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Title: Evaluating and benchmarking biodiversity monitoring: metadata-based indicators for sampling design, sampling effort and data analysis

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ABSTRACT

1. The biodiversity crisis has led to a surge of interest in the theory and practice of biodiversity monitoring. Although guidelines for monitoring have been published since the 1920s, we know little on current practices in existing monitoring schemes.

2. Based on metadata on 646 species and habitat monitoring schemes in 35 European countries, we developed indicators for sampling design, sampling effort, and data analysis to evaluate monitoring practices. We also evaluated how socio-economic factors such as starting year, funding source, motivation and geographic scope of monitoring affect these indicators.

3. Sampling design scores varied by funding source and motivation in species monitoring and decreased with time in habitat monitoring. Sampling effort decreased with time in both species and habitat monitoring and varied by funding source and motivation in species monitoring.

4. The frequency of using hypothesis-testing statistics was lower in species monitoring than in habitat monitoring and it varied with geographic scope in both types of monitoring. The perception of the minimum annual change detectable by schemes matched spatial sampling effort in species monitoring but was rarely estimated in habitat monitoring.

5. **Policy implications**: Our study identifies promising developments but also options for improvement in sampling design and effort, and data analysis in biodiversity monitoring. Our indicators provide benchmarks to aid the identification of the strengths and weaknesses of individual monitoring schemes relative to the average of other schemes and to improve current practices, formulate best practices, standardize performance and integrate monitoring results.

KEYWORDS

2020 target; assessment; biodiversity observation network; biodiversity strategy; citizen science; conservation funding; environmental policy; evidence-based conservation; statistical power; surveillance

1. INTRODUCTION

The global decline of biodiversity and ecosystem services led to the adoption of several ambitious goals by the international community for 2010 and then again for 2020. Monitoring of biodiversity is instrumental in evaluating whether these goals are met. Although literature on how monitoring systems should be organized has been published since at least the mid-1920s (Cairns and Pratt, 1993), interest in the theory and practice of biodiversity monitoring has surged since 1990 (Noss, 1990; Yoccoz et al., 2001) and culminated in comprehensive, theory-based recommendations for monitoring (Balmford et al., 2003; Lindenmayer and Likens, 2009; Mace et al., 2005; Pocock et al., 2015).

Despite this growing knowledge, significant concerns regarding current practices remain (Lindenmayer and Likens, 2009; Walpole et al., 2009). A consistently voiced concern is that monitoring is not adequately founded in theory because many schemes are not designed to test hypotheses about biodiversity change even though their primary objective, almost exclusively, is to detect changes in biodiversity (Balmford et al., 2005; Nichols and Williams, 2006; Yoccoz et al., 2001). Although not all monitoring schemes require hypothesis-testing given the variety of their objectives (Pocock et al., 2015), there is also a general concern over the ability of monitoring schemes to adequately detect changes in biodiversity due to biased sampling designs, inadequate sampling effort, or low statistical power to detect changes (Di Stefano, 2001; Mihoub et al., 2017). Legg & Nagy (2006) and Lindenmayer & Likens (2009) warned that these shortcomings may lead to poor quality of monitoring, and, ultimately, to a waste of valuable conservation resources.

There is little information, however, on the prevalence of these potential methodological weaknesses in current practices of biodiversity monitoring. Descriptions of current practices are available for monitoring schemes in North America (Marsh and Trenham, 2008), and for European schemes of habitat monitoring (Lengyel et al., 2008a) and bird monitoring (Schmeller et al., 2012), however, these descriptions do not evaluate strengths or weaknesses in monitoring. Monitoring schemes are rarely known well enough for a comprehensive evaluation of current practices (Henle et al., 2010a; Schmeller et al., 2009), partly because monitoring schemes are designed for many different objectives at different spatial and temporal scales (Geijzendorffer et al., 2015; Jarzyna and Jetz, 2016; Pocock et al., 2015). Therefore, the performance of biodiversity monitoring in terms of the criteria regarded by the critiques as insufficiently considered in monitoring has not yet been assessed. Consequently, little is known about whether and how performance varies among programs by spatial and temporal scales or socio-economic drivers. Moreover, it is rarely known whether and how programs evaluate their performance, either by expert judgement on their ability to detect trends or by estimating their statistical power to detect changes (Geijzendorffer et al., 2015; Nielsen et al., 2009). Hence, there is a need to provide monitoring coordinators with standard indicators of performance so that they can evaluate their programs and revise their practices to address potential weaknesses. A clear understanding of performance in existing monitoring schemes also provides crucial information to the institutions running and funding monitoring schemes as well as to policy-makers using information from biodiversity monitoring.

Here we present an overview of current practices in biodiversity monitoring in Europe by focusing on properties that have been frequently mentioned in critiques of biodiversity monitoring. We used metadata on monitoring schemes to develop indicators for sampling design, sampling effort and type of statistical analysis. While monitoring schemes have been established for many different purposes, these three properties are regarded as generally relevant in determining the scientific quality of the information derived from biodiversity monitoring (Lindenmayer and Likens, 2009; Nichols and Williams, 2006; Yoccoz et al., 2001). Sampling design, an indicator of how well the spatial and temporal distribution of data collection is founded in sampling theory (Balmford et al., 2003), is essential for accuracy, i.e., closeness of measured trends and real trends in biodiversity. Sampling effort, the number of measurements made, is central to precision, i.e., the ability to measure the same value under identical conditions. Finally, to translate collected data into information relevant for further use, such as conservation or policy, appropriate statistical analysis of data is required to detect changes or trends with a given level of uncertainty, and confidence in the estimates should be based on the ability of the scheme to detect changes (Legg and Nagy, 2006).

Although these three indicators are generally relevant in any type of monitoring, monitoring schemes differ in their objectives and many different types of monitoring schemes exist (Pocock et al., 2015). For example, schemes in Europe have been started as early as the 1970s, are motivated by different reasons, funded by different sources, and their geographic scope ranges from local to continental (Lengyel et al., 2008a; Schmeller et al., 2012). To account for these socio-economic differences and to increase the useability of our indicators in different monitoring schemes, we evaluated the variation in indicators as a function of starting year, funding source, motivation, and geographic scope. Finally, we show how our indicators can be used by coordinators as benchmarks to assess their schemes relative to the average practice and to identify options for improvement of their monitoring schemes. We present different benchmark values for the three indicators to be meaningful for schemes monitoring different species groups and habitat types.

2. METHODS

2.1. Definition and dataset

We used Hellawell's (1991) definition of "biodiversity monitoring" as the repeated recording of the qualitative and/or quantitative properties of species, habitats, habitat types or ecosystems of interest to detect or measure deviations from a predetermined standard, target state or previous status in biodiversity. We collected metadata on biodiversity monitoring schemes in Europe in an online survey (Henle et al., 2010a). The online questionnaire contained 8 general questions and 33 and 35 specific questions on species and habitat monitoring schemes, respectively (**Table S1**, **S2**). We sent more than 1600 letters with requests to fill out the questionnaire to coordinators of monitoring schemes, government officials, national park staff, researchers and other stakeholders at institutions involved in biodiversity monitoring. The information entered was quality-checked and organized into a meta-database (http://eumon.ckff.si/monitoring).

The survey response rate was 40% (646 schemes for 1600 letters), which was comparable to the only other questionnaire-based study of biodiversity monitoring (48%) (Marsh and Trenham, 2008). Response rate varied among countries and we evaluated this bias based on the logic of Schmeller et al. (2009) (**Supporting Information S1.1**). Our metadatabase is not, and cannot be, exhaustive to involve all monitoring schemes because the universe of all schemes is not known, however, it provides a cross-section of geographic scope (**Supporting Information S1.1**). The final dataset contained metadata on 470 species schemes and 176 habitat schemes, or a total of 646 schemes from 35 countries in Europe. Assessment of country bias showed no substantial differences from the usual publication bias for 25 (or 71%) of the 35 countries, overrepresentation for three countries and underrepresentation for seven countries (**Fig. S1**).

