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Manual "Best Practice for Monitoring Species and Habitats of Community Interest"

Deliverable 30 of EuMon's Work Package 6

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Summary

A best practice for monitoring of species and habitats addresses the design of monitoring schemes on one hand and the integration of results from monitoring schemes on the other. In all monitoring, clear objectives are essential to identify the relevant biodiversity components, the data types to collect, and the sampling design to deliver the appropriate data. The sampling design should take account of the spatial and temporal variability of the data and should quantify sampling errors. Only monitoring that explicitly covers potential causal factors may draw conclusions about the reasons for observed changes. Adequate replication of sampling is necessary to derive precise estimates, to allow appropriate conclusions about changes in observed variables over time or space. Analytical models should incorporate spatial and temporal heterogeneity and account for measurement errors. Advance analytical methods should be used where possible, as they will accommodate more realistic models and may handle missing data. Operational constraints for monitoring should be adequate to accommodate the objectives of the monitoring under suitable statistical conditions (unbiased, precise estimates) to allow clear conclusions about observed changes. Involvement of volunteers may be an effective way of securing adequate monitoring effort within realistic costs. Integration of information across existing monitoring schemes may be another effective way of strengthening conclusions from monitoring with minimal extra resource use. Integration may result in wider thematic, spatial or temporal coverage, as well as more general and robust conclusions. Meta-analysis is a particularly suitable tool for integration of information from quite variable monitoring schemes. Otherwise, the details of integration will depend on the biodiversity components and data types covered, commonalities in sampling design and data series, and, for habitats, compatibilities in habitat typology, spatial scale and spatial structure. Integration of existing monitoring schemes will be essential to mobilise adequate information about the state and trends of Europe's biodiversity.

1 Introduction

Member states of the European Union and other countries of Europe are committed to halt the loss of biodiversity by 2010 (Balmford et al. 2005). The Habitats and Birds Directives and the Community's other biodiversity policies (e.g. the Biodiversity Action Plans) are important instruments to ensure that this target is reached. However, to verify whether or not we are approaching the target of halting biodiversity loss, the monitoring of biodiversity components, specifically species and habitats of Community interest, is essential. It is a key aim of the EuMon project to contribute to improved practice for the monitoring of species and habitats. This document represents a summary of EuMon's recommendations for "best practice for monitoring species and habitats of Community interest".

By "best practice for monitoring species and habitats of Community interest" we mean that the monitoring should cover relevant components of biodiversity in a coherent manner across Europe. The sampling design, field and analytical methods should also make it possible to discover real changes with adequate sensitivity, i.e., to be able to distinguish real changes from natural variation and measurement errors in time for action. Overall, monitoring schemes should be set up to provide the best results from available resources. This also includes the possibility of integrating data or results from on-going monitoring activities.

Species and habitats of Community interest include species and habitats listed under the Habitats and Birds Directives, as well as other species and habitats of general European conservation interest. In general, the European interest will focus on the state and trends in distribution and amount of these species and habitats. For habitats, there may also be considerable interest in how habitat quality (measured by biotic or abiotic properties) is changing. Except perhaps for the rarest or most concentrated occurrences of species or habitats, where special targeted approaches may be needed, at a strategic level, recommended monitoring methods will be applicable for most species and habitats.

The "best practice for monitoring" as presented here builds on work presented in several other EuMon deliverables.

- Recommendations for monitoring design and data analysis are presented in deliverables D2 and D12 (with detailed case studies in the D12 annexes)
- Recommendations for the coherence, scientific quality and time and cost-effectiveness of monitoring schemes are presented in deliverables D17 and D20
- Recommendations for the integration of monitoring schemes are presented in deliverables D16, D18 and D19.
- Recommendations for operational approaches to participatory monitoring networks are presented in deliverable D24
- Summaries of these recommendations are given in the policy briefs for biodiversity monitoring (D26) and for monitoring and volunteers
- These various elements are presented in an integrated format for the internet through the BioMAT tool (cf deliverable D28)
- These topics are also elaborated in various articles for a special issue of Biodiversity and Conservation where a few of the most relevant articles are Henry et al. (2008) and Lengyel et al (2008)

Here we will give a coherent presentation of critical issues and key points from these recommendations.

2 Designing monitoring of species and habitats

When designing a monitoring programme or scheme several issues should be addressed (e.g., Elzinga et al. 2001, Yoccoz et al. 2001, Green et al. 2005, Teder et al. 2007):

- What is the purpose of the monitoring and what kind of observations or measurements will be needed to fulfil this purpose?
- How should the measurements be taken, i.e. what kind of sampling design will ensure unbiased measurements for the area and time period of interest?
- How should the data be analysed to provide the most information and the strongest conclusions that are relevant for the monitoring issues of interest?
- How should available resources be allocated and other operational conditions be set to produce the best data for the monitoring purpose?

These questions are strongly linked and there is rarely one optimal solution to design a monitoring scheme. Instead, users need to consider the various issues and make informed choices based on their priorities for biodiversity and geographical coverage, statistical requirements and resources available. We offer guidance through the various issues to be considered but no overall 'best' solutions. Figure 1 illustrates the relationship between the monitoring objectives, the data collection methods and the methods for analysis. In addition, there is the issue of allocation of resources and other operational aspects to optimise the results from monitoring.

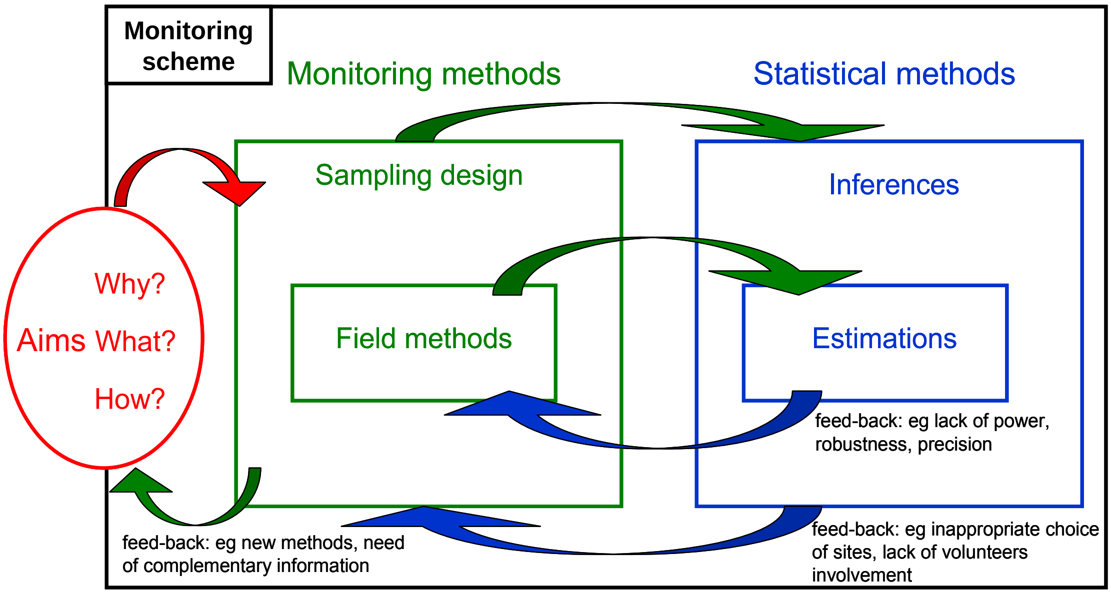

Figure 1 *Basic principles of a monitoring scheme. The aim of a monitoring scheme is to answer specific questions. Sampling designs and field methods (determining field data) are chosen according to objectives. Inferences that can be extracted from the monitoring scheme to answer the general questions of the scheme are directly dependent on the chosen sampling design; the same is true for the biological parameters that can be estimated from the collected data. Ideally, a monitoring scheme should be adaptive: monitoring methods are revised according to weaknesses identified during data analysis, and monitoring goals are revised according to new needs or more precisely defined questions. (From EuMon deliverable D2)*

2.1 Monitoring objectives and biodiversity components – what to observe?

The first step when designing a biodiversity monitoring scheme or programme is to clearly define the questions that the monitoring aims to answer. These questions will mainly cover one of the following three categories: which policy support, which management problem, or which scientific issue. These questions will then determine much of the characteristics of the monitoring: what and how to monitor?

With specific monitoring objectives, the monitoring scheme can be designed to deliver adequate results within an optimal use of resources. However, biodiversity monitoring often aims also to function as an early warning system for potential environmental pressures that are not yet identified. Then more general approaches to monitoring or surveillance must be designed, with a broader set of response variables and including various scales. For the long-term, multi-purpose surveillance will also be advantageous to address general questions, such as the state and trend of distribution or abundance for a set of species, and the causes for their changes. Narrowly targeted monitoring schemes may often end with a change in policy priorities, before they can yield the expected results.

