simplistic Tier 1 indicators. One of the major challenges with this approach is finding a way to incorporate variables of both high ecological significance and low cost. It is also important to note that the frequency of the measurements depends on the taxa being studied. Taxa with shorter life spans often require more frequent monitoring.

4.2.3 Choosing a Spatial Sampling Scheme

Despite recent advances in remote sensing for particular species (Pettorelli et al. 2014), for most taxa it is impractical to monitor an entire region at the one to five year intervals sought by many programs. Therefore, a spatial sampling scheme needs to be adopted for each monitored variable. We can broadly divide spatial sampling schemes in two major groups, extensive and site-based monitoring schemes (Fig. 4.1; Couvet et al. 2011). In extensive monitoring schemes a variable is observed at numerous sites over a large territory at regular time intervals, often using volunteers or citizen scientists (e.g., Breeding Bird Survey in North America, or the Pan European Common Bird Monitoring Scheme). In contrast, site-based or intensive monitoring schemes observe a range of variables at a limited number of sites, often associated to field stations of universities or organisations (e.g., the International Long Term Ecological Research Network—ILTER, the National Ecological Observation Network in the USA—NEON). Therefore a trade-off exists between the number of sites in a monitoring scheme (that is, its extensiveness) and the number of variables to be monitored or even the time intervals for the sampling (that is, the intensity of the monitoring effort). While extensive monitoring schemes have been very successful in providing long-term data on biodiversity change across large areas in developed regions, much of the data coming from developing regions is associated with site-based monitoring schemes (Proença et al. in press). Where volunteer capacity exists, the development of extensive national monitoring programs can be done very rapidly and it has been proposed that this model could also be applied in some developing countries (Pereira et al. 2010).

For both extensive and site-based monitoring schemes, the question of where to place the monitoring sites arises. This can be done using a systematic sampling design such as a grid, a random sampling design or a stratified random design

(Elzinga et al. 2001). One of the most common stratification schemes used is environmental stratification based on important habitat variables (Metzger et al. 2013). Sometimes a mixed design is used, for instance by systematically defining a grid and then randomly sampling inside that grid or within each habitat stratum of the grid. de Kruijter et al. (2006) provide a comprehensive guide to designing sampling frames.

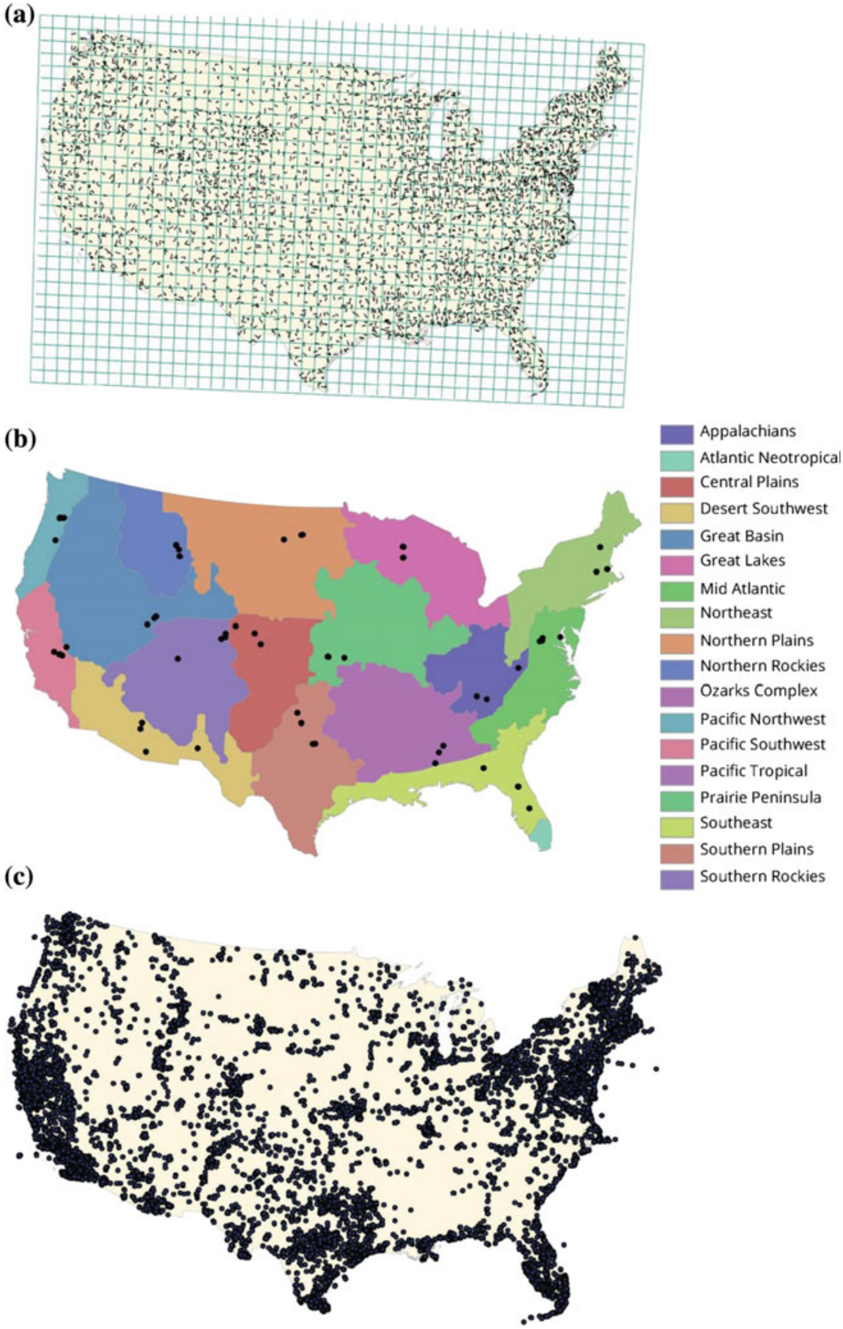
One type of spatial data that is becoming increasingly relevant is opportunistic data (Fig. 4.1c). Over the last century, much biodiversity data was collected for museums and natural history collections. For instance, the Global Biodiversity Information Facility (GBIF) indexed more than 500 million species occurence records as of 2015, many of them from such collections. More recently, the development of websites for recording and sharing species observations (Boakes et al. 2010) is mobilizing an impressive range of data almost in real-time. Despite opportunistic observations being vulnerable to multiple biases (e.g., they are often presence-only data, so it is difficult to distinguish true from false absences), Bayesian methods have been recently developed to use this data to track biodiversity change (van Strien et al. 2013). Furthermore, the interactive community features of the social web allows for mobilizing observers for biodiversity observations in novel ways.