2.2. Indicator development

To compute an indicator of sampling design, we scored seven design variables in both species and habitat monitoring schemes (**Table 1**). Scores were chosen to be higher for sampling designs that were better founded in sampling theory and/or that obtained more or better, e.g. quantitative rather than qualitative, information on species and habitats (further details: **Supporting Information S1.3**). Scores were determined for each scheme as a consensus among DSS, KH and SL. As a final output, we calculated a 'sampling design score' (*SDS*) indicator as the sum of the seven scores (range: 0-13 in species schemes, 0-10 in habitat schemes).

For sampling effort, we derived both a temporal and a spatial indicator. We used the following formula for the "temporal sampling effort" indicator:

$$SE_{temp} = \log(F_{by}(T^2 - 1)(T*F_{wy} - 2)), \qquad \text{(eqn 1)}$$

where F_{by} is the between-year frequency of sampling (value of 1 indicating monitoring in every year, 0.5 for monitoring every other year, etc.); T is the duration of monitoring in years; and F_{wy} is the number of sampling occasions (site visits) within a year. A derivation of equation 1 is given in **Supporting Information S1.4**.

For the "spatial sampling effort" indicator ($SE_{spatial}$), we used information on the number of sampling sites and the total area monitored. Assuming that more sampling sites in equal-sized areas indicate higher sampling effort, we calculated the residuals from an ordinary least-squares regression of the number of sites (log-transformed response) over the total area monitored (log-transformed predictor). Positive values (above the fitted line) indicate higher-than-average effort, whereas negative values (below the fitted line) indicate lower-than-average effort for equal-sized areas.

Each of these three indicators (SDS, SE_{temp}, $SE_{spatial}$) is negatively proportional to at least one source of variation (temporal, among-site, or within-site) that increases the variance of the trend estimate from monitoring. Hence the higher the values of the indicators, the better the sampling design, the higher the sampling effort, and the higher the precision of the trend estimate. The three indicators cannot be readily integrated but have the advantage that coordinators of monitoring schemes can easily calculate them based on Eq. (1) or the regression equations and can use them as benchmarks (see Results).

For the "type of data analysis" indicator, we used information on the analytical method as given by the coordinators. The single-choice options were (i) descriptive statistics or graphics, (ii) simple linear regression, (iii) advanced statistics, e.g. general linear models etc, (iv) other analyses, (v) data analyzed by somebody else, or (vi) data not analyzed. We considered options (i) and (vi) as evidence for the lack of inferential statistics and hypothesis-testing and considered all other options as signals for hypothesis-testing. Although the option 'data analyzed by someone else' could also involve descriptive statistics or graphics, i.e., no hypothesis-testing, this option was chosen for only 26 species schemes (<6% of 439 responses) and four habitat schemes (<3% of 154 responses), and pooling these into either group did not influence our results.

Finally, to evaluate the coordinators' expert judgement of the ability of their schemes to detect changes, we asked coordinators to estimate the precision of their scheme as the minimum annual change per year in the monitored property (e.g. population size, habitat area) that is detectable by their scheme (1%, 5%, 10%, 20%, or more). We then correlated these "precision estimates" with our temporal and spatial indicators of sampling effort to test whether coordinators correctly estimated the sampling effort of their schemes. We arbitrarily took 30% for responses of 'more than 20%'. We found that using different percentages (40%, 50% etc.) did not qualitatively affect our conclusions.

2.3. Socio-economic effects

We analyzed the variation in each indicator caused by four socio-economic factors: (i) starting year, (ii) main funding source (European Union [EU], national, regional, scientific grant, local), (iii) motivation (EU directive, other international law, national law, management/restoration, scientific interest, other), and (iv) geographic scope (pan-European, international, national, regional, local). These factors were chosen because they are fundamentally important in biodiversity monitoring and because knowledge of how these factors impact the indicators (e.g. "sampling designs are more advanced in schemes funded by certain types of donors") will influence how monitoring coordinators and institutions interpret and use the indicators.

To detect changes in certain time periods, we classified schemes by starting year in four time periods of European biodiversity policy: (i) period 1: years until the adoption of the Birds

Directive in 1979, (ii) period 2: from 1980 until the adoption of the Habitats Directive in 1992, (iii) period 3: 1993 until 1999, and (iv) period 4: since 2000 or the preparations of the 2010 biodiversity targets. For funding source, motivation, and geographic scope, we used the single-choice responses as given by the coordinators.

2.4. Data processing

The three indicators had heterogeneous variances and/or non-normal distributions, and the scales of the predictor and the response variables could differ so that comparisons based on parametric test statistics (e.g. means) would have an unclear meaning. Therefore, we present results using boxplots to illustrate differences and use Kruskal-Wallis tests to compare medians. Sample sizes differ because not all information was available for all schemes.

3. RESULTS

3.1. Sampling design and effort

In species monitoring, *SDS* was similar through time and geographic scope (**Fig. 1**; Kruskal-Wallis test, n.s.) but varied by funding source (H = 15.156, df = 5, P = 0.010) and motivation (H = 17.029, df = 5, P = 0.004). *SDS* was higher in schemes funded by scientific grants than in other schemes, and lower in schemes motivated by national laws than in other schemes (**Fig. 1**). SE_{temp} decreased with time (H = 261.088, df = 3, P < 0.0001) and varied by funding source and motivation (**Fig. 2**). SE_{temp} was higher in schemes funded by private sources than in other schemes (H = 32.173, df = 5, P < 0.0001) and was lower in schemes motivated by EU directives than in other schemes (H = 82.625, df = 5, P < 0.0001). $SE_{spatial}$ decreased with time (H = 12.817, df = 3, P = 0.005) and was lower in schemes motivated by international laws and higher in schemes motivated by ‘other reasons’ than in other schemes (**Fig. 3**, H =

11.554, $df = 5$, $P = 0.041$). $SE_{spatial}$ did not vary significantly by funding source and geographic scope (**Fig. 3**).

In habitat monitoring, *SDS* decreased with time ($H = 7.974$, $df = 3$, $P = 0.047$), but did not differ by funding source, motivation, or geographic scope (**Fig. 4**). SE_{temp} also decreased with time ($H = 51.324$, $df = 3$, $P < 0.0001$), but did not vary by funding source, motivation, or geographic scope (**Fig. 5**). Finally, $SE_{spatial}$ did not vary by any of the four predictors (**Fig. 6**).

3.2. Data analysis

The proportion of schemes using hypothesis-testing statistics was significantly lower (48%) in species schemes ($n = 439$) than in habitat schemes (69%; $n = 157$; $\chi^2 = 20.838$, $df = 1$, $P < 0.0001$). In species monitoring, this proportion did not differ by starting period (range: 40-52%) or funding source (36-53%; χ^2 -test, n.s.). However, hypothesis-testing statistics were more frequent in schemes motivated by scientific interest (56%, $n = 172$) than in schemes motivated by EU directives (28%, $n = 67$), other reasons (31%, $n = 26$), or international law (33%, $n = 15$), national laws (43%, $n = 107$), management/restoration (43%, $n = 82$; $\chi^2 = 18.267$, $df = 5$, $P = 0.003$). Hypothesis-testing statistics were also more frequent among schemes of European or international scope (63% each, $n = 8$ and 16, respectively) than in local schemes (32%, $n = 114$) (national: 49%, $n = 203$; regional: 45%, $n = 128$; $\chi^2 = 16.007$, $df = 4$, $P = 0.003$).

In habitat monitoring, hypothesis-testing statistics were more frequent in schemes started in period 2 and 3 (71% of $n = 17$ in period 2 and 74% of $n = 77$ in period 3) than in schemes started in period 1 (50%, $n = 8$) or period 4 (49%, $n = 72$) ($\chi^2 = 12.967$, $df = 3$, $P = 0.005$). In addition, these statistics were more frequent in schemes whose geographic scope was national

(60%, n = 35) and local (72%, n = 87) rather than regional (44%, n = 48; European and international schemes excluded due to low sample size; χ^2 = 11.855, df = 2, P = 0.003). The frequency of hypothesis-testing statistics did not differ by funding source (range 40-67%) or motivation (range 53-86%; χ^2-test, n.s.).

3.3. Precision estimates vs. sampling effort

Coordinators estimated the minimum annual change detectable by their schemes in 74% of species schemes (n = 470) and in only 36% of habitat schemes (n = 176). In species schemes, $SE_{spatial}$ correlated negatively with precision estimates, as expected (Spearman rho = -0.128, n = 309, P = 0.024), whereas SE_{temp} was not related to precision estimates. In habitat schemes, there were no correlations between SE_{temp} or $SE_{spatial}$ and precision estimates.