We may monitor species in order to describe their state and possible changes in geographical/spatial distribution or in abundance or population density. We may also be interested in the state and changes in the population's demographic structure or processes, or, for multi-species systems, in the community structure or processes. We may also wish to follow the condition of individuals or the population in terms of physiological, genetic or other properties. To address such monitoring issues, we will typically collect data for species in the form of presence/absence data, simple counts of individuals, more advanced possibilities for following individuals over time (e.g. capture-mark-recapture, telemetry), measures of population structure (by age or size), or measures of individual properties by anatomical, physiological or genetic variables. Table 1 indicates combinations of species monitoring issues (rows) and data types or measures (columns) that are likely to be useful. In EuMon we have primarily focused on species distribution and abundance, less on more complex issues related to within-population structure or aggregated measures for multi-species communities.

Whereas the species concept and the classification of species are reasonably well established, that is not the case for habitats. When studying habitats we can choose to work with primary attributes of habitats, i.e., properties that may be continuously distributed over the landscape without any specific link to given habitat types, such as terrain form, soil structure, vegetation density or 'greenness' etc. Often, however, we need to address units or patches of specific habitat types, and then need to relate these habitat types to a specific habitat classification or typology. Such habitat typologies may originate for different purposes and typically vary much among different countries. Some habitat typologies are developed at the European level, such as the Corine Land Cover typology, EUNIS habitat classification and the classification for the Habitat Directive's Annex I.

Habitat monitoring may focus on several issues. We may be concerned about how much there are of certain habitat types, in terms of area or the number of habitat patches. We may wish to assess the geographical distribution of the various habitat types (as area or patches) over a given target area. Or we may be interested in the size distribution within each habitat type. Even more complex habitat monitoring issues may relate to the spatial layout of the various patches of different habitat types, e.g. the degree of fragmentation of certain habitat types. Such an issue will typically require the mapping of the entire target area and the extraction of various metrics for landscape or patch spatial structure. Finally, we may be concerned about the habitat quality, measured in terms of biological (e.g. species

***Table 1** Data types commonly available in species monitoring and associated questions that may be addressed. The questions of interest will focus on status and trends of various properties at species or community levels.*

	Data types				
Monitoring issues	Presence/ absence	Counts of individuals (frequencies, ranks)	Age or size structure	Individual follow-up (e.g. CMR, telemetry)	Individual measurements of anatomy, physiology, genetics etc
Distribution	**Optimal**	Sub-optimal, rarely adequate data for large enough areas	Not used	Ideal but field intensive; often only on a local scale	Not used
Abundance	Appropriate but relatively low power to detect trends	**Optimal**	Not used	Ideal but field intensive; often only on a local scale	Not used
Demographic structure/ processes		Appropriate for estimation of abundance rate changes only	Appropriate	**Optimal**	Genetics may give optimal information on structure
Community structure/ processes	**Optimal**	Possible but often difficult to get correct detection probability for all species	Not yet adequately developed theory	Not yet adequately developed theory	Not used
Individual condition, quality	Not used	Not used	Appropriate for age/size and related measures	Not directly used but be coupled to individual measures	**Optimal**
Phenology	Appropriate for some measures	Appropriate for some measures	Appropriate for some measures	Not used	Appropriate for some measures
Causes of change	Needs monitoring coupled to potential causal factors, preferably through experimental design				

composition), chemical or physical properties of the habitat types of interest. As is the case for species monitoring, the kinds of issues we may address will depend on the types of monitoring data we have available for habitats. In EuMon we have focused mainly on issues related to the amount, distribution and/or quality of various habitat types.

The types of data we may generate in habitat monitoring will in principle come from either field samples of various properties of habitats or ecosystems or some kind of remote sensing of properties on the ground based on data from aerial or satellite-born instruments. With remote sensing data, both sampled data from the target area and complete coverage of the entire area may be possible, depending on the size of the target area (the spatial extent), the spatial resolution (grain) of the data, and the amount of data that can be handled. In field sampling, the data will most often represent samples of the properties of interest from the target area. With complete coverage of the target area, the data may be converted to maps showing the position of the patches of the various habitat types. EuMon does not cover methodological issues related to create such maps from primary attributes (but see, e.g., Nagendra 2001, Lillesand et al. 2003). Table 2 indicates combinations of habitat monitoring issues and data types or measures that are likely to be useful.

An additional monitoring concern will be to relate observed changes for species or habitats to some underlying cause. This can only be addressed by also monitoring possible causes of such changes, preferably in an experimental design to be able to verify any causal relationship (cf chapt. 2.2.1).

Recommendations

- Most relevant monitoring issues for species are state and trends in distribution and abundance of various species.
- Most relevant monitoring issues for habitats are state and trends in distribution, amount and quality of various habitat types.
- Different monitoring issues can best be addressed by specific types of data:
  - Distribution of species: presence/absence data or counts/relative frequency
  - Relative abundance of species: counts/frequencies/ranks
  - Habitat amount (per habitat type): remote sensing, complete coverage data; maps
  - Habitat distribution: remote sensing, complete coverage data; maps; field samples possible
  - Habitat quality: field samples

Table 2 *Data types commonly available in habitat monitoring and associated questions that may be addressed. The questions of interest will focus on status and trends of various properties for habitats classified to different types. RS data refers to data collected by remote sensing (collected by aerial or satellite sensors).*

	Data types				
Monitoring issues	Sampled field data	Sampled RS data	Complete coverage, field data	Complete coverage, RS data	Map of habitat patches
Amount, number of patches per habitat type	Not used	Not used	Suitable, only for limited areas	**Optimal**	**Optimal**
Amount, area per habitat type	Suitable	Suitable	Suitable, only for limited areas	**Optimal**	**Optimal**
Distribution of habitat types over focal area	Suitable	Suitable	Suitable, only for limited areas	**Optimal**	**Optimal**
Distribution of patch sizes per habitat type	Not used	Not used	Suitable, only for limited areas	**Optimal**	**Optimal**
Spatial structure of patches per habitat type	Not used	Not used	Suitable, only for limited areas	**Optimal**	**Optimal**
Habitat quality per habitat type	**Optimal**	Suitable for some habitat properties	Suitable, only for limited areas	Suitable for some habitat properties	Not used
Phenology	**Optimal**	Appropriate for some measures	Suitable, only for limited areas	Appropriate for some measures	Not used
Causes of change	Needs monitoring coupled to potential causal factors, preferably through experimental design				

2.2 Discovering real changes in time and space

Once the monitoring objectives are decided, which biodiversity components to focus on, and the measures that will be used to represent these components, we need to consider how the data will be collected and analysed. The aim of the data collection and analysis is to get as much information from the data as possible to allow sound conclusions about any observed changes in the biodiversity components that are monitored.

A range of possible field methods are available depending on the species and habitat types that are chosen for monitoring. Such methods are therefore not covered here. Several textbooks and reviews give introductions to standard field methods (e.g. Bookhout 1994, Elzinga et al. 2001). Instead, we provide generic advice on how data should be collected and analysed to make as clear inferences from the data as possible.

2.2.1 Ensuring unbiased estimates of biodiversity measures

Essentially, the generic format for all monitoring data will consist of (1) a specific measure of the biodiversity component of interest, (2) a particular site and (3) a specific point in time where the observations are made, and (4) some measure of the uncertainty of the observations (cf Buckland et al. 2000, 2005, Yoccoz et al. 2001, Magurran 2004, Nichols & Williams 2006). The first of these is related to the aims of the monitoring and the biodiversity components covered, and it is discussed in chapter 2.1 above. The latter three are related to the sampling design of the monitoring and are discussed here.

How to distribute samples in time and space is the crucial step of sampling design and is essential if we want to make reliable inferences from the collected data. It is fundamental for any kind of data collection, including monitoring, but is often neglected in many monitoring schemes. The most important considerations for the sampling design are:

- Where to monitor
- When to monitor
- How to account for errors in measurements

A different issue, which is also important when considering the sampling design, is whether we intend to relate our observations of biodiversity components to any assumed causal factors.

Where to monitor

When we distribute our samples over a certain area, we assume that these samples will in some way be representative also of those parts of the area that we do not sample. There are essentially two main concerns linked to the distribution of samples in space: (1) We need to make sure that we have defined our area of interest (focal or target area) and that our sampling will cover this in some way. (2) We also must make sure that our samples from this area will reflect the spatial variation in the biodiversity components of interest as correctly as possible, i.e., that the samples are as unbiased as possible. Otherwise we cannot be sure that our data will give a representative picture of the changes within our focal area.

The focal area for our monitoring may be defined implicitly as the area approximately covered by the sample points, or explicitly as the area where our monitoring results shall apply. In either case, there must be a reasonable overlap between the defined focal area and the coverage of samples. For habitats and slow-moving or resident species this may be enough to define the focal area adequately. Monitoring animals with high dispersal ability, however, we may not know whether we have contained the population within our focal area or not. There are standard methods for the estimation of such ‘open’ populations (e.g. Seber 1982).

Distributing samples across the focal area can be done in three main ways to ensure that the results will be reasonably representative for the focal area:

- Placing sample points over the focal area by some kind of random process is the statistically most robust sampling design if we have no prior knowledge of relevant spatial variation within the focal area. With random sampling, unbiased estimates of the biodiversity property across the focal area can be achieved by standard methods.