4.3 Taxon-Specific and Driver-Specific Examples

In this section we discuss methods available to monitor species EBVs (Table 4.1), particularly species distributions (also referred to as species occupancy or species occurrences) and species abundances. We emphasise species distributions and species abundances since some other EBVs (e.g., taxonomic diversity) can be inferred from those when data is collected for entire species assemblages. We use taxon-specific examples for mammals, amphibians, butterflies, and plants. We also include an example for monitoring a specific driver: wildlife diseases.

4.3.1 Mammals

Harmonizing monitoring schemes is likely to be more challenging for mammals than for other taxa (e.g., birds), because observation techniques used for mammals are often very species-specific (Battersby and Greenwood 2004) and reliability of techniques is likely to be affected by habitat type. It is advantageous to monitor mammal species that are common and easily observed as part of a global harmonised observation system. However, at a national level, it is also important to monitor less common species, particularly those of conservation concern, because of reporting requirements from international policy agreements and to assess nationally set targets.


(a)
(b)
Appalachians
Atlantic Neotropical
Central Plains
Desert Southwest
Great Basin
Great Lakes
Mid Atlantic
Northeast
Northern Plains
Northern Rockies
Ozarks Complex
Pacific Northwest
Pacific Southwest
Pacific Tropical
Prairie Peninsula
Southeast
Southern Plains
Southern Rockies
(c)

◀ **Fig. 4.1** Spatial sampling schemes for species data. **a** Extensive monitoring in the Breeding Bird Survey of the USA. Approximately 3000 routes are monitored yearly across the USA. The original routes were placed randomly for each 1° × 1° cell, but the system has since expanded to take advantage of the proximity of cities with large numbers of observers. **b** Site-based monitoring in the National Ecological Observation Network. Each site was placed in order to be representative of an environmental domain. **c** Point species occurrence data from the iNaturalist portal, mostly opportunistic observations contributed by citizen scientists

The Tracking Mammals Partnership (TMP), established in 2005 by the Joint Nature Conservation Committee (JNCC), provides an interesting case study of a mammal monitoring programme developed at the national level. Despite a long history of natural history recording in the United Kingdom (Flowerdew 2004), reviews in the 1990s suggested a paucity of data on population, abundance, and distribution data for British mammals, prompting a call for an integrated monitoring programme to track the status of British mammals (Harris et al. 1995). The TMP is a collaborative effort between 25 organisations and uses a diverse programme of monitoring schemes, collecting data on a range of species in both urban and countryside environments, and covering a number of species relying on specialist survey methods. The TMP aims to detect changes in species distributions and abundance over time, by using stratified sampling to also provide regional trends, thus ensuring geographical representativeness (Battersby and Greenwood 2004).