3.4. Benchmarking: how do single schemes perform?

Our indicators provide benchmarks against which single schemes can be compared. Coordinators can compute these indicators for their own schemes in three steps. First, the *SDS* indicator is calculated by selecting the response options of their own scheme for each of the seven variables in **Table 1**, reading the corresponding score value, and summing the seven score values, which can then be compared to the reference mean *SDS* value given in **Table 2** for major species groups and habitat types. Second, the SE_{temp} indicator is calculated by substituting the values of a given scheme into Equation 1, which then can be compared to the reference values given in **Table 2**. Finally, $SE_{spatial}$ is obtained by calculating the difference between the number of sampling sites in a given scheme and the mean number of sites predicted for schemes that monitor similar areas. The mean predicted number is determined by regression equations based on intercepts and regression coefficients in **Table 3**. For example, the mean number of sampling sites predicted for schemes monitoring higher plants in an area of 100 km^2 is given as $log(Y) = 0.47 + 0.34*log(100) = 1.15$ (where 0.47 and 0.34 are from **Table 3**), resulting in $Y \approx 14$. If the given scheme monitors higher plants at 20 sites in an area of 100 km^2, the value of $SE_{spatial}$ (scheme value − predicted value) is 6, indicating a higher-than-average effort than in other schemes. The regression equation for $SE_{spatial}$ in habitat schemes is $log(Y) = 0.51 + 0.36*log(X)$, where X is the area monitored in km^2 and Y is the predicted number of sites. Separate regressions for habitat types were not meaningful due to low sample size in several habitat types (**Table 2**).

4. DISCUSSION

4.1. General patterns in monitoring

This study is the first to provide a comprehensive evaluation of sampling design, sampling effort and data analysis in biodiversity monitoring based on indicators calculated from metadata on existing schemes. Despite limitations in the data (see **Supporting Information**), our evaluation is based on the most comprehensive dataset currently available on existing schemes. A full validation of the indicators is not yet possible due to the absence of quantitative estimates of statistical power and accuracy derived from monitoring data in existing schemes, which could provide an independent reference. For a correct interpretation, we note that our metadatabase showed overrepresentation for 9% of the countries and underrepresentation for 20% of the countries relative to the usual publication bias, therefore, not all our results apply equally to all 35 countries represented in the metadatabase.

Our results provide evidence that biodiversity monitoring varies with the socio-economic background. We found decreasing trends in SE_{temp} in species schemes and in *SDS* and SE_{temp} in habitat schemes over time. Hypothesis-testing statistics were also less frequently used in more recent species schemes than in earlier (1980s-1990s) ones despite several calls for hypothesis-testing (Balmford et al., 2005; Lindenmayer and Likens, 2009; Nichols and Williams, 2006; Yoccoz et al., 2001). Similar results were reported by Marsh & Trenham (2008), who found a recent increase in the percentage of North American species schemes that did not decide on statistical methods.

We also found higher *SDS* in schemes funded by scientific grants and higher SE_{temp} in schemes funded by private sources than in other schemes. The influence of motivation in species schemes was less expected, with lower *SDS* in schemes motivated by national laws, lower SE_{temp} in schemes motivated by EU directives, lower $SE_{spatial}$ in schemes motivated by international laws, and lower frequency of hypothesis-testing statistics in schemes motivated by EU directives and other international laws than in other schemes. Finally, the use of hypothesis-testing statistics increased with geographic scope in species monitoring, whereas it decreased from national to regional schemes in habitat monitoring. Each of the four socio-economic variables was associated with substantial variation in at least one of the indicators, suggesting that biodiversity monitoring is influenced by socio-economic factors (Bell et al., 2008; Schmeller et al., 2009; Vandzinskaite et al., 2010).

4.2. Promising developments

Our results draw attention to several promising developments in current biodiversity monitoring. First, *SDS* did not change substantially over time, indicating that despite the continuous growth in the number of schemes (e.g. Lengyel et al., 2008a), the quality of the sampling design used in schemes is not deteriorating. Second, we found less variation in indicators in habitat schemes than in species schemes. This is probably related to the fewer habitat schemes present in our sample. In addition, habitat monitoring is methodologically less heterogeneous, based mostly on field mapping and remote sensing (Lengyel et al., 2008a), than species monitoring, where different species groups are monitored with different methods even in single taxonomic groups, such as birds (Schmeller et al., 2012). Finally, the precision estimates given by monitoring coordinators corresponded with spatial sampling effort in species monitoring schemes as expected (i.e., more sites relative to area = higher precision).

4.3. Reasons for concern

Our survey also confirmed several concerns. First, while the number of schemes increases as general interest in biodiversity conservation increases (Henle et al., 2013), we found that sampling effort decreased over time, mainly because the number of temporal replicates per unit area decreased, both in species and in habitat schemes. This is especially alarming in species schemes where repeated observations over shorter time periods (i.e., within a season) are essential to estimate the probability of detecting individuals (Schmeller et al., 2015).

Second, we identified lower-than-average values for several indicators in species monitoring: in national schemes (*SDS*), and in schemes motivated by EU directives (SE_{temp}) and other international laws ($SE_{spatial}$). Furthermore, we found that data are less frequently analyzed in species schemes motivated by EU directives and other international laws and in habitat schemes that are local or regional. These results support the view that the policies guiding monitoring and the institutions providing funding should develop standard criteria for initiating/funding different schemes (Legg and Nagy, 2006). These criteria should include minimum requirements for sampling design and effort that ensure that the performance of the individual schemes moves towards the average of all existing schemes.

Third, precision estimates were much less frequently specified in habitat schemes (36%) than in species schemes (74%). On one hand, this is plausible as it is probably easier to specify precision in schemes that monitor one or a few species than in schemes that monitor entire habitat types, i.e., species communities. On the other hand, many habitat monitoring schemes use standardized methods to document spatial variation, e.g. field mapping or remote sensing, which should facilitate the evaluation of precision.

Finally, hypothesis-testing statistics were used in less than half of the species schemes and more than two-thirds of the habitat schemes. Thus, our results support previous concerns over the lack of a hypothesis-testing framework in biodiversity monitoring (Legg and Nagy, 2006; Lindenmayer and Likens, 2009; Yoccoz et al., 2001). The infrequent use of hypothesis-testing statistics and the large number of schemes for which no precision estimate was given by the coordinators also suggest that the ability of schemes to detect changes in biodiversity (statistical power) is rarely considered in monitoring design (Di Stefano, 2001; Marsh and Trenham, 2008).

4.4. Recommendations

The variation in indicators can potentially have serious consequences regarding the ability of monitoring schemes to detect trends or the reliability of the trend estimates detected, which can thus easily provide misleading information on changes in biodiversity. Our results provide insight into potential areas of improvement that can help to avoid such potential consequences. Generally, sampling design can be improved by applying levels associated with higher scientific quality to one or more of the variables listed in **Table 1**. An ideal habitat monitoring scheme should apply both remote sensing and field mapping to document spatial changes because the two approaches work best at different scales (Lengyel et al., 2008b). The introduction of an experimental approach in monitoring, with adequate controls, was proposed as the greatest potential for improvement as it provides an opportunity to establish causal relations between trends and possible drivers of the trends (Lindenmayer and Likens, 2009; Yoccoz et al., 2001). Because experiments may have limited external validity due to limitations in the scale at which experiments can be performed, they should be complemented by observational studies addressing the same issues at the relevant larger scale (Lepetz et al., 2009) or by studies using natural experiments that are not controlled for scientific or monitoring reasons (Henle, 2005).

In principle, sampling effort can be improved by increasing either the number of sites, site visits, samples, or the frequency of sampling. In contrast to sampling design, where there is often a trade-off between options, the spatial and temporal intensity of sampling can be increased simultaneously and independently. It is fundamental to have accurate (unbiased) and precise (low-variance) estimates for the trend of the habitats of interest by ensuring adequate spatial and temporal replication (Lindenmayer and Likens, 2009). Estimating the adequate number of replicates should be based on a quantitative evaluation of the ability of monitoring schemes to detect trends in explicit analyses of statistical power (Nielsen et al., 2009; Taylor and Gerrodette, 1993).