- Systematic sampling (e.g. at fixed points in a regular grid or along fixed transects) across the focal area may be easier to apply in practice than random sampling. Systematic sampling may be reasonably representative of the focal area, but we run the risk (although often small) that the pattern of sampling points may coincide with the pattern in some natural properties that may influence our measurements and make them biased.
- Stratified random sampling will be the most effective, unbiased sampling scheme if we have prior knowledge of natural variation of importance for our chosen biodiversity components. Such natural variation may be characterised as more or less homogeneous strata that may be identified and delimited for the whole focal area. Within each stratum, sample points can then be placed by random, in a adequate number to ensure a defined precision of measurements per stratum. Stratified random sampling will give a more effective distribution of sampling effort than straightforward random sampling in terms of representing each stratum at a defined precision for the estimates.

For some biodiversity components and focal areas of limited extent it may be possible to achieve a complete inventory or mapping of the biodiversity components of interest (e.g. all occurrences of a given habitat type). Such a form of data collection is not sampling and measures will, of course, be representative. However, for most biodiversity components and measures a complete inventory is impractical or impossible. The exception is habitat mapping based on remote sensing data that cover the entire focal area. If remote sensing data can be adequately linked to the biodiversity components of interest on the ground, then a complete inventory of these components is possible. Note, however, that there are considerable technical difficulties in ensuring that remote sensing data properly reflect reality on the ground and that these interpretations are stable over time (e.g. Nagendra 2001, Lillesand et al. 2003).

In many types of monitoring, the placement of samples is done by “expert judgement”. This may often be well considered, and from a subjective point of view give reasonably representative samples for a given focal area. However, from a statistical point of view subjective placement of sample points cannot be seen as unbiased and will therefore make it impossible to make proper statistical inferences from the collected data.

When original sampling has not been done in a representative way, it may be possible to apply ‘post-stratification’ of the data. The data are then related to strata of known natural variation after sampling, estimates are calculated for the properties of interest per stratum, and weighted estimates combined to get a common estimate for the whole focal area.

When to monitor

Many of the same concerns of sampling in space also apply in time. We need to make sure that our samples are taken in such a way that our inferences apply for the time period we want to represent by our measurements. Two types of natural variation then need to be considered: within-year, seasonal or phonological, variation and between-year variation.

Within-year variation in the biodiversity components of interest may be related to the overall seasonal progress of events through the year, governed by light or climate, or to more species-specific phenological manifestations of life history traits. In order to take account of such temporal phenomena in sampling design, we need *a priori* knowledge of these patterns and whether specific periods may be identified. We may then apply stratified random sampling by placing random samples within each temporal stratum. If we do not have such prior knowledge to guide the identification of temporal strata, systematic or random sampling at several times through the year may be the best way to ensure that any within-year natural variation does not bias the sampling results. With only one sample period per year, as is often the case in many monitoring schemes, it is recommended to sample at the same phenological stage each year, or if that proves difficult, to sample at the same calendar date each year. Any within-year seasonal bias should then at least be reasonably constant from year to year.

Many biological phenomena exhibit fluctuations from year to year. Where such fluctuations may be considered as more or less random, without any multi-annual pattern or trend, we may treat such fluctuations as random variation that may reduce the precision of our estimates but that should not introduce any bias. With systematic multi-annual variation the possibility of introducing biases in the sampling must be considered. As long as samples are taken at least once a year, such multi-annual variation is not a problem for the sampling but may be considered in the analysis of the data. For some species with long life cycles or for habitats, where changes may be expected to be rather slow, it may be possible to sample less frequently than every year. To avoid any systematic effects of natural multi-annual fluctuations on the results, two alternative sampling strategies may be considered: If the sampling is done 3-5 times per period of the natural fluctuations, the pattern of these fluctuations may be captured by the data and explicitly incorporated in the analytical model. Alternatively, sampling may be conducted at the same stage of the periodic fluctuation to treat this as a fixed effect. However, this assumes that we have near-perfect knowledge of the natural variation pattern, something which is unlikely to be the case.

In practical adaptation of the monitoring design, the distribution of samples in time and space may be combined, e.g. by randomly sampling only some sites every year to save resources. In addition, other statistical concerns and operational factors will have to be considered.

Accounting for measurement errors
Observing or measuring components of biodiversity in nature will never be perfect. Most measures will be associated with errors due to the observer, the equipment or the observation situation. In order to make reliable comparisons between observations or measurements at separate points in time or space, we need to quantify the precision in our observations or measurements by accounting for measurement errors. This requires replicated sampling for the session (the location and period of sampling) for which we want to produce an estimates for our biodiversity measure. The larger the number of samples per sampling session, the higher will be the precision of the estimates. To include more samples per session it may be tempting to expand the sampling session to include a larger area or a longer period of sampling. This will generally introduce greater heterogeneity in the observations and lead to poorer spatial and temporal resolution. However, an increase in the number of samples may often overcome this increased heterogeneity in terms of the precision (standard error) of the overall estimate (e.g., Hochachka et al. 2000, Hovestadt & Nowicki 2008).

A major source of measurement error in monitoring data is imperfect detection (detection probability < 1). In any monitoring, the recorded value is the product of the true value of the variable of interest and the detection probability. The detection probability may depend on the variable being observed and with the observer and the observation situation. A detection probability < 1 will bias the results if we are concerned with the state and trends of the true value of the variable rather than of a relative measure, or if the detection probability will vary with time or space (as is often the case). Then changes in the recorded values may not reflect the true changes in the variable but, instead, variations in detection probability. Hence, the sampling design should allow for the estimation of the detection probability, although this may require considerable field effort, e.g., by calibrating observations with more detailed studies that allow estimates of detection probability. An alternative may be to keep observation conditions at each site as constant as possible over time and to include a site factor in the analytical model. In some cases there may be independent estimates of the detection probability for some biodiversity measures (typically, for the census of common birds in certain habitats and regions).

Accounting for causal factors
As we have seen in chapter 2.1, monitoring may aim not simply to document changes in certain biodiversity components but also to link these changes to possible causal factors. This is mainly the case where we have reasonable *a priori* hypotheses of what may cause observed changes. Linking monitoring observations to causes will require that we explicitly incorporate potential causal factors in the monitoring design. A few alternatives are possible:

- An experimental design, with random assignment of treatments and controls, with sufficient replicates, is the most robust design to make clear inferences about causal factors. However, in most cases such a design will be impractical or impossible to implement (due to spatial, temporal or resource limitations).
- An alternative design is a before-and-after experiment, where the treatment is introduced partway through the monitoring period. This design will be improved if a non-treated control can be included. Such a design may be suitable for many landscape level treatments, but many types of causal factors may not fit within realistic spatial and temporal constraints.
- Independent measures of possible causal factors at suitable spatial and temporal scales may be related to biodiversity monitoring by comparative studies and correlation analyses. Such a design will not allow any definitive conclusions on causal factors, but strong correlations may strengthen the hypothesis of such a relationship. Such inferences may be strengthened further if some mechanistic model may be fitted to the data.

Recommendations
- Define the focal or target area that the monitoring data shall represent.
- Distribute samples in space according to a random pattern if there is no prior knowledge of important natural variation in space, by stratified random sampling if relevant spatial units (strata) may be defined. Alternatively, systematic sampling or a complete inventory (if possible) of the focal area may be applied.
- Distribute samples in time at random or regular intervals, taking account of natural temporal patterns in the variables measured to avoid bias. If there are regular natural temporal patterns in the variables measured, the frequency of sampling should be higher than that of the natural fluctuation period by a factor of 3-5. Sampling frequency should also be adapted to the requirements for response sensitivity in the measurements (cf below).
- To account for measurement error, replicated sampling within each sampling session is necessary. Considerable replicates may be needed to achieve high precision in the estimates.
- To account for possible causal factors, data on such factors must be incorporated in the monitoring or at least be available for analysis. An experimental monitoring design, with possible causal factors as treatments and a non-treatment control, will be the best basis for making strong inferences about causal factors.

2.2.2 Ensuring adequate response sensitivity

A key concern in monitoring is to discover a real change as quickly and with as high a probability as possible. However, due to natural variability in biodiversity variables, measurement errors and imperfect detection, any observed differences in the observation variables do not necessarily indicate that a real change has occurred. We need to be able to distinguish real changes from natural variation and noise in our observations. We may formulate this requirement as follows: *We wish to discover a change of a certain magnitude with a specific probability within a given time period*.

The general problem is related to discovering the real difference between two or more sampling sessions in time or space, or over several consecutive sampling sessions, e.g. in a time series. One sampling session is defined as a specific area and time period for which we want to characterise the state of our biodiversity variable. The statistical power to discover differences in the states of a biodiversity variable between sampling sessions will depend on the standard errors of the observations from each sampling session, consisting of
- The natural variability of the biodiversity variable or indicator of our choice. Variability that takes the form of regular (cyclic) fluctuations presents a particular problem and should be handled by adopting a sampling strategy that allows for incorporation of the fluctuation pattern in the analytical model (cf When to monitor, above).
- The standard errors of our measurements/observations
- The probability of detecting an occurrence of the biodiversity variable when it is present. This probability will generally be < 1, i.e., it will tend to negatively bias the estimates of the true value. The detection probability may also often vary with space and time.