Learning from monitoring efforts on bird populations was central to the development of the TMP, including through direct input from the ornithological community (Battersby and Greenwood 2004). For instance the British Trust for Ornithology (BTO) was involved in devising mammal tracking programmes such as the Winter Mammal Monitoring scheme. Specific lessons learnt included the importance of establishing long-term datasets of population indices through annual monitoring and the use of non-governmental conservation organisations and volunteers to collect data (Battersby and Greenwood 2004; Harris and Yalden 2004). While there is no single approach that suits all mammal species equally, it was suggested that a small number of monitoring techniques that can be applied to a large number of terrestrial mammal species could be integrated to form a multi-species monitoring programme (Harris and Yalden 2004). Most importantly, the chosen techniques should be applicable across a wide range of habitats to overcome biases established by past monitoring schemes focussing on specific habitat types (e.g., hedgerows, woodlands; Flowerdew 2004). By 2007, the TMP was reporting on annual trends for 35 species of terrestrial mammals.

While the TMP is less active at present, the constituent partner organisations are carrying out continued monitoring projects, some of which are run annually and cover multiple species (e.g., the National Bat Monitoring Programme run by the Bat Conservation Trust, the Breeding Bird Survey run by the BTO, and the Mini Mammal Monitoring run by the Mammal Society). Many of these monitoring schemes are based on line transects (for sightings of medium to large mammals and field signs) or live trapping transects (for small mammals) within specified grid squares (most often randomly selected 1 km^2 squares and involving two transects; e.g., Risely et al. 2012).

Transect counts are time-consuming. However, for large- to medium-sized mammals which occur at high densities in relatively open habitat, are relatively easily spotted (e.g., active at time of survey) or have field signs which are easily identifiable, transect counts can provide relatively robust estimates of species richness, relative abundances and habitat use. With help of specialist software such as DISTANCE (Thomas et al. 2010), estimates of absolute densities of species are also possible. Live trapping for small mammals has often been suggested as a key methodology for small mammal monitoring (Toms et al. 1999). Small annual changes in small mammal populations (e.g., 3–11 %) can be detected with 85 % power when monitoring is carried out for 10 years at a minimum of 50 sites (Flowerdew 2004). Other methodologies tested for use in the UK include road traffic casualties to monitor changes in relative abundance of several mammal species. With some refinement of the methodology, such as taking road type into account, the method may be sensitive enough to be used in national mammal monitoring schemes (Baker et al. 2004).

With the development of new technology, remotely monitoring mammals becomes more practical, often cutting down on man-hours spent in the field. In particular, camera trapping has been increasingly applied worldwide in monitoring and conservation (Fig. 4.2). It has been applied in a range of contexts from tracking specific species (e.g., the pygmy hippo in Sapo National Park; Collen et al. 2011), to multi-species monitoring, including tracking rare or elusive species in dense habitats such as tropical forests (Munari et al. 2011), monitoring small invasive

Fig. 4.2 Camera trapping is becoming one of the main methods to monitor medium to large mammals

mammals (Glen et al. 2013), and monitoring arboreal mammals (Cerbo and Biancardi 2012). Animal density estimation was previously only possible for species with individually recognisable markings; however, recent analytical developments have focussed on deriving methods and models to derive animal density estimates for species eliminating the requirement for individual recognition of animals (Rowcliffe et al. 2008; Chandler and Royle 2013). Methods have also been proposed to integrate data from camera trapping into biodiversity indicators (e.g., the Wildlife Picture Index; O'Brien et al. 2010; Beaudrot et al. 2016). Remote monitoring of mammals can result in large amounts of data, and the volunteer focus of traditional monitoring programmes is set to be turned into large-scale citizen scientist involvement to facilitate data processing (e.g., via species identification through mobile phone apps, such as Instant Wild; see www.edgeofexistence.org/instantwild/).

4.3.2 Amphibians

Assessing trends in amphibian populations can be challenging because they can fluctuate dramatically (Pechmann et al. 1989; Collins et al. 2009). In addition, many species often occur as meta-populations with some populations acting as 'sources' of individuals colonizing other places due to birth rates exceeding mortality rates, and some populations acting as 'sinks', receiving more animals than those that leave and where mortality rate exceeds birth rate. Therefore, it may be important to monitor the entire meta-population in order to produce meaningful results. Long-term studies have also shown that amphibian populations can vanish locally as a result of natural habitat changes that take place over decades (Collins et al. 2009).

As for other taxa, it is impossible to survey every habitat or catch every individual of a population, but ideally one should look to sample units that are separate and (statistically) independent. Sample units are usually individual animals for single population studies; they are quadrats, transects or habitat features like ponds and streams for community studies. Some monitoring programs focus on a handful of target species and report, in addition, all observations of rare species encountered during the surveys (e.g., Netherlands national monitoring scheme; Groenveld 1997).

A number of methods exist to survey species abundances and ranges for amphibians. Below we present very brief accounts of some of the most popular and promising ones:

- **Clutch counts** (also known as egg masses, spawn clumps, or batches) and **nest counts** are techniques that have been used to monitor population trends of some species and can also help to assess which factors are affecting populations. Egg mass counts have been used to assess population sizes of pool-breeding amphibians, particularly some explosive-breeding species, and they are relatively simple in that they only require surveying ponds repeatedly for clutches. Species whose eggs do not hatch very quickly (e.g., more than 10 days between laying and hatching) have higher detection probabilities (Crouch and Paton 2000). Nest counts have been used to estimate population size of some salamander species over long periods of time (e.g., Harris 2005).