To address the alarmingly rare use of hypothesis-testing statistics, we recommend that responsible international institutions and national agencies as well as funding agencies establish mechanisms, including procedural requirements and training opportunities, to facilitate a better use of the data collected. Because several schemes used other, unspecified statistics, it needs further study to determine the type of these analyses and to evaluate whether such unspecified statistics are appropriate for integration across monitoring schemes

(Henry et al., 2008; Mace et al., 2005). Using advanced statistics to analyze data from otherwise well-designed sampling is a straightforward way to improve the quality of information derived from monitoring data (Balmford et al., 2005; Di Stefano, 2001; Yoccoz et al., 2001).

4.5. Benchmarking: practical help for implementing recommendations

Although scientifically desirable, it may not be realistic to expect that monitoring schemes improve or change everything to have state-of-the-art practices given the many goals they pursue and the many constraints under which they operate (Bell et al., 2008; Marsh and Trenham, 2008; Schmeller et al., 2009). It is more realistic to provide the monitoring community with guidelines on how to improve schemes relative to the average practice (Henle et al., 2013). Our study provides a basis for such practical guidance in two ways. First, by revealing the impact of socio-economic factors on biodiversity monitoring, our study provides knowledge on the impacts of starting time, funding source, motivation and geographic scope on three general properties of biodiversity monitoring, which should ideally be explicitly considered in decisions made by monitoring coordinators and institutions. Second, our study provides three indicators and presents different indicator values for use in monitoring schemes that differ in their monitored object (**Tables 2** and **3**). Coordinators can thus identify the strengths and weaknesses in sampling design, effort and data analysis in their schemes relative to the average of existing schemes in a benchmarking approach. It will in turn enable coordinators to design and implement changes that may improve the ability of their schemes to collect more broadly useable data. By modifying the values of the indicators, coordinators can further assess which of the alternative options available to them would more efficiently increase the performance of their scheme.

Although the benchmarking proposed here does not provide a quantitative assessment of statistical power, its relative ease of use compared to a rigorous assessment of statistical power can make it widely applicable in many different monitoring schemes. We note that our benchmarking method is relative, i.e., the outcome for a single scheme will depend on the values of the other schemes. We aimed to minimize this variation by presenting different benchmark values for schemes monitoring different groups of species or types of habitat (**Table 2** and **3**). In addition, cooordinators and institutions should also look at how the four socio-economic factors modify the values of the indicators to develop a joint interpretation of the indicator values relative to the average practice and of the indicator values in schemes with similar socio-economic background. These two types of information will help coordinators and institutions to fine-tune the benchmarking of their monitoring schemes, to identify areas of strengths and weaknesses relative to the average practice and to address options for improving their own practice.

Ongoing efforts, both to build monitoring schemes from scratch and to improve existing schemes, such as regional and global Biodiversity Observation Networks (Wetzel et al., 2015), can benefit from the insight gained from comparing their plans with characteristics of existing schemes. Furthermore, the evaluation and benchmarks may be used in the integration of monitoring results in large-scale assessments of biodiversity and ecosystem services, e.g. under the Convention on Biological Diversity, assessments of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services or in citizen-science programs.

5. CONCLUSIONS

We acknowledge that a direct and full application of scientifically credible criteria to biodiversity monitoring practice may be overzealous and inadequate and that other approaches may be more appropriate. Our study, however, suggests that while there are many promising developments in biodiversity monitoring that do not deserve the critique sometimes voiced against monitoring, there is also a need to improve current practices in sampling design, sampling effort and data analysis. Such concerns have been voiced in several previous studies based mostly on anecdotal data or personal observations. Our study provides the first comprehensive evaluation of actual practices to back up these concerns and to show where these are little justified and offers a practical framework based on benchmarking to address several of these concerns.

6. AUTHOR CONTRIBUTIONS

KH, PYH, SL and DSS designed the study. KH, PYH, BK, MK, SL and DSS collected data. BK, SL and YPL analysed and interpreted data. BK and SL wrote the first draft and all authors contributed to final manuscript writing.

7. DATA ACCESSIBILITY

All metadata used are available for browsing or download upon request from the DaEuMon database at http://eumon.ckff.si/about_daeumon.php.

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9. REFERENCES

Balmford, A., Crane, P., Dobson, A., Green, R.E., Mace, G.M., 2005. The 2010 challenge: data availability, information needs and extraterrestrial insights. Philosophical Transactions of the Royal Society B-Biological Sciences 360, 221-228.

Balmford, A., Green, R.E., Jenkins, M., 2003. Measuring the changing state of nature. Trends in Ecology & Evolution 18, 326-330.

Bell, S., Marzano, M., Cent, J., et al., 2008. What counts? Volunteers and their organisations in the recording and monitoring of biodiversity. Biodiversity and Conservation 17, 3443-3454.

Cairns, J.J., Pratt, J.R., 1993. A history of biological monitoring using benthic macroinvertebrates, in: Rosenberg, D.M., Resh, V.H. (Eds.), Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman & Hall, New York, pp. 10-27.

Di Stefano, J., 2001. Power analysis and sustainable forest management. Forest Ecology and Management 154, 141-153.

Geijzendorffer, I.R., Targetti, S., Schneider, M.K., et al., 2015. How much would it cost to monitor farmland biodiversity in Europe? Journal of Applied Ecology 53, 140-149.

Hellawell, J.M., 1991. Development of a rationale for monitoring, in: Goldsmith, F.B. (Ed.), Monitoring for Conservation and Ecology. Chapman & Hall, London.

Henle, K., 2005. Lessons from Europe, in: Lannoo, M. (Ed.), Amphibian Declines: the Conservation Status of United States species. University of California Press, Berkeley, pp. 64-74.

Henle, K., Bauch, B., Auliya, M., Külvik, M., Pe'er, G., Schmeller, D.S., Framstad, E., 2013. Priorities for biodiversity monitoring in Europe: A review of supranational policies and a novel scheme for integrative prioritization. Ecological Indicators 33, 5-18.

Henle, K., Bauch, B., Bell, G., Framstad, E., Kotarac, M., Henry, P.Y., Lengyel, S., Grobelnik, V., Schmeller, D.S., 2010a. Observing biodiversity changes in Europe, in: Settele, J., Penev, L., Georgiev, T., Grabaum, R., Grobelnik, V., Hammen, V., Klotz, S., Kotarac, M., Kuhn, I. (Eds.), Atlas of Biodiversity Risk. Pensoft Publishers, Sofia.

Henle, K., Kunin, W., Schweiger, O., et al., 2010b. Securing the conservation of biodiversity across administrative levels and spatial, temporal, and ecological scales research needs and approaches of the SCALES project. Gaia - Ecological Perspectives for Science and Society 19, 187-193.

Henry, P.Y., Lengyel, S., Nowicki, P., et al., 2008. Integrating ongoing biodiversity monitoring: potential benefits and methods. Biodiversity and Conservation 17, 3357-3382.

Jarzyna, M.A., Jetz, W., 2016. Detecting the multiple facets of biodiversity. Trends in Ecology & Evolution 31, 527-538.

Legg, C.J., Nagy, L., 2006. Why most conservation monitoring is, but need not be, a waste of time. Journal of Environmental Management 78, 194-199.

Lengyel, S., Déri, E., Varga, Z., et al., 2008a. Habitat monitoring in Europe: a description of current practices. Biodiversity and Conservation 17, 3327-3339.

Lengyel, S., Kobler, A., Kutnar, L., Framstad, E., Henry, P.Y., Babij, V., Gruber, B., Schmeller, D., Henle, K., 2008b. A review and a framework for the integration of biodiversity monitoring at the habitat level. Biodiversity and Conservation 17, 3341-3356.

Lepetz, V., Massot, M., Schmeller, D.S., Clobert, J., 2009. Biodiversity monitoring: some proposals to adequately study species' responses to climate change. Biodiversity and Conservation 18, 3185-3203.

Lindenmayer, D.B., Likens, G.E., 2009. Adaptive monitoring: a new paradigm for long-term research and monitoring. Trends in Ecology & Evolution 24, 482-486.