We need to replicate the sampling within each sampling session to quantify the standard errors of our biodiversity variables. The more replicates, the lower the standard errors, and the more precise the estimate for the true value of the biodiversity variable. Within a confined sampling session in time and space, the natural variability in our observation variable is likely to be lower than over a more extensive sampling session (wider area or longer time period), and so are the measurement errors and any variation in the detection probability. But as indicated above (Accounting for measurement errors), increased sampling may overcome this increased heterogeneity. Also, with very confined sampling sessions there is a need for many more sampling sessions to ensure adequate representation in space and time. Whether a sampling session should be confined (more homogeneous) or broad (more heterogeneous), and thus how sampling effort should be distributed within versus between sampling sessions, will depend on the monitoring objectives and the particulars of the monitoring situation (which variables to measure, area and time period to cover, natural variability in time and space). Hence, only rather general advice is possible here.

Recommendations
- Replicate sampling within the defined sampling session (i.e., the area and time period for which we want an estimate of the value of our biodiversity variable).
- Define the sampling session to take account of the monitoring objectives and the natural variability of the biodiversity variables of interest. A confined sampling session is likely to be more homogeneous, with a need for fewer replicates for a specified standard error, compared to a broad sampling session.
- Distribute the total sampling effort to get a good balance between the need for precision in estimates per sampling session and the need to represent any variability across the focal area and during the monitoring period.

2.2.3 Getting the most information from the data

Once monitoring data have been collected within the framework of the monitoring objectives and the sampling design, we would like to extract as much information from the data as possible. However, the sampling design and the types of data collected will largely constrain the information that any analysis can extract.

Analysis of monitoring data will generally be concerned with
- What is the state of the biodiversity variable for an individual sampling session? The state for individual sampling sessions may then be aggregated across several sites to give an estimate of the state for a wider area (similar aggregation in time is less common).
- What are the differences in the biodiversity variable between individual sampling sessions, in time or space?
- What are the trends in the biodiversity variable across two or more sampling sessions, in time or space?

In most monitoring, the main interest is on trends in time, but it is equally valid to investigate any spatial trends or patterns. There may also be interactions between temporal and spatial patterns, e.g., with different sites having different temporal patterns.

Here we will assume at least that the focal area for the monitoring is defined and that sampling across this focal area is reasonably representative. What kind of analysis that can be performed will then much depend on the type of data collected and whether replicated sampling has been done to allow for the quantification of measurement errors and the natural variability of the biodiversity variable of interest.

Distribution and abundance of species
Data for analysing distribution patterns of species over wider spatial scales may be presence/absence data or counts of individuals. In either case, non-detection may be a problem as it may falsely indicate absence. If possible, the detection probability should be incorporated into the analytical model. Counts of individuals may provide information on spatial patterns in occurrence that presence/absence may not. In the analysis, it will be important to account for any such spatial heterogeneity in the data, e.g. by relating the observations to spatial variation in observation characteristics (observer, weather, time etc), or to relevant habitat or geographical patterns. If the detection probability is known or can be estimated, true distribution estimates can be derived, otherwise only estimates of the minimum distribution (based on positive observations) can be made. For distribution patterns of individuals or local populations, capture-mark-recapture (CMR) and various telemetry methods may give very detailed data on animal distributions and there is a large literature available on the analysis of such patterns (e.g., Bennetts et al. 2001). However, such methods are generally too resources demanding for distribution monitoring except for some species of particular interest (e.g., large carnivores, ungulates).

Counts of individuals are the most common data type for estimates of state and trends in relative abundance of species. Non-detection of individuals is also a concern here, and the detection probability should be incorporated into the analytical model when possible. Alternatively, we must calculate relative abundance values and trends, taking appropriate care of any spatial heterogeneity in the observations (cf above). Temporal variation (or spatial variation between geographical strata) in observed numbers will affect the precision of the estimates and may require some form of weighting (e.g. by the inverse of the standard errors). With more complex data types that allow the follow-up of individuals (like CMR data and telemetry), any spatio-temporal patterns may be explicitly accounted for in the analysis and true densities and population vital rates may be estimated.

Distribution, amount and quality for habitat types
Data on habitat types may range from presence/absence or sampled frequency data of individual habitat types within sample sites, to complete coverage data, typically based on remote sensing data, and complete habitat maps for the entire focal area. Various methods exist for classifying and mapping habitat types from primary attributes like data on land cover and other information from field samples or remote sensing instruments. EuMon has not covered such methods (but see, e.g., Nagendra 2001, Lillesand et al. 2003)..

Measures of distribution or amount for given habitat types can easily be extracted for habitat maps, for the time period covered by the map. Measures of uncertainty will originate from different activities in the process from data gathering to final map construction, and the aggregated uncertainty may be difficult to quantify. Data extracted with current GIS software will often be treated as 'true values', in spite of the underlying uncertainty. However, by appropriate field controls of the map, the overall uncertainty related to classification and delimitation errors can be quantified.

Complete coverage data from remote sensing instruments will have some of the same characteristics as maps, but here the data will enter as primary attributes in a process of classification of habitat types and delimitation of habitat patches. In this process, the classification and delimitation errors may be combined with error estimates for the original remote sensing data to give aggregated measures of mapping uncertainty. Generally, field control is central in the classification process and can contribute to estimation of the mapping uncertainty.

Sampled habitat data from the field will have many of the same characteristics as similar data for species. The same kinds of concern with sampling error and spatial and temporal heterogeneity will apply and similar remedies must be executed. If these data are then used as input in a habitat classification procedure (in spite of their limited spatial coverage), additional errors will be introduced and must be accounted for (cf remote sensing data).

General analytical approaches
Several text book and journal articles describe various aspects of theory, technical methods and software to design sampling schemes and analyse the resulting data (cf, e.g., Seber 1982, Buckland et al. 2000, 2005, Yoccoz et al. 2001, Magurran 2004). The general analysis approaches include General Linear Models, General Additive Models, and various forms of time series analysis, with special methods for more advanced or information-rich types of data (e.g.CMR data). The TRIM software for analysis of counts data and the MARK software for CMR data are typical examples of widely used software for such analyses. Various image analysis and Geographical Information System software is available to help in the classification and delimitation of habitat patches, although algorithms linking the primary data to a given habitat classification will often be quite specific to the habitat classification and the primary data. The specific choice of methods will depend on the particular circumstances of the case in question and is therefore not presented here.

In Deliverable D12, appendices 1-3, several case studies related to species monitoring are described in detail to illustrate both considerations of sampling design and analysis. These cases present realistic examples of the trade-offs and imperfections that will often be unavoidable in practice. They also indicate how these studies may be improved in the future.

Recommendations
- Data for analysis should include replicated measures per sampling session and should account for spatial and temporal differences among sampling sessions, e.g., by incorporating relevant covariates or stratifying the data for sampling sessions with common characteristics.
- The analytical models should incorporate replicated samples, spatial and temporal differences between sampling units, and any potential causal factors.
- Advanced analytical methods should be applied where appropriate, as these will accommodate more realistic models and will be able to handle missing data in time or space.
- Insight from the analyses should be used in feedback to the monitoring objectives and the sampling design to improve these where possible.

2.3 Effective use of resources

The operational constraints on monitoring schemes for biodiversity may include many and complex issues, from institutional arrangements, commitment of sponsors and site managers, ownership and access to monitoring sites, to resources, staff and infrastructure available to conduct the monitoring (e.g. Parr et al. 2002). Such a wide range of issues will be difficult to incorporate systematically in an overall framework for monitoring, as operational factors must be balanced against e.g. monitoring objectives or statistical requirements, but some general recommendations may be offered (below). The main topics of interest in EuMon has been on resources, staff and expertise required for monitoring on the one hand, and the opportunities for effective involvement of volunteers on the other.

Recommendations
Favourable operational constraints for biodiversity monitoring may be listed as follows:
- Clear and explicit monitoring objectives
- A defined focal or target area and identified biodiversity components with relevant variables or indicators measured with scientifically sound and established methods
- Specified and published sampling protocols, quality standards and quality assurance protocols
- Long-term commitment from management and funding agencies, as well as monitoring operators, to ensure that adequate data series are accumulated to capture long-term dynamics.
- A clear commitment to public dissemination of information about the monitoring scheme/ programme and its results
- Adequate funding and other material resources (equipment, infrastructure) to ensure that sufficient sampling can be conducted to satisfy the requirements of representative sampling of the focal area for the time period necessary, as well as the replicated sampling needed to quantify sampling errors
- Adequate number of staff with appropriate training to ensure consistent application of the sampling protocol
- Using volunteers in monitoring will increase sampling frequency and/or coverage for a given resource input and will improve public commitment to monitoring. The increase in sampling activity will in many cases more than compensate for the possible increase in sampling variability.

2.3.1 Resources for monitoring

It is difficult to assess in general how much resources, in terms of money or staff, that will be needed to conduct a specific type of monitoring, and to weigh such resource measures against various statistical requirements. Resource needs will vary strongly with type of monitoring, institutional framework and country. Through analysis of the information recorded in EuMon's database on monitoring schemes in Europe (cf deliverables D17 and D20), we may get some impression of such resource needs. This may then form a basis for recommendations on necessary resources for various types of monitoring.