- **Trapping** animals over time is a common method, either by using passive traps or by attracting animals to a trap (active traps). Nearly all passive traps for amphibians are either funnel traps or pitfall traps. Funnel traps have a funnel-shaped entrance that guides animals to a larger holding chamber, while pitfall traps consist of some type of container sunk into the ground with the rim level with the surface, and deep enough that the animals that fall into it cannot climb out (Gibbons and Semlitsch 1981). Traps are often used in combination with drift fences, which are vertical barriers that curtail the options of animals on the move and guide them towards a trap. The combination of drift fences and traps has proved very successful in some places (e.g., southern U.S.) but not in others (e.g., forests in NE Australia).
- **Area-based surveys** are used to estimate the abundance and density of a species or survey the amphibian fauna of a site. One needs to define small units within a larger area (plots or transects) that are sampled for amphibians, and, from the data collected, inferences are made about the larger area. The data can be used to compare species among habitats or to study how communities change over ecological gradients or over time. The literature indicates plots are generally square or rectangular, with median dimensions of 25 × 20 m (range 4–400 × 2–240 m); transects are narrow plots intended to be explored by a single person at a time, and their median dimensions are 100 × 2 m (range 7–2000 × 1–8 m) (Marsh and Haywood 2010). Though plots and transects are often surveyed visually, sometimes they can be sampled by registering calls. The final choice of the size, shape, and number of units to sample depends on the questions that the survey is intended to address.
- **Auditory monitoring** is a relatively efficient method for assessing frogs and toads. The method has proven a useful tool for anurans because many are more easily heard than seen and it is widely used in the U.S. and Canada (Weir and Mossman 2005). This is a good method for monitoring changes in anuran occupancy or for rough species inventories. Nevertheless, it has some limitations, as it relies on detecting singing males (and thus misses females and sub-adults), and cannot be applied to the non-singing salamanders and caecilians. More recently, automated systems, or frogloggers, are being used to collect data at single sites. Such automated systems may be the most efficient way to monitor threatened species or those with unpredictable breeding seasons in the future.
- **Environmental or e-DNA** is a promising technique that will likely be useful for detection of rare freshwater species (Ficetola et al. 2008; Thomsen et al. 2012). This technique relies on DNA obtained directly from small water samples of lakes, ponds and streams. It has been tested successfully in temperate systems for detection of amphibians, but to our knowledge is not yet being used for amphibian monitoring.

4.3.3 Butterflies

Contrary to most other groups of insects, butterflies are relatively well-documented, easy to recognise and popular with the general public. Butterflies use the landscape at a fine scale and react quickly to changes in management, intensification or abandonment. Furthermore, a sustainable butterfly population relies on a network of breeding habitats scattered over the landscape, where species exist in a meta-population structure. This makes butterflies especially vulnerable to habitat fragmentation. Moreover, as ectotherm animals, many butterflies are highly sensitive to climate change.

At the national scale the following monitoring techniques can be used to monitor species ranges and species abundances of butterflies:

- Unvalidated, **opportunistic data** can only be used for coarse distribution maps. Species distribution modelling including habitat and climate variables can be used to refine the species ranges from opportunistic data (Jetz et al. 2012). If the quantity of observations is high enough and the quality of visits can be established, the Frescalo method (Hill 2012) and occupancy modelling can be used to establish distribution trends (Isaac et al. 2014).
- **Standardised day-lists** can be used for occupancy modelling (van Strien et al. 2011). An advantage of this method is that it can work with co-variates (e.g., the Julian date, as butterflies typically have a limited flight period). Occupancy modelling with day-lists also addresses the problem of detection probability. Occupancy modelling can also produce colonisation and persistence trends, population parameters that can be very helpful to identify the causes of observed occupancy changes. It is important to note that the statistical methods for occupancy modelling are data and computation intensive.
- **Standardised counts** following a protocol is ideal for population abundance monitoring. For instance, in Europe although field methods differ to some degree across countries, most counts are conducted along fixed transects of about 1 kilometre, consisting of smaller sections, each with a homogeneous habitat type (van Swaay et al. 2008). Visits are only conducted when weather conditions meet specified criteria. Site selection varies from random stratified designs (only in a few countries), to grid design (only in Switzerland), to free observer choice (most countries). Countries use a software package called TRIM to analyse and supply trend information at the national level. Trend data are then integrated to create European population indices for species and multi-species indicators.