Mace, G.M., Delbaere, B., Hanski, I., Harrison, J., Garcia, F., Pereira, H., Watt, A., Weiner, J., Murlis, J., 2005. A User's Guide to Biodiversity Indicators. European Academy of Sciences Advisory Board, Liege.

Marsh, D.M., Trenham, P.C., 2008. Current trends in plant and animal population monitoring. Conservation Biology 22, 647-655.

Mihoub, J.B., Henle, K., Titeux, N., Brotons, L., Brummitt, N., Schmeller, D.S., 2017. Setting temporal baselines for biodiversity: the limits of available monitoring data for capturing the full impact of anthropogenic pressures. Scientific Reports 7, 41591.

Nichols, J.D., Williams, B.K., 2006. Monitoring for conservation. Trends in Ecology & Evolution 21, 668-673.

Nielsen, S.E., Haughland, D.L., Bayne, E., Schieck, J., 2009. Capacity of large-scale, long-term biodiversity monitoring programmes to detect trends in species prevalence. Biodiversity and Conservation 18, 2961-2978.

Noss, R.F., 1990. Indicators for monitoring biodiversity: a hierarchical approach. Conservation Biology 4, 355-364.

Pocock, M.J.O., Newson, S.E., Henderson, I.G., et al., 2015. Developing and enhancing biodiversity monitoring programmes: a collaborative assessment of priorities. Journal of Applied Ecology 52, 686-695.

Schmeller, D.S., Henle, K., Loyau, A., Besnard, A., Henry, P.Y., 2012. Bird-monitoring in Europe - a first overview of practices, motivations and aims. Nature Conservation 2, 41-57.
Schmeller, D.S., Henry, P.Y., Julliard, R., et al., 2009. Advantages of volunteer-based biodiversity monitoring in Europe. Conservation Biology 23, 307-316.
Schmeller, D.S., Julliard, R., Bellingham, P.J., et al., 2015. Towards a global terrestrial species monitoring program. Journal for Nature Conservation, 51-57.
Taylor, B.L., Gerrodette, T., 1993. The uses of statistical power in conservation biology - the Vaquita and the Northern Spotted Owl. Conservation Biology 7, 489-500.
Vandzinskaite, D., Kobierska, H., Schmeller, D.S., Grodzińska-Jurczak, M., 2010. Cultural diversity issues in biodiversity monitoring - cases of Lithuania, Poland and Denmark. Diversity 2, 1130-1145.
Walpole, M., Almond, R.E.A., Besancon, C., et al., 2009. Tracking progress toward the 2010 biodiversity target and beyond. Science 325, 1503-1504.
Wetzel, F.T., Saarenmaa, H., Regan, E., et al., 2015. The roles and contributions of Biodiversity Obervation Networks (BONs) in better tracking progress to 2020 biodiversity targets: a European case study. Biodiversity 16, 137-149.
Yoccoz, N.G., Nichols, J.D., Boulinier, T., 2001. Monitoring of biological diversity in space and time. Trends in Ecology & Evolution 16, 446-453.

10. SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

S1. Supplementary Methods
S1.1. Country bias
S1.2. Questionnaire variables
S1.3. Rationale for scores for sampling design
S1.4. Theoretical underpinning for the temporal indicator of sampling effort
S2. Supplementary Results
Country bias and other potential biases
S3. Supplementary Figure
S4. References

TABLES

Table 1. Scores allocated to different levels of variables describing the sampling design used in species and habitat monitoring schemes in Europe. Please see Supporting Information for justification of score values.

Object monitored	Variable	Response option	Score
Species	Monitored property	Population trend	0
		Distribution trend	1
		Community/ecosystem trend	2
		Population + distribution trend	1
		Population + community trend	2
		Distribution + community trend	3
		All three of the above	3
	Data type	Presence/absence	0
		Age/size structure	1
		Phenology	1
		Counts	2
		Mark-recapture	3
	Information on population structure	No	0
		Yes	1
	Stratification of sampling design	No	0
		Yes	1
	Experimental design	Not used	0
		Before/after comparison	1
		Controlled experiment	2
		Before/after plus control	3
	Selection of sampling sites	Expert/personal knowledge or other criteria	0
		Exhaustive, random, or systematic	1
	Detection probability	Not quantified	0
		Quantified	1
Habitats	Monitored property	Species composition (quality)	0
		Distribution (quantity)	1
		Both of the above (quality and quantity)	2
	Data type	Species presence/absence	0
		Species abundance	1
	Documentation of spatial variation	Not reported / no spatial aspect	0
		Field mapping	1
		Remote sensing	2
	Extent of monitoring	Certain habitat types in an area	0
		All habitat types in area	1
	Stratification of sampling design	Not stratified	0
		Stratified	1
	Experimental design	Not used	0
		Used	1
	Selection of sampling sites	Expert/personal knowledge or other criteria	0
		Exhaustive, random, or systematic	1

Table 2. Means ± standard deviations (S.D.) of sampling design score (*SDS*) and the temporal sampling effort index (SE_{temp}) in species and habitat monitoring schemes; *N*: number of schemes with metadata.

	SDS			SE_{temp}		
Monitored object	**Mean**	**S.D.**	**N**	**Mean**	**S.D.**	**N**
Taxon group in species monitoring						
Lower plants	4.9	1.63	22	3.3	0.77	20
Higher (vascular) plants	4.8	2.14	41	3.4	1.07	39
Arthropods (mainly insects)	5.1	2.00	34	3.6	1.05	27
Butterflies	5.0	1.97	38	4.1	1.26	34
Fish and macroinvertebrates	5.3	1.93	27	3.2	0.99	23
Amphibians and reptiles	5.2	1.83	43	4.0	0.91	40
Birds in general	5.2	1.74	59	4.2	1.15	54
Birds of prey	5.8	2.19	21	4.4	1.00	20
Waterbirds	4.8	1.66	53	4.5	1.03	52
Songbirds	5.4	1.82	27	4.3	0.78	27
Bats	4.1	2.07	23	3.3	0.77	22
Small mammals	4.6	1.91	28	3.7	0.93	27
Large mammals	4.5	1.69	40	3.7	1.03	34
Multiple taxon groups	5.7	1.77	14	3.9	0.79	10
All taxon groups combined	5.0	1.89	470	3.9	1.08	429
EUNIS category in habitat monitoring						
A marine only	5.3	1.92	12	3.4	0.16	3
AB marine and coastal	5.6	1.75	11	3.7	1.31	2
B coastal only	6.5	2.83	16	3.0	0.92	10
C wetlands	4.2	2.09	11	3.7	1.20	4
D heaths and fens	5.7	3.01	13	3.3	0.64	10
E grasslands	5.5	2.37	16	3.2	0.62	15
F scrubs	6.8	2.48	6	4.0	0.37	3
G forests	5.2	1.66	41	3.4	1.01	25
H caves	6.5	0.71	2	5.4	–	1
I arable land	5.5	0.71	2	3.7	0.45	2
X habitat complexes	6.0	2.14	8	3.2	0.80	7
All habitat types in an area	5.0	2.35	38	3.3	1.37	22
All EUNIS habitat categories combined	5.4	2.23	176	3.3	0.98	104

Table 3. Parameters estimated from an ordinary least-squares regression of the number of sampling sites over the area monitored in species monitoring schemes targeting major taxonomic groups

Taxon group	**Intercept**	**Slope**	**S.E. slope**	**R^2**	**t**	**p**
Lower plants	1.40	0.15	0.149	0.056	0.977	0.343
Higher (vascular) plants	0.47	0.34	0.083	0.336	4.148	0.000
Arthropods (mainly insects)	0.46	0.30	0.070	0.397	4.292	0.000
Butterflies	0.52	0.35	0.077	0.411	4.579	0.000
Fish and macroinvertebrates	0.89	0.15	0.099	0.108	1.515	0.146
Amphibians and reptiles	0.82	0.22	0.105	0.119	2.139	0.040
Birds in general	1.42	0.13	0.090	0.050	1.465	0.151
Birds of prey	0.84	0.12	0.177	0.024	0.669	0.512
Waterbirds	1.55	0.04	0.101	0.005	0.420	0.677
Songbirds	0.45	0.20	0.079	0.216	2.516	0.019
Small mammals	0.33	0.25	0.070	0.351	3.601	0.001
Bats	0.88	0.15	0.112	0.091	1.339	0.197
Large mammals	0.21	0.34	0.088	0.343	3.895	0.001
Multiple groups	0.49	0.59	0.137	0.696	4.284	0.003

FIGURE LEGENDS

Figure 1. Sampling design score (SDS) in species monitoring schemes vs. starting period (A), funding source (B), motivation (C) and geographic scope (D). Boxplots show the median (horizontal line), the 25th and 75th percentile (bottom and top of box, respectively), minimum and maximum values (lower and upper whiskers) and outliers (dots). Abbreviations: (B): EU - European Union, nat - national, reg - regional, sci - scientific grant, priv - private source, oth - other; (C) dir - directive, intl - international law, nlaw - national law, sci - scientific interest, mgmt - management/restoration, oth - other reason; (D) EU - European, intl - international, nat - national, reg - regional, loc - local.