Analysis of the EuMon database on monitoring schemes (based on data per September 2007) gives us the following key findings:
- There is great variation in the resource use, in terms of person-days and money spent per year, for different kinds of schemes. This is partly related to the spatial and ecological extent of the schemes, their taxonomic or habitat focus, country where they are based, their scientific complexity, and various operational aspects. But considerable variation remains even after these factors are accounted for.
- The annual effort (person-days per yr) per monitoring scheme tends to be far higher (2-5 times) for habitat than species monitoring. This may partly be due to the impact of a few very large and costly habitat monitoring schemes covering large areas and involving a large number of people. Habitat monitoring schemes also tend to have much higher non-personnel costs, probably related to high equipment and running costs.
- For habitat monitoring, 10-20% of the total annual effort per scheme seems to be devoted to data gathering, the rest for data processing, analyses, and management.
- For species monitoring, median annual effort varied from 45 and 55 person-days per scheme for vascular plants and mammals to 122 and 150 for butterflies and birds. Calculated per species monitored, the vertebrates used 12-17.4 person-days per species, butterflies 10.7 and plants 3.1.

- The use of volunteers is much more common in species than in habitat monitoring. However, there is considerable variation among schemes for different species groups in the use of volunteers: amphibian/reptile and bird monitoring schemes use volunteers the most, followed by invertebrates and mammals, with plant monitoring making the least use of volunteers.
- Monitoring schemes in western Europe tend to use volunteers more than in central and eastern Europe. Apparently, this allows schemes in western Europe to keep costs of monitoring lower while keeping a considerable effort in person-days and sampling intensity, thereby maintaining (or improving) scientific quality.

In assessing the cost effectiveness of monitoring schemes, we need to relate the resource needs (or efforts), in the form of staff and costs of materials and equipment for monitoring, to some measure of the 'potential production' of the monitoring scheme (Production/Effort). 'Potential production' may be formulated in terms of area and/or taxonomical/ecological content covered, and scientific quality of the monitoring. Criteria for scientific quality may be related to

- Clear objectives for the monitoring and coherence of objectives, methods and the type of data collected (cf tables 1 and 2) *
- Spatial and statistical representativity of the data collected *
- Ability to detect trends/response sensitivity: statistical power & measurement precision *
- Sampling design refinement: accounting for spatial and temporal variability
- Scientific knowledge required to conduct the monitoring ¤
- State-of-the-art field and statistical methodology applied ¤
- Potential causal factors included in the design

(key issues are grouped by cost effects: * priority criteria with no direct effect on costs, ¤ criteria with likely effect on costs)

Adapting the recommendations of deliverables D17 and D20 somewhat, an evaluation of cost effectiveness of monitoring schemes may involve the following successive points:

1) Geographical extent of monitoring (area monitored per yr)
2) Ecological/taxonomical extent of monitoring (number of species/taxa/habitats monitored per yr)
3) Coherence between monitoring objectives and data types
4) Primary indicator for scientific quality based on combination of assessment of representativity, statistical power, and precision
5) Secondary indicators of scientific quality based on coverage of causes of change, requirement of scientific knowledge, and state-of-the-art methodology
6) Calculate indicator for 'Potential production' based on items 1-5 in some combination
7) Calculate indicator for Effort from personnel needs (time), personnel (costs), material costs, or some combination of these two items
8) Build a composite P/E measure, contrasting 'Potential production' or some of its components to the measures of Effort (resource needs)

We should stress that exploration of these properties for schemes in the EuMon database indicates that a broad comparison of schemes with these indicators is likely to results in great variation in scores, reflecting the great variety of schemes and not necessarily any critical quality differences. It will be most useful to apply these criteria to schemes that have similar objectives and scope.

For planned schemes with given objectives and scope, users may apply the criteria listed above to score schemes and then adjust one or more of the criteria to see if an improved 'Potential production' to Effort ratio can be achieved, remembering that adjusting some quality criteria may also affect the resource needs.

2.3.2 Volunteers in monitoring

Monitoring of species and habitats requires the participation of a large number of people that far outstrip the capacity of professional scientists. Even if there were sufficient numbers of skilled professionals to cover large geographical areas in high enough numbers during peak monitoring periods, the financial costs would be too high. Significant numbers of volunteer naturalists are needed to contribute to the wide range of activities connected to the collection and analysis of biodiversity.

As well as contributing their skill and time, volunteers support monitoring organisations through subscriptions and donations and represent a core of citizens who are committed to nature conservation and management. There is much variation in the amount and types of volunteer monitoring and the organisations in which it takes place. These organisations are defined as *Participatory Monitoring Networks (PMNs*), a broad term that includes a host of very different arrangements and involves collaboration between a range of nature specialists, both professional and amateur. Here are some conclusions from EuMon's work on volunteers in monitoring (cf deliverable D24).

Factors for successful volunteer involvement in biodiversity monitoring

- Socio-political background influences levels of participation
- Different strategies needed for recruitment and retention of volunteers
- Inform volunteers about the use of their collected data
- Several factors motivate volunteers
- Carefully consider relations between professionals and volunteers
- Collaboration with other PMNs adds value to monitoring

Socio-political background influences levels of participation

In European countries the willingness of citizens to undertake voluntary activities of any kind has to be understood in relation to a country's social, political and economic situation. Voluntarism thrives in EU member states with a relatively undisturbed tradition of democratic political institutions and where voluntary associations have long formed a significant portion of civil society. In post communist countries historical circumstances mean that social, political economic and religious factors can prevent the expansion of voluntarism. In seeking to understand the different social contexts affecting volunteer biological monitoring in European countries it is also important to consider the historical and cultural significance of natural history and the kinds of roles played by amateurs. For example, social status attached to being an amateur naturalist varies from country to country and from one era to another. In trying to sustain a viable volunteer base each PMN must adapt to changing social, political or economic circumstances.

Different strategies needed for recruitment and retention of volunteers

Strategies for both recruiting and retaining volunteers differ according to the types of PMN in question. A general measure for success is the extent to which the PMN's attitudes towards volunteering match the desires and aspirations of its volunteers. PMN's can use a range of publicity material and media connections to recruit volunteers. However, any PMN should be wary of recruiting more volunteers than it can manage. The degree of effort needed to bring in new volunteers while consolidating and motivating the existing body of participants requires lots of effort and inventiveness. Interpersonal interactions are important for the retention of volunteers. If numbers become unmanageable, communication and interaction can suffer leading to negative experiences and de-motivation amongst volunteers. Good communication is a key attribute of vibrant PMNs.

Inform volunteers about the use of their collected data

The responsibility and commitment that volunteers often bring to their monitoring activities means that they care about what happens to the data that they produce and what it is used for. There is an extensive spread of expertise, skill and commitment required from volunteers according to the types of PMNs in which they participate. Volunteers are willing to take responsibilities beyond recording work preparing it for publication in bulletins, reports and atlases produced to extremely high standards. All biological records collected by amateur volunteers are personalised to some degree, because they hold unique meanings for the person who on their own accord went out and collected them. PMNs need to inform volunteers about the fate of their data and consult them about decisions relating to data. The greater the internal solidarity within a PMN the less chance there is that its recorders feel separated from the records that represent the nature to which they are so closely attached. Many PMNs depend on websites to reveal the results of their surveys and hold themselves accountable to their volunteers. They also communicate results through reports and other forms of media.

Several factors motivate volunteers

The motivation of volunteers involves a combination of wanting to learn, passion for nature and the desire to be with other like-minded people. PMNs need to cater for the combination of these factors and find creative ways of addressing them. The desire to learn is a hallmark of serious volunteer recorders who demonstrate a hunger for increasing their knowledge and skills. Volunteers talk about their enjoyment of being outdoors and feeling close to nature. This sense of intimacy with the natural world relies on developing a better understanding of how nature works; a goal that is sought through mutually supportive learning partly by engaging with others who share similar enthusiasms. PMNs need to find ways to harness volunteer naturalists' desire to follow their love of nature through the acquisition of knowledge and skills. But they must also ensure that volunteers - driven by interest and passion - have opportunities to become ever more skilful at collecting data. There is a marked tendency among volunteer monitors to want the scientific knowledge they contribute to be placed at the service of conservation. Volunteers also want assurance that their work carries a sufficient degree of scientific legitimacy to have influence in policy domains.

Carefully consider relations between professionals and volunteers
While professionalisation can benefit certain types of PMNs, potential negative effects need to be acknowledged and managed to create a balanced relationship between professionals and volunteers so that neither category of people feel under valued or isolated. Our research suggests that the balance between professionalisation of a PMN whilst retaining the respected status of volunteers is often difficult to achieve and may swing backwards and forwards across the history of an organisation. Problems can arise when processes of professionalisation are allowed to degrade the amateur status and make it appear an inferior version of professional practice. These circumstances can lead to lack of opportunities for amateurs to build expertise through participation in monitoring projects, creating disillusionment among an organisation's membership, dissent between amateurs and professionals and eventual institutional decline.