4.3.4 Plants

Plants, as primary producers, are effectively the basis of life on earth, and fundamental not only to many millions of species, known and unknown, but also our own. However, our knowledge of the world's flora remains limited, despite over 250 years of scientific research. In 1753 when Linnaeus published his Species Plantarum, some 5573 plant species were included; at that time, he was convinced the number would never exceed 10,000. Today, the total of known species stands at $\sim$380,000 (Paton et al. 2008) out of a total of more than 890,000 published names for plant species, with almost 2000 newly described species published annually (www.ipni.org/stats.html). Centres of plant diversity (Davis et al. 1997) and hotspots of threatened plants (www.conservation.org/hotspots) have been identified. There are many permanent forest plots that have received one or more complete censuses (e.g., the CTFS network; www.ctfs.si.edu/plots). However, this is collectively only a very small proportion of the total land area of the Earth and for many individual species there is little available data beyond the natural history collections, herbarium specimens and their original description.

Recent attempts to consolidate existing knowledge, from which EBVs and hence global biodiversity indicators must be derived, have been largely driven by international policy objectives. The botanical community has galvanised around the Global Strategy for Plant Conservation, adopted by the Convention on Biological Diversity. This Strategy has a set of targets to be achieved by 2020, including Target 1 which is to produce '*an online flora of all known plants*' and Target 2 which is to undertake '*an assessment of the conservation status of all known plant species, as far as possible, to guide conservation action*'.

Formal assessments of the conservation status of most plant species are still lacking. Only 19 728 plant species have been assessed by the Red List (www.iucnredlist.org), totalling less than 5 % of the world's flora (as of November 2014). Of those assessed, about 54 % (10,584 plant species) have been classified as threatened. The assessment of extinction risk is based on objective and quantitative criteria that capture one or more EBVs (e.g., species distribution and species abundance). This can be based, in the first instance, on opportunistically-collected herbarium specimen data and published botanical literature (Brummitt et al. 2008; Rivers et al. 2011), followed by verification and validation in the field (Brummitt et al. 2015). It is important that assessments are based on a verifiable trail of data, from maintained long-term databases, preserved herbarium specimens, or published literature sources.

Field-based monitoring techniques for plant EBVs are many and varied, including:

- **Quadrats** can be used to survey plants, as it is a particularly effective method for sessile organisms. Quadrats can be of different sizes, depending on the size of the plants and the structure of the vegetation, but need to be consistent within the study. Typically they are a few times larger than the mean size of the organisms being monitored. Quadrats should be placed at random and should be permanently marked to allow repeated measures through time. In addition, there should be a sufficient number of replicates to ensure statistical power. Within each quadrat, species can be recorded as actual counts, as some measure of cover (see below) or density or frequency, or occasionally biomass (dry weight).

Species can be grouped into higher taxonomic units such as genera or families or as functional ecological units such as graminoids (grasses and grass-like plants), forbs (herbaceous plants), shrubs, trees, and climbers. The standardised plot surveys of the Centre for Tropical Forest Science, in which each individual tree is identified, tagged, and mapped on a repeated cycle, are perhaps some of the largest quadrats ($\sim$50 ha in size) being measured with standard protocols around the world.

- **Transects** of varying width, are often employed over longer distances, especially against an environmental gradient or gradient of disturbance that intentionally includes the range of floristic variation within the area. Along each transect, each species may be recorded including information on numbers of individuals, distance from transect, cover, biomass, density or frequency.
- Placement of **quadrats along transects** has several advantages. First, quadrats along a line can be easier to relocate than if scattered across an area. Second, quadrats allow for more vegetated space to be measured along the line than compared to points along a transect line. Finally, the advantage of a transect is maintained (i.e., covering more space, thus incorporating more variability, and enabling spatial analysis).
- **Cover** can be assessed using different methods, such as the DAFOR (Dominant, Abundant, Frequent, Occasional or Rare), Braun-Blanquet (5 classes up to 100 % cover, not of equal size) and Domin (10 classes up to 100 % cover, not of equal size) scales. Each can be used with existing sampling techniques such as quadrat or transect of defined length and width. The classes for the DAFOR scale can be interpreted by the user relative to the particular situation, as long as this is consistent and stated within each study. Assessments of extinction risk under IUCN Criterion A require estimates of population size and its change over time from 'an index of abundance appropriate to the taxon', using any of these cover assessment methods across the species range, as long as this is stated and applied consistently between time points.
- **Counts** of all individuals of conspicuous plants at low densities are possible, although this is time-consuming and it can be difficult to avoid double counting. Counts are particularly challenging for densely-growing plants and clonal plants. In those situations measures of cover, of numbers of ramets (modular, repeating, connected units of the plant) or numbers of stems or reproducing stems may be used instead. For Red List assessments under IUCN Criterion C, actual counts of numbers of individuals are required, but the thresholds for threatened categories are low in value. Therefore this is a feasible technique for species of known conservation concern, although it is not generally viable for widespread and less threatened species. Frequency of presence/absence in quadrats of known size can be related to population density.
- **Mapping** vegetation over larger areas is possible using GPS points or tracks and a pre-defined habitat classification such as the National Vegetation Survey of the UK, the Braun-Blanquet vegetation types, one specified by the user, or from remotely-sensed data. Available satellite imagery can detect fine spatial resolution and variation within vegetation, even detecting characteristic individual

tree species with LIDAR data, to which image-recognition algorithms can be trained. Care needs to be given to seasonality for vegetation mapping, including the tropics where seasons tend to be defined by rainfall rather than temperature, even within apparently uniform rain forest. The combination of different methods is extremely useful in vegetation mapping, as remotely-sensed data needs validation and ground-truthing through on-the-ground observations from quadrats, transects or point surveys.