Figure 2. Temporal sampling effort (SEtemp) in species monitoring schemes. (Abbreviations: Fig. 1)

Figure 3. Spatial sampling effort (SEspatial) in species monitoring schemes. (Abbreviations: Fig. 1)

Figure 4. Sampling design score (SDS) in habitat monitoring schemes. (Abbreviations: Fig. 1)

Figure 5. Temporal sampling effort (SEtemp) in habitat monitoring schemes. (Abbreviations: Fig. 1)

Figure 6. Spatial sampling effort (SEspatial) in habitat monitoring schemes. (Abbreviations: Fig. 1)

FIGURES

Fig. 1

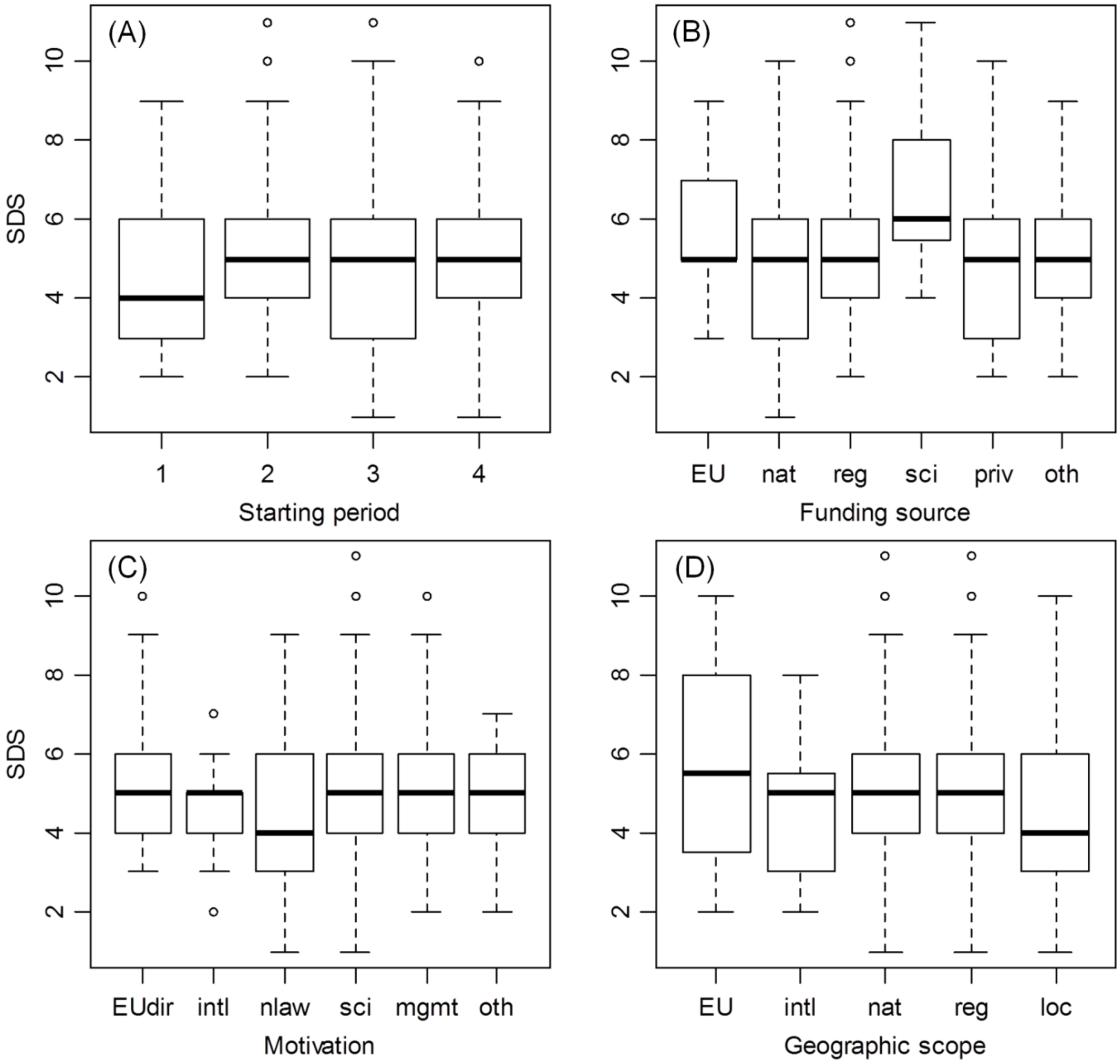

Fig. 2

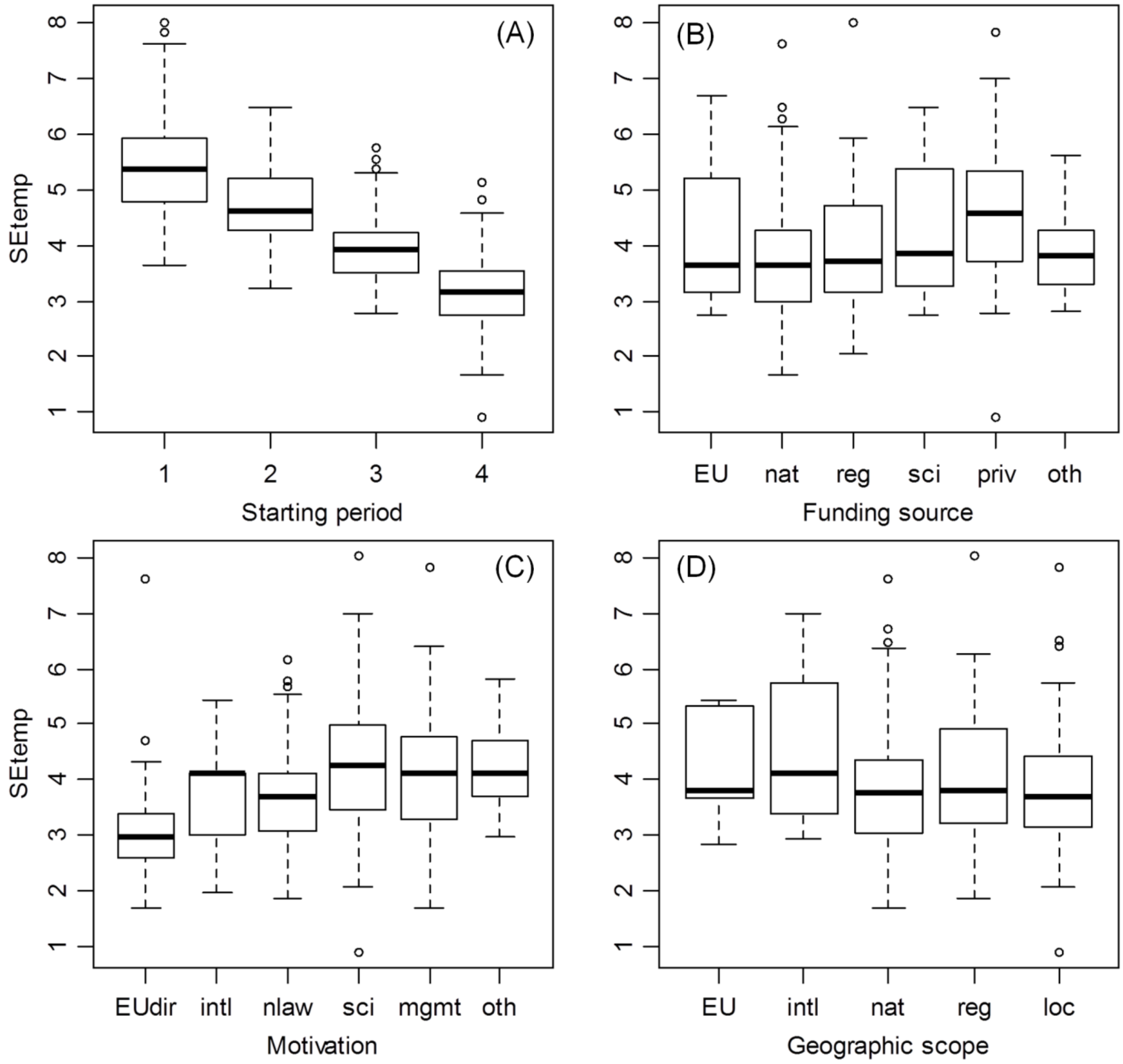
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Fig. 3