Collaboration with other PMNs adds value to monitoring
Collaboration with other organisations has many benefits and can be an efficient and cost-effective tool for monitoring programmes. Most PMNs have connections with other organisations for the sharing and management of data and some collaborate for other purposes. Monitoring programmes that are interrelated or even nesting within one another (e.g. monitoring several species and/or habitats) could prove to be extremely effective model for future monitoring in Europe. Collaboration provides a means for PMNs to pool volunteers and expertise of their staff and share the financial burden. There is also an added benefit of creating wider networks for greater dissemination of information and results.

3 Integration of on-going biodiversity monitoring

3.1 Why is integration important?

Biodiversity monitoring is needed to verify the state and trends for biodiversity and the effects of policies to maintain or improve the state of biodiversity. Many biodiversity monitoring initiatives have been launched over the last 20 years, making an increasing number of time series on species, communities, habitats and other components of biodiversity available. However, most of these time series appear to be focused on local areas and/or limited biodiversity components and do not directly indicate general trends of biodiversity (Balmford et al. 2003, Mace 2005). Hence, to get a more robust and representative picture of the state and trends of biodiversity, there is a need to integrate information from single monitoring schemes into indicators that can provide information on broader patterns of biodiversity change (Parr et al. 2002, Nichols & Williams 2006).

Both top-down or bottom-up approaches may lead to integration of biodiversity monitoring. Top-down approaches are based on international monitoring networks, with standardised monitoring variables and protocols for sampling, analysis and quality assurances, as well as common data access, analysis and reporting. However, there are no top-down, global monitoring networks for biodiversity and any such network face formidable practical and governance challenges (Parr et al. 2002). Hence, bottom-up approaches, such as combining available ongoing monitoring schemes, are currently the only realistic possibility for assessing the global state and trend of biodiversity. Despite its importance, integration of information across existing biodiversity monitoring schemes is still poorly developed.

Nevertheless, most biodiversity monitoring schemes have some basic elements in common: they produce indicators of biodiversity components for defined units of space and time, and this can be a basis for integration. Combining monitoring output across schemes may compensate for the three main weaknesses of ongoing biodiversity monitoring (Mace 2005; Pereira & Cooper 2006): (1) fragmentary biological and spatial coverage, (2) no direct compatibility of data sets among initiatives, and (3) insufficient integration of biodiversity monitoring.

Both similarities and complementarities among biodiversity monitoring schemes are of interest when integrating monitoring output from different schemes. If different taxa, countries, or habitats exhibit similar responses to the same environmental change, similarity among schemes indicates that we can make strong inferences on biodiversity states and trends. If biodiversity responses differ in intensity or in direction across schemes, taxa, or habitats, different schemes carry complementary information that may give us better insight in the processes responsible for the changes.

Here, we present the key points about the integration of biodiversity monitoring and give advice on practical issues to be considered when combining existing monitoring schemes. We focus on three topics: (1) the benefits of integrating information among monitoring schemes, (2) the integration of monitoring schemes with different sampling designs, and (3) commonly used statistical methods for integration of monitoring data. Although basic issues will be the same for species and habitat monitoring, we will examine these issues from both species and habitat monitoring perspectives.

3.2 Integration from a species monitoring perspective

3.2.1 Benefits of biodiversity monitoring integration

Improving precision of estimates
As discussed above, the precision of an estimate depends on the sample size and the natural variability of the observed or measured biodiversity property, and the ability of monitoring to detect a significant change depends on the precision of the estimate. Combining information from different monitoring schemes will increase sample size, the precision of estimates, and, hence, statistical power, without increasing sampling effort per scheme. Although natural variability and measurement error may increase with samples from a larger area, longer time period or with different observers, an increase in sample size can still improve the precision of the overall estimate.

Increasing biological coverage of monitoring
Combing information from monitoring schemes that cover the same biological properties (e.g. abundance) for the same set of species will potentially yield estimates with higher precision and wider generality of results. Depending on the type of data collected and the sampling strategy employed (cf below), data or results may be more or less directly combined.

Monitoring schemes covering different biological properties or measures for the same set of species represent complementary information that may add insight to the underlying processes of an observed pattern, e.g. by identifying population processes at a local scale that may assist in interpreting abundance patterns at broader scales.

Monitoring of the same biological properties across different sets of species will indicate if observed patterns have broad taxonomic relevance or if particular species have a deviant pattern. All species may be treated the same in the analyses. By differentiation by functional types, taxonomic group, habitat relations or trophic level, additional insight may be gained. However, theory to guide the analyses of such groups is poorly developed (Buckland et al. 2005, Nichols & Williams 2006).

Many monitoring schemes aim to relate their observations to possible causes of change. Combining information from schemes addressing the same causes of change will increase the robustness of any observed relationships. Meta-analysis methods (cf below) are suitable for such integration, even when the original data and estimates may be difficult to merge directly. If patterns of responses to causes of change appear to differ between species or sites, combined analyses may throw additional light on the mechanisms behind the observed changes. A combination of proper experimental studies with broader correlative studies may expand and strengthen the confidence in a causal relationship.

Monitoring of species and habitats covers different components of biodiversity, often at somewhat different scales. Integration of species and habitat monitoring will give a broader picture of biodiversity patterns and changes. Information of the extent and quality of habitats indirectly tells us something of the potential for associated species, and information about the species will indicate something about habitat quality. Monitoring of specific habitats may give a broader (as well as more fine-grained) geographical coverage than what can be achieved when monitoring species directly.

Increasing spatial coverage
Integration of existing monitoring schemes through space has three main benefits:

- Increasing spatial coverage without increasing sampling effort: Individual monitoring schemes generally have moderate spatial coverage. Integration of local, regional and national schemes will expand spatial coverage without increasing overall sampling effort. This is illustrated by the European bird and butterfly indicators, which are based on national or regional monitoring schemes.
- Accounting for spatial variation in biodiversity components: All biodiversity components vary in space and time. To make correct inferences about the overall state and trends, this variability must be accounted for, either as properly weighted overall averages or by explicitly incorporating the spatial (and temporal) variability in the analytical models. Expanding the spatial coverage of monitoring by integrating schemes over larger areas and/or with more detailed coverage within the target area, will allow more robust extrapolations of the inferences of the state and trends for the biodiversity components of interest.
- Facilitating optimal placement of new monitoring schemes: Integration existing monitoring schemes in space may help to identify gaps in monitoring that can be filled by new schemes. A network of integrated schemes may also function as a framework for exchange of expertise about monitoring to make new schemes more effective.

Increasing temporal coverage
Integration of monitoring schemes that have been running for different time periods may help to expand the length of the period for which inferences about trends in biodiversity components can be made. Integration of results from long-running schemes with infrequent sampling, with results from schemes operating at higher frequency but perhaps for shorter periods, may help to extrapolate information about short-term dynamics in biodiversity components over a longer period. This is particularly critical when monitoring species with cyclic population dynamics.

Recommendations

- Integrating data or information from several monitoring schemes (addressing the same biodiversity phenomena) will in general contribute to increase the sampling frequency and thereby the precision of estimates, without increasing the overall sampling effort.
- Combining information from monitoring schemes varying in coverage of species, biological properties and other variables will contribute to different kinds of benefits:
  - same biological properties & same species: increase sample size and precision of common estimates
  - different biological properties & same species: add insight about processes of observed pattern
  - same biological properties & different species: indicate if observed patterns have broad taxonomic relevance or if particular species have a deviant pattern
  - causes of change: schemes addressing same causes of change will increase robustness of observed relationships
  - species and habitats: give a broader picture of biodiversity patterns and strengthen inferences possible for both species and habitats
- Integration of information from monitoring through space will allow inferences to be made for larger areas, with better coverage of spatial variation in biodiversity components, and will improve spatial and thematic placement of new monitoring schemes to cover identified gaps in monitoring coverage.
- Integration of information from monitoring schemes covering different time periods or having different sampling frequencies may contribute to an increase in the overall period covered or to extrapolation of short-time dynamics over longer periods.

3.2.2 Integration of monitoring schemes with different sampling designs

The sampling design defines how samples will be distributed in space and time to deliver the data about the biodiversity components of interest needed to fulfil the monitoring objectives. Inadequate sampling design can seriously impair the strength of the conclusions drawn from monitoring data. Combining information from monitoring schemes with different sampling designs may partly compensate for potential defects in the design of some schemes. Here we consider ways to take advantage of differences among schemes in three major components of sampling designs: (1) accounting for spatial variation, (2) handling missing data in time series, and (3) measurement error.

Spatial variation and choice of sampling sites
Information from monitoring schemes using site selection methods that objectively account for spatial variation, can be combined without any correction. This pertains to schemes that monitor all sites or that select a subset of sites randomly or systematically. Most monitoring schemes, however, appear to select sites freely or by expert knowledge. These monitored sites may yield biased information for the target area. In this case, data have to be transformed *a posteriori* (by spatial post-stratification and weighting) so that estimates and conclusions derived from the data are as unbiased as possible. To optimize sampling effort while maintaining unbiased site selection, samples may be distributed randomly within specified spatial units (strata) but with unequal frequency between strata, e.g. to represent defined habitats, regions or frequency of the biodiversity components of interest (e.g., targeting rare species hot-spots). When analysing data from integrated monitoring schemes with different stratification designs, the inverse weighting of the stratification must be applied.