- **Environmental DNA (eDNA)** approaches, in which estimates of species richness and species abundances may be obtained from next-generation sequencing of leaf litter or soil samples, offer considerable promise for rapid ground-truthing of satellite imagery, if a suitable DNA library exists against which to compare the species.

Few plant species have sufficient data at the global or regional levels for the majority of the Essential Biodiversity Variables (Table 4.1). However, much is already known: there is a draft global species checklist (www.theplantlist.org), with synonymy and distributions for each species; species ranges are available for many vascular plants in some regions (e.g., Europe, USA); weight is one of the main traits compiled in the TRY database (Kattge et al. 2011); phenology, at least for flowering and often fruiting, can be inferred from herbarium specimens (collections are usually only made if a species is in flower or fruit, and collecting date is given on the label) and taxonomic literature; dispersal mode if not distance can be similarly inferred from fruit and seed morphology. What generally is not known for the overwhelming majority of plant species is how these variables are changing over time. Furthermore, data on local abundances and population structure is only being compiled at some research sites, such as the aforementioned forest plots (e.g., CTFS), and data on individual trophic interactions is even less available. Still, available plot data was recently used to provide a global assessment of changes in local species richness over the last few decades (Vellend et al. 2013), with the surprising result that no net change on species richness was found on the set of plots analysed.

The capacity for developing countries to undertake repeated measurements of the EBVs for which base data already exists, such as species ranges, populations, and phenology, is limited. Therefore measuring and monitoring EBVs for plants is inherently also a capacity-building exercise. Knowledge of the plants themselves and the ability to accurately identify them is of utmost importance. There is an ever-increasing availability of digital specimen data through GBIF (www.gbif.org) or other platforms, or crowd-sourced specimen databasing and georeferencing. Rapid, standardised satellite imagery can be used to monitor habitat loss and vegetation change. But it is essential to develop training workshops in assessment and monitoring techniques for local experts, provide easy-to-use identification tools and field guides, and develop long-term partnerships. Many of these approaches come together in work conducted for the IUCN Sampled Red List Index for Plants

(Brummitt and Bachman 2010) (www.threatenedplants.myspecies.info), where observable change in range size or population size is measured to re-assess the Red List status of a broadly representative sample of plant species from around the world.

4.3.5 Monitoring Diseases

Infectious wildlife diseases are emerging globally, and their adverse effects are becoming more and more visible (Fisher et al. 2012). It is therefore important to include disease surveillance or pathogen monitoring into global, regional, and national biodiversity monitoring strategies. The three main questions faced when designing a disease monitoring scheme, i.e. why, what, and how to monitor, are also relevant here. The answer to *why* to establish disease surveillance is straightforward: the adverse effects of non-native emerging infectious diseases can throw entire ecosystems out of balance and have major impacts on humans, livestock and crops (Keesing et al. 2010). The question of *what* to monitor is a bit more challenging, as one could monitor the symptoms of a disease, the disease itself, or the pathogen. Considering that disease monitoring should also be an early warning system, it might be suboptimal to monitor the symptoms of a disease or the disease itself. It is preferable to monitor the presence of a pathogen, but then, what are the EBVs needed to describe the status of a pathogen? Finally, the question of *how* to monitor pathogens needs to consider different sources of error such as the representativeness and detection probability. Random selection or stratified random selection of monitoring sites ensures that the sample will be representative for the larger area from which the sites are selected (Yoccoz et al. 2001). However, other questions might demand a different site selection strategy. Imperfect detection, or detection probability (Kéry and Schmidt 2008; Archaux et al. 2012), is of particular interest in pathogen monitoring, as pathogens are often difficult to detect (McClintock et al. 2010).