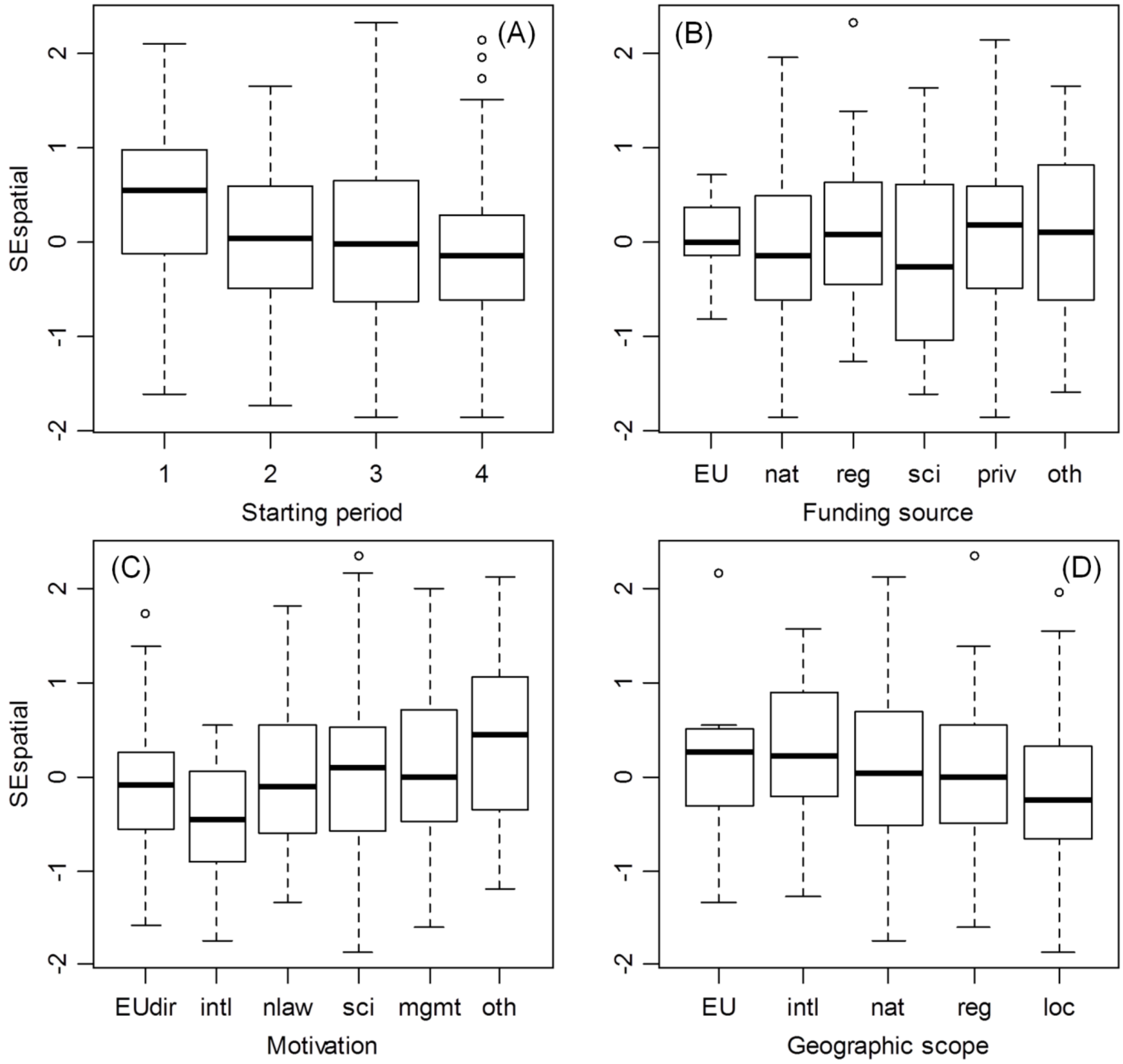
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Fig. 4

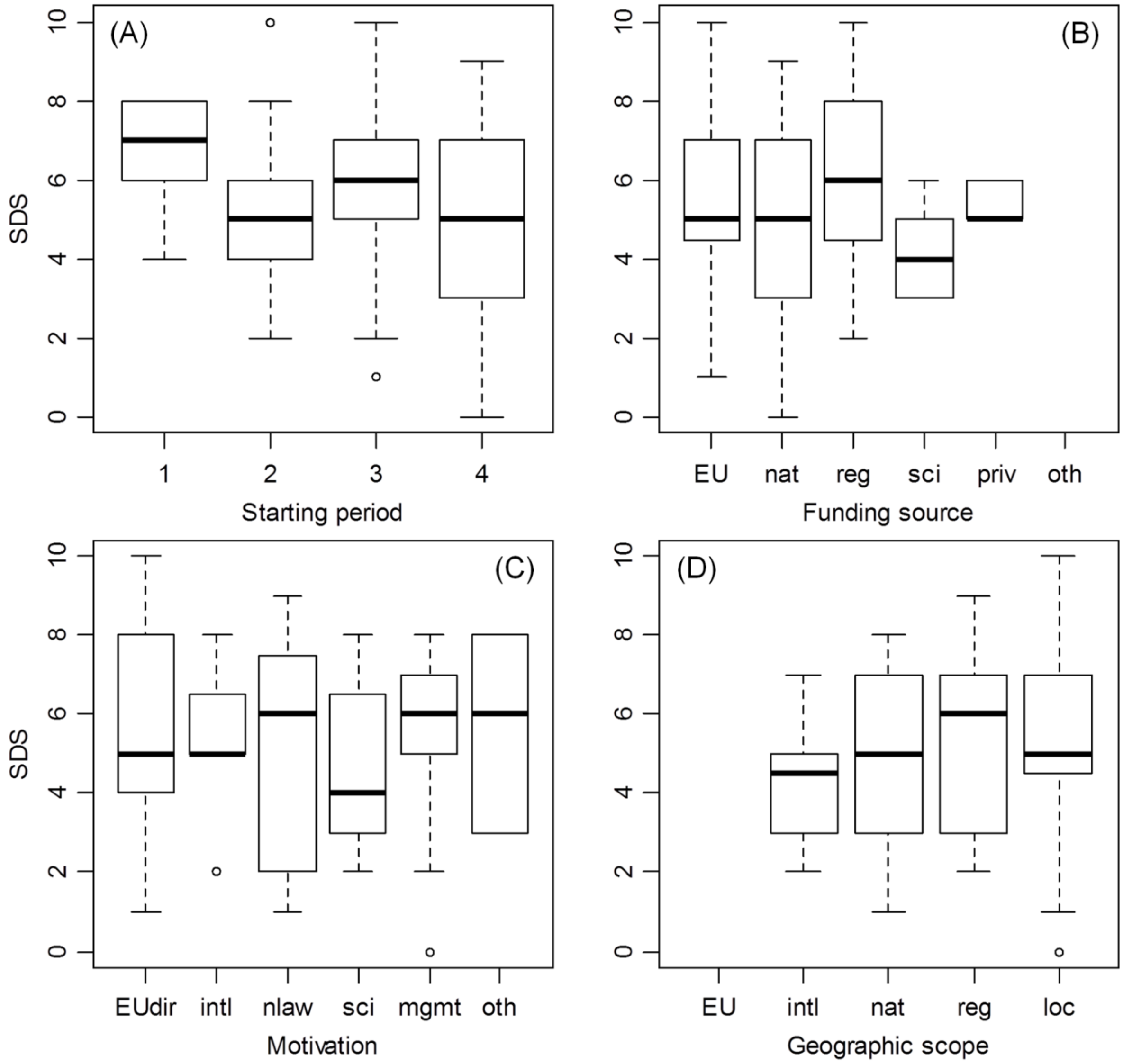
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Fig. 5