Temporal design and missing data
Different existing monitoring schemes will usually differ in the start or end year of monitoring or in their within or among-year sampling frequency. There may also be discontinuities in a monitoring programme due to modifications in the sampling schedule. To compensate for incomplete time series in monitoring, one option is to use statistical models that account for missing data, e.g., generalized linear models, with appropriate selection of data distribution, link-function and parameterization of the effects of schemes and year, intrinsically accounting for heterogeneity among schemes and through time. When combining only a few different monitoring schemes, another solution is to calibrate data across schemes from overlapping portions of the time series.

Accounting for measurement errors
Accounting for measurement errors is essential to quantify the precision of the estimates produced from observations of biodiversity components. An important source of measurement error is imperfect detection (i.e., not all occurrences of the property are discovered). When integrating information from monitoring schemes with and without a design to account for detection probability, it is common practice to ignore detection probability. This is only reasonable if detection probabilities can be considered constant (or vary randomly) in space and time, although this is rarely the case. Alternatively, uncertainty in the biodiversity measure can be quantified by additional information, e.g., from extra fieldwork, or by independent estimates of the error in the joint analytical model. Estimates of detection probability can be extracted from monitoring schemes with appropriate sampling designs and incorporated in analyses of data from monitoring schemes with inappropriate sampling designs.

Recommendations
- Information from schemes monitoring all sites or selecting sites randomly or systematically, may be combined without correction. For monitoring sites selected by stratified random sampling, information may be combined after taking account of unequal weighting of strata (inverse weighting of data). For subjectively placed monitoring sites, post-stratification (and weighting) may be used to ensure less biased representation of spatial variability.
- To compensate for incomplete time series, statistical models that account for missing data should be applied. Cross-calibration among schemes may be possible where time series for the schemes overlap.
- Accounting for measurement errors is essential to allow appropriate common inferences from combined monitoring schemes. For schemes without information on measurement errors, or detection probability, independent estimates of measurement error may be used.

3.2.3 Statistical methods of integration

Integrating information from different monitoring schemes can be done by combining data or combining estimates. It is possible to combine raw data into a single dataset when data are compatible, i.e., quantifying the same biodiversity property by the same measurement unit (or can be reduced to the same unit). When the schemes cover the same biodiversity property but data types differ, separate estimates of the property can be combined across datasets.

Combing data sets
When the monitoring schemes cover the same biodiversity properties with the same data type (measurement unit), raw datasets can be combined directly. Simultaneous analysis with the same paramet-

ric statistical model requires that the data follow the same theoretical distribution (otherwise, non-parametric methods should be used). Combined data can then be jointly analysed to produce an estimate across all monitoring data for the property of interest, and summary statistics can be derived from the integrated dataset. Combining heterogeneous data, the general model may not give a satisfactory fit to all data, and combining all data in a single analysis will not be warranted. Estimates of the biodiversity property of interest should then be extracted separately from each dataset and combined with meta-analysis methods.

When the collected data types differ among monitoring schemes, the simplest method for data combination is to reduce the complexity of information to the lowest common data type, e.g., converting counts of individuals to presence-absence data. However, much of the original information and precision contained in the data is lost and combining estimates instead of the raw data would make more optimal use of the information collected (cf below).

If no common data type (or currency) can be defined due to high data heterogeneity, available information on states and trends can be synthesised into standardized ratings, e.g., by assessment by independent experts based on standard criteria. These ratings are then used as raw data for biodiversity assessment.

Combing estimates – and effects

When data types are too different to modify to a common currency, parameter estimates rather than original data can be integrated. With estimates, measurement errors for the estimates, quantified by their standard errors, are generally known. An analysis using estimates as dependent variables should simultaneously account for estimates of the mean and of the standard error. Such integrated estimates can characterize state or trend of a biodiversity component or the response of this state or trend to an external factor.

When measurement units differ, information from each monitoring scheme can be summarized as the estimate of a single biological property for each separate data set. Integration is achieved by analysing these estimates with a single statistical model. For summary statistics from combined estimates, the recommended method is the geometric mean (instead of the arithmetic mean). A relevant example is the estimation of the average trend of breeding bird populations per major habitat in Europe, where up to 18 EU countries contributed data from national breeding bird surveys that counted individuals per species but with different methods (Gregory et al. 2005).

Rather than integrating information on the biodiversity property directly, one can combine information about the responses of this property to external factors by ***meta-analysis***. The idea of meta-analysis is that results of independent studies are treated as input units for the analysis of a general pattern (Gurevitch et al. 2001). This allows a combination of information from various monitoring schemes regardless of the differences in their sampling designs, objects monitored, data characteristics, and to some extent, even statistical methods applied. If statistical analyses applied on each dataset include the same effect (the same independent variables), then the average effect can

Meta-analysis

Meta-analysis methods use the effect-size concept to integrate estimates of effects across analyses. The effect size is a standardized estimate of the magnitude of the effect of an explanatory variable. A common metric of effect size is the estimate of the slope for the explanatory variable, divided by the standard error of the slope estimate. Effect size is computed independently for each monitoring scheme, and the mean effect size is computed by summing effect size estimates from all monitoring schemes and dividing this sum by the square root of the degrees of freedom (i.e., number of monitoring schemes – 1). If the supposed cause of change has an effect, the mean effect size will depart from 0. Whatever the magnitude of the true effect in each monitoring scheme, the expectation of the test statistic will be negative if there is a general negative effect or positive if there is a general positive effect. The statistical power of the resulting meta-analysis will depend on the magnitude and precision of the effects in the various monitoring, but power should be reasonable in the case of small to moderate effects in all monitoring schemes. A meta-analysis has a good probability of detecting the effect of the cause of change over all observations, which is not the case for separate tests on each single dataset. Another important advantage of meta-analysis is the possibility to identify different patterns of response across monitoring schemes with tests of homogeneity of effect size. This statistical framework allows estimation of average trends across monitoring schemes, as well as discriminating sets of regions with contrasting trends. When only qualitative information is available for the tested effect or cause of change (e.g., significant positive, non-significant, significant negative), non-parametric tests can be used to identify whether the proposed cause of change has, on average, a significant effect over all monitoring schemes.

be computed to infer the average pattern across all datasets. However, meta-analysis cannot compensate for all faults of the data, such as biases in data availability. Meta-analyses should be planned carefully to secure accurate contributions to biodiversity assessment. Accounting for the effects of time or causes of change as explanatory variables for observed trends in the biodiversity properties of interest, is particularly suitable for meta-analysis across many types of studies. However, so far meta-analyses seem to be little employed in making inferences from combined results form different monitoring schemes.

Compensating for differences

When we are combining information from different monitoring schemes, not all contributions may have the same importance for the overall result. Differences in area covered, sampling intensity, precision of estimates, or *a priori* 'value' of some biodiversity components may all indicate that contributions (data or estimates) from some schemes should be given a higher weight than others. Statistically, weighting of data or estimates may adjust for differences in precision or may compensate for biases in the sampling. Using weights to adjust estimates of different precision, the weights should be the inverse of the squared standard errors ($1/SE^2$). If standard errors are not available, surrogates for precision, such as number of sampling units per scheme, may be used instead. Compensating for bias in sampling, weighting will depend on the nature of the bias. Differences in habitats covered or spatial heterogeneity in the response of the sampled biodiversity property may be compensated for by weights proportional to the area of each habitat type or for each region with a specific response in the property of interest. Combining different biodiversity properties, and assigning different importance (weights) to these, is a subjective process that should be well motivated by the monitoring objectives.

Cross-validation

For a given data set, a final model will eventually be selected that supposedly makes the best compromise between a good description of the data and parsimony (low number of parameters). To evaluate the robustness of the conclusions and the generality of the model, cross-validation may be applied. A part of the data is used for identifying the best statistical model, and the remaining part of the data is used to challenge this best model. This process is repeated several times with different data for model selection and testing, and cross-correlations calculated to quantify the departure between model predictions and observed data. When integrating data from different monitoring schemes, several datasets are available. By using one or more data sets to fit the model and the remaining sets to test it, the external validity of the model can be evaluated. This is particularly useful for assessing the robustness of spatial interpolations of biodiversity measures. If cross-correlation is high, the selected statistical model has a high predictive power. We may then conclude that the biodiversity states or trends are similar across schemes. If cross-correlation coefficients are low, however, important causes of biodiversity variation (i.e., effects) are still missing in the final statistical model.

Recommendations

- Results from monitoring schemes covering the same biodiversity properties with the same data types can be readily combined in common analyses. Different data types may be simplified to the lowest common type and then combined in common analyses.
- For heterogeneous data where no common data type can be found, it will be better to make separate estimates and combine these. Standard errors of the estimates are essential for appropriate combination of estimates. Summary statistics for combined estimates should use the geometric rather than the arithmetic mean.
- Meta-analysis is an effective method for combining information on the responses of biodiversity properties to external factors for schemes that differ in design, monitored objects, data types etc. Estimates of effect size are computed independently for each monitoring scheme, and the mean effect size is computed by summing effect size estimates from all monitoring schemes and dividing this sum by the square root of the degrees of freedom.
- Weighting of data or estimates should be applied whenever there are differences in the precision of estimates, biases in sampling or spatial heterogeneity of effects. Weights for precision should be proportional to the inverse of the squared standard errors. Weights for bias in sampling may be proportional to areas sampled or areas of specific effects, or other measures reflecting the underlying bias.
- Cross-validation, using part of the data sets to develop the model and the remaining sets to test it, is a valuable tool to test the robustness of the inferred relationships across results from several monitoring schemes.