As pathogens depend on their host, pathogen monitoring often starts with monitoring of the host. In many cases, a pathogen is only detected after disease outbreaks and when negative effects on the host population become evident (Berger and Speare 1998; Blehert et al. 2009). Monitoring species distribution can detect a change in a host population linked to disease outbreaks and the presence of pathogens. Species abundance is more sensitive, but it is also more difficult to conduct over large regions. Pathogen monitoring should be conducted at the same sites (or a random subset of them) to establish the occurrence pattern of the pathogen in both space and in time and to track disease outbreaks. Once the occurrence of a pathogen has been detected, infection prevalence (the proportion of infected individuals in a population) needs to be recorded, followed by infection intensity. These two state variables will inform about the extent of the infection and will give information on the future dynamics of the disease, especially if prevalence is above a 5–10 % threshold (Knell et al. 1998). Above such a threshold, epidemics often occur. In case pathogen occurrence is clustered or when unusual mortality rates are observed, it is advisable to conduct more detailed surveys with more specific questions. This may include delineation of clusters, identification of areas of host population declines, determination of the involved variants of the pathogen, and investigating the taxonomic, seasonal and temporal variation of prevalence and infection intensity. Such information can then feed into a risk analysis for the host population(s).

Care needs to be taken that the same host species is monitored across different sites and different years to yield robust information on the pathogen. It is also important to have sufficient sample sizes when conducting detailed surveys, as otherwise false negatives may not allow delineating the distribution of the pathogen. The necessary sample size is dependent on the minimum prevalence expected if the population/specimen were infected. For example, the common prevalence of a resident disease in a population is approximately 5 %. With that level of prevalence, at least 90 specimens need testing for the likely detection of one or more positive individuals to reach 99 %. An approximation to the number of individuals that need to be tested to be 95 % certain of detecting at least 1 positive individual is $n = 3/p$ (for 99 % certainty it is $4.5/p$), where p is the prevalence expressed as a proportion (Walker et al. 2007). In case no visible symptoms of a disease can be detected, such as in the amphibian disease chytridiomycosis, detection and quantification of a pathogen might need quantitative molecular tools such as PCR (e.g., for *Batrachochytrium dendrobatidis*; see Boyle et al. 2004; Hyatt et al. 2007) or Next-Generation Sequencing.

4.4 From Species Monitoring to Ecosystem Services

Biodiversity plays several roles along the process chain that links ecosystems to human well-being and which includes ecosystem processes, final ecosystem services (i.e., services that directly underpin or give rise to goods), and the (material and non-material) goods generated by those services (Mace et al. 2012). As species may contribute to all these stages, the application of species monitoring data to ecosystem services should take into account their position in this process chain. Establishing these connections between species monitoring and ecosystem services is important to support the work of the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES; Díaz et al. 2015).

If species constitute final ecosystem services or goods, that is, if species are directly linked to services, then species population data can be directly used to monitor ecosystem services. This is usually the case of provisioning services (i.e., material ecosystem outputs that can be directly used) and cultural services (i.e., non-material ecosystem outputs with cultural or spiritual significance). Examples of provisioning services provided directly by species include, among others, food (e.g., game birds, wheat, mushrooms), fibres (e.g., cork oak, timber trees, sheep) and medicines (e.g., *Aloe* spp., medicinal herbs, poison dart frogs). Examples of cultural services include, among others, charismatic species (e.g., monarch butterflies, primates, orchids) and species inspiring technology (e.g., *Morpho* butterflies, lotus plants). Therefore, a decrease in the species abundance or species range of a game bird or a primate species corresponds to a decrease in the supply of the associated provisioning or cultural service.

In other situations, species do not constitute final services or goods, but are known to play a facilitator or intermediary role in the ecosystem processes underpinning the services. This is particularly true for regulating services (i.e., non-material ecosystem outputs not directly used by people but that affect human well-being) such as water run-off regulation or pollination, but also for some provisioning or cultural services such as clean water provision and landscape character. While individual species may play a dominant role in ecosystem processes generating services, for example, fruit tree pollination by honey bees, in most cases, ecosystem processes are affected by multiple species in a community (Díaz et al. 2007; Hillebrand and Matthiessen 2009; Lavorel et al. 2011). In these situations, data on species abundance and distribution obtained through monitoring schemes can be complemented with data on species traits (i.e., morphological, physiological and life history attributes), in order to compute community-aggregated metrics that characterise the community regarding traits of interest for a particular function. For example, data on root size and architecture can be used to assess the contribution of plant communities to water regulation and soil stability, and data on body size and feeding habits can be used to assess the pollination potential of insect communities (de Bello et al. 2010).

Species traits can also be applied in the identification of species functional groups relevant to monitoring provisioning, cultural or regulating ecosystem services. For instance, protein content could be an indicator of plants' forage value in pastures (Lavorel et al. 2011), production of medicinally important compounds, such as antioxidants and alkaloids, could be an indicator of medicinal value (Canter et al. 2005), and structural complexity could be an indicator of existence value (Proença et al. 2008).