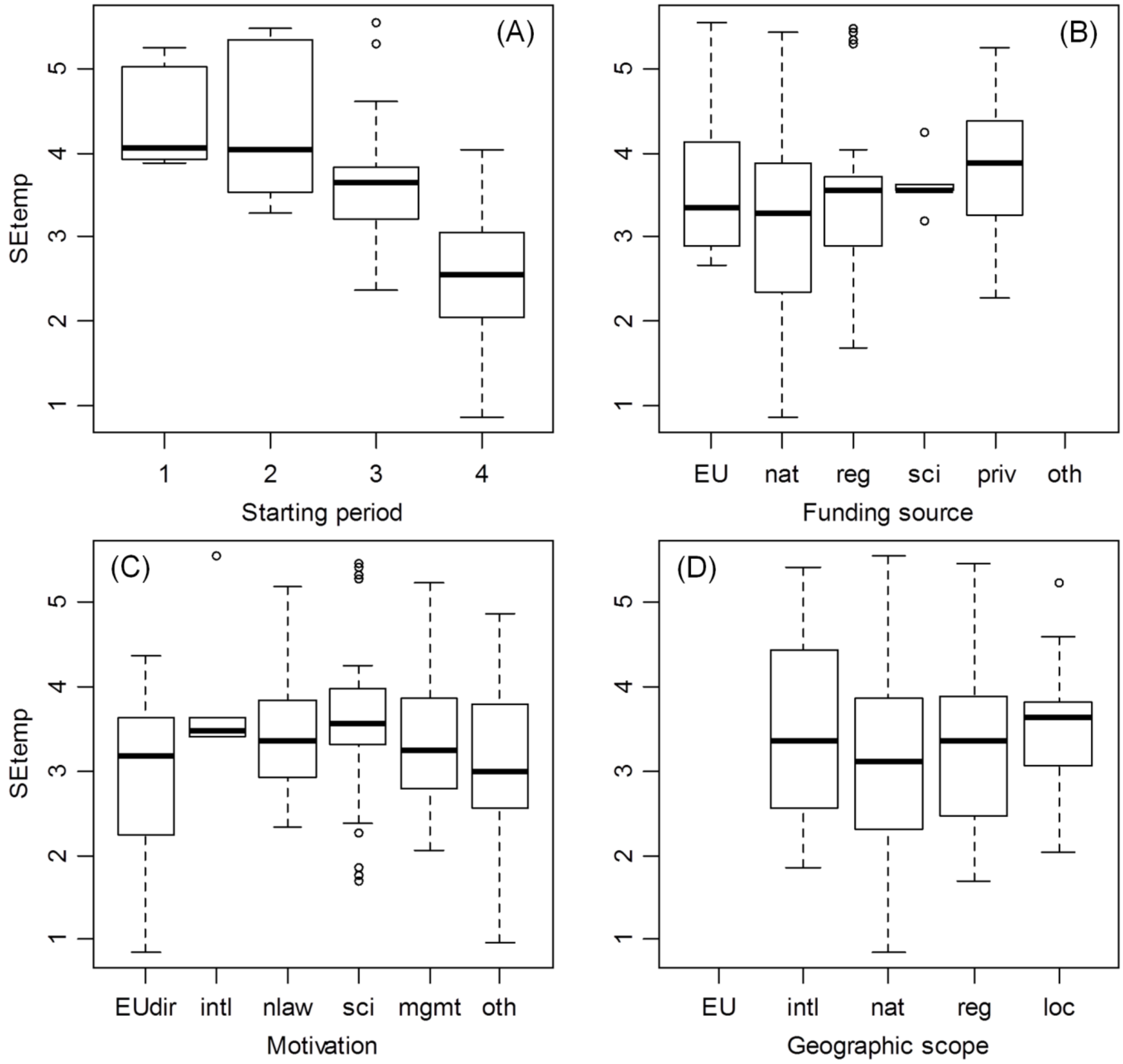
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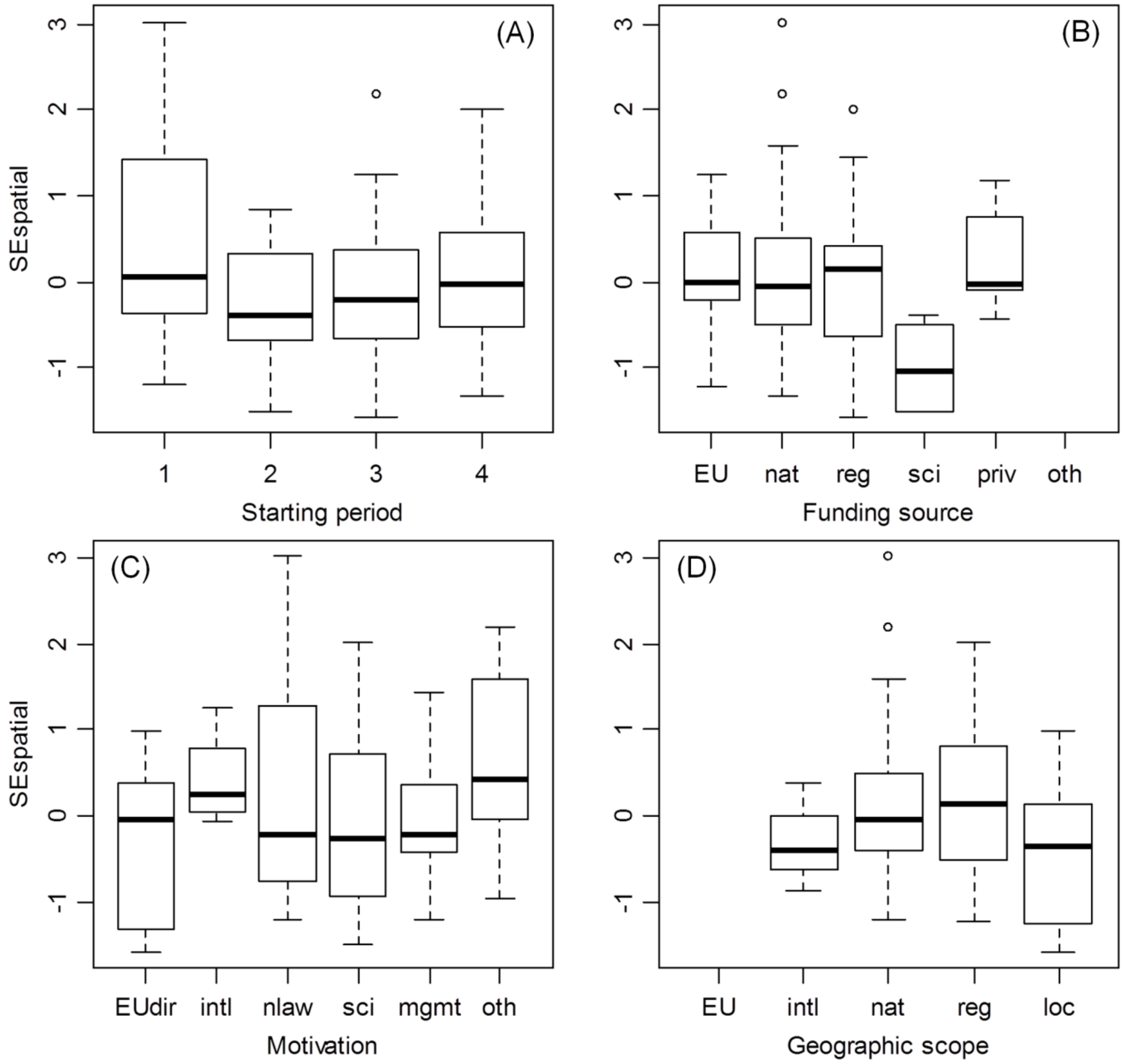
Fig. 6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