3.3 Integration of habitat monitoring

Integration of information from habitat monitoring schemes have many common elements with integration of species monitoring schemes but also some specific issues. The key differences relate to the spatial nature of habitats and the data types and properties that characterise habitats.

By habitats, we refer to the physical, chemical and biological components of a defined geographical area (cf also chapter 2.1). Habitats are characterised by a typology relating the various units to a specific classification and each habitat patch to a specific type, characterised by:
- the number of patches for each habitat type and the size distribution of habitat patches
- the spatial structure or layout of the patches and the geographical relationships between the patches; this may be described as habitat maps or by a variety of spatial statistics or indices
- habitat quality, i.e., the internal properties of individual habitat patches

Habitat monitoring will be concerned with the state and trends in any or a combination of these properties.

The data types collected in habitat monitoring will generally consist of sampled data, mainly in the form of field samples but sometimes by remote sensing, and data covering the entire target or focal area, mainly collected by remote sensing methods (primary data from aerial or satellite-based instruments). For sampled habitat data in particular, many of the same issues as for species monitoring (above) will apply. Monitoring based on remote sensing data will generally be concerned with the spatial aspects of habitats (how much of each type, patch structure and layout), whereas this is not always the case for monitoring based on field sampling. Monitoring schemes may also have a holistic approach, aiming to cover all habitat types in the focal area, or a target approach, focusing on only one or a few habitat types.

Issues of integration for habitat monitoring
When integrating information from different habitat monitoring schemes, we need to clarify whether raw data or some processed information is the basis for integration. Raw data may involve non-processed scenes from remote sensing, whereas processed information can be data already classified to habitat types in the form of a map or estimates of changes in certain properties of the habitats.

If raw data is the basis for integration, then their thematic coverage (one or all habitat types), spatial extent and scale or resolution will be important. A special concern is the integration of data of different spatial scales, as data of fine scales may not be representative at larger scales. With maps or estimates as input, commonalities in habitat typology, spatial extent and scale/resolution will be relevant. How the integration should be handled on the basis of this input will depend on the objective of integration, as certain types of input will be suitable for some objectives but not for others (cf table 2).

Specific issues for integration of information from habitat monitoring involve:
- relating habitat information to the same typology (habitat classification), either by having the same basic typology and habitat types, or by transforming the habitat units into a common typology, possibly at a more aggregated level of classification
- evaluating whether comparable spatial scales are used to identify and measure habitat types or whether they can be converted to comparable scales
- ensuring that characterisations of spatial structure address the same spatial phenomena and that quantifications of these phenomena can be made comparable
- ensuring that aspects of habitat quality address the same quality phenomena and that quantifications of these phenomena can be made comparable.

Avenues of integration
Habitat monitoring schemes may be grouped into 6 classes based on whether they cover spatial aspects or not, use field or remote sensing data, or target one/few habitat types (targeted) or all types (holistic). Integration can take place between schemes within one of these classes or in different classes.

Remote sensing-based monitoring schemes with a holistic approach are highly appropriate for integration (Nagendra 2001). These schemes have a common 'currency' in the form of geo-referenced, remotely sensed spatial information for entire spatial units. The compatibility of such monitoring schemes depends on their technical properties:

- comparable sampling intensity in space (all parts equally measured in the focal area) and time (seasonally and/or according to phenological changes of the habitat types),
- comparable sensors and spectral resolution, comparable conditions for input imagery (acquisition date/frequency, cloud cover etc.),
- comparable mapping scale or spatial precision: the minimum mapping unit (for vector maps) or the spatial resolution (for raster maps) should be similar,
- comparable mapping accuracy, consisting of thematic accuracy (percent of correctly classified habitats) and spatial accuracy (habitat delineation errors),
- compatible map projections and geo-referencing,
- comparable sensitivity to changes,
- compatibility of habitat nomenclatures (habitat classification systems), compatible level of habitat nomenclature hierarchy.

If all of these criteria are fulfilled, the input data sources can be combined for analysis (see relevant reviews etc for concrete methods and technical advice; e.g., Nagendra 2001; Lillesand et al. 2003). Integration may increase the extent and/or the resolution of the area where all habitats are monitored. A common map can be prepared, and common estimates can be calculated. If these criteria are not fulfilled, calibration and interpretation of differences is essential prior to a direct combination of remotely sensed data. If such calibration is not possible, separate maps and separate parameter estimates can be used.

Integration of *remote sensing-based, targeted schemes* will often combine disjunct areas in order to increase the monitored area of focal habitat types. In addition to the criteria presented above, all schemes should cover the same or at least comparable sets of habitats. If monitored habitat types differ between schemes, they may be aggregated to a common level of the habitat classification. The integration can be based on combination of remotely sensed input data (when all the above criteria apply) or on using the input data and/or map results of the scheme with higher spatial and thematic resolution to support and validate results in the less detailed scheme. A special case is when several monitoring schemes each monitoring a different target habitat type within some common area are integrated, thereby broadening the spectrum of habitats monitored.

Field mapping-based, holistic schemes are common, but usually cover widely different geographical areas. Even if spatial coverage is close to 100%, several issues deserve attention:

- the proportion of the focal area actually sampled and refinement of the sampling strategy (e.g. site selection randomly or systematically)
- the use of permanent plots/quadrates/transects in subsequent sampling occasions
- constancy of sampling intensity in space and time, across habitats and habitat types
- method of obtaining information for non-sampled areas (extrapolation, other sources etc.)
- comparability of precision (ability to detect trends or changes in the habitats) and error rates (e.g. measurement of observer biases)
- quantification of errors in mapping and data processing, e.g., by inherent variability of the attribute vs. accuracy/precision of measurement
- habitat classification system used

If these issues can be resolved, integration will lead to an increase in the area monitored. However, the results may not be relevant for non-sampled areas or large spatial scales.

Field mapping-based, targeted schemes are concerned with one or a few habitat types, monitored in several distinct sites with similar or different mapping methods. Integration of such schemes is straightforward if the schemes monitor the same (group of) habitat types. Then only differences in field mapping methodology are important from the perspective of integration. If different habitat types are monitored within some common area, integration can be used to broaden the spectrum of the habitats monitored.

Integration across classes: Combination of remote sensing-based and field mapping-based schemes with a holistic approach may be advantageous when both are complementary in habitat attributes covered or when the combination is more cost and time-efficient. High precision field survey data may support interpretation of remotely sensed data or validate the remote sensing-based mapping and monitoring results (ground-truthing). Field mapping can provide additional information on environmental variables not accessible to remote sensing. Remote sensing may complement or adjust spatial information obtained by field mapping, by providing information on spatial patterns of the habitats (e.g. fragmentation, connectivity) that are difficult to detect in field mapping. Criteria for such integration are as follow:

- comparable areas and spatial scales used in each scheme,
- compatibility of habitat nomenclatures (habitat classification schemes), compatible depth of habitat nomenclature hierarchy, exhaustiveness of field mapping,
- comparable thematic precision,
- comparable monitoring/mapping accuracy,
- comparable sensitivity to changes (ability to detect trends),
- common data formats, compatible data management systems (the latter is not necessary if a scheme is only used to validate the results of the other scheme).

Combining holistic and targeted schemes, the latter can complement the holistic scheme in the common area, where the holistic scheme does not adequately cover certain habitats. A set of targeted schemes that is complete enough over a common area can be combined into a holistic scheme. If the set of targeted schemes is incomplete for a common area, it can still be used to provide additional spatial and thematic detail in some important parts of the common area.

Recommendations

- Integration of habitat monitoring schemes must address
    - common typology (habitat classification),
    - comparable spatial scales
    - same spatial phenomena
    - same phenomena of habitat quality
- Integrating remote sensing-based monitoring schemes with a holistic approach is based on a common currency of spatial information, that can be the basis for common maps and estimates. Ideally, the information should have comparable sampling intensity, data quality, spatial precision, mapping accuracy, and compatible mapping projects and habitat nomenclatures.
- Integration of remote sensing-based targeted schemes may expand the total area covered, but schemes should cover comparable habitat types.
- Integration of field mapping-based, holistic schemes may increase the area covered, but may not be relevant for larger areas. Schemes must be considered in terms of proportion of focal area covered, repeated sampling from permanent sampling units, sampling intensity in time and space, comparable precision and error rates, and habitat classification system.
- Integration of field mapping-based, targeted schemes may expand the area covered, if they cover the same habitat types, or broaden the habitats covered, if different habitat types in the same area are covered.
- Data from remote sensing-based schemes over larger areas may be combined with more detailed data from field mapping-based schemes and provide calibration of remote sensing data as well as additional data on environmental variables. Schemes must be considered in terms of comparable areas and spatial scales, compatible habitat nomenclatures, comparable thematic detail, monitoring accuracy and sensitivity to changes.

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