In addition to the traits determining species contribution to ecosystem processes, final services or goods (effect traits), species can also be characterised by traits shaping their responses to pressures (response traits). These two categories of traits provide complementary information regarding species interaction with their environment, that is, species responses to external drivers and species input to ecosystem processes and services. Response traits, such as fire response traits (e.g., resprouting ability, serotiny) and habitat specialisation, can be used to assess or predict the impacts of drivers of change or conservation measures on species populations and communities. The borderline between the two categories is not strict, as some effect traits may also be response traits. For example, leaf area has an effect on evapotranspiration, and hence on water regulation, but it can also respond to drought or nutrient availability. Response traits are not only reactive to pressures, providing a way of tracking their impacts on a certain area, but also to the variation of abiotic conditions across a landscape or region (Lavorel et al. 2011). Therefore, data on abiotic variables, such as climate and physiography, are also needed when monitoring ecosystem services using species data, since abiotic factors indirectly affect ecosystem processes through effects on species functional attributes. Moreover, the contribution of species or functional groups to the processes underpinning ecosystem services should be weighed against the direct influence of abiotic factors on these processes.

4.5 Scaling from Local Observations to the Global Monitoring of Biodiversity Change

Perhaps the main challenge facing the development of EBVs at the species level is the scaling from the temporally and spatially scattered local observations to the global level. Data collection, mobilisation, sharing and harmonisation are key steps in addressing this challenge, but two additional stages are important: the development of indicators and the development of models of EBV responses to drivers of biodiversity change (Akçakaya et al. 2016).

Over the last decade significant advances have been made in developing indicators of biodiversity change as assessment and communication tools (Sparks et al. 2011; Collen et al. 2013). Indicators are able to synthesise the wealth of data in a given EBV, for example, the abundance of each species i at time t in location $[x, y]$, into a single scalar number, such as geometric mean abundance at time t. This can confer statistical robustness to indicators: when individual observations are brought together, statistics such as means and variances can be calculated. Naturally the statistical power of indicators is completely dependent on the representativeness of the underlying data, and it has been argued that indicators used in recent assessments are spatially, temporally and taxonomically biased (Pereira et al. 2012; Akçakaya et al. 2016). Indicators also allow to communicate the evolution of a particular aspect of biodiversity (e.g., mean species abundance) to the public, which can be compared to targets set by managers and policy makers (Jones et al. 2011; Geijzendorffer et al. 2016). Several species based indicators where recently used to assess international progress towards the 2020 Aichi Targets of the Convention on Biological Diversity, including the Red List Index, the Living Planet Index, the number of mammal and bird extinctions, the Wild Bird Index, and the cumulative number of alien species introduction events (Tittensor et al. 2014).

Indicators are powerful communication tools that can help to transmit succinct information about the status of biodiversity, but they may be insufficient to uncover the drivers of biodiversity change. In order to understand what is driving biodiversity change, the indicators, or even better, the EBV data itself, needs to be analysed and modelled in relation to datasets on drivers of change such as land-use change, climate change, harvest or hunting pressure, and pollution. As an example, Rittenhouse et al. (2012) found a strong response of bird species richness and abundance to land-cover changes between 1992 and 2001, using correlative models. The PREDICTS project has reviewed studies of the impact of different types of land-use change on different metrics of biodiversity using over 1 million records of species abundance and over 300,000 records of species occurrence or richness (Newbold et al. 2015). They estimated a global reduction of 10 % in local species richness based on global models of land use in relation to a historical baseline (Newbold et al. 2015). An alternative approach is to develop indicators of the effect of a driver on biodiversity, such as the indicator of the impact of climate change on European Bird populations (Gregory et al. 2009) or the community temperature index (Devictor et al. 2012).

The development of models connecting responses of EBVs such as species distribution and species abundance to drivers such as land-use or other biophysical variables that can be measured using remote sensing is particularly important to address this upscaling challenge. Such models could allow the extrapolation of point observations resulting from *in situ* monitoring into continuous variables in space and time. Species distribution models are already capable of producing spatially explicit projections, at global scale, of how a species range might respond to climate change based on a limited number of point-based observations (Peterson et al. 2011) and wall-to-wall climate data. Similar correlative models have also been used to project species distributions for different scenarios of land-use change (Jetz et al. 2007; Rondinini et al. 2011).

With the support of CSIRO, Map of Life, PREDICTS and others, GEO BON is now developing several global biodiversity change indicators (GEO BON 2015) that build on the EBV framework concept (Pereira et al. 2013). The idea is that EBVs such as species distributions can be modelled continuously in space by integrating point-based species observations, remote-sensing of habitat cover, and other biophysical data such as elevation (Jetz et al. 2012). The availability of annual updates on the distribution of global forest cover, allows one to also estimate species ranges of forest dependent species over time. Finally, for any spatial region (e.g., a country or part of a country) an indicator of the total area of suitable habitat for each species can be calculated and averaged across a taxonomic group of interest (e.g., threatened birds).

As these examples illustrate, the collaboration between volunteers and professionals collecting biodiversity data, the scientists analysing the data, and the managers acting on the data, will be critical to address the on-going biodiversity crisis. We hope the EBV framework will help harmonise and integrate the work across these different communities.

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